Transactions of the Sixty-ninth North American Wildlife and Natural Resources Conference

Conference Theme:
Resource Stewardship in the 21st Century: A Voyage of Rediscovery

March 16 to 20, 2004
DoubleTree Hotel and Spokane City Center
Spokane, Washington

Edited by
Jennifer Rahm

Published by the
Wildlife Management Institute
Washington, DC
2004

These *Transactions* are reviewed and proofread by the Wildlife Management Institute. Unless peer review for scientific accuracy is initiated by the author(s) or session chair, no such detailed editorial treatment is provided or implied. Conference presentations may not be included in the *Transactions* if the written papers do not follow the prescribed guidelines or if their content is deemed by the editor to be unsuitable.

Additional copies may be procured from:
WILDLIFE MANAGEMENT INSTITUTE
1146 19th Street, NW, Suite 700
Washington, DC 20036

The Wildlife Management Institute acknowledges special assistance at the 69th North American Wildlife and Natural Resources Conference by *Celeste Ruth*, *Mandy Scott* and *Matt Dunfee*

Transactions of the
69th North American Wildlife
and Natural Resources Conference
ISSN 0078-1355

Printing by Sheridan Books, Inc.
Ann Arbor, Michigan

2004 COSPONSORS
of the
69th North American Wildlife and Natural Resources Conference

Platinum

The Nature Conservancy
Rocky Mountain Elk Foundation

Gold

National Wild Turkey Federation
Sierra Club
National Shooting Sports Foundation
U. S. Geological Survey
U.S.D.A. Animal and Plant Health Inspection Service
U. S. Fish and Wildlife Service
U. S. Bureau of Reclamation

Silver

Boone and Crockett Club
U. S. Bureau of Land Management
U.S.D.A. Forest Service
U.S.D.A. Cooperative State Research, Education and Extension Service
Recreational Boating and Fishing Foundation

Bronze

The Conservation Fund
Ducks Unlimited, Inc.
Theodore Roosevelt Conservation Partnership, Inc.
The Wildlife Society
Federal Cartridge Company
Izaak Walton League of America
National Rifle Association
Congressional Sportsmen's Foundation
Safari Club International

The Wildlife Management Institute
appreciates and respectfully acknowledges the special partnership, assistance and cooperation of these cosponsors.

Contents

Opening Session. On the Trail of Lewis and Clark: The Quest for Conservation

Special Session One. Managing Mammalian Predators and Their Populations to Avoid Conflicts

Special Session Two. Our Water Resources: A Candidate for Listing?

Special Session Three. Producing Energy and Conserving Wildlife

Special Session Four. Fire Management: Burning Issues

Special Session Five. Wildlife on Wheels: The Marketing of Today's Outdoor Experience

Special Session Six. Policy Implications from Long-term Studies of Mule Deer and Elk: A Synthesis of the Starkey Project

Opening Session.
On the Trail of Lewis and Clark: The Quest for Conservation

Chair
Rollin D. Sparrowe
Wildlife Management Institute
Washington, DC

Cochair
C. Thomas Bennett
International Association of Fish and Wildlife Agencies
and Kentucky Department of Fish
and Wildlife Resources
Frankfort

Opening Remarks of the 69th North American Wildlife and Natural Resources Conference

Rollin D. Sparrowe
Wildlife Management Institute
Washington, DC

Welcome to the 69th North American Wildlife and Natural Resources Conference. This is the first time this conference has met in Washington or in Spokane. The conference theme, once again, focuses on our historical legacy—this time the historic "voyage of discovery" by Lewis and Clark. The 200th anniversary of that unique exploration of the Northwest has captured wide attention.

The "Quest for Conservation" byline for the plenary session is appropriate. The Lewis and Clark expedition led to rapid settlement of the western United States and to the application of the fabled Jeffersonian ideal—that every human and family should be working the land—that paved the way for both success and failure. We still deal with these realities now. For example, the recent Klamath Basin disputes over water, endangered fish, migratory birds, the Endangered Species Act and the needs of farmers have their roots in what happened after Lewis and Clark explored the Northwest. People were encouraged to settle and develop land and water in ways that now do not easily balance human needs with resource sustainability. We are still on that quest for conservation to learn how to manage and to live on the arid landscape of the West. The Endangered Species Act appears as the devil to some, as ironclad responsibility to others, but it just cannot replace the broader burden of responsibility for the western landscape; it is ours.

This is the third year of the new format of this conference that intermingles committee work and formal sessions during the week. As many as 150 sessions of working groups, councils, coalitions, committees of the International Association of Fish and Wildlife Agencies, and federal agency staff meetings are negotiating progress in the leadership business of wildlife management and conservation. The diversity of these professional activities continues to grow.

Please, note the rich history of this conference on page 24 of your program. Since the first National Game Conference in 1915, this meeting has been sponsored and administered by the Wildlife Management Institute (WMI). Our philosophy at WMI is that it belongs to all of you as participants, cosponsors, presenters, program committees and fellow natural resource professionals. Next year the conference follows custom and is in the Washington, DC area; several years beyond 2005 are already under contract (also noted in the program) as we move geographically around North America to afford professionals a chance to participate.

Formal conference sessions focus on issues of contemporary importance. On this plenary session, we are happy to welcome agency leaders and representatives of organizations engaged in conservation. Sessions on mammalian predators, water resources, energy and wildlife, fire management, all-terrain vehicle use, and an all-day symposium on results of long-term research on elk, mule deer, their habitats and their public use are some of the issues of the day that cross international boundaries into Canada and Mexico. These topics hold great interest here in the West, but they are conservation issues equally important across all of North America. You may have to manage your time to go between work sessions and formal conference sessions, but it will be worth it. We hope you will choose to celebrate another successful conference with us at the banquet on Friday evening.

At last year's conference in North Carolina, WMI and its agency and organization partners developed a field demonstration of the benefits of controlled burning to enhance habitat. Media were invited and the experiment worked well. This year a field trip and media event before this formal session has highlighted the huge threat to fish and wildlife habitats and to ranching and farming that is posed by invasive species. Some of you recall a plenary presentation at the 65th conference by Jerry Asher of the U. S. Bureau of Land Management on this huge threat that, by some accounts, is reducing habitat productivity by thousands of acres each day. In 2000, when he gave this presentation, there were an estimated 70 million acres (28,327,995 ha) of invasive, exotic weeds in eleven western states. The solutions, he said, are up to the people in this room. We will continue to select such issues in a renewed effort to cast light on them.

Insights on modern resource management across North America in the often contentious public arena can be gained by a focus on the tremendous energy that has unfolded in the past few years in the hunter/angler/conservation community. This connection between responsible use of fish and wildlife and human willingness to conserve them has been a part of this conference since it began. At the 67th conference, we presented a day-long combination of two sessions that explored why people hunt and the relevance of hunting to conservation. That same year the newly-organized American Wildlife Conservation Partners, catalyzed by the venerable Boone and Crockett Club and an impressive array of 37 hunter-conservation groups that signed on, delivered a broad conservation agenda to the President and resolved to better connect to the Administration and the President personally. A year later, the Theodore Roosevelt Conservation Partnership linked hunters and anglers in an action mode on select issues, with intent to engage the grass roots.

These groups formed from the premise that access to presidents and their cabinets had become politicized and that our legitimate and important voice was not being heard. Contrary to some interpretation, this resolve was, and remains, to get the conservation agenda of hunters and anglers before every Administration and President—not just one.

There has been considerable publicity, many questions, media treatment (that seems mainly to be looking for political spin) and outright curiosity about a historic meeting in December 2003, between the President, some of his Cabinet and staff members, and twenty organizations referred to by the press as "the hook and bullet crowd."

Equal curiosity and speculation has been generated over a decision the President made shortly after the historic meeting to reaffirm his Administration's policy of seeking "no net loss of wetlands." This was a hugely important, positive step for wetland conservation. Speculation centers on whether hunters and anglers somehow forced the President to make this decision. Some immediately said it meant nothing unless detailed orders were issued to all federal agencies. On balance, many sat up and took notice. As one who was there, I heard the President say he would soon make a personal decision that would conserve wetland, and he did.

One of the questions I refer to is, "What is of interest to hunters and anglers?" Some issues of importance to hunters and anglers that have been taken to the President and to the current Administration, include:

- the need for conservation tax incentives for private land
- the value of conservation easements to preserve farm and ranchland and lifestyles
- continued conservation of wetland to meet the no net loss goal
- caring for fish and wildlife, while seeking energy independence
- support for conservation provisions in transportation legislation reauthorization
- access to public and private land for hunting and fishing
- state funding for wildlife programs
- fishable water initiatives
- fish, wildlife and recreational considerations in forest planning
- careful implementation of Healthy Forest legislation.

These are solid, centrist issues about conservation of land, fish and wildlife, and they embrace the human presence on the landscape.

What unifies this array of diverse organizations is a desire for positive outcomes for fish and wildlife resources. There is a strong focus on reengaging hunters and anglers at the grassroots level. Hunting and angling conservation groups focus strongly on partnership with agencies to get work done on the ground. Most are not prone to litigation and seek legislation only when dialogue and joint work fails. They have decided the best way to get things done is to be prepared to work for positive outcomes on a short-term and long-term basis, as needed.

In a recent discussion with leaders in the Council on Environmental Quality, we found that implementation matters. Getting a bill passed, a decision made, or a positive statement from a chief executive asserted are all essential steps. The rest of the story is that favorable implementation comes through hard work on the ground with agencies. This is the role that we at WMI have played for decades, working with many of you attending this conference.

Another insight is that many of the groups who met with President George W. Bush had actively supported passage of the Healthy Forest Initiative legislation. We did not all agree that legislation was needed, but all agreed that a more active approach to forest management was essential. It was obvious that some sort of legislation would pass, so we supported it with a clear desire to be a part of the solution, including having to work for careful and effective implementation. Whether to make good on the no net loss objective for wetland or to manage forests to reduce fire hazards to people and to create wildlife habitat, implementation is what matters.

What is next on the agenda for hunters and anglers? At that meeting with the President, we heard him say that he believed that energy development should occur in an environmentally sound manner. Seeking positive attention to the welfare of fish and wildlife resources as the country seeks energy independence is a task that will take work and direct help from the Administration. It will not be an easy task, but clearly the door is open, and we intend to step through it. You have already heard that the list of resource issues important to hunters and anglers is long, and it will transcend this Administration and others to come. Although recent focus is on events in the United States, wetland, energy and many of the issues I mentioned affect Canada and Mexico.

I emphasize this resurgent movement by hunters and anglers for several reasons. Much of this activity is predicated by a belief that strident advocacy driven by litigation is pulling the conservation movement apart. Many perceived gains occur without buy-in by affected public and are not the best road to enduring public policy. The treatment of human presence and the responsible use of resources as something separate from our wildlife heritage is not reasonable, and it will not generally be accepted by society. Finally, there are many hunters and anglers to impact if issues are kept in front of presidents, members of Congress, and the public. I assure you that we intend to do that from now on.

Finally, let me thank the diverse array of agency and nongovernment cosponsors of this conference. You will find their names listed at the front of your program. Please join them and WMI in making good use of the fine program ahead for the next several days and, once again, help make progress for conservation.

Remarks of the U. S. National Park Service Director

Fran P. Mainella
U. S. National Park Service
Washington, DC

Thank you so much for the invitation to address this opening session of the North American Wildlife and Natural Resources Conference. As Director of the U. S. National Park Service, I feel particularly drawn to the theme of this year's meeting: Resource Stewardship in the 21st Century: A Voyage of Rediscovery.

The national park idea was born in the 19th century when Yellowstone National Park was set aside by Congress. It was refined with the U. S. National Park Service's Organic Act in the early 20th century, when many of our early national parks were established. The national park idea shares its roots in conservation with the hunting and fishing communities. Hunters and anglers were the first conservationists and their actions over the past century and a half have formed the basis of the United States' conservation successes. The U. S. National Park Service is grateful and appreciates what hunters and anglers do for conservation today. Although most of the parks are not open to hunting, many do offer fishing opportunities. More importantly, national parks provide habitat and security for all wildlife, including those species that provide outstanding hunting and fishing opportunities on the public and private land that surrounds these parks. With the support of the hunting and fishing community and of others, the national park idea was born.

Later in the century we added the national seashores, national recreation areas, wild and scenic rivers, and national preserves; there are now 387 special places in the National Park System. But, it is in this century that the national parks are being rediscovered, and that is what I want to share with you today.

Being entrusted with these parks is both a great honor and a great responsibility. Improving our national parks is one of our priorities. One of the key tenets of U. S. National Park Service's philosophy and mission is to protect and to manage parks both for the enjoyment of the North American people and to prevent the impairment of park resources. To do this important job well, we need to understand these resources better, to know the condition that they are in, and to restore resources when they become degraded.

At the beginning of this 21st century, Congress funded the U. S. National Park Service's Natural Resource Challenge. The "Challenge," as we refer to it, has been our charter of rediscovery in the national parks. We are now in the final years of the Challenge, and we have already learned an enormous amount about the United States' parks. We are immersed in conducting the 12 basic natural resource inventories called for by the Challenge. These inventories range from documenting which plants and animals live in parks to mapping the soils that sustain them. Resource-based inventories from air to geology to water and biological resources are also part of the Challenge.

The National Park System is now organized into 32 ecological networks that are facilitating "vital signs" monitoring of the parks. Vital signs monitoring is just what the name implies—taking the pulse of park resources, so we get early warning of potential problems. This process, with its extensive public input, is now bearing fruit. We will have better information to take to the public in our management plans and environmental compliance documents.

And, we are doing much more than just rediscovering what we have in the parks, we are taking active steps to protect those resources. In 2003, for example, we have spent almost $23 million to help individual parks protect endangered species and to control invasive species.

Let's now take a moment to look at how we are facing specific challenges, starting with the threat of invasive plants. As you know, those spectacular scenes that Lewis and Clark observed two centuries ago were, in fact, complex communities of native plants and animals that had evolved over millions of years. The natural balance within these communities is now threatened by the invasion of nonnative plants. For example, leafy spurge, an import from Eurasia, easily replaces the grassland of the northern Great Plains. And in my adopted home state of Florida, melaleuca trees, from Australia, threaten to replace the wet prairies of the Everglades and all of the animals that depend on them. Last year, I had the honor of chopping the last known melaleuca tree in Big Cypress National Preserve. In 1995, there were about 100,000 melaleuca-infested acres (about 42,000 ha) in the preserve.

Today, invasive plants infest approximately 2.6 million acres (1.1 million ha) in the National Park System, reducing the natural diversity of these special places. Drawing funds from the Challenge, the National Park Service has established rapid response exotic plant management teams (EPMTs) to control invasive plants.

Modeled after the approach used in wildland fire fighting, EPMTs provide highly trained, mobile strike forces of plant management specialists who assist parks and cooperators in the control of invasive plants. The 16 EPMTs—from the partnership with Florida, based at Everglades National Park, to the Columbia Cascades EPMT, here in Washington—have been lauded for their eradication efforts. The U. S. Fish and Wildlife Service is now establishing similar teams, and we are working to identify ways that we can collaborate to tackle this huge problem.

I know that invasive species are not the only issue where we share pressing concerns; another one is the spread of wildlife diseases. Let's discuss chronic wasting disease (CWD) in deer and elk. CWD occurs in Rocky Mountain National Park, in Colorado, and Wind Cave National Park, in South Dakota, and CWD has been detected in areas adjacent to several other national parks. I have instructed our park staff to be vigilant in monitoring for the disease and to prevent the introduction or spread of the disease. Until we have a better understanding of the distribution of the disease, we will tightly control relocating deer and elk into or out of parks. Parks will coordinate and cooperate with state agencies in CWD monitoring and management. We were pleased to receive a national award from the International Association of Fish and Wildlife Agencies (IAFWA) in 2002 for our work with Colorado in addressing CWD research and control. To meet the needs of parks and their neighbors, we have doubled our spending on CWD to $1.3 million in 2004.

The U. S. National Park Service is also committed to reducing the risk of brucellosis to livestock in the Greater Yellowstone Area, while conserving wild and free-ranging elk and bison herds. This year we will implement a pilot project in which we will vaccinate young, nonpregnant bison against brucellosis if they are captured at the U. S. National Park Service trap site near the north boundary. Meanwhile we will continue the implementation of the Interagency Bison Management Plan as a means to reduce the risk of brucellosis transmission from wildlife to livestock.

It is not only big game diseases that have our attention. Parks are cooperating with the U. S. Department of Agriculture (USDA) and Texas, so gray foxes in Big Bend National Park are vaccinated for rabies. Disease issues such as these have made us realize that today wildlife management is frequently a socially charged and controversial endeavor for the national parks.

Because I believe that communication on controversial issues is so important, last fall I issued a director's order, which recognizes that the public plays an essential stewardship role in taking care of national parks for the enjoyment of present and future generations. By working closely with the public at the earliest stages of planning for projects and programs, we can build strong support for and understanding of the decision-making process and sometimes even for the outcome that is produced.

During the 19th and 20th centuries, the U. S. National Park Service concentrated on protecting and defending park boundaries. We now realize that this approach alone will not truly protect parks; we also need to work better with our neighbors and visitors. For example, did you know that recreational hunting is currently allowed by legislation in about 60 places in the National Park System? Some people might be surprised by this number, but I know that you are not because all hunting on park land is conducted in accordance with applicable federal and state laws and regulations and in consultation with state agencies, which usually set the seasons, bag limits and general regulations. And, recreational fishing is allowed in all parks except where it has been specifically prohibited by federal law.

Recently, there has been a concerted effort on the part of animal rights groups to challenge the hunting programs in national parks. We have been and will continue to vigorously defend these programs, not only as being legal activities authorized by the establishment of legislation for these parks but as sound resource management and conservation tools conducted in cooperation with state wildlife management agencies that are consistent with sound biological data and goals. As the number and frequency of these lawsuits increase, and we have every reason to believe they will, the U. S. National Park Service will rely heavily upon you, our state fish and wildlife management partners, for data and science to support these legal and biologically sound management practices. These are your hunting programs, so, as the saying goes, "Take part or get taken apart."

By way of an update, we, in cooperation with the Massachusetts Division of Fish and Wildlife, are beginning the process of preparing an environmental impact statement for the complete hunting program at Cape Cod National Seashore in response to a court order. At New River Gorge National River, the U. S. National Park Service is drafting special regulations, in cooperation with West Virginia, to validate our ongoing hunting programs. And recently, we were successful in defending the New Jersey black bear hunt at Delaware Water Gap National Recreation Area.

Now, thanks to U. S. Department of the Interior Secretary Norton, we have additional funding for partnership efforts through the Cooperative Conservation Initiative, or the CCI. The CCI funds projects that restore natural resources or expand habitat for wildlife. They call for an equal match of federal and nonfederal funds. In the first year alone, 74 projects were initiated with the cooperation of 200 partners; those partners contributed over $8 million.

State governments are key partners, with over one-quarter of this funding going to projects involving 19 state agencies. Projects this year include the repopulation of native fish and plants, as well as the restoration of coastal wetland and riparian areas. Currently, this program must focus on our land. In the future, we hope to expand our partnership to include projects beyond park boundaries.

Our Fire Management Program is among our most active partnerships with states. The landscapes surveyed by Lewis and Clark on their voyage of discovery have been greatly affected by our fire management practices. We will continue to work hard with other agencies and with our neighbors to restore healthy forests and wildlife habitat through fuel reduction, the restoration of fire as a natural process and the rehabilitation of burned areas where necessary. In fact, we frequently use wildlife as an indicator of our success in these endeavors.

Our ecosystem restoration program is developing new initiatives to address major fire-related resource issues, including the restoration of native understory components of our eastern forests and western shrubland. We are also holding a workshop this spring, to chart a U. S. National Park Service role in efforts to restore the American chestnut; both of these efforts have great implications for wildlife in these ecosystems.

An example of our partnerships in science is the Cooperative Ecosystem Studies Units (CESU) Network. The CESU Network provides research, technical assistance and education to federal resource management, environmental and research agencies, and their partners. Thirteen federal agencies from five departments—The U. S. Department of the Interior, U. S. Department of Agriculture, U. S. Department of Commerce, U. S. Department of Defense, and U. S. Department of Energy, as well as the Environmental Protection Agency and the National Aeronautic and Space Administration—now participate. The 17 CESUs include over 120 universities and colleges, and 22 of these universities and colleges are minority institutions.

CESU projects range from small monitoring projects to a million-dollar restoration effort. Many involve several federal agencies working together and side-by-side with university faculty and students. The shared expertise is extraordinary, and CESUs are a valuable addition to the "science tool kit" of the U. S. National Park Service and our partners. For those of you who wish to learn more about the CESU Network there is a session here tomorrow at 11:00 a.m.

Soon after I became Director of the U. S. National Park Service, I said that we should strive to make national parks some of the best places to restore threatened and endangered species. We have acted on this pledge. Ever year, we continue to improve the status of federally listed plants and animals that occur in our parks.

Last year a condor chick successfully fledged at the Grand Canyon. For several years, visitors have watched condors soaring above the rim of the canyon. This success comes from the kind of cooperation we now strive for. The park has worked with Arizona Department of Game and Fish, The Peregrine Fund, the U. S. Fish and Wildlife Service and the U. S. Bureau of Land Management to bring the condors back to Arizona. Several times, birds have had to be brought back into rehabilitation because of lead intake from the environment. But, with the birth of this chick, the condor recovery program has turned a corner. I am optimistic that condors will continue to thrill future generations of park visitors.

Plants comprise the largest category of listed species in the national parks, and we are putting increasing emphasis on plant restoration. Last year the Mauna Loa Silversword and eight other plant species were restored at Hawaii Volcanoes National Park. This amazing achievement is one that we want to repeat across the country.

What about species that are rebounding spectacularly due to recovery efforts? We are all familiar with the recovery of peregrines, bald eagles, grizzly bears and gray wolves. These successes mean that both the U. S. National Park Service and state agencies now have to address a changing set of wildlife management issues. Two of those issues are monitoring the effects of reestablished predators and managing human-wildlife interactions. The U. S. National Park Service will be working with Wyoming to ensure an efficient and effective exchange of information on wolf numbers in and around parks.

Recently Yellowstone National Park and Rocky Mountain CESU sponsored a workshop on the management of bears that have become habituated to humans. Every state with grizzlies sent its bear biologists to this meeting. We need both partnerships and the best science available to protect visitors, to preserve bears and to enhance the experiences of the public. You can hear about the results of this meeting at the symposium on human-wildlife conflicts this afternoon.

Today I have shared with you some of the examples of how the Challenge has led to a rediscovery of our values and the rejuvenation of the national parks. The Lewis and Clark expedition was the beginning of an east-west migration of humans to explore the vast unknown, to document the natural resources found there and to connect the continent. Today, as we rediscover our stewardship responsibilities for all species, we acknowledge that discovery and documentation are important functions of our agency and that collaboration is essential if our conservation efforts are to be successful.

It is appropriate to note that Fort Clatsop, where the Corps of Discovery encamped during the winter of 1805 to 1806, is now a national memorial and part of the National Park System. Just as William Clark spent his time at Fort Clatsop correcting the record and filling in the blanks on his map of the West, the U. S. National Park Service continues the process of correcting the record and filling in the blanks.

Thank you for letting me share with you today some of the successes of the Challenge and thank you for joining with us as we rediscover what is special in the United States' national parks, which represent a portion of the vast and wonderful natural heritage of this nation. We look forward to continuing our work with you, and we wish you every success for this conference.

A Balancing Act: Partnering to Provide Water and Power in an Environmentally Sensitive World

John W. Keys, III
U. S. Bureau of Reclamation
Washington, DC

Introduction and Overview

Good morning. It is a pleasure to address this esteemed group and to be here alongside many of my colleagues from the U. S. Department of the Interior.

For those not familiar with the U. S. Bureau of Reclamation (Reclamation), I want to tell you who we are and what we do. Reclamation works in the 17 western states, managing, developing and protecting water and water-related resources in an environmentally and economically sound manner. We were founded by President Theodore Roosevelt and by Congress in 1902 to bring water to a dry West, to make the desert bloom. And, more than 100 years later, we are still doing just that! We are the fifth largest electric utility in the West, and we are the United States' largest wholesale water supplier, administering 348 reservoirs. We are most commonly known for our landmarks and engineering marvels, such as the Hoover, Grand Coulee and Glen Canyon dams. However, our presence is felt throughout the West, and our efforts benefit the entire nation.

Our projects provide water for farmers, cities, recreation, refuges, fish, wildlife and tribes. However, when you take all of those entities—who sometimes have competing interests—and when you add over-allocated water supplies, limited water supplies, multiple years of drought and population growth in the West, you get us where we are today.

Today's Reclamation is dealing with tough and tumultuous issues. However, we are facing these head on! Under the guidance of this Administration, we are partnering to protect wildlife while upholding our mission. It's these partnerships that I want to focus on today.

Problem Solving: Using the Four Cs

I am privileged to be serving in this Administration, under the leadership of the U. S. Department of the Interior Secretary Gale Norton. She is someone who truly understands our issues, and that makes all of the difference in the world when you're in a job like this one. If you've ever heard Secretary Norton speak, then you've heard her talk about her four Cs philosophy—conservation through cooperation, communication and consultation. And, I'm here to tell you that it's not just talk; the four Cs represent the way we do business, especially when it comes to these tough and oftentimes controversial issues. It is a philosophy that means common sense, stewardship and citizen-centered government. We are taking a proactive approach to address issues head-on and to try to get ahead of the problems.

Let me show you how that approach works on something that gets a lot of attention—the Endangered Species Act (ESA). It means improving and streamlining the process and working to avoid listings in the first place. We're addressing issues from a multispecies aspect rather than individually. And, we're addressing concerns early, before it's too late. We're taking a broader ecosystem approach, looking at the water basin point of view. We're developing an internal ESA handbook, and we're continuously developing training for our people on ESA issues. And, we're doing all of this with the help of the agency partners, other stakeholders and the U. S. Fish and Wildlife Service (FWS).

Now the last point may not seem like a big deal, but it is. There was a time when the head of Reclamation would not even speak to the head of the FWS and vice versa. Today, we have a great relationship with FWS Director Steve Williams and the staff at the FWS. For example, we are working closely with FWS on the Fish and Wildlife Coordination Act to develop new policy and standards of compliance. We are talking with it about Reclamation's role under the ESA, and each year our employees are invited to go to training on the ESA.

We're also bringing stakeholders to the table and listening to what they have to say. This Administration believes that decisions are best made at the local level by the people who are most familiar with the issues. That's why our stakeholders play such an important role in our problem-solving approach. These partnerships are a vital part of how we do business. Let me highlight some of our other successful collaborations.

Success Stories

Klamath

I don't think I have to tell you that the Klamath Basin in Oregon is an area filled with conflict and competing water interests. However, it is also an area where we are making great strides and employing partnerships to address tough issues.

For instance, The Conservation Implementation Program, or CIP, is an effort that Reclamation is leading to collaboratively resolve short- and long-term water issues in the Klamath Basin. The CIP is a partnership between all levels of water users and interested parties—federal, state and local stakeholders. The goals of the program include:

- recovering the endangered short-nose and Lost River suckers and substantially contributing to the recovery of the coho salmon
- continuing sustainable operation of existing water management facilities and future water resources improvements for human use
- contributing to the tribal trust responsibilities of the federal government.

The CIP would establish resources and authority to implement these goals.

President George W. Bush's fiscal year 2005 budget request calls for unprecedented help for the Klamath Basin. The proposed budget calls for investing more than $100 million (among several agencies) in habitat restoration and in water improvement projects and programs. This unprecedented level of commitment will help Klamath Basin communities restore their watershed and avoid future water supply crises.

Upper Colorado River Recovery Implementation Program Management Workgroup

I also want to touch on two success stories from the Upper Colorado River Basin. These are two examples where environmental values have been brought to the forefront, where we have achieved significant success and where we have protected the significant benefits of our projects for the communities they serve.

The Upper Colorado River Endangered Fish Recovery Program is making significant progress for the humpback chub, the bonytail, the Colorado pikeminnow and the razorback sucker by:

- restoring more natural river flows
- reconnecting fragmented habitat with fish passage structures
- restoring degraded habitat
- preventing fish entrainment and associated mortality in canal systems with fish screens
- implementing aggressive stocking operations.

The key to the program's success lies with the involvement of states, federal government and nongovemment partners.

That's the same case with the Glen Canyon Dam Adaptive Management Program and Workgroup (AMWG). This group of 26 members works together to recommend adjustment and adaptation of dam operations as a result of ongoing scientific monitoring and experimentation. The work of this group over the last five years has resulted in the evaluations of previous flow releases, the creation of program goals and objectives and—most recently—the current experimental flow program related to humpback chub benefits and trout management. It comes down to a commitment by all parties to seek solutions. Are we to have confrontation or consensus? Conflict or cooperation?

The commitment to adaptive management is the only way that I know of in which multiple interests can set aside their differences to do what's right for a resource while preserving the project benefits that are critical to the western states. Without it, we are back to win-lose confrontations fought in Congress, in the courts and before the public. The AMWG demonstrates that we can, and are, doing better than that.

Multispecies Conservation Program

I spoke earlier about our work with ESA, and a related success story is in the Lower Colorado River Basin with the Multispecies Conservation Program, or MSCP. MSCP is being developed by a multistakeholder steering committee, made of state, federal, tribal, environmental and other interests. The goal of the program is to implement conservation measures that will conserve and move endangered and threatened species toward recovery and will decrease the need for future listings while accommodating current and future water and power operations on the lower Colorado River. Successful achievement of these goals will allow Reclamation to uphold its mission of delivering water and power while complying with ESA, conserving habitat and reducing the likelihood of additional species being listed under ESA.

Hatcheries and Fisheries

Reclamation continues its support in studies and activities related to fish hatcheries and their role in the recovery of threatened and endangered species. In the Pacific Northwest, we participate on the Federal Hatchery Team, providing financial support for activities at the Leavenworth Fish Hatchery

Complex and for interagency efforts to reduce the detrimental impacts of artificial production of salmon on wild stocks.

Another great example of partnerships that benefit fisheries is the Lake Havasu Fisheries Improvement Program Partnership. It is, perhaps, the largest and most comprehensive warm water fishery project ever undertaken in the United States. This alliance of seven conservation-minded agencies and a nonprofit organization was established in 1992 to enhance the lake's fish habitat, to improve a threatened and an endangered native fish population, and to provide universally accessible shoreline piers and amenities for nonboating anglers. Since its inception, the partnership has developed six fully accessible shoreline-fishing piers, has stocked 60,000 endangered razorback suckers and bonytail chubs, and has installed 42 improved habitat sites at the bottom of the lake, covering an area that exceeds the size of 600 football fields and all with recyclable, donated materials.

Water 2025

Our newest and most publicized partnership effort is Water 2025. Secretary Norton launched Water 2025 last year—her vision for the future of management of water in the West. Water 2025 is a problem-solving initiative to manage scarce water resources by focusing on areas of the West where conflict and crises over water can be predicted. Water 2025 highlights five tools that we can use today to alleviate and prevent crisis over water:

- conserving
- collaborating
- researching more cost effective water treatment technologies, including desalination of salt and brackish water
- removing institutional barriers to efficient water management
- improving interagency coordination of programs.

Water 2025 has been given a warm reception all over the West. Many of you were part of 3,000 water users, managers, stakeholders and interested citizens who attended one of our meetings across the West last summer and fall. Water 2025 has raised important western water issues to national prominence. Reclamation is currently accepting proposals as part of a challenge grant program that is focused on achieving the goals identified in Water 2025, particularly in water conservation, efficiency and markets, and collaboration.

Water 2025 focuses on Reclamation's role—preventing water crises before they occur, including preventing another Klamath disaster or something worse in the future.

Water 2025 is a perfect example of partnering to get ahead of a problem before it occurs and of working together to meet the needs of all parties involved by partnering with states, counties, cities, irrigation districts, environmental groups and federal agencies. In the final analysis, long-lasting solutions to chronic water shortages will not come from the federal government but from the local level, from the people whose lives are most affected.

Recreation Partnerships

I've talked a lot today about our policies as they relate to fish and wildlife issues; however, I want to close by mentioning, our recreation partnerships. Reclamation is responsible for 8.5 million acres (21,003,500 ha) of land and 13,000 miles (20,917 km) of shoreline, the majority of which are available for public recreation. There are about 310 recreation acres (766 ha) on Reclamation projects that attract more than 9 million people annually. These visitors contribute $6 billion to local economies and provide 87,000 nonfederal jobs.

Wherever possible, Reclamation enters into management agreements with nonfederal entities. The fish and game on our land and in our waters are managed by the state fish and game departments. And, the state park departments manage the majority of the recreation areas. There are over 200 state parks on Reclamation land. We work cooperatively with our federal and state partners to make water-based recreation available. We are dedicated to honoring our commitments with these groups, such as keeping boat ramps and recreation areas operational, especially during droughts.

And let me put in a word here for the Cast a Special Thrill, or CAST, program. There are 38 CAST events across the West every year to give diabled kids the opportunity to fish for a day; this is an outstanding partnership in action.

Conclusion

As I wrap up here, I want to thank you again for giving the opportunity to address you this morning. I am honored to be serving in an Administration that values partnerships and that seeks action and progress. I hope I've given you a taste of the good work that Reclamation is doing and of the efforts we are making to protect wildlife while providing water and power to a thriving West. Look for opportunities to work with us at Reclamation.

The Quest for Quantifying Conservation Reserve Program Benefits

Michael Yost
U. S. Farm Service Agency
Washington, DC

Thank you for that kind introduction. And, thank you for inviting me to Spokane. It is a pleasure to be here . In April, we are celebrating the 34th anniversary of Earth Day, which was started by a Harvard University professor back in 1970. Earth Day is a reminder of how important conservation is to everyone in this nation. To some, it wasn't surprising when President George W. Bush increased conservation funding by 80 percent when he signed the 2002 Farm Bill. Today, we are examining how we can use the bill to further enhance wildlife conservation through the Conservation Reserve Program (CRP). I welcome opportunities like these. CRP isn't simply my mission; it's what I, as a farmer, practice. I've farmed since 1979, raising corn, soybeans, wheat and alfalfa in Minnesota. From 1979 to today, we've gone from producing 120 bushels per acre to 180 bushels per acre (4,229–6,343 l/acre) of corn. In the meantime, we've reduced our pesticide volume and the farm's tillage. Not only do we farmers love the land, but we also love to reduce our costs. The traditions of conservation come naturally to farmers because it's easy to recognize the benefits.

In the early 1990s, I signed my land up for CRP. Filterstrips have increased our bounty of wildlife and have cleaned our water. And, I'm pleased to mention that my two sons and my father feel as strongly about conservation as I do. So, when I talk to you about the accomplishments of CRP, I'm speaking from years of family farming experience, as well as from my recent appointment as Associate Administrator of the U. S. Farm Service Agency (FSA).

We have a great tradition of private land conservation in this country, along with a tradition of voluntary, incentive-based conservation. Over the past several decades, improvements to our environment and to our quality of life, especially on private land, have yielded quantifiable results, many of these from U. S. Department of Agriculture (USDA) programs, such as the CRP, but most of all from the producers who've participated. CRP's role in enhancing wildlife habitat and in protecting North America's natural resources is now widely recognized. The CRP is the USDA's largest conservation program with 34.6 million acres (1.4 million ha) enrolled and a $2 billion per year budget. Participants remove environmentally sensitive land from agricultural production by entering into 10- to 15-year contracts. I don't know of any group of people who could possibly better understand and appreciate CRP's evolution than you who are gathered here today. Participants have gone from retiring highly erosive cropland to targeting sensitive agricultural land to maximizing the benefits through protecting soil productivity and to improving water, air quality and wildlife habitat. For that reason, I would like to take this opportunity to talk about the need to quantify and communicate CRP benefits, and I'd like to discuss what FSA is doing in this regard.

Let me begin by talking about where CRP started, where it is now and where it is going. Originally CRP accepted acres based on whether the land offered was highly erodible, if it had a crop history and whether the rental-rate offer was acceptable. Congress measured success by whether the program met the statutory acreage enrollment goals. From the onset of the program, it was clear that CRP resulted in substantial reductions in soil erosion. By 1990, the water quality and wildlife benefits generated by the CRP were widely recognized, leading to the adoption of an environmental benefits index (EBI). The EBI helped to select land offered for enrollment, maximizing the conservation benefits for soil, water and wildlife.

In 1996, FSA recognized that certain conservation practices, such as riparian buffers and grass filters, benefited to landowners going through a general sign-up without the bidding process. This allowed landowners to install practices on a continuous basis, creating immediate conservation benefits. In 1997, USDA began to develop collaborative CRP initiatives with states to address specific conservation concerns.

The CRP now includes four programs:

- General CRP, which uses the EBI to select offers during sign-up periods
- Continuous CRP, which accepts acreage outside of the sign-up process; eligible producers must install highly beneficial conservation practices, such as riparian buffers, grassfilters, bottomland hardwood and wetland restoration
- Conservation Reserve Enhancement Program (CREP), which is a state and federal partnership
- Farmable Wetlands Program, in which small wetland on cropland can be enrolled to provide wildlife benefits.

What have the participants enrolled in CRP accomplished through these programs? CRP has:

- played a major role in USDA helping farmers reduce soil erosion by more than 40 percent since 1982
- restored more than 1.8 million acres (0.7 million ha) of wetland and wetland buffers
- installed more than 1.5 million acres (0.6 million ha) of riparian buffers and grassfilters
- improved wildlife habitat to increase prairie upland duck, pheasant, many grassland birds and other wildlife populations
- developed into the largest, federal carbon sequestration program.

But what does this mean? How does this communicate the benefits of conservation? These are the questions that we need to answer. President Bush has said, "What gets measured gets done." He's also said that, unless you can measure your accomplishments against your stated goals, your programs won't get funded. If CRP is going to continue to be funded, we need to be able to measure and communicate CRP goals and accomplishments outside of the agricultural and the conservation community.

To some, CRP's primary achievement is reduced soil erosion. But, what is the importance of the reduced soil erosion? Obviously, keeping the soil in place maintains the productivity of the soil, but it also keeps sediment out of our rivers, streams and lakes, thereby improving water quality. Reduced sedimentation improves the habitat for fish, mollusks and other aquatic species.

But, that is not all that is going on when participants enroll in CRP. A conservation cover of grass, forbs or trees is established. This provides a habitat for wildlife and sequesters carbon in the soil. The challenge FSA and USDA face is how to communicate all of the conservation benefits that occur for each conservation practice? How do they communicate the aggregate effect of these practices?

FSA has undertaken a research program to help quantify CRP conservation accomplishments and to improve CRP accountability. Our intention is to communicate meaningful conservation measures so everyone can read our measures and answer for themselves, "Is CRP making things better?" To do this, FSA proposes better identification of the changes that take place when conservation covers are established on cropland. FSA wants to change how we report progress.

- Rather than simply saying how many acres of wetland were restored, we want to tell how much nitrogen and phosphorus are not getting into our streams, rivers and lakes. How much erosion was prevented? How many ducks (and other wildlife species) were bred in wetlands? How much carbon was sequestered? And, how can wetlands reduce flood levels?
- Rather than saying how many wildlife acres were established, we want to be able to talk about changes in wildlife populations. I just spoke about estimating changes in duck populations. We also are going to estimate changes in pheasant and northern bobwhite populations, and we are going to identify how other species populations have changed in association with duck, pheasant and bobwhite populations.
- Rather than saying how many acres of riparian buffers and grass filters have been installed, we want to talk about how much nitrogen, phosphorus and sediment they intercept and keep out of our surface waters.

We also want to talk about the effects of different grassland management activities, such as haying and grazing, on vegetative vigor and wildlife populations. We need to be able to provide the best science available, leading to better decisions and management.

For these reasons, FSA has entered into research contracts with land grant universities, including Oklahoma State University and Iowa State University, and with other agencies, including the U. S. Geological Survey, as well as with nongovernment organizations, including the Food and Agricultural Policy Research Institute. I believe that we are starting on the right track with these partnerships.

As you know, FSA state committees and the state technical committees are designed to work together to provide sound conservation and farm programs. The FSA state committees decide on FSA programs using guidance from the state technical committees. State technical committees are designed to ensure that science-based conservation is used for USDA programs and that stakeholders' concerns are heard. The 2002 Farm Bill created an opportunity by permitting midcontract management, or the option to return rental payments in exchange for economic use of CRP land, such as haying and grazing. FSA is working to ensure the committee process supplements CRP midcontract management practices. These practices offer an opportunity to implement conservation systems to enhance wildlife populations, to increase vegetative vigor and diversity, and to stabilize farms and ranches.

Midcontract management also creates opportunities to manage CRP land in a manner that enhances habitat. For instance, we are vigorously pursuing means to manage habit for the northern bobwhite quail. FSA needs to ensure that conservation management practices adopted are science-based and that producer concerns are addressed. For instance, last year a proposal was made to allow earlier haying by shortening the Indiana bird-breeding season. If implemented, the proposal would have had a significant negative impact on pheasant and other breeding bird populations on CRP land. The state technical committee reviewed this proposal. The committee strongly recommended that it be declined. The state FSA disagreed and asked the FSA Deputy Administrator for Programs for an exemption. The exemption was reviewed and turned down, based on the technical committee's recommendation. That's exactly how we want it to work.

FSA is working to ensure the committee's process remains effective while implementing CRP midcontract management practices. We want to assure you that recommendations from conservationists will be heard and addressed in a science-based venue. Those of you who serve on state technical committees already know this. For technical committees to work, the agricultural and conservation communities need to communicate their concerns to the committee.

There's one more topic I want to address because it's fundamental to all of our work. As budget pressures increase, we can expect even more competition for discretionary funds. Agencies that cannot tell their story and to justify their budget requests will be less likely to get funding requests granted. What this boils down to is when we're talking about competing for limited federal dollars, we need to make sure that we can measure results. FSA is one of the first agencies in USDA to identify goals with measurable outcomes.

How this applies to CRP is especially interesting. First, we are using scientific evidence to corroborate and to document the increase in wildlife habitat brought into CRP. Second, we are using scientific evidence to determine which additional species are thriving. Third, we are determining the additional gains, also scientifically, from stopping erosion to clearing streams to sequester carbon.

In the effort to quantify our assumptions, we are focusing on improved decision-making and on communicating how far we've come and where we'd like to go. These actions are clearly in line with the presidential management agenda for enhanced measurable outcomes.

CRP has built its success through the momentum of partnerships, one step at a time, one buffer strip at a time, one creek, one river and one watershed at a time. The power of CRP lies in the cumulative grassroots strength of people working together with common goals and shared insights about how to achieve mutual goals in conservation of our land, water, air and wildlife.

In closing, everyone is a stakeholder in our natural resources, and everyone can play a role in conservation. Conservation is a global issue,a national issue, a local issue and a personal issue. When I walk my land, I take great pleasure in the changes I see. And, I take pleasure in that fact that these changes are happening throughout the nation. We are doing this for ourselves and for our children's children. I'm proud to be a part of CRP.

All of you have been key supporters of conservation. I want to thank you for your hard work and continued support. I hope you are inspired to do more because there's more we can do. I want to hear your ideas, dreams and accomplishments, and I look forward to continued work with you in the future.

Appendix

Conservation Reserve Program (CRP) Research Projects and Funds

Funds have been committed to three research projects:

- Wetland Functions ($250,000)
- Grassland Management Systems ($250,000)
- Wetland Filters Restoration ($108,000).

The remaining funds ($392,000) will be distributed in the near future for developing measures of wildlife enhancements.

The associated deliverables are being developed in cooperation with the U. S. Natural Resource and Conservation Service (NRCS) to augment the USDA's Conservation Effects Assessment Project (CEAP). The research projects should recommend improved CRP policy and program management within 1 year of initiation.

Wetland Functions. The U. S. Geological Survey (USGS) Northern Prairie Wetland Research Center has received $250,000 to identify critical wetland functions and to estimate changes in these functions when prairie pothole wetland is restored on previously cropped land. The project will focus on the impacts of introduced grasses and on native grasses (CP1 and CP2), wildlife habitat (CP4), wetland restoration (CP23) and on CRP conservation practices in the Prairie Pothole Region. This research will enable USDA to set wetland restoration goals and to report progress toward restoring wetland functions. The USGS project will:

- estimate acres of farmable wetland enrolled
- estimate potential reduction in movement of sediments and nutrients entering the wetland
- apply soil-loss models to estimate potential reduction in soil erosion within catchment basins
- estimate carbon sequestered
- estimate water storage volumes
- summarize the potential to offset greenhouse gas emissions, based on existing research (e. g., methane and nitrous oxide)
- estimate wildlife enhancements
- assess the current status of restored CRP wetland surveyed in 1997 by the center.

Grassland Management Systems. The Farm Security and Rural Investment Act of 2002 directs the Secretary of the U. S. Department of the Interior to develop vegetative management requirements for grassland. To meet these requirements, Oklahoma State University (OSU) researchers are being funded to identify appropriate conservation management systems for CRP grassland. The grassland management systems will help the FSA and CRP participants to identify appropriate practices leading to more vigorous and diverse grassland stands, which will provide increased forage for livestock and increased environmental benefits (e. g., enhanced wildlife habitat and improved water quality and soil productivity). The OSU project will:

- develop various practices, incorporating NRCS technical guide information with minimum management criteria for maintenance of improved grass stands
- determine the value of management systems relative to comparable yield rental rates; these systems will be regionally specific and will include variations in grassland type, precipitation, growing season, haying

practices, grazing practices, prescribed burning practices and other unique landscape features
- recommend appropriate management practices to assist local FSA staff to assess alternative bids and to discuss various management systems with producers resulting in improved CRP outcomes.

Wetland Filters Restoration. Tile drainage systems are a primary source of nutrient runoff into streams and rivers. Excess nitrogen runoff in the Mississippi River Basin has been blamed for causing an oxygen deprived dead zone in the Gulf of Mexico, often phrased the hypoxia problem. Scientists believe that restored wetland filters can filter excess nitrogen from tile drainage systems thereby reducing the area of hypoxia. Iowa State University (ISU) researchers have determined that nitrogen delivered to streams via tile drains can be reduced 50 percent by wetland restoration. ISU is being funded $108,000 to identify are as suitable for this activity. The ISU study will:
- provide a brief background on the problems associated with nitrates (hypoxia, drinking water quality, etc.) and how agriculture poses a major nitrate source, especially in tile-drained areas
- identify suitable areas in the upper Mississippi River Basin for filtering tile drainage systems with restored wetland
- predict total nitrate reduction that could be achieved using wetland as nitrogen sinks in tile-drained regions as a function of total wetland area
- predict total nitrate reduction that could be achieved within major sub-basins as a function of total wetland area
- develop guidance for citing and designing criteria to maximize nitrate reduction benefits of wetland and that can be broadly applied rather than limited to a narrow geographic area.

Wildlife Enhancements. Discussions on the project are underway within USDA and with USGS, the U. S. Fish and Wildlife Service (FWS) and other wildlife organizations. These discussions are focused on developing an agreement between NRCS, the Cooperative State Research Education and Extension Service (CSREES) and FSA for interagency sponsorship of a national wildlife conservation reporting framework.
- The project will consist of an interagency effort for developing a framework to estimate and to report USDA conservation program

effectiveness on enhancing wildlife populations. The two components are a national framework and a regional implementation with a tentative funding level of $150,000, divided equally among FSA, NRCS and CSREES.

- FSA has an immediate need for wildlife performance measures that are not met by the multiyear timeframe of the broad CEAP project. Thus, FSA will conduct research compatible to the CEAP effort to develop national data capturing the CRP wildlife benefits that can be estimated in 1 year. This research will move the interagency efforts forward by developing data and a platform that will provide a building block for measuring broader wildlife benefits. The January 15 to 16 meeting with NRCS, CSREES, U. S. Fish and Wildlife Service and other wildlife interests identified strategies to assure coordinated wildlife research efforts.
- The RFA will request proposals to identify and analyze data, providing information on CRP conversion of cropland to conservation covers. The RFA will seek to bring together expertise in multiple wildlife disciplines, but it will assure the research of each participant, integrating it into a national framework and providing consistent wildlife population estimates.
- The successful proposal will use NRI, CRP contract data, and other databases (including the Breeding Bird Survey, state game records, FWS waterfowl data, USGS grassland bird research records, et. al.) to develop estimates of the CRP effectiveness in increasing wildlife populations.

***Estimating Reduced Nutrient and Sediment Off-farm Movements*.**

FSA and the Office of Risk Analysis and Cost Benefits Analysis (ORACBA) entered into a cooperative agreement with the Food and Agriculture Policy Research Institute (FAPRI). FAPRI received $150,000 to estimate how much nitrogen, phosphorus and sediment runoff is reduced when cropland enters the CRP. Because nitrogen, phosphorus and sediments are primary agricultural agents contributing to water degradation, reduced nutrient and sediment nutrient runoff is an important measure of CRP effectiveness. The results of this study will provide a more direct estimate of CRP's effect on water quality.

The FAPRI buffer study will estimate predominate soils, climatic patterns, cropping practices and conservation covers, including:

- reduced soil erosion at the edge of the field
- reduced nitrogen and phosphorus in surface runoff
- reduced nitrogen and phosphorus leached beyond the root zone
- changes in soil carbon
- changes in surface water runoff, as cropland is placed in conservation cover under the CRP.

The analysis will examine differences between native and introduced grass cover to provide better information regarding applicability of different conservation practices.

Buffer Effects and Reduction of Nutrient and Sediment Delivery to Streams.

FSA entered into a cooperative agreement with FAPRI. FAPRI received $150,000 to estimate reduced nitrogen, phosphorus and sediment delivery to waterways when stream-side buffers are installed. Riparian buffers and grassfilters have been installed along over 2 million miles (3.2 million km) of streams to intercept sediment, nitrogen and phosphorus runoff from cropland. Buffers can reduce delivery of nutrients and sediment to waterways, but empirical estimates have not been available. The results of this study will provide a more direct estimate of CRP buffer on water quality.

The FAPRI buffer study will estimate predominate soils, climatic patterns, cropping practices and conservation covers, including:

- sediment intercepted
- nitrogen and phosphorus intercepted
- nitrogen and phosphorus by-passing the buffer beyond the root zone
- effect of tile-drainage systems on buffer effectiveness
- changes in soil carbon
- surface water intercepted as cropland is placed in conservation cover under the CRP.

The analysis will identify effective and ineffective soils and covers to provide better information regarding the applicability of buffer conservation practices.

Session One.
Managing Mammalian Predators and Their Populations to Avoid Conflicts

Chair
James E. Miller
Department of Wildlife and Fisheries
Mississippi State University, Mississippi

Cochair
Kenneth A. Logan
Colorado Division of Wildlife
Montrose

Opening Remarks

James E. Miller

Managing wildlife resources that are a public trust, whether existing on private or public land, with diverse public and special interest opinions about how these resources should be managed throughout North America is a difficult and often unappreciated task. The diversity of opinions expressed make the task complex, arduous and often controversial. Our challenge as wildlife researchers, wildlife managers, educators, agencies, organizations or institutions is to balance wildlife conservation for the public good with sustainability of the natural resource base and resolution of the conflicts that predators often create with human interests. There is probably no group of wildlife species that invokes greater challenges to both the science and policy of management than mammalian predators, particularly those larger species, such as coyotes, pumas, wolves, bears and the mesopredators that prey on waterfowl and other ground-nesting birds.

The presentations that follow these opening remarks will address some of the most controversial wildlife management issues of modern times. These issues make it increasingly difficult for state and federal agencies to establish science-based resource management and policy decisions, and they often create situations in which political pressure, expediency and actions taken ignore or overshadow good science. Throughout North America, but particularly in the West, policy makers, wildlife administrators and managers must interact with landowners, community leaders and other public members who often have contrasting views about the value of predators and the ecological and social implications of changing predator management regulations. Clearly, as we have learned over the years in dealing with predators and the controversies related to their management, not everyone will be pleased with the final decisions about their management, nor will everyone agree on the science that has documented their positive or negative impacts on prey species and human interests.

These presentations by professionals with many years of experience studying and managing predators will provide some insight and recommendations about the management of mammalian predators and their populations, including conflicts that you may not have anticipated hearing. It is important to learn from the studies that have been conducted in the past as well as from the expertise of professionals who have been dealing with predator issues for some time, and it is important to acknowledge the gaps in our knowledge that hinder our management capabilities for these species. It is also important that we recognize that the restoration of some large predator populations, although surely a wildlife success story, also brings issues that may include biological, economic, social and political problems that must be addressed. In some areas of North America, the essence of the majority of predator issues is that of resource allocation, the competition between predators and humans for prey. If these issues are to be appropriately addressed, it may require new management models, new adaptive management strategies and greater input from diverse stakeholders.

Admittedly, there are several known reasons for the recent increase of predator attacks on humans, including human population growth, suburban sprawl and protection of predator species that previously were harassed and suppressed by hunters, trappers and landowners. However, we also conclude that some of these species appear to be more adaptable to urban/suburban habitats that are rich in resources. Our increasing urban mentality of feeding wildlife, supporting feral cat colonies and owning small pets that substitute well for natural prey likely have also contributed significantly to these behavior changes. Some of the increased protection for predators has come from ballot initiatives, political edict, the lack of organized support for professional wildlife management recommendations and the increased emotional public perceptions falsely created by some media efforts that have significantly influenced public opinions about predators, e. g., Gentle Ben.

Although some individuals and interest groups may disagree with the recommendation that regulated public harvest of some species of these predators may be necessary, these recommendations have not come from knee-jerk reactions but from extensive studies of these predators and from feedback from diverse stakeholders. The management of habituated grizzly bears in our national parks is a classic example of conflicting purposes and expectations. On the one hand, the U. S. National Park Service must provide for public enjoyment of the parks, which includes watching bears. Yet, it must also prevent people from negatively impacting bears or from habituating bears, while attempting to prevent bears from attacking or threatening people. Should management strategies be based on what is best for the bears or on the public expectations of seeing bears when they visit the parks? This is one of those serious dilemmas that must be addressed by resource managers, agency administrators and policy makers. Is there a place for collaborative models between management agencies and nongovernment organizations, and what do such cooperative examples tell us about the need to address the variety of stakeholders' interests? Has our profession sometimes ignored the need for predator control in the management of wildlife resources because it is not a socially acceptable form of management, or have we been so indoctrinated that appropriate habitat management is the answer to most wildlife issues that we fail to recognize that wildlife damage management is an integral part of wildlife management? These are just some of the issues that will be addressed by today's speakers.

The following presentations will focus on managing mammalian predator populations that we feel will be beneficial to the profession as we move further into the 21st century. Clearly, we must recognize that these conflicts are not likely to diminish; they are not likely to become any less controversial. How effectively our profession addresses these issues will be a part of its legacy.

I must express my sincere appreciation to each of the authors and coauthors for their diligence and their promptness in submitting abstracts and preparing manuscripts, and I express my appreciation to each of those professionals who will make the following presentations in this session. I thank the Wildlife Management Institute Conference Steering Committee for including this session in the program, and I thank the cochair, Dr. Ken Logan, for his able assistance and encouragement as we organized this session over the past several months and prepared for the following presentations. We hope you enjoy the session, and we anticipate some interesting questions and discussions will be generated.

Reconciling Science and Politics in Puma Management in the West: New Mexico as a Template

Kenneth A. Logan
Hornocker Wildlife Institute
Moscow, Idaho

Linda L. Sweanor
Hornocker Wildlife Institute
Moscow, Idaho

Maurice G. Hornocker
Hornocker Wildlife Institute
Moscow, Idaho

Western Puma Management and Stakeholder Values

Puma (*Puma concolor*) management in the West evolved from unregulated killing and organized predator control during the period of modern settlement to legal protection in all of the western states and provinces (except for Texas) since 1965 (Young 1946, Nowak 1976). A probable result of regulating human-caused death of pumas, along with apparent increases in prey populations (i. e., white-tailed deer, mule deer, elk) in much of the West during the 1980s to early 1990s was that puma populations increased from historically low numbers, reclaimed historic geographic range and may have expanded into areas modified to their advantage by humans.

Although the present status of the puma in the West is a wildlife conservation success story, there are problems that are biological, economic, social and political. As puma numbers increase, so do chances for depredation on livestock, pets and hobby animals (Smith et al. 1986, Cunningham et al. 1995, Torres et al. 1996), depression of some small populations of bighorn sheep (Logan and Sweanor 2001), and dangerous puma-human encounters (Torres et al. 1996). These problems are partly associated with the concomitant increase in the human population into habitats of pumas and their prey. Sport-hunting pressure on pumas has also increased in recent years in most western states and provinces (Logan and Sweanor 2000, Dawn 2002), but so has protective advocacy by some on behalf of the puma. These controversies have led to a variety of outcomes, including increased hunter kill of pumas, localized control of pumas to manage depredation and threats to people, endangered ungulate populations, and ballot initiatives that banned puma hunting in California (1990) and use of dogs to hunt pumas in Oregon (1994) and Washington (1996)(Minnis 1998).

Political and legal efforts toward greater protection for pumas in recent years reflect the shifting stakeholder values from the predator control and utilitarian views prevalent during the 1800s to mid-1900s to greater appreciation for large carnivores from the 1960s to the present. A conceptual framework, modified from Kellert and Smith (2001), identifies nine values that reflect human biological relationships toward large mammals that help to explain the diversity of stakeholder values toward pumas that we experience today (Table 1).

New Mexico has grappled with puma management issues familiar to the other western states and provinces. This paper presents information on the evolution of puma management in New Mexico and the effort of the New Mexico Department of Game and Fish (NMGF) and contracted puma researchers in the Hornocker Wildlife Institute, from 1985 to 1998, to research and then develop a robust, adaptive management structure that considers puma biology and ecology, limitations to reliable information, and the varied stakeholder values that influence puma management.

Puma Management in New Mexico

The puma in New Mexico was bountied for $5.00 from 1867 until 1923 (Nowak 1976). By the early 1930s, pumas were severely reduced in numbers and geographic range (Hibben 1937, Young 1946). In 1971, the puma was placed on the list of New Mexico's protected wildlife species with the NMGF assuming authority to manage hunting seasons and nuisance pumas (Evans 1983). Along with legal protection and regulated take came the potential for puma numbers to increase.

Puma hunting regulations varied since legal protection was granted (by NMGF hunting proclamations from 1971 to 1995). In 1971, three-quarters of New Mexico was closed to puma hunting. The southwestern quarter was opened for 4 months with a bag limit of one puma per hunter, and females followed by cubs and cubs less than 1 year old were protected thereafter. In subsequent years, more areas of New Mexico were progressively opened to puma hunting,

Table 1. Peoples' values toward puma (see Kellert and Smith 2001).

Value	Description
Naturalistic	Emphasizes personal experiences that people have with animals. Those experiences engage human curiosity and imagination and invoke feelings of adventure, exploration, discovery and satisfaction of skill in the process of getting close to large animals to either hunt them or to observe them.
Scientific	Pertains to direct study and understanding of animals which fosters intellectual growth about nature that can result in practical advantages to people and promote an attitude of caring for nature.
Aesthetic	Refers to the physical attraction of nature to people. Puma are often featured in art (e. g., photographs, paintings, sculptures) and other visual media. Puma invoke impressions of nature's refinement and beauty. Aesthetic perceptions of nature may have evolved in humans through our connection with animals and habitats that gave us sustenance and safety and caused people to hone survival skills.
Utilitarian	Focuses on the practical and material value. Puma hunting currently provides direct economic value to outfitters and guides, ranging from $2,000 to $3,000 per hunt. Economic benefits accrue to rural communities and specialists (e. g., taxidermists) that provide hunting-related services. Wild landscapes support puma and prey populations, and they provide essential ecosystem services, including watersheds, clean air and extensive outdoor recreation opportunities, including consumptive and nonconsumptive uses. Puma also contribute to the integrity of wild ecosystems (Logan and Sweanor 2001:366).
Humanistic	Acknowledges the emotional connection of people to nature that fosters affection and concern. For puma, this has been demonstrated in the increasing protection for puma since the mid-1960s and the formation of organizations devoted to protecting puma (e. g., Mountain Lion Foundation, Cougar Fund).
Dominionistic	Refers to the human inclination to subdue nature. This includes controlling puma to make the environment safe for people, livestock, pets and hobby animals. Hunting puma to either kill them or partake of them in a nonconsumptive way, like observation or photography, can demonstrate an ability to function in challenging conditions and express strength, vigor and boldness. This includes people that value puma hunting for pure enjoyment and its competitive opportunities.
Moralistic	Pertains to the ethical responsibility that people have to conserve puma and to treat puma with respect. This has been demonstrated by the greater acceptance of protection of puma and regulations governing the treatment of puma (see Minnis 1998, Gigliotti et al. 2002, Teel et al. 2002).
Negativistic	Emphasizes the fear and aversion toward puma and anxiety about the risk of attack, particularly to one's self and family, but may extend to livestock and game animals that represent a source of direct and indirect sustenance. Those feelings may also promote awe and respect for the animal.
Symbolic	The figurative significance of puma in modern society expressed in children's books, toys, marketing, advertising and as symbols for educational institutions and the professional sports industry.

and the season length was extended to 11 months in duration. From 1979 to 1983, almost all of New Mexico was opened to an 11-month puma hunt season with an increased bag limit of two pumas per hunter. After 1979, hides of all pumas taken had to be tagged by the NMGF.

In 1983 the protected status of the puma was challenged. Members of the agricultural industry concerned with depredation on livestock attempted to return the puma to its former varmint status by introducing a bill to New Mexico's House of Representatives. The bill was tabled in committee, but the legislature requested more information from the New Mexico Game Commission and the NMGF. The NMGF responded by producing the first in-depth report on pumas in the state; it was titled, *The Puma in New Mexico—Biology, Status, Depredation of Livestock, and Management Recommendations* (Evans 1983). The report concluded that puma numbers probably had declined during the previous 11 years (1972–1983). Management recommendations in the report, bolstered by public sentiment, resulted in more conservative puma hunting regulations.

In 1984, the hunting season was reduced to 3 months throughout almost all of New Mexico. But five hunt units (two in the southwest, three in the southeast) had harvest quotas of 10 to 17 animals and the season was extended 2 months in order to kill more pumas where depredation on livestock was problematic. From late 1984 to 1997, puma hunting regulations were uniform. Almost all game management units in New Mexico were open to puma hunting for 4 months (December 1 to March 31) with a bag limit of one puma per hunter.

During 1984to 1997, the number of sold puma hunting licenses increased by about 121 percent, from 443 to 980, and the number of puma harvested per year increased by 113 percent, from 79 to 168. However, hunter success declined (Weybright 1993). The number of puma hunting permits issued per year explained 80 percent of the variation in the number of pumas harvested per year (Logan and Sweanor 2001:373).

Because of unusually high puma depredation on domestic sheep on up to five ranches in one game management unit in the Guadalupe Mountains in southeastern New Mexico, the NMGF initiated a special preventive control program in 1988. The program allows the killing of up 14 pumas per year to prevent puma predation on sheep. Pumas involved in depredation incidents are also killed. Throughout the remainder of New Mexico, puma depredation incidents were relatively few. An average of 17 depredation incidents occurred each year from 1984 to 1995, resulting in an average of seven pumas killed per year.

The general status of the puma population in New Mexico is crudely assessed from the numbers of pumas killed for harvest or control and from talking to residents, such as hunters and ranchers. Otherwise, puma populations in New Mexico have not been monitored in the field.

A major result of the review of the state of knowledge of pumas in New Mexico in 1983 was that the New Mexico Game Commission and the NMGF recognized a lack of information on pumas. There was little information to address puma-related issues and to develop management strategies and prescriptions that would contribute to a self-sustaining puma population. In order to fill that void, the Hornocker Wildlife Institute (HWI) was contracted to execute 10-years of ecological research on pumas and to assist the NMGF to develop a statewide puma management plan and educational materials.

Puma Research

HWI studied a puma population for 10 years (1985–1995) on the 795-square mile (2,059-km^2) San Andres Mountains (SAM) in southern New Mexico. HWI had three main objectives: (1) to describe the structure and dynamics of the puma population, (2) to describe the behavior and social organization of pumas and (3) to describe the relationships of pumas to desert mule deer and desert bighorn sheep. To address specific hypotheses related to those objectives and other scientific hypotheses (Logan and Sweanor 2001), HWI divided the study area into a treatment area (TA) of 271 square miles (703 km^2) and a reference area (RA) of 524 square miles (1,356 km^2). The puma population in the TA was experimentally reduced during December 1990 to June 1991; HWI removed 53 percent of the adults and 58 percent of the independent pumas, i. e., adults plus subadults (Logan and Sweanor 2001).

During the 10-year project, HWI studied the biology of 294 pumas, 241 of which it captured and tagged, and it radio-collared 126. In addition, HWI studied 175 radio-collared mule deer and 36 radio-collared desert bighorn sheep that lived on the TA. Data on deer and sheep demography gathered by personnel of NMGF, U. S. Fish and Wildlife Service (USFWS) and HWI was also used to model prey population dynamics and effects of puma predation on those populations (Logan and Sweanor 2001).

Puma Population Dynamics

Densities of adult pumas in HWI's study population ranged from 0.8 to 2.1 per 38.6 square miles (100 km^2). Densities of all pumas, including adults, subadults and cubs, ranged from 1.7 to 4.3 pumas per 38.6 square miles (100 km^2)(Logan and Sweanor 2001:162). These density ranges were similar to other puma populations that have been studied intensively in Alberta (Ross and Jalkotzy 1992), British Columbia (Spreadbury et al. 1996), Idaho (Seidensticker et al. 1973), and Wyoming (Logan et al. 1986). The high range of adult puma density was the second highest density quantified for North America (Logan and Sweanor 2001:167).

The SAM puma population had been severely depressed as a result of predator control actions prior to our research (1980–1984). The subsequent protection of the population for our research and the experimental reduction of pumas in the TA enabled us to quantify rates of population increase, the first ever documented. The average annual rate of increase for adults in the TA was 0.21 and 0.28 units for the pretreatment (1988–1991) and posttreatment (1992–1995) spans, respectively. The rate of increase for adults in the fully protected RA was 0.11 during the 7-year span from 1989 to 1995. When we analyzed the RA's rate of increase in two 4-year spans, the rate of increase for adults was 0.17 during 1989 to 1992 and 0.05 during 1992 to 1995. The slower rate of increase in the RA during 1992 to 1995 was attributed, partially, to density dependence and a declining mule deer population that was affected by a drought that began in 1992 and persisted to the end of the study (Logan and Sweanor 2001).

Metapopulation Dynamics

The SAM puma population was a source of pumas for other subpopulations throughout southern New Mexico, and its own growth depended on immigrants. The SAM population produced an estimated average of 8.6 emigrants per year to other subpopulations throughout southern New Mexico. On the other hand, the SAM population recruited an average of 4.4 immigrants per year and 4.1 progeny (i. e., pumas born on the study area) per year. Other puma populations in the state were increasing, stable or declining, depending upon local habitat conditions and human exploitation rates. Hence, the puma population in New Mexico probably formed a demographic source-sink metapopulation structure (Sweanor et al. 2000, Logan and Sweanor 2001).

Puma Behavior and Social Organization

HWI found that the social organization best fit a *reproductive strategies hypothesis*, where adult male pumas maximized individual reproductive success by being territorial and competing directly with other males for access to mates. Sexual selection, apparently, selected for large male pumas with greater individual reproductive success. Adult females, however, were not territorial. Instead, they maximized individual reproductive success by avoiding other pumas in general, which increased their survival rate and that of their offspring, and by raising as many progeny to independence as possible during their lifetime. Philopatry also favored female reproductive success. Male and female pumas with the highest reproductive success were generally those that had the longest lifespans (Logan and Sweanor 2001).

Puma-Mule Deer Relationships

Desert mule deer were the most important prey for pumas. Based on analyses of over 800 puma scats, deer comprised 86 percent of the diet (frequency of occurrence) and about 92 percent of biomass consumed (Logan and Sweanor 2001:306–7). Puma predation was the principal, proximate, limiting factor affecting deer population growth. Still, during a period of moderate rainfall and high fawn production, and during a period when the deer population was apparently below carrying capacity (1987–1991), the deer population increased. But, during the drought period (1992–1995), when the deer population was below carrying capacity and fawn production was so low that recruitment was about zero, puma predation rates on deer 1 year old or older increased, and puma predation hastened the decline in the deer population. Hence, weather and puma predation interacted to modulate mule deer population dynamics. HWI inferred that the deer population was ultimately limited by food (Logan and Sweanor 2001:332–3).

Puma-Desert Bighorn Sheep Relationships

Puma predation rates on the state-listed endangered desert bighorn sheep were sporadic and independent of puma density. However, an individual puma caused severe mortality in the population, which still declined significantly after the offending puma was removed. More sheep died from nonpredation causes (10 vs. 16). Psoroptic mange (i. e., scabies) occurred in 60 percent of the sheep killed by pumas and 62 percent of the sheep that died of other causes. The high prevalence of disease contributed directly and indirectly to the relatively high rate of mortality in adult sheep (Logan and Sweanor 2001). Thus, the sheep population, which numbered about 40 animals, was unstable and highly vulnerable to extinction. In fact, the demise of the sheep population was linked to the sharply declining mule deer population. As the deer declined, puma predation rates on the sheep increased, and puma predation drove this remnant sheep population to extinction (i. e., only one adult ewe survived in December 1997) (Logan and Sweanor 2001:354–5).

Puma Management Plans

Now that we had biological and ecological information on puma in New Mexico, the next step was to integrate it into puma management. During 26 public and agency meetings held throughout the state of New Mexico during January to May 1997, we informed the public and wildlife professionals of our research findings and how they related to puma studies in other parts of North America. Eighteen of those meetings were sponsored by the NMGF and were designed to directly solicit the public and wildlife managers to identify their issues in managing pumas in New Mexico.

Management Issues

Ten major issues identified through those meetings reflected a broad range of stakeholders' values toward pumas. They included: (1) pumas kill livestock, (2) pumas kill deer that could be taken by hunters, (3) puma predation threatens desert bighorn sheep survival, (4) too many pumas threaten human safety, (5) sustained puma hunting is desirable, (6) puma harvest should focus on taking males and protecting females and cubs, (7) hunting pumas with dogs is undesirable, (8) puma hunting is undesirable, (9) increase human development threatens New Mexico's ability to support puma populations and (10) diverse stakeholder values make puma management difficult.

Management Strategies

We identified seven key management strategies that might be useful for addressing those issues identified by the stakeholders and managers. In addition, we used our research findings and other reliable scientific finding on pumas in the West to guide those strategies (Logan and Sweanor 2001). Strategies included:

1. Control of puma populations is achieved when off-take exceeds the rate of population increase. Depending upon the demography of the focal population, off-take may have to exceed 5 to 28 percent of the adult pumas (Logan and Sweanor 2001:170) and could be achieved with direct control or sport-hunting.
2. Sport-hunting opportunity is sustained through quota-regulated harvest. HWI's data suggested that harvest rates might range between 5 to 28 percent of adult pumas, depending upon local puma population dynamics (Logan and Sweanor 2001:170). Moreover, HWI recommended protecting female pumas and cubs. Harvest rates could be adjusted later relative to data from monitoring population trends (i. e., adaptive management).
3. Translocation of nuisance pumas is done selectively. HWI's data suggested that independent pumas that were 2 years old or less were the best candidates for translocation. However, HWI recommended that older pumas and those involved in depredation of domestic stock or direct threats to humans should be euthanized (Ruth et al. 1998).
4. Protection of puma populations is due to recognition of critical risks, unknowns and uncertainties in puma management (see below), the need for naturally functioning puma populations, and metapopulation dynamics. Because pumas are extremely cryptic and live at very low densities in complex landscapes, they are difficult to study and manage. Protected populations function as robust, biological savings accounts that contribute to population resilience by countering management-related mistakes that are going to be made from time to time in human impacted populations (Logan and Sweanor 2001). HWI's empirical data suggested that protected areas should encompass at least (1,160 square miles [3,000 km^2]) of puma habitat (Logan and Sweanor 2001:386). Human exploitation may also disrupt traditional patterns of natural selection, the long-term effects of which are unknown (Logan and Sweanor 2001, Murphy 1998). Dispersal of pumas from the protected areas would provide potential numeric and genetic augmentation to human-impacted puma populations. Ultimately, self-sustaining puma populations in the West are dependent on public and private lands that are managed to provide for the vital needs of pumas (i. e., prey, cover, security, wild landscape linkages).

5. Monitoring puma populations is essential if managers are to know if management prescriptions reach management objectives and to educate wildlife professionals and stakeholders about management actions. Managers can use field techniques to directly estimate puma population trends. Methods and guidelines for puma track counts have been developed to index puma population abundance on snow-covered (Van Sickle and Lindzey 1991) and dry terrain (Van Dyke et al. 1986, Beier and Cunningham 1996). However, current methods are imprecise and their long-term reliability is unknown. Because it is not economically or logistically feasible to monitor all management units in the state, representative units could be chosen for monitoring. Monitoring should focus on areas representing management prescriptions with highest priority, such as areas with puma control, high harvest or high puma-human conflicts. Furthermore, monitoring in protected areas or refuges could provide valuable references (i. e., experimental controls) for comparison with human impacted puma populations.
6. Research findings should be adapted into management. We have already demonstrated some examples above. Ongoing research will be necessary to gain reliable knowledge, develop theory and refine management practices and monitoring methods. Whenever possible, research and monitoring should be incorporated into experimental management prescriptions. Moreover, research findings are the basis for educating the public and wildlife professionals. Focal topics for puma research include: population dynamics, interactions with prey, habitat use, interactions with humans, genetics, population monitoring techniques and methods of domestic animal husbandry that reduce conflicts with pumas.
7. Education is essential to the success of any puma management approach. Education will have to be an ongoing process that includes public stakeholders and wildlife managers. Managers informed about pumas may improve management prescriptions and results. And, informed publics may take a more active and understanding role in how pumas are managed by agencies. In today's atmosphere of ballot initiatives and litigation regarding carnivore management and conservation, informed and responsive managers and an informed and engaged public may render those kinds of actions unnecessary.

Management Unknowns and Uncertainties

In the process of developing these management strategies, managers realized a set of unknowns and uncertainties they would have to deal with in most of New Mexico, where intensive puma research or population monitoring was not possible. Those included the following.

1. The number of pumas in local populations is unknown.
2. Local puma population trends are uncertain; there are no reliable quantitative data available to ascertain this attribute.
3. Puma population growth rates (the main parameter for setting harvest rates) are unknown.
4. Puma population responses to management prescriptions are uncertain.
5. Effects of hunter selection (on natural selection, population trend, infanticide and orphaning of cubs) are unknown.
6. Basic statistics and parameter estimates for modeling puma populations are unknown or uncertain, including:
   a. density distributions
   b. population age and sex structure
   c. reproductive rates
   d. age-specific survival rates
   e. immigration rates
   f. emigration rates
   g. validity of puma population simulation models.

Therefore, one of our major dilemmas was how to successfully integrate the desires of concerned stakeholders and managers with the reliable scientific information yet recognize the unknowns and uncertainties that influence puma management.

Zone Management

HWI's efforts to resolve stakeholders' issues with management strategies tempered with reliable science, while recognizing the unknowns and uncertainties, resulted in an adaptive landscape-level puma management structure that HWI called *zone management* (Logan and Sweanor 1998, 2001:384–8, New Mexico Department of Game and Fish 1997a, b). The plan has three main zones, each designed to address regional puma management objectives.

1. Control zones are areas where it is deemed necessary to control pumas to protect private property (i. e., livestock, pets, hobby animals), human safety, endangered species or game animals. Control zones would function as puma population sinks. Zones of puma control or exceedingly high harvest quotas with the operational intent to suppress puma populations to achieve such objectives should be managed as experiments that test if puma population reduction actually achieves the desired objectives. This means structuring testable hypotheses and attendant sets of predictions.
2. Sport-hunting zones sustain puma hunting opportunity through quotas that would annually limit the number of pumas that can be killed by sport-hunters. In these zones, there is an emphasis on protecting females and cubs. Female subquotas or sex-specific limited entry permits can also be instituted to more closely regulate total and female off-take. Depending upon the productivity of the puma population, these zones may function as puma source populations. Zones with low, conservative harvest quotas could be used to link movements of dispersing pumas between refuge zones and other human-impacted puma populations (i.e., control and sport-hunting zones).
3. Refuge zones are large areas where pumas are not hunted. To be functional, refuges should be at least 1,160 square miles (3,000 km^2) (Logan and Sweanor 2001:386). Refuge zones would help protect the stability of the statewide or regional puma populations from management mistakes that may result from the biological unknowns and uncertainties that influence management. When refuge zone puma populations are stable or increasing, they function as puma source populations. In addition, refuge zones allow natural selection to be paramount. Genotypes and individuals produced in refuge zones disperse and immigrate into human exploited zones. Thus, refuge zones contribute pumas to exploited zones augmenting them both genetically and numerically. We recommended at least two puma refuge zones in New Mexico, one in the northern and one in the southern part of the state. Although pumas would not be hunted for sport in the refuges, individual pumas that caused depredation on private property or threatened human safety could still be killed per direction of state policy.

Figure 1 illustrates how zone management might look spatially in New Mexico. Note that control zones are not adjacent to refuge zones, and conservatively harvested sport-hunting zones can link to refuge zones. This structure uses the source-sink metapopulation concept (Sweanor et al. 2000, Logan and Sweanor 2001) to manage pumas in an adaptive management approach that addresses local stakeholder interests and management needs while conserving the biological stability of the puma population statewide. Flexibility in this approach makes it possible for a variety of zone management configurations to be developed with changing stakeholder and management needs.

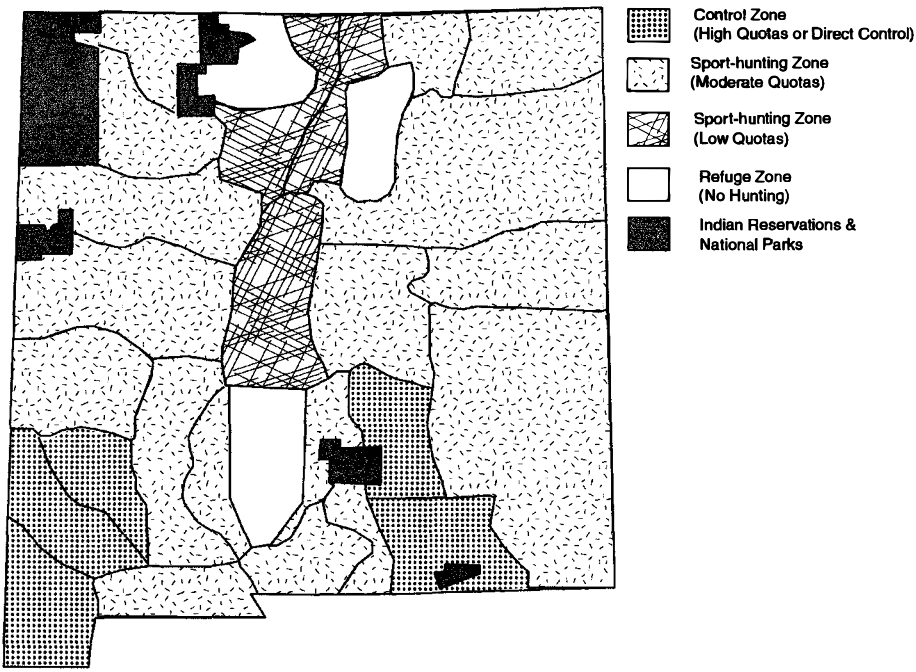


Figure 1. A conceptual model of zone management in New Mexico.

Conclusions

Zone management uses the best scientific information available on pumas to develop a robust, adaptive management structure that can be tailored to local conditions and stakeholder interests by using a broad range of options that include experimental control, sport hunting and protection. It also recognizes the array of unknowns and uncertainties that plague puma management. Hence, zone management is an approach that can be defended in the public arena by wildlife management agencies.

Development and implementation of a zone management structure is going to take sincere professional commitment and, for some wildlife agencies, a reevaluation of traditional values and operations. It will require administrative leadership and informed, responsive game and wildlife commissioners working with an informed and caring public. We believe zone management provides the broadest range of management options and is responsive to the broad range of stakeholder values in pumas. Zone management may also be useful in managing bears and wolves.

Reference List

Beier, P., and S. C. Cunningham. 1996. Power of track surveys to detect changes in puma populations. *Wildlife Society Bulletin*. 24:540–6.

Dawn, D. 2002. *Cougar management in the western United States: An analysis of hunting harvest strategies*. M. S. thesis. San Jose State University.

Cunningham, S. C., L. A. Haynes, C. Gustavson, and D. D. Haywood. 1995. *Evaluation of the interaction between mountain lions and cattle in the Aravaipa-Klondyke area of southeast Arizona, Technical Report 17*. Phoenix, Arizona: Arizona Game and Fish Department.

Evans, W. 1983. *The puma in New Mexico: Biology, status, depredation of livestock, and management recommendations. Response to House Memorial 42*. Santa Fe, New Mexico: New Mexico Department of Game and Fish.

Gigliotti, L. M., D. Fecske, and J. Jenks. 2002. *Mountain lions in South Dakota: A public opinion survey—2002*. Pierre, South Dakota: South Dakota State University, Brookings and South Dakota Game, Fish and Parks.

Hibben, F. C. 1937. A preliminary study of the mountain lion (*Felis oregonensis* ssp.). *University of New Mexico Bulletin, Biological Series*. 5(3):1–59.

Kellert, S. R., and C. P. Smith. 2001. Human values toward large mammals. In *Ecology and management of large mammals in North America*, eds.

S. Demarais, and P. Krausman, 30–63. Upper Saddle River, New Jersey: Prentice Hall.

Logan, K. A., L. L. Irwin, and R. Skinner. 1986. Characteristics of a hunted mountain lion population in Wyoming. *Journal of Wildlife Management.* 50:648–54.

Logan, K. A., and L. L. Sweanor. 1998. Cougar management in the West- New Mexico as a template. In *Western proceedings, 78th annual conference western association of fish and wildlife agencies*, 101–11. Jackson, Wyoming: Western Association of Fish and Wildlife Agencies.

———. 2000. Puma. In *Ecology and management of large mammals in North America*, ed. S. Demaris and P. R. Krausman, 347–77. Upper Saddle River, New Jersey: Prentice-Hall.

———. 2001. *Desert puma: Evolutionary ecology and conservation of an enduring carnivore.* Washington DC: Island Press.

Minnis, D. L. 1998. Wildlife-policy making by the electorate: An overview of citizen-sponsored ballot measures on hunting and trapping. *Wildlife Society Bulletin.* 26:75–83.

Murphy, K. M. 1998. *The ecology of the puma* (Puma concolor) *in the northern Yellowstone ecosystem: Interactions with prey, bears, and humans*. Ph.D. dissertation, University of Idaho, Moscow, Idaho.

New Mexico Game and Fish Department 1997a. *Long range plan for the management of puma in New Mexico, 1997–2004. Federal aid in wildlife restoration W-93-R38, project 1, job 5.5.* Santa Fe, New Mexico: New Mexico Department of Game and Fish.

———. 1997b. *Action plan for the management of puma in New Mexico, 1997–2004, Federal aid in wildlife restoration W-93-R38, project 1, job 5.5.* Santa Fe, New Mexico: New Mexico Department of Game and Fish.

Nowak, R. M. 1976. *The cougar in the United States and Canada.* Washington DC: U. S. Department of the Interior, Fish and Wildlife Service and New York Zoological Society.

Ross, P. I., and M. J. Jalkotzy. 1992. Characteristics of a hunted population of pumas in southwestern Alberta. *Journal of Wildlife Management.* 56:417–26.

Ruth, T. K., K. A. Logan, L. L. Sweanor, M. G. Hornocker, and L. L. Temple. 1998. Evaluating puma translocation in New Mexico. *Journal of Wildlife Management.* 62:1,264–75.

Seidensticker, J. C. IV, M. G. Hornocker, W. V. Wiles, and J. P. Messick. 1973. Mountain lion social organization in the Idaho Primitive Area. *Wildlife Monographs*. 35:1–60

Smith, T. E., R. R. Duke, M. J. Kutilek, and H. T. Harvey. 1986. *Mountain lions* (Felis concolor) *in the vicinity of Carlsbad Caverns National Park, New Mexico and Guadalupe Mountains National Park, Texas*. Alviso, California: Harvey and Stanley Associates, Inc.

Spreadbury, B. R., K. Musil, J. Musil, C. Kaisner, and J. Kovak. 1996. Puma population characteristics in southeastern British Columbia. *Journal of Wildlife Management*. 60:962–9.

Sweanor, L. L., K. A. Logan, and M. G. Hornocker. 2000. Cougar dispersal patterns, metapopulation dynamics, and conservation. *Conservation Biology*.14:798–808.

Teel, T. L., R. S. Krannich, and R. H. Schmidt. 2002. Utah stakeholders attitudes toward selected cougar and black bear management practices. *Wildlife Society Bulletin*. 30:2–15.

Torres, S. G., T. M. Mansfield, J. E. Foley, T. Lupo, and A. Brinkhaus. 1996. Mountain lion and human activity in California: testing speculations. *Wildlife Society Bulletin*. 24:451–60.

Van Dyke, F. G., R. H. Brocke, and H. G. Shaw. 1986. Use of road and track counts as indices of mountain lion presence. *Journal of Wildlife Management*. 50:102–9.

Van Sickle, W. D., and F. Lindzey. 1991. Evaluation of a puma population estimator based on probability sampling. *Journal of Wildlife Management*. 55:738–43.

Weybright, D. 1993. *Mountain lion management information. Performance report, P-R proj. W-93-R-35. Job 9*. Santa Fe, New Mexico: New Mexico Department of Game and Fish.

Young, S. P. 1946. History, life habits, economic status, and control, Part 1. In *The puma, mysterious American cat*, eds. S. P. Young, and E. A. Goldman, 1–173. Washington, DC: The American Wildlife Institute.

Predator Management in Alaska: Insight into a Historically Intractable Issue

Cynthia A. Jacobson
Alaska Department of Fish and Game
Anchorage

Daniel J. Decker
Cornell University, Human Dimensions Research Unit
Ithaca, New York

Tommy L. Brown
Cornell University, Human Dimensions Research Unit
Ithaca, New York

Jeffrey H. Hughes
Alaska Department of Fish and Game
Anchorage

Among the issues facing wildlife professionals, predator management is arguably the most contentious. Predator management, specifically predator control (i. e., killing of predators to minimize predator-induced impacts), evokes strong sentiments, ranging from utilitarian (e. g., predator control is necessary to ensure healthy game or livestock numbers) to protectionist (e. g., predator control is unacceptable under any circumstances). Interest in predator management issues tends to transcend geographic boundaries, confounding the management environment within which decisions are made. For example, predator management elicits the interests of those most directly affected by management policies (e. g., ranchers, local hunters) and those geographically distant (e. g., members of national interest groups, nonresident hunters) who believe they have a relevant stake in the outcome of predator management decisions.

Insight into the human dimensions of predator-prey issues, particularly those regarding large charismatic species, such as wolves (*Canis lupus*) and moose (*Alces alces*), offers some hope of understanding the human component of the management environment needed to weigh competing interests in management. Human dimensions research has provided a wealth of information about people's general attitudes and values regarding predators (Hook and Robinson 1982, Kellert 1985), tolerance for risks associated with predators (Riley and Decker 2000, Zinn and Pierce 2002) and acceptance of predator control measures (Manfredo et al. 1998, Zinn et al. 1998). While this information provides a foundation for understanding the human component of predator management, more specific information would help managers assess the social environment within which decisions about predator management are made. We believe essential information for this purpose includes data on people's attitudes and perceptions about predator-prey dynamics, predator-related impacts on humans and acceptance of predator management actions to minimize those impacts. Further, an understanding of whether an issue is primarily local or statewide can help managers anticipate the level of controversy and activity (Minnis and Peyton 1993) that will emerge.

The purpose of this paper is twofold: (a) to offer a pragmatic approach to conceptualizing the essence of predator-prey management, which highlights the human component in terms of perceptions, preferences and attitudes about management actions and (b) to explore the effect of issue proximity on salience of public perceptions of predator-prey imbalances.

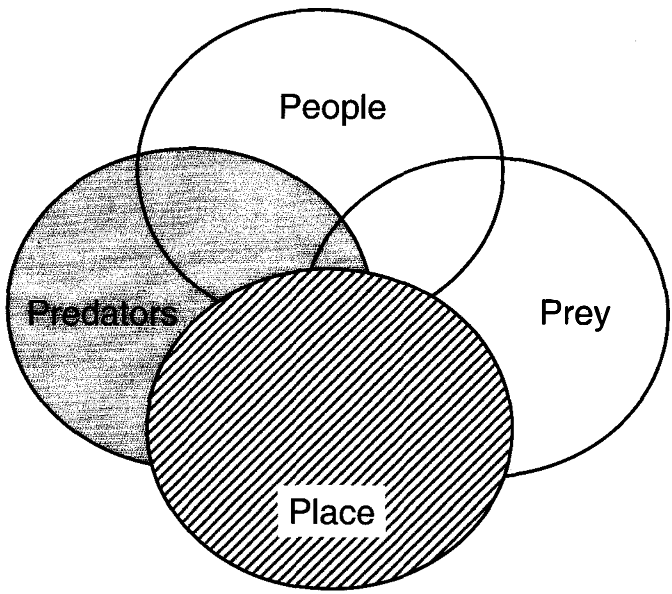
Manager's Model of Predator Management

Although claims about public attitudes and values regarding large predators, such as wolves, abound in research papers and popular journals (Nie 2003, Kellert 1985), they tend to be sweeping, generalizations of broad sectors of the American public. They reveal that: (a) attitudes about predators have been changing, primarily among the growing urban population, which has developed a romantic image of large predators; (b) some predators appeal to many people, most of whom have never seen, let alone experienced, the impacts of wild predators; and (c) attitudes about predator management: (i) range widely, (ii) cause positions to polarize and harden when predator management is considered and (iii) reflect the growing urban-rural divide with respect to human-land relationships.

Although contributory to understanding the social-values backdrop for management of predators, these sweeping generalizations fall short of the specific human dimensions insight needed by state wildlife managers. Managers need more specific human dimensions information to serve statewide or local management decision making. Although this can be approached in many ways, it is often useful to articulate the core management system to focus human dimensions information needs. This can be accomplished through a manager's model, which is basically a mental model of the managerially significant aspects of the management system.

In this paper we conceptualize the typical predator management scenario as a system having four main elements: place, prey, predators and people (Figure 1). These elements interact with one another in a variety of ways, but an overarching objective in predator management is to maintain an acceptable balance in the condition, or well-being of all four elements. This is a simplification of an ecologically and sociologically complex phenomenon that approximates the fundamental elements that can be realistically addressed by management. Depending on the specific situation, additional elements might be included, but in every case, place (habitat and ecological context generally), prey, predators and people are the core of the system—they are the starter set of considerations for the wildlife manager.

Figure 1. Four main elements of a manager's model of the predator-prey management system.



Referring to the set of interactions that can be managed as simply predator management misrepresents the system to be managed because it exclusively focuses on, therefore grossly overemphasizes, the predator element of the system. The real management challenge often goes unstated; to wit, *the core resource allocation issues are the essence of most predator management issues*. The focus of predator management does not arise from the dynamics between wildlife species (predators and prey) but from competition between predators and humans for utilization of prey, be that game animals or livestock. Typically it is the availability of prey for human use that draws management's attention. Nevertheless, we will use the phrase predator management here, with the reader recognizing the broader system to which we are referring.

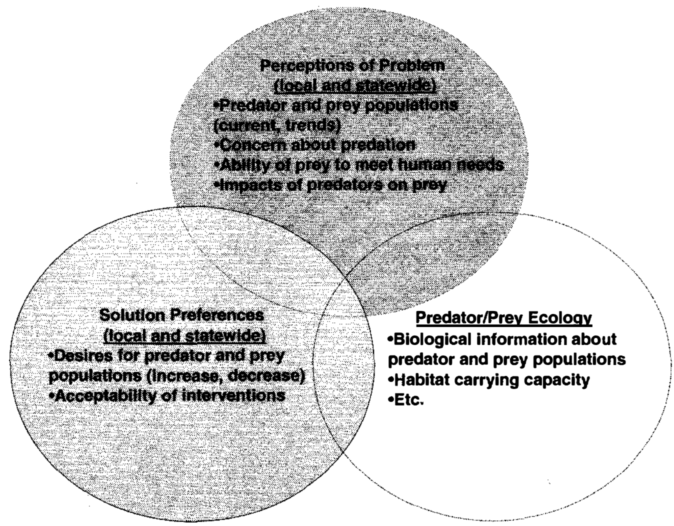
In adaptive impact management terms (Riley et al. 2003), the generic *fundamental objective* of predator management is to meet the needs of people who utilize prey for food, traditional cultural activity or recreation. The generic (but perhaps not the only) *enabling objective* is to maintain a necessary level of prey available (i. e., numbers of animals and accessibility to them) for human use. In this conceptualization of the system, the manager is faced with just a few elements of the system that might be influenced by management interventions. That is, the number of prey available for human consumption can be increased by:

- improving place (habitat improvements, e. g., setting intentional fires, improving escape cover for prey)
- decreasing mortality caused by nonhuman predation (e. g., instituting predator control)
- increasing access and effectiveness of human users such that they out compete nonhuman predators
- setting priorities for types of human use (e. g. subsistence, traditional lifestyle, recreational hunting), which may have different prey consumption needs and expectations and, therefore, different requirements for prey availability.

Human Dimensions Insight Needs to Support the Model

Identifying the insight needed to inform predator managers is aided by the manager's model; from it, a more specific conceptualization of predator management that is useful to management decision making can be developed (Figure 2). Both biological-ecological dimensions and human dimensions information needs are evident. The human dimensions needs can be divided into

Figure 2. Manager's model for predator management insight needs (colored circles are the human dimensions components)



two components: perceptions of a management situation (often referred to as an issue or problem) and solution preferences (for both fundamental and enabling objectives). These perceptions and preferences might be examined from both the local and statewide perspectives, for reasons explained earlier.

The matter of scale is often based on agreeing jurisdictional authority. For this example, we assume a state wildlife agency perspective, in which two scales are of interest: (a) people living in the region of management concern within the state and (b) residents statewide. The latter is important because: (a) residents from outside the local area utilize prey species in the region of concern and (b) the controversial history of predator management and the possibility of litigation or ballot initiatives could forestall or limit management possibilities, respectively. Thus, it would be desirable to have human dimensions information to characterize statewide residents and local residents.

The information a manager might seek from these human populations varies, depending on situation specifics and resources available for an inquiry. For example, if the situation is one where biologists indicate that the predator population is increasing and the prey population is declining, one has a predator-prey imbalance. Knowing the extent to which this imbalance is perceived in the local and statewide resident populations and knowing whether predator populations are perceived to be appropriate is valuable in assessing the likely reaction to management interventions aimed at predator reduction.

These kinds of data help managers evaluate public understanding of several of the key elements depicted in the manager's model (Figure 1). Managers also need an indication of the acceptability of the proposed fundamental objective of predator management, assuring the well-being of people who utilize prey for food, cultural activity or recreation. Equally important is determining the acceptability of the proposed objective and maintaining a necessary level of prey available (i. e., the number present and accessibility) for human use through specific management interventions to minimize predator impacts. Determining public attitudes about these anthropocentric objectives will reveal a great deal about the management environment. Estimating the extent of agreement with such objectives is an essential task of these inquiries to support management's decision . Data was used from a study of Alaska residents' attitudes regarding the management of predators (i. e., wolves and grizzly bears [*Ursus arctos*]) and prey (i. e., moose and caribou [*Rangifer tarandus*]) to illustrate the manager's model.

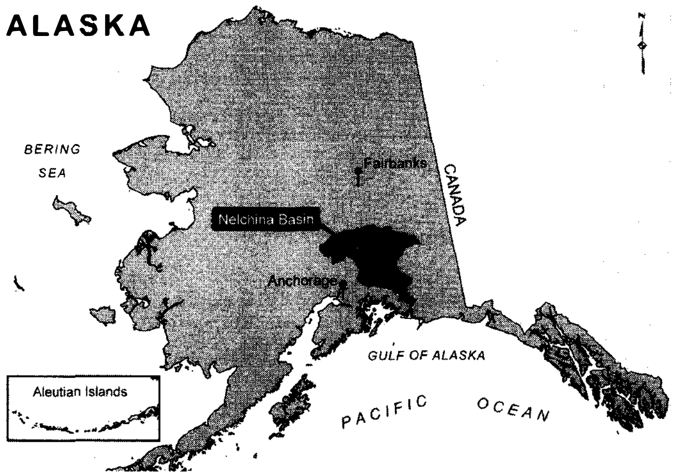
Predator Management in Alaska: Applying Human Dimensions Inquiry to Serve the Manager's Model

Background

Predator control in Alaska has been controversial since its inception in the early 1900s and remarkably so during the last 40 years. This issue has been the subject of ballot initiatives, travel boycotts, lawsuits and protests (Regelin 2002). The controversy centers on whether, when and how it is appropriate to kill predators to increase the number of prey for human harvest. Some factions of the public want the Alaska Department of Fish and Game (ADF&G) to control predators, particularly wolves, to allow prey, primarily moose and caribou, to increase. Others believe that policies should facilitate the public's ability to conduct predator control (e. g., allow land-and-shoot taking of predators). Still others believe that predators should not be persecuted and that nature should be left to take its course without human intervention.

The Nelchina Basin (Figure 3), located in southcentral Alaska, is one area where concerns about the impact of predators on moose and caribou populations resulted in intensive management regulations for wolves and grizzly bears to restore the abundance or productivity of moose and caribou. In response, ADF&G has expanded its research efforts to provide additional information

Figure 3. Alaska, Nelchina Basin in dark gray



about predator-prey relationships in the area, and the Alaska Board of Game has taken steps to increase mortality of wolves and grizzlies; it has instituted liberal wolf trapping and wolf and bear hunting regulations.

The challenges of managing socially and ecologically viable prey populations, predator populations and human uses of prey are vividly illustrated in Alaska. Acceptability of allocating, at a renewable and sustained level, proportions of the harvestable prey resource (such as moose and caribou) to large predators (such as wolves and grizzlies) and people (both subsistence and recreational users) is an ongoing, perhaps intractable, issue. Nevertheless, ADF&G recognized an opportunity to influence management decision making by including human dimensions inquiry as part of a public involvement for predator management in the Nelchina Basin.

Predator management in the Nelchina Basin closely reflects the manager's model described above. At the core, the question is finding an acceptable balance in allocation of prey species to wild and human predators. ADF&G wildlife managers were concerned about the scale of public perception (locally or statewide) of a predator-prey imbalance as well as of issues in the Nelchina Basin. That is, ADF&G staff believed they should answer the following questions.

- What are the perceptions of residents, both local and statewide, regarding predator and prey populations (current status, recent trends, etc.) as well as regarding predator-prey dynamics in the Nelchina Basin?

- Are area-specific, predator-prey imbalances local concerns, or are they recognized more broadly?
- Do local or statewide residents believe that current populations of prey can meet human needs?
- What management interventions for predators are acceptable to Nelchina Basin residents and statewide residents?

ADF&G collaborated with Cornell University's Human Dimensions Research Unit to conduct a survey of the Nelchina Basin and other Alaska residents to: (a) explore the effect of proximity on salience of public perceptions of predator-prey imbalance and (b) determine whether perceptions of predator or prey population conditions trigger expectations for management intervention. The manager's model is a framework for the analysis and discussion that follow.

Methods

After preliminary discussions with ADF&G staff and other stakeholders, a survey was drafted, reviewed, tested and revised. The first section of the questionnaire addressed predator management in general. The second section addressed residents' interest in predators and prey in the Nelchina Basin; it assessed their attitudes about caribou, moose, wolf and grizzly populations and trends. It questioned concerns about the impacts of predation, and it explored support for various methods of managing predators in the Nelchina Basin.

A random sample of 2,600 names (1,300 for the statewide stratum and 1,300 for the Nelchina Basin stratum)—generated from white pages of telephone directories—was purchased from Genesys, Inc. A cover letter asking the adult (greater than or equal to 18) with the most recent birthday to complete the questionnaire was used to help minimize age and gender bias inherent in drawing samples from telephone directories (Dillman 2000).

The survey was implemented in February 2003 using a modified version of the total design method outlined by Dillman (2000). A random selection of nonrespondents (50 in each stratum) were interviewed by telephone to determine their demographic backgrounds, participation in wildlife activities and general questions about predator and prey management in Alaska.

Results

For the statewide stratum, 483 questionnaires were returned out of 1,028 that were deliverable (47%). For the Nelchina Basin stratum, 700 questionnaires were returned out of 1,129 that were deliverable (62%). Sixty-five percent of the statewide respondents and 88 percent of the Nelchina Basin respondents indicated interest in Nelchina Basin wildlife management topics and answered the set of questions dealing with predator management there.

Extent of Nonresponse Bias

Nonrespondents within the statewide stratum, compared to respondents, were more likely to be female, were less interested in wildlife, were less likely to hunt and were less likely to belong to a conservation organization. Their level of education, age and number of years lived in Alaska did not differ significantly from that of other respondents.

Nelchina Basin nonrespondents, compared to Nelchina Basin respondents, were less educated, were less interested in wildlife, were less likely to feed birds, hunt, fish or trap, and were less likely to be interested in Nelchina Basin wildlife topics. On the other hand, these nonrespondents took trips or made outings to view or photograph wildlife, ride snow machines, and use game for food in similar proportions to respondents. Their age and number of years lived in Alaska also did not differ significantly from that of respondents. Nelchina Basin nonrespondents rarely differed from respondents with respect to perceptions about predator-prey interactions in Nelchina Basin.

Respondent Profiles and Comparison of Statewide Respondents to Alaska Residents

Many respondents from both the statewide and Nelchina Basin samples had lived in Alaska for over a quarter century, and few of either group had lived in Alaska 5 years or less. On average, respondents had posthigh school education and were in their early 50s. Over 40 percent of respondents statewide, and over 60 percent of Nelchina Basin area respondents grew up in rural areas or in small villages of under 5,000 people, while the others grew up in cities with larger populations. The respondents were predominately male (80% in both strata) and Caucasian (85% in the statewide and 90% in the Nelchina Basin strata), indicating likely gender-related and ethnicity-related bias in the results reported herein.

Perceptions of Existing Wildlife Populations and Trends

Respondents were asked whether they would describe the existing populations of caribou, moose, wolves and grizzlies as too high, about right or too low. Over one-third of the statewide respondents indicated they didn't know the population conditions for caribou and moose, and nearly one-half indicated they did not know about the populations of wolves and grizzlies. Nelchina Basin respondents indicated greater familiarity with these populations; the "don't know" rate was less than 15 percent for prey species and about 21 percent for predators. Similar percentages of "don't know" responses were found in the questions regarding population perceptions. Fewer statewide (less than 20%) than Nelchina Basin (less than 8%) respondents replied "don't know" when asked whether predator and prey populations should increase, decrease or stay the same. In the subsequent discussion and tables, the "don't know" responses have been removed.

A majority of respondents of both strata believed that the current prey populations in the Nelchina Basin were too low (Table 1). The majority of Nelchina Basin respondents thought both wolf and grizzly populations were too high. Most statewide respondents thought wolf populations were too high but were evenly divided as to whether grizzly populations were too high or about right.

Table 1. Percentage of respondents with an opinion from each stratum who thought existing prey and predator populations were too high, about right and too low in the Nelchina Basin.

	Too High		About Right		Too Low		n	
	Nelchina Basin	State-wide	Nelchina Basin	State-wide	Nelchina Basin	State-wide	Nelchina Basin	State-wide
Caribou	0	1	28	27	72	71	524	182
Moose	0	0	17	26	83[a]	74	549	194
Wolves	64	66	29	26	7	8	472	152
Grizzlies	60[a]	47	35[a]	46	5	7	478	157

[a]Significant difference between Nelchina Basin and statewide respondents (χ^2 test, $p \leq 0.05$)

Two-thirds of both statewide and Nelchina Basin respondents thought prey populations had been declining and that predator populations had been increasing recently (Table 2). Significantly more statewide respondents thought the wolf population had been increasing, and significantly more Nelchina Basin respondents thought the grizzly population had been increasing.

Table 2. Percentage of respondents with an opinion from each stratum who thought existing recent trends in prey and predator populations were growing, stable and declining in the Nelchina Basin.

	Growing		Stable		Declining		n	
	Nelchina Basin	State-wide	Nelchina Basin	State-wide	Nelchina Basin	State-wide	Nelchina Basin	State-wide
Caribou	5	7	26	22	69	71	499	163
Moose	1	1	16[a]	25	83[a]	75	522	175
Wolves	63[a]	72	29[a]	19	8	9	454	140
Grizzlies	61[a]	55	33	38	6	7	445	140

[a]Significant difference between Nelchina Basin and statewide respondents (x^2 test, $p \leq 0.05$)

Predator-Prey Relationships

Most respondents perceived an imbalance regarding wolves and prey species in the Nelchina Basin. A consistent 70 percent of statewide and Nelchina Basin respondents thought wolves were causing caribou numbers to decline. About 25 percent (27% for Nelchina Basin respondents) thought wolves were having little effect on caribou numbers, and the remainder thought caribou numbers were increasing despite predation. Slightly more Nelchina Basin respondents (76%) than statewide respondents (71%) thought wolves were causing moose numbers to decline.

Fewer respondents, but still a majority, thought grizzlies were causing prey populations to decline in the Nelchina Basin; 61 percent of Nelchina Basin and 57 percent of statewide respondents thought grizzlies were causing caribou numbers to decline, and 72 percent of Nelchina Basin and 62 percent of statewide respondents thought grizzlies were causing moose numbers to decline—a significant difference.

Concern about Predation Impacts

The majority of respondents were either moderately or strongly concerned about the effects of wolves on prey populations. Nelchina Basin respondents showed significantly stronger concern; 52 percent showed strong concern compared to 41 percent of statewide respondents. The level of concern shown for grizzly predation was similar to that regarding wolves for Nelchina Basin respondents (52% were strongly concerned; 22% were moderately concerned). The concern of statewide respondents was less strong (35% were strongly concerned; 25% were moderately concerned), yet a majority expressed either strong or moderately strong concern.

The majority of respondents of both strata indicated that the caribou population is not large enough to meet human use needs (71% of Nelchina Basin and 67% of statewide respondents). Moreover, larger proportions indicated that the existing moose population is not large enough to meet human use needs (78% of Nelchina Basin and 77% of statewide respondents).

Management Preferences

Roughly 80 percent of respondents from each sample wanted to see caribou and moose populations increase (Table 3). Over 60 percent of Nelchina Basin respondents wanted grizzly and wolf populations to decrease. A majority of the statewide respondents (59%) wanted the wolf population to decrease. However, statewide respondents were split as to whether the grizzly population should decrease or remain the same.

Table 3. Percentage of respondents with an opinion who wanted predator and prey populations to increase, stay the same, or decrease.

	Increase		Stay the same		Decrease		n	
	Nelchina Basin	Statewide	Nelchina Basin	Statewide	Nelchina Basin	Statewide	Nelchina Basin	Statewide
Grizzlies	6	13	32[a]	44	62[a]	43	550	235
Wolves	8	11	25[a]	30	67[a]	59	546	233
Caribou	82	80	18	18	0	2	572	250
Moose	84	80	15	19	1	1	577	249

[a]Significant difference between Nelchina Basin and statewide respondents (x^2 test, $p \leq 0.05$)

Over 90 percent of those who wanted the grizzly population reduced favored accomplishing it by a liberal hunting season (Table 4). Allowing ADF&G to kill problem bears was supported by approximately half of respondents (51% for the Nelchina Basin stratum and 46% of the statewide stratum). The use of contraceptives was the least popular alternative for population reduction.

Similar results were found for actions to reduce the wolf population. The vast majority of those who wanted the wolf population reduced also favored accomplishing it through liberal hunting and trapping seasons (Table 5). Just over half found it acceptable for ADF&G to kill wolves from aircraft. Less than 20 percent of all respondents in each stratum found the use of contraceptives acceptable.

Table 4. Percentage of respondents (by stratum) who found various means of population control acceptable for reducing the grizzly population in the Nelchina Basin (n = 908).

	Percentage of all respondents supporting		Percentage of support from those favoring population reduction	
	Nelchina Basin	State-wide	Nelchina Basin	State-wide
Maintain a liberal hunting season	83[a]	67	95	99
Allow ADF&G to kill problem grizzlies	43[a]	34	51	46
Use of long-term, permanent contraceptives	14	15	20	17
Use of temporary contraceptives	15	18	24	18
Population reduction is not needed	17[a]	25	NA	NA

[a]Significant difference between Nelchina Basin and statewide respondents(x^2 test, $p \leq 0.05$)

Table 5. Percentage of respondents (by stratum) who found various means of population control acceptable for reducing the wolf population in the Nelchina Basin (n = 910).

	Percentage of all respondents supporting		Percentage of support from those favoring population reduction	
	Nelchina Basin	State-wide	Nelchina Basin	State-wide
Maintain a liberal hunting season	78[a]	64	93	92
Maintain a liberal trapping season	74[a]	56	87	81
Use of long-term, permanent contraceptives	16	19	18	27
Use of temporary contraceptives	14	15	17	22
Population reduction is not needed	15[a]	22	NA	NA
Allow ADF&G to kill wolves	46[a]	38	55	54

[a]Significant difference between Nelchina Basin and statewide respondents (x^2 test, $p \leq 0.05$)

Discussion

Our data suggest that respondents, particularly those who lived in the study area, perceived a predator-prey imbalance. Survey respondents generally believed that prey populations were low and predator populations were high in the Nelchina Basin. Predation, especially by wolves, was believed to be a cause of prey decline. Respondents overwhelmingly wanted moose and caribou numbers to increase. Consistent with this desire, our data indicate prey populations were believed to be below desirable levels for various human benefit considerations. Despite concerns about predation, it seems that the desire to increase prey is stronger than the desire to decrease predators; one could argue that, if these respondents are reflective of the population, Alaskans' concern was focused more on supply of prey than reduction of predators. If this is true, it seems that any public deliberations on similar predator-prey issues might best be framed in terms of the number of prey animals needed for reasonable availability for human use, not in terms of target numbers of predators to be killed or left remaining in the extant population.

The data indicate that respondents not only believed a predator-prey imbalance existed in Nelchina Basin, but also that it warranted attention. This assessment of respondent concern is supported by responses to multiple questions. Two-thirds of respondents indicated moderate to strong concern about predation. Half to two-thirds of respondents wanted predators to decline, and four-fifths wanted prey populations to increase. Also, two-thirds to three-quarters of respondents believed prey populations were inadequate to meet human needs in the Nelchina Basin. The decline in prey was attributed to predators by most respondents. It is remarkable that so many Alaskans were interested in predator-prey relations in the Nelchina Basin. Moreover they were significantly concerned about predator-prey imbalance there, especially considering many respondents did not reside in or near Nelchina.

These findings have implications for predator and prey management in the Nelchina Basin. Nelchina Basin residents may have a special stake in predator and prey management in the area, but considering this a local or even regional issue would be too. Alaskans from outside the area may expect to be involved in any decision-making processes leading to management of the predators in the Basin.

With the exception of long-term or temporary contraceptives, significant differences existed between Nelchina Basin and statewide respondents regarding acceptance of management interventions. Nelchina Basin residents were more supportive of lethal management actions than statewide respondents for both grizzlies and wolves. Statewide respondents also were less likely to think population reduction for grizzlies or wolves was necessary; although, relatively few respondents from either stratum made that assertion. Of the population reduction methods inquired about in the survey, respondents favored the use of liberal hunting (for wolves and bears) and trapping (for wolves) seasons by a wide margin. There appears to be enough opposition to ADF&G staff killing predators that the public should be given considerable information about ADF&G using its staff to reduce predators even to supplement liberal hunting and trapping seasons for predators. Contraception techniques, either permanent or temporary measures, were not well received by respondents. The study did not delve into detail about acceptability of the allowable kinds of liberalized hunting or trapping regulations that might be considered.

Conclusion

The manager's model provides a useful conceptual tool to focus human dimensions inquiry regarding predator management issues. The model identifies three sets of information needs that are likely to be important to managers when making decisions about predator management. The need for sound biological and ecological information in the management of predators and prey is clear, so it is a component of the model. Our focus, however, is on the remaining two components: (a) perceptions of predator-prey issues (e. g., is there a perceived predator-prey imbalance, and is this a concern); and (b) preferences for predator and prey populations (i. e., desire for populations to increase or decrease) and solutions (i. e., management actions) to achieve optimal numbers of predators and prey. We sought to understand these components at the local and statewide levels to determine whether proximity influences strength of attitudes and salience of predator management issues.

In applying this model to management of predators (wolves and grizzlies) and prey (moose and caribou) in Alaska, we found that differences between the two strata were a matter of degree when it came to perceptions of the problem and solution preferences. Although most management problems have a specific geographic focus, many regulatory decisions are made on a statewide basis. Thus it is important for managers to know if disparities in problem assessment exist between local—people more likely to be impacted by predators or management decisions—and statewide residents. An understanding of whether a problem has statewide as well as local significance can help managers anticipate the level of controversy that might emerge, determine the extent and kind of information or education needs that exist, design citizen participation approaches and identify the scale of public interest in the outcome of any management decisions that are made.

Wolf and grizzly management in Alaska may be intractable issues in their own right, but we can learn a great deal from it that is useful for conceptualizing the human dimensions of predator management. The general manager's model and the manager's model of insight needs that we present are practical, conceptual tools, likely applicable to many predator management situations in North America. We hope to conduct additional inquiries to evaluate the models. We also hope other wildlife managers review these ideas and, if they choose to experiment with them, relate their experiences in applying them in other contexts.

Reference List

Dillman, D. A. 2000. *Mail and Internet surveys: The tailored designed method.* New York, New York: John Wiley & Sons, Inc.

Hook, R. A., and W. L. Robinson. 1982. Attitudes of Michigan citizens toward predators. In *Wolves in the World*, eds. G. H. Harrington, and P. C. Paquet. Park Ridge, New Jersey: Noyes.

Kellert, S. R. 1985. Public perceptions of predators, particularly the wolf and coyote. *Biological Conservation.* 31:167–89.

Manfredo, M. J., H. C. Zinn, L. Sikorowski, and J. Jones. 1998. Public acceptance of mountain lion management: A case study of Denver, Colorado and nearby foothills areas. *Wildlife Society Bulletin.* 26:964–70.

Minnis, D. L., and R. B. Peyton. 1993. Cultural carrying capacity: Modeling a notion. *Proceedings of urban deer: A manageable resource?* St. Louis, Missouri: 55th Midwest Fish and Wildlife Conference.

Nie, M. A. 2003. *Beyond wolves: The politics of wolf recovery and management*. Minneapolis, Minnesota: University of Minnesota Press.

Regelin, W. L. 2002. *Wolf management in Alaska with an historic perspective*. Fairbanks, Alaska: Alaska Board of Game.

Riley, S. J., and D. J. Decker. 2000. Wildlife stakeholder acceptance capacity for cougars in Montana. *Wildlife Society Bulletin*. 28(4):931–9.

Riley, S. J., W. F. Siemer, D. J. Decker, L. H. Carpenter, J. F. Organ, and L. T. Berchielli. 2003. Adaptive impact management: An integrative approach to wildlife management. *Human Dimensions of Wildlife.* 8(2):81–95.

Zinn, H. C., M. M. Manfredo, J. J. Vaske, and K. Wittman. 1998. Using normative beliefs to determine the acceptability of wildlife management actions. *Society & Natural Resources.* 11:649–62.

Zinn, H. C., and C. L. Pierce. 2002. Values, gender, and concern about potentially dangerous wildlife. *Environment and Behavior.* 34(2):239–56.

Coyote Attacks: An Increasing Suburban Problem

Robert M. Timm
University of California, Hopland Research & Extension Center
Hopland

Rex O. Baker
California State Polytechnic University-Pomona (retired)
Corona

Joe R. Bennett
U. S. Department of Agriculture, Animal
and Plant Health Inspection Service, Wildlife Services
Taft, California

Craig C. Coolahan
U. S. Department of Agriculture, Animal
and Plant Health Inspection Service, Wildlife Services
Sacramento, California

Introduction

Coyote (*Canis latrans*) attacks on humans, once thought to be rare, have increased in frequency over the past decade. In expanding suburban areas, such as those found in several counties in southern California, residential developments are often near steep, brushy wildland areas. Coyotes inhabiting such wildlands are drawn into suburban landscaped environments, which can support an abundance of rodents and rabbits, where they can utilize water sources, pet food, household refuse and even house cats and small dogs as prey.

Our observations indicate that, in the absence of harassment by residents, coyotes can lose their fear of people and associate humans with this safe, resource-rich environment. This problem is exacerbated by people who intentionally feed coyotes. In such situations, some coyotes have begun to act aggressively toward humans, chasing joggers and bicyclists, confronting people walking their dogs, and stalking small children.

We queried representatives of various federal, state, county and city agencies as well as private wildlife control companies about coyote attacks on humans occurring in southern California during the past three decades, giving particular attention to localities where attacks previously had been verified (see Howell 1982, Baker and Timm 1998). From the information gathered, we now list 89 coyote attacks in California (incidents when one or more coyotes made physical contact with a child or adult or attacked a pet while in close proximity to its owner) (Table 1). In 56 of these attacks, one or more persons suffered an injury (Figure 1). In 77 additional encounters (not listed), coyotes stalked children, chased individuals, or aggressively threatened adults. In 35 incidents (not all listed), where coyotes stalked or attacked small children, the possibility of serious or fatal injury seems likely if the child had not been rescued. Because no single agency maintains data on such attacks and because some agencies and organizations are reluctant to discuss such incidents, we do not have data on all attacks that have occurred.

Table 1. Coyote attacks in California, 1978 to 2003, listed chronologically.

Date	Location	Attack details
May 1978	Pasadena	5-yr-old girl bitten on left leg while in driveway of home.
May 1979	Pasadena	2-yr-old girl attacked by coyote while eating cookies on front porch; grabbed by throat and cheek.
June 1979	Pasadena	Adult male bitten on heel while picking up newspaper from front yard.
July 1979	Pasadena	17-yr-old female's leg lacerated by coyotes while attempting to save dog being attacked.
July 1979	Pasadena	Coyote bit adult male on legs while jogging; climbed tree to escape.
Aug. 1979	La Verne	Coyote grabbed 5-yr-old girl and attempted to drag her into bushes; suffered deep bites on neck, head, and legs before saved by father and a neighbor.
July 1980	Agoura Hills	13-month-old girl grabbed and dragged off by coyote. Suffered puncture wounds to midsection before being saved by mother.
Aug. 1981	Glendale	3-yr-old girl killed in front yard by coyote; massive bleeding and broken neck.
Aug. 1988	Oceanside	4-yr-old boy nipped and bruised by coyote while playing in yard. (Morning)
Aug. 1988	Oceanside	8-yr-old girl approached by coyote while roller-skating after she had fallen. Coyote tugged at her skate, and was scared off by two women who threw rocks. (Morning)
Aug. 1988	Oceanside	Coyote grabbed 3-yr-old girl by the leg and pulled her down, then bit her on head and neck. Coyote chased off by mother and neighbors. (7 p. m.)
Oct. 1988	San Diego	Adult female bitten by coyote in back yard while talking on phone. (Daytime)

Table 1 (continued). Coyote attacks in California, 1978 to 2003, listed chronologically.

Date	Location	Attack details
June 1990	Reds Meadow	5-yr-old girl attacked and bitten in head while in sleeping bag at campground. (3 a. m.)
June 1990	Reds Meadow	One person bitten on foot through sleeping bag; one bitten on hand; same campground as above.
Sept. 1991	Laguna Niguel	Man chased, and his poodle was ripped from his arms; poodle taken by coyote.
Mar. 1992	San Marcos	Adult female attacked and bitten on face while rescuing pit bull pup from attack in her yard.
Apr. 1992	Fallbrook	Grove worker bitten by coyote.
May 1992	San Clemente	5-yr-old girl attacked and bitten several times on her back, climbed swing set to get away; mother chased off coyote. (Daytime)
Oct. 1992	Fallbrook	10-yr-old boy attacked and bitten on head while asleep on back porch of residence. (4 a. m.)
Oct. 1994	Griffith Park	Man with no shirt or shoes bitten by coyote. (5 p. m.)
Mar. 1995	Griffith Park	Man with no shirt bitten by coyote. (12 p. m.)
Mar. 1995	Griffith Park	Coyote stalked and then knocked down 5-yr-old girl twice; mother rescued child. (Daytime)
June 1995	Griffith Park	Woman in shorts, barefoot, preparing food, bitten by coyote. (Daytime)
June 1995	Laguna Niguel	Man attacked while lying on chaise lounge, bitten on bare foot. (Night)
June 1995	Laguna Niguel	Man bitten on bare foot while getting newspaper from yard. (Mid-morning)
June 1995	UC Riverside	Three boys chased; 7-yr-old bitten. (Late afternoon)
July 1995	Griffith Park	Man bitten by coyote while sleeping on lawn. (2:45 p. m.)
July 1995	Griffith Park	Man bitten by coyote while sleeping on lawn. (4 p. m.)
July 1995	Griffith Park	Coyote was chased away once; then returned to attack 15-mo-old girl in jumpsuit; child suffered bites to leg. (4 p. m.)
Sept. 1995	Fullerton	3-yr-old girl attacked in yard, bitten on face, head, and thigh. (6:30 p. m.)
Nov. 1995	UC Riverside	Children chased while playing; 3-yr-old boy bitten.
June 1996	Los Altos	Coyote grabbed 3-yr-old boy by hand and dragged him toward bushes; treated for bites on scalp and hand. 15-yr-old brother scared coyote away. (8 PM)
Jan. 1997	San Juan Capistrano	Two women attacked; one bitten twice on left ankle and pulled to ground. Both yelled, used alarm device, and swung handbag.
Jan. 1997	San Juan Capistrano	Coyote attacked adult female, grabbed lunch pail and ran.
Jan. 1997	San Juan Capistrano	Coyote charged adult female, took purse containing lunch.
Jan. 1997	San Juan Capistrano	Coyote charged adult female and took purse.

Table 1 (continued). Coyote attacks in California, 1978 to 2003, listed chronologically.

Date	Location	Attack details
Jan. 1997	San Juan Capistrano	Coyote attacked man, bit shoe, no injury. Coyote refused to retreat. (Before daylight)
Jan. 1997	San Juan Capistrano	Coyote jumped on back of man, biting his backpack. Was knocked off and retreated.
Feb. 1997	South Lake Tahoe	Man attacked and bitten on hand while feeding coyote. (Late morning)
Feb. 1997	South Lake Tahoe	4-yr-old girl in yard attacked and severely bitten; heavy snowsuit protected all but face. Father rescued child. Coyote stayed in unfenced yard until shot by police. (Late morning)
Sept. 1997	Pomona	Man was stalked, then attacked by two coyotes and bitten on ankle. (Early evening, daylight)
Nov. 1998	San Mateo County	Coyote approached group of 4 women hikers and bit woman on buttocks.
Nov. 1998	San Mateo County	Coyote approached 3 women hikers, grabbed one by pant her leg, let go, attempted to attack again.
Spring 1999	South Lake Tahoe area	Two adults bitten by coyotes.
Spring 1999	South Lake Tahoe area	Woman bitten by coyote in parking lot of motel.
May 1999	Canyon Country	Coyote attacked dog in yard, and would not cease attack; man scratched in melee. (Night)
Aug. 1999	Green Valley Lake	Coyotes attacked woman and her dogs in yard; one dog bitten. Woman and dogs escaped to vehicle; coyotes jumped aggressively on car and scratched it. (8:30 a. m.)
Aug. 1999	San Antonio Heights	Three coyotes attacked and killed dog being walked on leash by elderly man.
Oct. 1999	Ventura County	Six coyotes attacked man on bicycle with his dog; dog bitten.
Nov. 1999	Hollywood Hills	Coyote attacked and killed pet dog in man's presence; coyote would not leave. (Morning)
Feb. 2000	Calimesa	Adult male attacked in back yard by coyote while attempting to rescue dog; suffered cuts, scrapes and bruises. (9 p. m.)
May 2000	La Mesa	3-yr-old boy bitten on his side; treated for 4 puncture wounds. (7 p. m.)
May 2000	Dublin area	Coyote killed small dog while woman was walking it.
Oct. 2000	Oildale	Pair of coyotes treed woman's pet cat, then turned aggressively on her.
April 2001	Pomona	54-year-old woman fought, using an axe handle, with a large coyote that had attacked small poodle in back yard. Received bite on leg, and despite her efforts, the coyote killed the poodle and jumped over fence carrying the carcass. (4:30 p. m.)
June 2001	Frazier Park	22-yr-old female camp counselor sleeping in open awakened by coyote sniffing and pawing at her head. (2 a. m.)

Table 1 (continued). Coyote attacks in California, 1978 to 2003, listed chronologically.

Date	Location	Attack details
June 2001	Northridge	7-year-old girl attacked and seriously injured by a coyote, despite mother's attempts to fight off the coyote. (7 p. m.)
July 2001	Thousand Oaks	Five coyotes attacked large dog in yard, and aggressively threatened residents attempting to rescue dog; would not leave area despite two visits by sheriff.
July 2001	Irvine	3-yr-old boy bitten by coyote in leg while playing in yard; attack interrupted by father, who was 10-20 ft. away at time of bite. (8:15 p. m.)
July 2001	Tustin	Coyote bit woman.
July 2001	Encinitas	Coyote attacked and took dog, while it was being walked on leash by woman. (4 p. m.)
Aug. 2001	Hollywood Hills	Coyotes bit man 8 times as he was defending his dog against their attack. (11:50 p. m.)
Aug. 2001	Irvine	Woman walking poodle on leash bitten by coyote while attempting to remove dog from coyote's mouth. (4:30 p. m.)
Aug. 2001	Chatsworth	Two coyotes came into yard and took pet cat out of hands of 19-mo-old toddler.
Sept. 2001	Agoura	Woman attacked by coyote when she attempted to stop its attack on her small dog. (7:15 a. m.)
Sept. 2001	Lancaster	Man walking encountered 4 coyotes, which crouched, circling him, attempting to attack; fought off with walking stick, hitting one square across the face. (Morning)
Oct. 2001	San Clemente	Coyote attacked children on schoolyard; 8-yr-old girl bitten on back of neck and scratched; 7-yr-old boy bitten on back and arm. Third student attacked but coyote bit backpack. (12:15 p. m.)
Nov. 2001	San Diego	8-yr-old girl bitten in leg by coyote that family had been feeding at their apartment. (1:30 p. m.)
Nov. 2001	La Habra Heights	Coyote on golf course ran up to woman, jumped on her back, and bit her on right forearm. (Daytime).
Dec. 2001	San Gabriel	Coyote bit 3-yr-old girl in head; grabbed her shoulder in an attempt to drag her off. Father chased coyote off. (7:30 p. m.)
May 2002	Anza Borrego St. Park	Coyote bit boy in sleeping bag on the head.
May 2002	Los Angeles	Coyote attacked man walking his dog.
July 2002	Woodland Hills	Adult female attacked by coyote, bitten on arm. (6 a. m.)
July 2002	Woodland Hills	Adult male bitten on boot by coyote when he inadvertently came upon it between car and garage.
July 2002	Canoga Park	Woman walking 2 large dogs accosted by 3 coyotes; fell backward and fended coyotes off.
July 2002	Carlsbad	Woman walking Labrador retriever accosted by 8 to 10 coyotes, which bit at her legs and pants after she tripped and fell; her dog fought off the coyotes until she could escape. (10 p. m.)

Table 1 (continued). Coyote attacks in California, 1978 to 2003, listed chronologically.

Date	Location	Attack details
Aug. 2002	Mission Hills	Coyote approached couple walking dog, attempting to snatch dog out of man's arms; left only after being kicked. (4 a. m.)
Nov. 2002	Carbon Canyon	Coyote came into trailer park and took dog in presence of its owner. (3 p. m.)
Nov. 2002	Woodland Hills	Coyote scaled 6-ft. wall into yard, attacked and killed small dog in presence of owner; in melee, woman kicked coyote, then fell and fractured her elbow and was attacked and scratched by coyote. (1 p. m.)
Dec. 2002	East Highland	Utility worker attacked by coyote, which tore his trousers. (Evening)
Dec. 2002	East Highland	Coyote attacked adult male. (Evening)
Feb. 2003	Lake View Terrace	Jogger bitten (tooth scrape on ankle) by coyote after jogging past neighborhood coyote feeding station.
May 2003	Woodland Hills	Coyote acted aggressively toward man after he intervened during its attack on his dog.
May 2003	Highland	Coyote came into neighbor's garage after 2-yr-old girl, biting her on arm. (10 p. m.)
May 2003	Woodland Hills	Coyote came into residence to attack small pet dogs. (2 p. m.)
July 2003	Granada Hills	Boy walking family's 2 dogs attacked by 3 coyotes; one dog was killed and the other injured; rescued by father.
July 2003	Alta Loma	Coyote grabbed her small dog while woman was walking it; she was able to rescue it.
Aug. 2003	Apple Valley	4-yr-old boy attacked on golf course; bitten on face and neck; saved by father. (Late afternoon)
Nov. 2003	Claremont	Man and his dog attacked by 3-4 coyotes; he defended himself, hitting several coyotes with his walking stick. (8 a. m.)

Figure 1. Four-year-old Lauren Bridges suffered multiple wounds to her face, of which 16 required stitches, when attacked by a coyote in the yard of a South Lake Tahoe, California residence in February 1997. Photo credit: Steve Bridges, father of the victim

We also questioned representatives of agencies and private firms about the results of their corrective and preventive actions taken in relation to coyote attacks. We summarize and discuss this information as a contribution toward improved strategies to deal with this wildlife-human conflict.

The Changing Suburban Environment

Urban sprawl throughout southern California, now extending across valleys and flat lands adjacent to mountain slopes and arroyos thickly vegetated with chaparral and mountain scrub, provides miles of habitat edge between residential developments and wildlands. Driven by new landscape ordinances, increased affluence and desire to create lush and attractive landscapes in new developments, humans have now created (within as few as 6 years) rich landscapes that are more attractive to rodents, rabbits and other wildlife (Baker 1984). These new habitats, as well as landscaped freeway rights-of-way, may develop significant populations of rabbits (*Sylvilagus* spp.), pocket gophers (*Thomomys bottae*), ground squirrels (*Spermophilus beecheyi*), meadow voles (*Microtus* spp.) and commensal rodents (*Rattus* spp. and *Mus musculus*). Such areas serve as corridors for coyote movement within suburban areas, and they are sufficiently rich in resources to serve as permanent coyote habitat.

Urban Coyote Ecology

Coyotes in wildland environments typically feed on numerous small mammals, birds, reptiles, arthropods, fruit, seeds, other plant materials and carrion (Bond 1939, Sperry 1941, Young and Jackson 1951, Ferrel et al. 1953). Many investigators have concluded that coyotes are omnivorous feeders and opportunistic predators (Van Vuren and Thompson 1982), varying their diet with seasonal availability but perhaps relying on learned behaviors. While rodents and rabbits are typically main components of a coyote's diet, local food habits often reflect the composition of the local prey base (Fichter et al. 1955, Knowlton 1964).

Suburban coyotes consume many human-related foods as partial substitutes for natural food items. Recent studies of suburban coyotes (MacCracken 1982, Wirtz et al. 1982, Shargo 1988, McClure et al. 1995) confirm that these animals rely heavily on food items present in the suburban landscape (e. g., "garbage," chicken, rabbit, melons, avocado, zucchini).

Analyses of coyote scat collected near Claremont, California revealed that coyotes relied heavily on pets and rabbits in winter and spring (Wirtz et al. 1982); similarly, in Malibu, domestic cat was found in 13.6 percent of coyote scats (Shargo 1988). Historian and storyteller, J. Frank Dobie, quotes early naturalist Vernon Bailey as having said that coyotes take "special delight" in killing domestic cats (Dobie 1949:71). At one location in southern California near the site of a coyote attack, coyotes were relying on a feral cat colony as a food source. Over time, the coyotes killed most of the cats and then ate the cat food placed daily at the colony site by citizens who were maintaining the cat colony (Baker and Timm 1998).

Complaints of coyote attacks and predation on pets received by U. S. Department of Agriculture, Wildlife Services (USDA-Wildlife Services), mainly from suburban areas in California, have increased during the last decade. Such reports rose from 17 incidents in federal fiscal year (FY) 1991 to 149 incidents in FY1997 and to 281 incidents in FY 2003. These attacks were reported from nearly all of 39 counties having cooperative programs with USDA Wildlife Services. Recent newspaper reports of coyote attacks on pets have also come from Las Vegas, Nevada; Tulsa, Oklahoma; St. Louis, Missouri; Eastham, Massachusetts; and Greenwich, Connecticut. Officials in the Vancouver, British Columbia Ministry of Environment, Lands and Parks documented a 315-percent increase in coyote complaints from 1985 to 1995 (City of Vancouver 1995). Coyote attacks on pets reported in Texas rose more than four-fold during the last decade (66 attacks in FY 1994 vs. 284 attacks in FY 2003) (Gary L. Nunley, personal communication 2004).

Food abundance regulates coyote numbers by influencing population density as well as reproduction, survival, dispersal and space-use patterns (Gier 1968, Todd and Keith 1983, Gese et al. 1996, Knowlton et al. 1999). Where resources are plentiful, coyote territories and home ranges are significantly smaller than where resources are scarce. Male coyotes in the wild generally have home ranges from 8.1 to 16.1 square miles (21–41.6 km^2) and females 3.1 to 3.9 square miles (8–10 km^2) (Gipson and Sealander 1972, Chesness and Bremicker 1974); although, home ranges of dominant, territorial coyotes on a northern California sheep ranch have been measured at 1.2 to 2.9 square miles (3.0–7.4 km^2) with an average of 1.9 square miles (5.0 km^2) in what was regarded as a food-rich rangeland environment (Neale et al. 1996, Sacks 1996). Estimates of coyote densities throughout the West and the Midwest are typically 0.2 to 1.5

coyotes per square mile (0.5–3.9 per km^2) but with occasionally 5 to 10 coyotes per square mile (13–26 per km^2) reported (U. S. Fish and Wildlife Service 1978). Suburban coyotes in Southern California were found to occupy home ranges of only 0.25 to 0.56 square mile (0.64–1.44 km^2) (Shargo 1988). This suggests that suburban environments are extraordinarily rich in resources for coyotes, which leads to high densities. Following the lethal attack on a three-year-old girl in Glendale in August 1981, authorities removed 55 coyotes from within one-half mile (0.8 km) of the attack site over a period of 80 days (Howell 1982).

Changes in Coyote Behavior

Young and Jackson (1951:69) relate a 1947 report from Yellowstone National Park in which park staff described two coyotes habituated to tourists. They noted that, while in the past, park visitors, "were lucky to even see a glimpse," of a coyote, now these two animals were extensively observed begging for food and posing for pictures, causing tourist traffic jams along the main park highway, an occurrence, "until now unheard of in Yellowstone's colorful history." Parker (1995) describes two instances in which coyotes bit visitors to Cape Breton Highlands National Park in Nova Scotia. In both cases, he noted that the coyotes responsible had grown accustomed to tourists feeding them, even though such feeding is strictly prohibited.

The typical activity pattern of coyotes in the absence of human harassment seems to be crepuscular and diurnal, but, when predator control activities are undertaken, coyotes shift their activity mainly to nighttime to avoid humans (Kitchen et al. 2000). Conversely, a lack of human harassment coupled with a resource-rich environment that encourages coyotes to associate food with humans can result in coyotes losing their "normal" wariness of humans. Howell (1982:21) stated that this sort of environment, which had developed in hillside residential areas of Los Angeles County, produced, "abnormal numbers of bold coyotes." At that time, he noted it was not unusual for joggers, newspaper delivery persons and other early risers to observe one to six coyotes daily in such residential areas. By the late 1990s, Baker and Timm (1998) noted that coyotes in this area commonly could be observed feeding in late mornings and afternoons, and residents saw coyotes in yards, on streets (Figure 2), on parks and on golf courses throughout the day. More recently, coyotes have been observed during midday on school grounds. Such behavioral changes appear to be directly associated with increased attacks on humans.

Figure 2. An urban coyote strolls through West Hills, a suburb of Los Angeles, California, in July 2002. Photo credit: Troy Boswell, Department of Animal Regulation, City of Los Angeles.

Based on an analysis of coyote attacks previously described, there is a predictable sequence of observed changes in coyote behavior that indicates an increasing risk to human safety (Baker and Timm 1998). We now define these changes, in order of their usual pattern of occurrence, as follows:

1. an increase in observing coyotes on streets and in yards at night
2. an increase in coyotes approaching adults or taking pets at night
3. early morning and late afternoon daylight observance of coyotes on streets and in parks and yards
4. daylight observance of coyotes chasing or taking pets
5. coyotes attacking and taking pets on leashes or in close proximity to their owners
6. coyote chasing joggers, bicyclists and other adults
7. coyotes seen in and around children's play areas, school grounds and parks in midday
8. coyotes acting aggressively toward adults during midday.

Carbyn (1989) analyzed 10 attacks on humans documented in Canadian and U. S. national parks from 1960 through 1988, concluding that they were predatory in nature; that is, the coyotes, having lost their fear of humans, regarded small children as prey. This opinion has been shared by others who have investigated such attacks (see Baker and Timm 1998). Carbyn noted that of the four most serious attacks, all were on children and three occurred during the season when pups were whelped or were being fed. He speculated that the coyotes' boldness was related to food stress. He also noted the occurrence of additional aggressive responses to humans, at various seasons, that did not fit this pattern (e. g., chasing cars, biting at tires, slashing tents and nipping at campers in sleeping bags), concluding that there may not have been a common basis for these additional aberrant behaviors. The motive for attacks by coyotes is not always hunger (Connolly et al. 1976) or protection of dens. Movement, particularly escape behavior, is a key stimulus for eliciting orientation and attack (Lehner 1976); children's play and running behavior, particularly when running away from a coyote, may provide a strong stimulus for attack.

An Increasing Problem

As far as we know, the first reported coyote attacks on humans in California not involving rabies-induced aggression occurred in the late 1970s, and we document a total of 89 attacks in the state between that time and December 2003. Approximately 79 percent of these have occurred in the last 10 years, indicating that this problem is increasing (Table 1, Figure 3). Of the persons suffering injury, more than half (55 percent) have been adults.

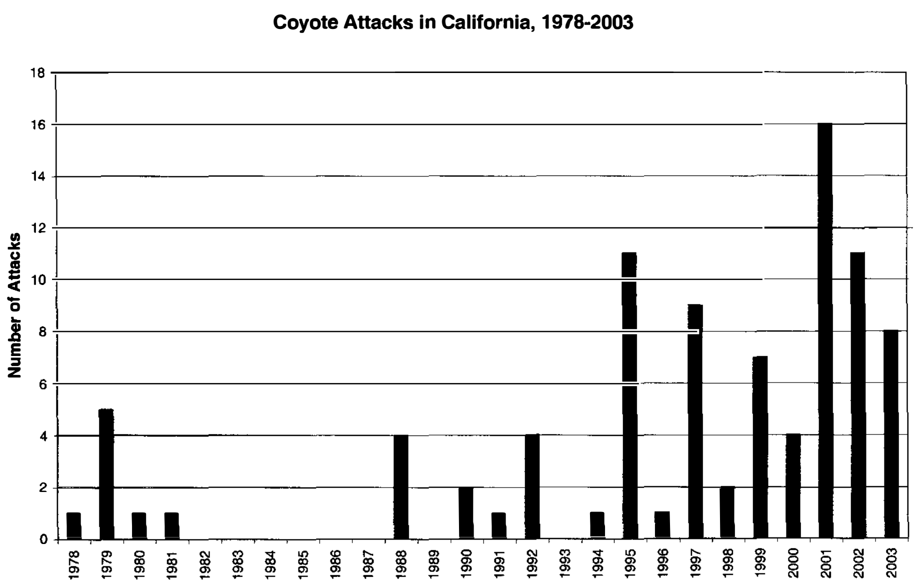


Figure 3. Coyote attacks on humans in California by year, 1978 to 2003.

Of the attacks on children and adults listed in Table 1, 63 percent occurred during the season when adult coyotes would most likely be provisioning pups or experiencing increased food demands because of the female's gestation (March through August), while 37 percent of attacks occurred during the other six months of the year (September through February). When only those attacks directed against children (less than or equal to 10 years of age) are considered, 72 percent occurred during the reproductive season. This lends support to Carbyn's (1989) hypothesis that such attacks may be related to food demands. Alternatively, this seasonality in attacks could be related to other behaviors associated with territoriality, reproduction and defense of den sites or pups.

While most of the coyote attacks on humans in California have occurred in southern California (Los Angeles, Orange, San Diego, San Bernardino and Riverside counties), we list similar attacks that have occurred in Alameda, El Dorado, Kern, Madera, San Mateo and Ventura counties. In recent years, coyote attacks are also reported from Stateline, Nevada; Oro Valley, Scottsdale and Lake Havasu City, Arizona; Durango, Colorado; Eminence, New York; Sandwich, Massachusetts; Vancouver, British Columbia; and Cape Breton, Nova Scotia. Loven (1995) described the way in which coyotes are adapting to the excellent habitat found in many suburban areas throughout Texas, and he noted the recent marked increase in suburban coyote complaints received by offices of the Texas Animal Damage Control Service.

In addition to the human safety issue, coyotes' presence in close association with humans can represent a potential health risk to people and their pets. Rabies, if it were to become established in suburban coyote populations, could easily put humans and domestic animals at risk. An episode of rabies in 16 dogs in Los Angeles in 1921 was suspected to have originated with coyotes or other wildlife. Another rabies outbreak in 1959 to 1960 in the border areas of Mexicali Valley, Baja California, and in Imperial Valley, California is described by Cocozza and Alba (1962). Many newborn calves were lost, and there were multiple coyote attacks on humans, cattle and dogs. Between 1950 and 1995, 28 coyotes were confirmed positive for rabies in California (Ryan 1997). Coyotes also carry the dog tapeworm (*Echinococcus granulosus*), which can cause hydatid cyst disease in humans. Further, coyotes can serve as reservoirs for the canine heartworm (*Dirofilaria immitis*), which is spread to dogs by mosquito vectors (Sacks 1998). They can also serve as hosts for the mite (*Sarcoptes scabiei*) that causes sarcoptic mange in canids.

Discussion and Management Implications

Several factors may have led to the recent increases in predator attacks on humans in North America. Among them are human population growth, suburban sprawl and protection of predator species that were once harassed and suppressed by hunters, trappers and landowners. The number of incidents between humans and coyotes in southern California seems to be related to the human population (or some function that correlates with human population); counties with larger populations have experienced the greatest number of coyote attacks (Table 2).

Table 2. Coyote attacks on humans in California by county versus human population and land area.

County (number of attacks)	Human population[a]	Land area, mi^2 (km^2)
Los Angeles (36)	9.52 million	4,752 square miles (12,308 km^2)
Orange (15)	2.85 million	948 square miles (2,455 km^2)
San Diego (12)	2.81 million	4,526 square miles (11,722 km^2)
San Bernardino (9)	1.71 million	20,150 square miles (52,189 km^2)
Riverside (3)	1.55 million	7,303 square miles (18,915 km^2)

[a] 2000 U. S. Census

Southern California's residential developments in recent years have extended dramatically into landscapes that provide considerably more "edge" between brushy wildlands and the suburbs. This habitat change, which can enrich carrying capacity for coyotes, is partly responsible for growing predator populations in close proximity to humans. One estimate suggests that more than 5,000 coyotes live within the city limits of Los Angeles (Ryan 1997), an area of 469 square miles (1,216 km^2), for an average of 10.7 coyotes per square mile (4.1/km^2).

Reduced coyote control efforts by federal or county agencies, as well as by landowners, may have led to increased coyote attacks in two ways: local coyote numbers are no longer suppressed and coyotes' fear of humans is no longer reinforced by lethal control efforts (i. e., shooting and trapping). Coyote control programs, viewed largely by citizens as agricultural or rural services, have declined as southern California became urbanized and as political and financial support for control programs waned. Concurrently, sport hunting and target shooting in this region have declined as well, severely restricted by municipal, county or state ordinances. These factors have contributed to coyotes' loss of wariness.

Changes in predator management have paralleled a marked change in our society's attitudes toward large predators. Once nearly exterminated from much of their native ranges within the United States, many large predators (e. g., wolves [*Canis lupus*], mountain lions [*Felis concolor*] and alligators [*Alligator mississippiensis*]), now afforded nearly complete protection, have seen significant population growth and range expansion. The recent increase in attacks on humans is not unique to coyotes; half of the 20th century's 14 known deaths from mountain lion attacks in North America occurred in the 1990s. There were 110 attacks on humans by alligators in the United States between 1990 and 1995, compared to 78 alligator attacks in the 1980s, and only 5 recorded alligator attacks between 1830 and 1969 (Lowy 2001). More strikingly, during the past 5 years, several towns and cities in coastal Queensland, Australia, have seen a sharp increase in large packs of dingoes (*Canis lupus dingo*) roaming their suburbs, attracted to these localities by abundant food sources. This has been accompanied by attacks on pets and humans, including a fatal attack in April 2001 on a nine-year-old boy near a tourist campground on Fraser Island (Fleming et al. 2001, Roberts 2001, Rural Management Partners 2003).

Lethal Control

Lethal removal of problem coyotes by use of either leghold traps or shooting has been effective in solving problems when coyotes lose their fear of humans and begin to behave aggressively (Baker and Timm 1998). Number 3 Victor Soft Catch® or other padded leghold traps, when used by experienced trappers, can be quite effective. Pan tension devices can prevent capture of smaller species. When modified with double swivels, shock springs and a short (12- to 16-inch [30–40-cm]) chain, the risk of injury to captured animals is minimal. Twice-daily trap checks in suburban areas will decrease stress on captured animals as well as permit prompt release of any nontargets; captured coyotes typically are humanely euthanized at the site of capture. Frequent trap checking also reduces the opportunity for someone to approach a trapped coyote. Such traps can be used in California, under the provisions of the 1998 antitrap initiative, only when a public health or safety emergency exists. The initiative thus limits the use of padded leghold traps in preventing attacks on humans.

Shooting coyotes has limited feasibility in urban and suburban areas, and it must always be coordinated with local law enforcement agencies. Only experienced personnel should be involved in control measures where shooting is used. Several varmint-type rifles and shotguns can be effective. Night-vision equipment, infrared illumination, laser sights, sound suppressors on rifles and safer ammunition can make shooting operations more efficient and less disturbing in residential areas.

Of all available techniques used to date, trapping has had the greatest observed effect of reinstilling a fear of humans into the local coyote population (Baker and Timm 1998). Where two to five coyotes are trapped in a problem locality, the remaining coyotes will often disperse. Although, this is partially dependent on the size of the area, the number of coyote family units resident and the level of wariness in the animals. At locations where leghold trapping has been used successfully, coyote problems typically have not reoccurred for at least 2 years and usually longer. Presumably the use of other capture devices, such as the Collarum® and foot snares, would have a similar effect. There have also been some observations that shooting to remove problem coyotes can correct bold behaviors in other problem coyotes present in the immediate area (Thompson 1990).

Despite the demonstrated efficacy of lethal control measures in such situations, municipalities are often reluctant to authorize use of traps or shooting because of fear of adverse media coverage or litigation by animal welfare groups. Loven (1995) noted that in many cases in Texas, the tools needed to solve coyote problems in urban areas were not allowed by local authorities. Segments of the public that oppose lethal predator control have erroneously claimed that removal of coyotes subsequently leads to higher coyote populations. Knowlton et al. (1999) state that following removals, populations return to precontrol levels, which are largely controlled by food resources.

Nonlethal Control and Education

Public education efforts to inform citizens about wildlife and habitat are an integral part of programs designed to prevent coyote-human conflicts. Suburban residents need to have a basic understanding of the problem and of its root causes, and only then will there be sufficient public support for taking the actions necessary to prevent most suburban coyote attacks. An effective educational program, combined with use of lethal removal only as a last resort, was very effective in solving coyote-human conflicts in Glendale, California (Baker and Timm 1998).

Educational materials should discuss how residents can avoid attracting all wildlife (not just coyotes, but also their prey) into their yards and the importance of maintaining a fear of humans in wild animals. Neighborhood sanitation, in terms of keeping food sources and water unavailable to coyotes, is very important. Specifically, residents need to understand that coyotes will use pet food, improperly stored household refuse, various fruits and seeds accessible from gardens and from backyard trees, and compost piles as food sources. Backyard bird feeders may attract rodents and rabbits, as will certain kinds of lush landscaping, which in turn attract coyotes. Tall or thick vegetation needs to be cleared, wherever possible, to prevent coyotes from using it for cover near residences. Small pets need to be kept indoors or in well-fenced kennels when they are outdoors. Exclusion methods using fencing can be helpful in dissuading coyotes, as well as rabbits and other prey, from coming into yards, garden areas or other attractive sites. Where coyotes have already begun to be a problem, educational materials should include information on how to react when approached or attacked by a coyote.

Bounds and Shaw (1994) reported, from a survey of 188 U. S. national parks, that, where aggressive coyotes were present, feeding of coyotes by visitors was significantly more commonplace than in parks that did not have aggressive coyotes. In general, intentional feeding of coyotes has often been practiced at locations where subsequent coyote attacks occurred. Therefore it is critical that cities and municipalities develop statutes that prohibit intentional feeding of mammalian wildlife and require adequate sanitation for bird feeders. Many towns have developed such ordinances, but they are difficult to enforce. Some also require that refuse containers have lids that fasten securely and have devices to prevent them from being tipped over; some prohibit placement of refuse containers at the curb before the morning of collection. Neighborhood and homeowner association informational meetings can be helpful in changing attitudes toward predators through peer pressure and shared vigilance. Well-meaning individuals must come to understand that intentional feeding of coyotes dooms them to subsequent lethal control; a fed coyote is a dead coyote. People should be informed that feeding also puts neighborhood children and pets at risk of serious injury or death, as well as increasing risks to humans and pets from coyote-vectored diseases. Where bold coyotes are accustomed to being fed or to finding ample food in a neighborhood, abrupt removal of those food sources may result in aggression toward people or an increased likelihood of attacks on pets or small children. In such instances, it may be prudent for the coyotes to be removed prior to making food unavailable.

Residents can reduce their vulnerability to coyote attack by carrying a walking stick or a canister of pepper spray as a defensive measure, particularly when walking pets. Daily routines and walking routes should be altered, as coyotes will learn and take advantage of people's routines. Exercising pets in midday may be safer than in early morning or late evening when coyotes are most active.

Hazing and Aversive Conditioning

Some educational materials recommend that people harass or attempt to scare coyotes away from residential areas by such techniques as shouting, acting aggressively, waving arms and throwing rocks (U. S. Department of Agriculture 2002). Other techniques, such as shooting starter pistols and pellet guns or such as blasting air horns, have been used with varying degrees of success in the early stages of coyotes' adaptation to suburban settings. It is generally recognized that while some nonlethal approaches to controlling predator damage work well, they may be applicable only to certain situations and some may be of only temporary effectiveness (General Accounting Office 2001). Various methods of hazing coyotes may, when combined with modifications to the environment, reduce the chance that coyotes will lose their wariness of humans. However, once coyotes have begun acting boldly or aggressively around humans, it is unlikely that any attempts at hazing can be applied with sufficient consistency or intensity to reverse the coyotes' habituation. In these circumstances, removal of the offending animal(s) is probably the only effective strategy.

Carbyn (1989) has suggested that coyotes' loss of fear of humans in national parks and in urban areas is linked to predators' association of humans with food at campgrounds and, therefore, is analogous to habituation by bears (Ursidae) to human-provided food sources (Gilbert 1989, Herrero 2002). McCullough (1982) has noted that bears and other wild animals can habituate to stimuli (e. g., attempts at hazing) in the absence of a punishment. That is, the animal will, after repeated exposure to the stimulus, cease responses that are inappropriate or not adaptive (i. e., the animal will not expend time and energy in escape behavior). This concept would seem to apply to coyotes: "Bears can make complex evaluations of benefits and risks. For example, instead of simply fleeing from an encounter [with a human], a bear may back off and wait and, by persistence, obtain the food reward. Thus persistence and a variety of strategies for obtaining food in the face of risks are learned because they are rewarded.

Indeed, ingenuity is fostered. In the absence of punishment, the bear becomes habituated to the human, and its declining perception of risk leads to a greater frequency of obtaining the reward, a self-reinforcing process" (McCullough 1982:29).

McCullough goes on to state that when habituated bears become a problem, negative conditioning is needed: "successful negative conditioning must involve fear, perhaps pain" (McCullough 1982:31). However, "it would be difficult to punish bears severely enough to overcome behavior positively reinforced for long periods of time . . . . Bears in long contact with humans are likely to remain incorrigible and will likely have to be removed in most cases" (McCullough 1982:31). While Jonkel (1994) describes successful efforts in Montana to reinstill fear of humans into problem grizzly bears (*Ursus horribilis*), the cost of such treatments—involving capture, treatment and release—can reach $6,000 per animal and, therefore, would be prohibitive to apply to suburban coyotes.

Preventing Future Attacks

While it can be argued that, at present, risk of human injury as a result of coyote attack is very small in comparison to risk of dog bite, it is also true that humans have tremendous exposure to dogs. One estimate states there are 665,000 domestic dogs (*Canis familiaris*) within Los Angeles (Wolf 2003), compared to perhaps 5,000 coyotes (Ryan 1997). It is impossible to prevent all dog attacks because dogs live in close association with humans, including children. But, we believe it may be possible through management to reduce coyote attacks in suburban areas to nearly zero. We maintain that feasible management strategies can substantially reduce risk of suburban coyote attacks on both humans and pets, and they should be applied before the problems get out of hand. When it is possible to prevent the pain, suffering and potential tragedy associated with such attacks, we believe this should be done.

As coyotes continue to adapt to suburban environments and as their populations continue to expand and to increase throughout North America, coyote attacks on humans can be expected to occur and to increase. To reverse this trend, authorities and citizens must act responsibly to correct coyote behavior problems before they escalate into public health and safety risks for children and adults. It is our experience that when appropriate preventive actions are taken before coyotes establish feeding patterns in suburban neighborhoods, further problems can be avoided. However, this requires aggressive use of scare devices and hazing, as well as correction of many environmental factors that have attracted coyotes to the neighborhood.

Once attacks on pets have become frequent or once other neighborhood or public use area food sources have been used by coyotes for an extended period of time (i. e., for several months or more), lethal control techniques will likely be required to prevent continued attacks on pets or future attacks on children or adults. Following use of padded leghold traps (or other capture devices) or shooting, educational efforts must be emphasized in order to change the neighborhood habitat factors that have precipitated the problem, to prevent the problem's reoccurrence. Such proactive coyote management to prevent human safety risks typically cannot be carried out until residents understand the problem and its causes as well as understand the predictable consequences of inaction. Sadly, such understanding is sometimes not achieved until after an attack has occurred.

Reference List

Baker, R. O. 1984. Commingling of Norway and roof rats with native rodents. In *Proceedings, 11th vertebrate pest conference*, ed. D. O. Clark, 103–11. Davis, California: University of California.

Baker, R. O., and R. M. Timm. 1998. Management of conflicts between urban coyotes and humans in southern California. In *Proceedings, 18th vertebrate pest conference*, eds. R. O. Baker, and A. C. Crabb, 299–312. Davis, California: University of California.

Bond, R. M. 1939. Coyote food habits on the Lava Beds National Monument. *Journal of Wildlife Management*. 3(3):180–98.

Bounds, D. L., and W. W. Shaw. 1994. Managing coyotes in U. S. national parks: Human-coyote interactions. *Natural Areas Journal*. 14(4):280–4.

Carbyn, L. N. 1989. Coyote attacks on children in western North America. *Wildlife Society Bulletin*. 17:444–6.

Chesness, R. A., and T. P. Bremicker. 1974. *Home range, territoriality, and sociability of coyotes in north central Minnesota, November 14*. Denver, Colorado: Coyote Research Workshop.

City of Vancouver. 1995. *Administrative report to standing committee on planning and environment, subject: Environmental grants*. Vancouver, British Columbia: City of Vancouver.

Connolly, G. E., R. M. Timm, W. E. Howard, and W. M. Longhurst. 1976. Sheep killing behavior of captive coyotes. *Journal of Wildlife Management*. 40(3):400–7.

Cocozza, J., and A. M. Alba. 1962. Wildlife control project in Baja California. *Public Health Reports*. 77(2):147–51.

Dobie, J. F. 1949. *The voice of the coyote*. Boston, Massachusetts: Little, Brown, & Co.

Ferrel, C. M., H. R. Leach, and D. Tillotson. 1953. Food habits of the coyote in California. *California Fish and Game*. 39(3):301–41.

Fichter, E., G. Schildman, and J. H. Sather. 1955. Some feeding patterns of coyotes in Nebraska. *Ecological Monographs*. 25(1):1–37.

Fleming, P., L. Corbett, R. Harden, and P. Thomson. 2001. *Managing the impact of dingoes and other wild dogs*. Kingston ACT, Australia: Bureau of Rural Sciences, Deptartment of Agriculture, Fisheries and Forestry.

General Accounting Office. 2001. *Wildlife services program: Information on activities to manage wildlife damage, report GAO-02–138, November.* Washington, DC: General Accounting Office.

Gese, E. M., R. L. Ruff, and R. L. Crabtree. 1996. Social and nutritional factors influencing the dispersal of resident coyotes. *Journal of Wildlife Management*. 53:174–81.

Gier, H. T. 1968. Coyotes in Kansas (revised). *Kansas State College Agricultural Experiment Station Bulletin*. 393:1–118.

Gilbert, B. K. 1989. Behavioral plasticity and bear-human conflicts. *Bear-people conflicts: Proceedings of a symposium on management strategies*, ed. M. Bromley, 1–8. Yellowknife, Northwest Territories: Northwest Territories Department of Renewable Resources.

Gipson, P. S., and J. A. Sealander. 1972. Home range and activity of the coyote (*Canis latrans frustror*) in Arkansas. In *Annual Conference Proceedings of Southeastern Association of Game and Fish Commission*. 26:82–95.

Herrero, S. 2002. *Bear attacks: Their causes and avoidance (revised)*. Guilford, Connecticut: The Lyons Press.

Howell, R. G. 1982. The urban coyote problem in Los Angeles county. In *Proceedings, 10th vertebrate pest conference*, ed. R. E. Marsh, 21–3. Davis, California: University of California.

Jonkel, C. 1994. Grizzly/brown bears. In *Prevention and control of wildlife damage*, eds. S. E. Hygnstrom, R. M. Timm, and G. E. Larson, C-17–C-23. Lincoln, Nebraska: University of Nebraska Cooperative Extension.

Kitchen, A. M., E. M. Gese, and E. R. Schauster. 2000. Changes in coyote activity patterns due to reduced exposure to human persecution. *Canadian Journal of Zoology*. 78:853–7.

Knowlton, F. F. 1964. *Aspects of coyote predation in south Texas with special reference to white-tailed deer*. Ph.D. dissertation Purdue University, West Lafayette, Indiana.

Knowlton, F. F., E. M. Gese, and M. M. Jaeger. 1999. Coyote depredation control: An interface between biology and management. *Journal of Range Management*. 52:398–412.

Wolf, K. 2003. The list 2003: More lists—The animals. *LA Weekly, Supplement*. December 26. http://www.laweekly.com/ink/04/05/the-morelists.php.

Lehner, P. N. 1976. Coyote behavior: Implications for management. *Wildlife Society Bulletin*. 4(3):120–6.

Loven, J. E. 1995. Coyotes in urban areas: A status report. In *Coyotes in the southwest: A compendium of our knowledge, symposium proceedings*, eds. D. Rollins, C. Richardson, T. Blankenship, K. Canon, and S. Henke., 65–7. San Angelo, Texas: Texas A & M University.

Lowy, J. 2001. Fatal encounters with wildlife took leap upward in '90s. *Rocky Mountain News*. August 25.

MacCracken, J. G. 1982. Coyote foods in a southern California suburb. *Wildlife Society Bulletin*. 10(3):280–1.

McClure, M. F., N. S. Smith, and W. W. Shaw. 1995. Diets of coyotes near the boundary of Saguaro National Monument and Tucson, Arizona. *Southwestern Naturalist*. 40(1):101–4.

McCullough, D. R. 1982. Behavior, bears, and humans. *Wildlife Society Bulletin*. 10(1):27–33.

Neale, J. C. C., B. N. Sacks, M. M. Jaeger, and D. R. McCullough. 1996. Resource use by sympatric bobcats and coyotes in northern California (Abstract). In *Abstracts of the 3rd annual conference of the wildlife society*, 129. Cincinnati, Ohio: The Wildlife Society.

Parker, G. 1995. *Eastern coyote: The story of its success*. Halifax, Nova Scotia: Nimbus Publishing Ltd.

Roberts, G. 2001. Warning ignored on killer dingo. *Sydney Morning Herald.* May 1:1.

Rural Management Partners. 2003. *Economic assessment of the impact of dingoes/wild dogs in Queensland, report to Department of Natural Resources and Mines, project LP02/03NRM, October.* Milton, Queensland: Rural Management Partners.

Ryan, P. 1997. Coyotes and practitioners. *PULSE*. July:8–9

Sacks, B. N. 1996. *Ecology and behavior of coyotes in relation to depredation and control on a California sheep ranch*. M. S. thesis, University of California, Berkeley.

———. 1998. Increasing prevalence of canine heartworm in coyotes from California. *Journal of Wildlife Diseases*. 34(2):386–9.

Shargo, E. S. 1988. *Home range, movements, and activity patterns of coyotes* (Canis latrans) *in Los Angeles suburbs*. Ph.D. dissertation, University of California, Los Angeles.

Sperry, C. C. 1941. Food habits of the coyote. *Wildlife Research Bulletin*. 4:1–69.

Thompson, R. A. 1990. *Memo to Western Regional Director: July 10*. Washington, DC: U. S. Departent of Agriculture, APHIS, Animal Damage Control.

Todd, A. W., and L. B. Keith. 1983. Coyote demography during a snowshoe hare decline in Alberta. *Journal of Wildlife Management*. 47:394–404.

U. S. Department of Agriculture. 2002. Urban and suburban coyotes. *U. S. Department of Agriculture, APHIS, Wildlife Services Factsheet*. April:1–2.

U. S. Fish and Wildlife Service. 1978. *Predator damage in the West: A study of coyote management alternatives*. Washington, DC: U. S. Fish and Wildlife Service, Department of the Interior.

Van Vuren, D., and S. E. Thompson, Jr. 1982. Opportunistic feeding by coyotes. *Northwestern Science*. 56(2):131–5.

Wirtz, W. O., M. A. Keller, and W. G. Meikle. 1982. Urban coyotes in southern California: A progress report. In *62nd Annual Meeting, American Society of Mammalogy*. Snowbird, Utah: American Society of Mammalogists.

Young, S. P., and H. H. T. Jackson. 1951. *The clever coyote*. Washington, DC: Wildlife Management Institute.

Restoration and Conflict Management of the Gray Wolf in Montana, Idaho and Wyoming

Edward E. Bangs
U. S. Fish and Wildlife Service
Helena, Montana

Joseph Fontaine
U. S. Fish and Wildlife Service
Helena, Montana

Thomas Meier
U. S. Fish and Wildlife Service
Kalispell, Montana

Carter Niemeyer
U. S. Fish and Wildlife Service
Boise, Idaho

Michael Jimenez
U. S. Fish and Wildlife Service
Lander, Wyoming

Douglas Smith
U. S. National Park Service
Yellowstone National Park, Wyoming

Curt Mack
Nez Perce Tribe
McCall, Idaho

Val Asher
Turner Endangered Species Fund
Bozeman, Montana

Larry Handegard
U. S. Department of Agriculture, Wildlife Services
Billings, Montana.

Mark Collinge
U. S. Department of Agriculture, Wildlife Services
Boise, Idaho

Rod Krischke
U. S. Department of Agriculture, Wildlife Services
Casper, Wyoming

Carolyn Sime
Montana Fish, Wildlife and Parks
Kalispell, Montana

Steve Nadeau
Idaho Department of Fish and Game
Boise, Idaho

Dave Moody
Wyoming Game and Fish Department
Lander, Wyoming

Introduction

The gray wolf (*Canis lupus*) was once distributed throughout North America (Nowak 1995). Wolves kill livestock and compete with human hunters for wild ungulates (Young 1944, Fritts et al. 2003). They rarely threaten human safety, but many people still fear them. These conflicts and the historic public hatred of wolves resulted in extirpation of wolf populations in the western United States by 1930 (Mech 1970). Restoration of mule deer (*Odocoileus hemionus)*, white-tailed deer (*O. virginianus*), elk (*Cervus elaphus*), moose (*Alces alces*) and bighorn sheep (*Ovis canadensis*) populations began in the early 1900s, but large predators, particularly wolves, continued to be persecuted. In 1974, wolves became protected by the federal Endangered Species Act (ESA) and their recovery became the responsibility of the U. S. Fish and Wildlife Service (USFWS).

Wolf restoration in the western United States began in 1986 when a Canadian pack, denned in Glacier National Park (Ream et al. 1989). Management in northwestern Montana emphasized legal protection and building local, public tolerance of nondepredating wolves (Bangs et al. 1995). Wolves from Canada were reintroduced to central Idaho and Yellowstone National Park in 1995 and 1996 to accelerate restoration (Bangs and Fritts 1996, Fritts et al. 1997). The wolf population grew to 756 wolves in the northern Rocky Mountains (NRM) of Montana, Idaho and Wyoming by 2004 (Figure 1, Table 1)(U. S. Fish and Wildlife Service et al. 2004). Many people opposed wolf restoration because of concerns over human safety, potential land-use restrictions, livestock depredations and competition with hunters for wild ungulates. Resolving conflicts, both perceived and real, between wolves and people has been the primary focus of our wolf management program (Bangs et al. 2001).

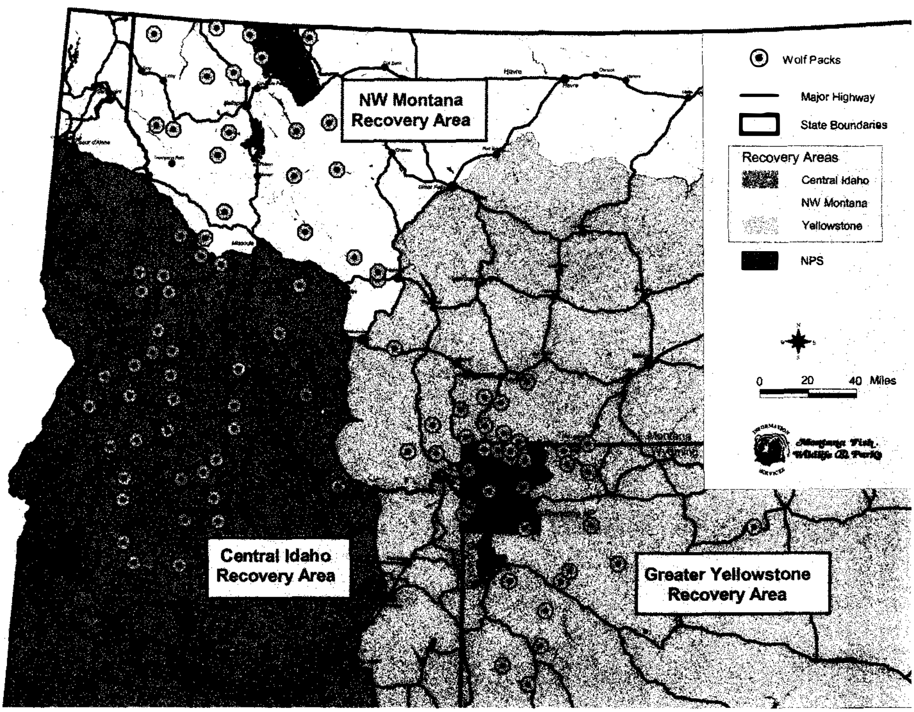


Figure 1. Wolf recovery areas and wolf pack (groups of two or more wolves) distribution in Montana, Idaho and Wyoming, United States, in 2003.

Table 1. Minimum fall wolf population, confirmed wolf depredation[a] and wolf removal in Montana, Idaho and Wyoming, United States, 1987 to 2003.

	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	Total
Montana																		
cattle	6	0	3	5	2	1	0	6	3	10	19	10	20	14	12	20	25	156
sheep	10	0	0	0	2	0	0	0	0	13	41	0	25	7	50	84	86	318
dogs	0	0	0	1	0	0	0	0	4	1	0	1	2	5	2	5	1	22
other[b]	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	5	0	9
total wolves	10	14	12	33	29	41	55	48	66	70	56	49	74	97	123	183	182	
wolves moved	0	0	4	0	3	0	0	2	8	22	20	0	14	6	17	0	0	96
wolves killed[c]	4	0	1	1	0	0	0	0	0	5	18	4	19	7	8	26	34	127
Wyoming																		
cattle	0	0	0	0	0	0	0	0	0	0	2	2	2	3	18	23	34	84
sheep	0	0	0	0	0	0	0	0	0	0	56	7	0	25	34	0	7	129
dogs	0	0	0	0	0	0	0	0	0	0	0	3	6	6	2	0	0	17
other[b]	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	10	11
total wolves	0	0	0	0	0	0	0	0	21	40	86	112	107	153	189	217	234	
wolves moved	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1
wolves killed[c]	0	0	0	0	0	0	0	0	0	0	2	3	1	2	4	6	18	36
Idaho																		
cattle	0	0	0	0	0	0	0	0	0	1	1	9	11	15	10	9	6	62
sheep	0	0	0	0	0	0	0	0	0	24	29	5	64	48	54	15	118	357
dogs	0	0	0	0	0	0	0	0	0	1	4	1	7	0	2	4	4	23
otherb	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
total wolves	0	0	0	0	0	0	0	0	14	42	71	114	156	187	251	263	340	
wolves moved	0	0	0	0	0	0	0	0	0	1	0	3	5	10	1	0	0	20
wolves killed[c]	0	0	0	0	0	0	0	0	0	1	1	0	3	11	7	14	7	44
Total, 3 States																		
cattle	6	0	3	5	2	1	0	6	3	11	22	21	33	32	40	52	65	302
sheep	10	0	0	0	2	0	0	0	0	37	126	12	89	80	138	99	211	804
dogs	0	0	0	1	0	0	0	0	4	2	4	5	15	11	6	9	5	62
other[b]	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	5	10	20
total wolves	10	14	12	33	29	41	55	48	101	152	213	275	337	437	563	663	756	
wolves moved	0	0	4	0	3	0	0	2	8	23	21	3	19	16	18	0	0	117
wolves killed[c]	4	0	1	1	0	0	0	0	0	6	21	7	23	20	19	46	59	207

[a] Numbers of animals confirmed killed by wolves in calendar year.

[b] Includes 1 foal in 1999, 4 llamas in 2001, 5 llamas in 2002, and 10 goats in 2003.

[c] Includes 13 wolves legally shot by ranchers. Others killed in government control efforts.

Study Sites

The NRM Wolf Recovery Plan identified preferred wolf habitat as large areas of public land with adequate year-round wild prey and few livestock (U. S. Fish and Wildlife Service 1987). Based on those criteria, northwestern Montana, central Idaho and the Greater Yellowstone Area (GYA), were recommended for wolf restoration (Figure 1). Each area has a large core of national park or U. S. Forest Service (FS) wilderness, where livestock grazing is limited. Other mountainous habitat is undeveloped federal public land, managed for multiple uses, such as forestry, mining, hunting, recreation and summer livestock grazing. Lower elevation valleys are often private agricultural lands. Wild ungulates, numbering between 100,000 to 250,000 per recovery area, typically disperse to higher elevations in summer, but many winter at lower elevations. Consequently, many wolves also use private land at least part of the year. The proximity of the core habitats to one another and the public land corridors between them, ensure enough connectivity so that wolves in the NRM form one metapopulation (Fritts and Carbyn 1995, Oakleaf 2002). However, core habitats alone are not large enough to support a viable wolf population in the NRM. Wolves must live in and disperse through areas occupied by people and livestock to ensure the population's long term viability. While human-caused mortality has not prevented wolf population recovery, it has affected wolf distribution. Wolves have not been able to persist in open prairie habitat in Montana, Idaho or Wyoming (Figure 1), where wolves are most vulnerable. There, wolves have the highest potential to attack livestock and are least tolerated by local society (Bangs et al. 1995). Except in large blocks of public land in the GYA and in the central Idaho wilderness, wolf pack territories will generally not be contiguous because of conflicts with livestock, steep topography or the seasonal availability of wild prey. The fragmentation of wolf habitat will moderate any potential impacts of the wolf population outside of the core habitats.

Methods

The USFWS-led interagency wolf recovery program has four basic components: monitoring, control, research and public outreach; its data are published annually (U. S. Fish and Wildlife Service et al. 2004). Standard techniques are used to monitor the wolf population but radio telemetry monitoring is emphasized (Bangs et al. 1998). Nearly 550 wolves were captured, by foot-hold trapping, helicopter darting and netgunning, or by coyote trappers. The wolves were radio-collared and monitored from aircraft or the ground, two to four times per month. Law enforcement agents immediately investigated all wolf deaths because the ESA prohibits killing wolves and penalties can be up to one year in jail and a $100,000 fine.

U. S. Department of Agriculture, Wildlife Services (WS) field specialists investigate reports of suspected wolf attacks on livestock within 24 hours, using standard necropsy techniques (Paul and Gipson 1994). A wide variety of wolf control techniques were authorized by the USFWS and were implemented by the WS under a cooperative agreement, when wolf depredation was confirmed. Under certain circumstances, private individuals may harass or kill wolves (Bangs and Fritts 1996, U. S. Fish and Wildlife Service 1994, 2003). Livestock producers are compensated 100 percent for confirmed wolf depredations and 50 percent for probable wolf depredations on livestock, herding and guard animals; this compensation is provided by a private program (Fischer 1989). We also cooperate with several private programs to assist livestock producers to voluntarily reduce the potential for wolf depredation (Bangs and Shivik 2001).

The USFWS initiated and funded research on a wide variety of wolf-related projects. Nearly all of that research involved multiple partners, but it was usually conducted by graduate students. Research focused on the concerns of local residents (i. e. conflicts between wolves and livestock and the potential effect of wolf predation on wild ungulate populations).

All recovery activities involved extensive public information and outreach components, so accurate information was available to other agencies, special interest groups and the public. We accepted hundreds of invitations to give public presentations and participated in thousands of media interviews. Wolves were routinely featured in regional, state, national and international newspapers, in magazines, in books, and on radio, film and television shows. Whether people liked or disliked wolves, everyone appeared eager to read, watch, listen or discuss wolves and wolf management.

Results and Discussion

Human Safety

Wolves in fairy tales and contemporary media are often portrayed as dangerous (Boitani 1995). Attacks on humans are remarkably rare and were often the result of rabies or some human-caused factor, such as food habituation (Linnell et al 2002). In North America, healthy wild wolves have not killed anyone (McNay 2002) and, of 80 wolf-human encounters, 39 cases contained elements of aggression among healthy wolves, 12 involved known or suspected rabid wolves and 29 documented fearless behavior in nonaggressive wolves. In six cases in which healthy, wild wolves acted aggressively, the people were accompanied by dogs. Aggressive nonrabid wolves bit people in 16 cases; 6 bites were severe but none were life threatening. There is no reliable documentation of a person being attacked in the NRM. Cases of rabid wolves infecting people in North America are rare, and the last human fatality was in Alaska in 1945 (Ritter 1981). We believe that accurate public information, zero tolerance by agencies for any wolf that threatened human safety and regulations that allow anyone to immediately kill any wolf that threatens humans resolved that concern for most people.

Land-use Restrictions to Promote Wolf Recovery

Wolves are habitat generalists and can be remarkably tolerant of human activities. They need ungulates to eat, and they need protection from excessive killing by humans. Public surveys indicated that people were much more tolerant of wolf restoration in wild areas if traditional human uses were not restricted (Bath 1992). USFWS has not implemented restrictions on human recreational or commercial activities on private or public land to enhance wolf survival and restoration. Additionally, ESA Section 7 consultations are not required outside of national parks and wildlife refuges within the experimental population areas (U. S. Fish and Wildlife Service 1994). However, there have been sporadic instances where human activities have been modified by public land managers because of the presence of wolves. Currently, M-44 cyanide devices (coyote getters) are prohibited by ESA in areas used by resident wolves. In national parks, people have been prohibited from approaching highly visible wolf dens on foot, primarily to preserve road-side wolf viewing opportunities. In national parks, traffic was slowed to protect human and wolf safety after denning wolves were routinely seen crossing roads and wolf-viewing traffic-jams occurred. In less than ten instances, the FS and private timber companies delayed opening seasonally closed backcountry roads to motorized vehicles. Wolves had denned near those then-closed dirt roads in mid-April. The roads, that normally opened to vehicles on June 1, were kept closed up to an extra month to allow the pups to become mobile. Once, when wolves in northwestern Montana were still listed as endangered, the FS did not permit spring commercial mushroom picking near a new active wolf den. Noncommercial use continued without negative impact. USFWS publicized and implemented a policy whereby traditional public uses of wildlands were not restricted because of wolf restoration (U. S. Fish and Wildlife Service 1987, 1994, 2003). We believe this has increased local tolerance of wolf recovery by natural resource extraction industries and local rural residents.

Wolf Conflict with Livestock

Wolves are effective predators on livestock, and throughout history this has been the overwhelming cause of conflict between humans and wolves (Lopez 1978, Fritts et al. 2003). In the late 1800s, as native prey populations diminished, wolf depredation on livestock was perceived as a major problem (McIntrye 1995), but, while significant, it was likely exaggerated (Gipson et al. 1998). Predictably livestock producers have the strongest dislike of wolves compared to other segments of U. S. society (Boitani 1995, McIntyre 1995, Kellert et al. 1996, Williams et al. 2002).

From 1987 to 2003, wolves in the NRM killed a minimum of 302 cattle, 804 sheep, 9 llamas, 62 dogs, 10 goats, and 1 foal. Cattle and sheep were typically killed in summer when livestock were dispersed on remote grazing pastures and when young livestock were least protected. One third of the cattle and one half of sheep killed by wolves were on public land (Bradley 2004). Wolf depredations were dispersed and sporadic, and few livestock producers had chronic losses. While insignificant to the local livestock industry, wolf depredations can affect the economic viability of a few individual producers, primarily those dependent on remote, public, land grazing allotments.

Sixty-two dogs, primarily those used for hunting or livestock herding and guarding, have been killed by wolves in the NRM. Although animal shelters in each state euthanize thousands more dogs than wolves kill, wolf depredation on dogs is a serious social issue because of the strong emotional attachment of people to their pets and because of the high value of trained working and hunting dogs. Attacks on dogs near homes also raised fears for human safety and anger over the perceived violation of personal space. In recognition of this problem, changes were made to wolf management regulations outside the experimental population areas. In those areas (Washington, Oregon, California, Nevada, northern Idaho, northwestern Montana, northern Utah and northern Colorado), wolves that are attacking herding and guarding animals on private land can be legally shot by landowners (U. S. Fish and Wildlife Service 2003).

We reviewed statistics on livestock numbers, losses and predation in 2000 to assess the relative importance of wolf depredation to the livestock industry (National Agricultural Statistics Service 2001). There were 2,210,000 sheep, 9,300,000 cattle and 437 wolves in Montana, Idaho and Wyoming in 2000 and ranchers reported losing 195,000 sheep and 235,000 cattle from all causes that year. Reportedly, 82,200 sheep (42%) and 10,300 (4.4%) cattle were killed by predators, and coyotes caused 70 percent of that damage. In 2000, wolves killed 80 sheep and 32 cattle; that is 3 and 1 in every 10,000 of overall losses, and 1 and 3 of every 1,000 predator-caused losses, respectively. While wolf depredation was rare overall, seven cattle might be killed and not documented for every one confirmed wolf depredation (Oakleaf et al. 2003).

USFWS stated that depredating wolves would not be the foundation for wolf restoration (U. S. Fish and Wildlife Service 1987, 1988, 1994, 2003). We relocated problem wolves 117 times and killed 207 to reduce conflicts (Table 1). In addition, regulations empowered local landowners and livestock producers to help to control problem wolves (U. S. Fish and Wildlife Service 1994, 2003). Livestock producers are also routinely provided radio telemetry receivers, so they can monitor nearby radio-collared wolves. People can harass wolves in a noninjurious manner at any time. Any wolf seen attacking livestock on private land can be legally shot, and six wolves have been killed in that way. Over a dozen livestock owners obtained USFWS permits to shoot wolves seen attacking livestock on public grazing allotments, but no wolves have been shot. On ranches with chronic livestock depredations, landowners received permits to shoot wolves on-sight. Since the first permits were issued in 1999, seven wolves have been killed. After a brief training course, over 100 landowners were issued permits and less-than-lethal munitions (12-gauge shotgun cracker shells and rubber bullets) to harass wolves near their livestock or dwellings. Several wolves were hit but none seriously injured, and those residents reported wolves seemed more wary. Biologists disturbed soon-to-be-active den sites or harassed wolves from rendezvous sites causing them to relocate their pups away from livestock.

Defenders of Wildlife (Defenders) started a private livestock compensation fund that has paid ranchers nearly $308,000 for confirmed and probable wolf-caused damage to livestock, livestock herding and guard animals from 1987 to 2003 (Fischer 1989, U. S. Fish and Wildlife Service 2003).

Compensation is based on WS field investigations of livestock death. The USFWS, Defenders and other private organizations also assisted livestock producers to voluntarily reduce the potential for wolf depredation by technical assistance or cost-sharing: guard animals, fencing, scare devices, extra livestock surveillance, disposal of livestock carcasses, alternative grazing pastures in lower risk areas, purchase or retirement of public land grazing allotments in areas of chronic conflict and funding research on nonlethal methods to reduce conflicts. While these nonlethal efforts reduced some conflicts, many were expensive to implement, and none has been proven widely effective (Bangs and Shivik 2001). Lethal control of chronic depredating wolves is still widely used and has reduced livestock losses (Bradley 2004).

Wolves are a minor cause of livestock death; correspondingly, we kill relatively few wolves, but both issues are inordinately controversial. Nearly every NRM wolf depredation and control action becomes a local, state or, sometimes national, news story. This high level of publicity exaggerates the effect of both wolf depredation and control. Several county and state governments passed resolutions declaring the wolf an unacceptable species, calling for their extirpation. The USFWS routinely responds to the concerns of local, state and federal elected officials. Control of depredating wolves generates angry correspondence from advocates for animal rights, antipublic land grazing, prowolf factions, and others. The management program has certainly not pleased everyone, but it allowed the wolf population to recover and kept both livestock losses and wolf removals below predicted levels (U. S. Fish and Wildlife Service 1994). The USFWS senses that the public generally views the livestock investigation and wolf control program as professionally administered, fair and responsive, which has helped maintain the tricky political balance between adequate local tolerance for nondepredating wolves and adequate general public tolerance for agency control of problem wolves.

Competition with Hunters for Wild Ungulates

The restoration of ungulate populations by hunters and state game agencies was one of the most remarkable achievements of modern wildlife management. Without it wolf restoration would be impossible. The average adult wolf eats over 12 pounds (5 kg) of prey per day, and wolf predation was a major concern of hunters and state wildlife management agencies. The effect of wolf predation on ungulate populations and hunter harvest remains a major public concern, especially since wolf numbers are now at nearly twice their predicated levels and since some ungulate herds and hunter harvests have declined since wolves became established.

Wolf predation may or may not reduce ungulate populations and hunter opportunity, depending on a wide number of variables (Boyce 1995, Kunkel 1997, Mech and Peterson 2003, Smith et al. 2004). In anticipation of potential conflict, the regulations for wolf reintroduction allowed for the relocation of wolves if ungulate populations were being significantly impacted. To date, no wolves have been moved because there has been no documented need, and any proposed translocations of wolves to benefit ungulate populations would likely encounter vigorous opposition from residents near any potential release site.

The most famous elk herd in North America—the migratory northern range herd, a portion of which winters outside of Yellowstone National Park (YNP) in Montana—illustrates this complex issue. The herd has fluctuated between 10,287 during 1990 to 1991, 19,359 during 1993 and 1994 and less than 9,000 elk in 2003 to 2004. The Montana Department of Fish, Wildlife and Parks administers an antlerless late hunt to manage elk numbers on winter ranges outside of YNP, so elk herds don't exceed the carrying capacity of winter range, cause long-term changes in plant communities or cause unacceptable damage to private property. Since 1988, that hunt removed an average of 1,400 antlerless elk annually. While the herd's current size results from a variety of natural and human-caused factors, including harvesting, wolf predation is often solely blamed. Several organizations, with reportedly thousands of members, have recently formed around rumors that most NRM ungulate populations are being decimated by wolf predation. Even national hunting magazines have recently echoed this rhetoric. Despite evidence that wolves only effect prey populations (not extirpate them) (Mech and Peterson 2003), wolf opponents will attribute all declines in ungulate populations or hunter harvest to wolf predation; wolf advocates will just as adamantly deny any declines in ungulate populations as being wolf-caused.

To address public concerns and biological uncertainty, cooperative research on wolf-ungulate interactions has been continuously initiated and funded by USFWS and other agencies since the 1980s. These multiyear studies reported everything, from wolves having a significant effect to having little measurable effect. Several researchers suggested that ungulate managers should anticipate some population declines, reduced hunting of females and more intensive monitoring to detect changes early (U. S. Fish and Wildlife Service 1994, Kunkel 1997, Smith et al. 2004). The overall effect of wolf predation on state-wide ungulate populations and subsequent hunter harvest in the NRM appears low, as predicted. However some herds, especially where wolf densities are highest and pack territories are most contiguous, are declining and wolf predation is certainly a factor. As a consequence, hunting opportunity for antlerless elk has been reduced for a few herds.

The full effect of wolf predation on ungulate populations has been very difficult to reveal despite the intensive research conducted to date. This is by far the most difficult issue to clarify and resolve because ungulate populations are often affected by such a wide variety of factors that it is nearly impossible to expose the exact role of wolf predation in changes to some ungulate herds. Ungulates are also affected by a multitude of other variables including: predators (most areas with wolves also have coyotes [*Canis latrans*], mountain lions [*Puma concolor*], brown bears [*Ursus arctos*], black bears [*U. americanus*], domestic dogs [*Canis familiaris*], and humans), summer drought, severe winters, access to high-quality agricultural forage, habitat changes brought about by rapidly increasing human development and fires or lack thereof, vehicle collisions and diseases. Mitigating the wolf's role in these circumstances may require that ESA protections be removed. State and tribal wildlife agencies already manage wolf prey. They have a much closer relationship with the public and can be more responsive to the concerns of local residents than the federal program. The states and tribes can also incorporate regulated public harvest into the array of wolf management techniques. We believe that local, state or tribal management, in combination with hunter participation in wolf management, will increase tolerance for wolf conservation just as it has for mountain lions and black bears.

Symbolism of Wolves

Perhaps the most interesting aspect and significant effect of wolves on people are the strong emotions that wolves illicit (Fritts et al. 2003). Humans have used wolves as very powerful symbols in many cultures for thousands of years (Lopez 1978, Boitani 1995). Today, wolves have little effect on most people, but many identify with their symbolism. Wolves can negatively impact some people by causing them to fear for their safety or to perceive that outsiders are telling them what they can do. Wolves can make some people's lives worse by killing livestock, pets or the elk that they or their client could have gotten.

Conversely, some people enjoy the opportunity to hunt, trap, view, hear and photograph wolves. Some people enjoy the knowledge that wolves are "out there" or that wolves enhance the ecological integrity and wildlife diversity of wildlands (Smith et al. 2003). Economic analysis predicted that wolf restoration, primarily associated with tourism in YNP, would generate up to $23,000,000 in economic activity in the NRM annually (U. S. Fish and Wildlife Service 1994). While specific follow-up studies aren't complete, the predictions appear accurate (J. Duffield, personal communication 2003). Most wolves are still generally wary of people, but high prey density, open habitat, and a highway in the northern portion of YNP provide outstanding wolf viewing opportunities. On June 26, 2002, a park naturalist announced that over 100,000 YNP visitors had seen wolves since 1995 (R. McIntrye, personal communication 2003). Several commercial wildlife viewing tour operators have started since 1995, and wolf viewing is a cornerstone of their businesses.

Conclusion

Wolf extermination deferred to other social objectives. Societal values changed, probably fueled most by urbanization. Society set aside public lands and eventually assigned some of them purposes other than commodity production. State wildlife agencies restored ungulate populations, and ultimately other species were conserved. Some predator control practices, like widespread poisoning, became socially unacceptable. Human values about nature changed, and these conservation-oriented perspectives were reinforced by popular media. Wolves also benefitted from the fact that few modern people actually had first-hand experience of the real challenges associated with living with them (Williams et al. 2002).

Resolution of wolf-caused conflicts will be required to maintain public tolerance (Fritts and Carbyn 1995, Mech 1995, Fritts et al. 2003). A viable wolf population can persist in the NRM because large areas of suitable habitat are secure in public ownership or will remain undeveloped. Conflict with livestock will remain low overall because sheep or other highly vulnerable types of livestock are unlikely to return to their former abundance because of global markets and changing social values about acceptable uses of public land. Agency management will continue to resolve the relatively few localized wildlife-agricultural conflicts. State and tribal wildlife management agencies will continue to implement ungulate conservation programs, ensuring an adequate prey base for wolves. We believe the most important next step in wolf management will be delisting of wolf populations, so they can be managed by the respective state and tribal wildlife agencies. Regulated public harvest will be one of the many tools necessary for wolf conservation in the future. Professional wildlife managers should minimize wolf-caused problems to reduce the likelihood of a backlash of public opinion against wolves. Such a backlash could again result in widespread vigilantism or renewed public calls for government extermination programs (Mech 1995). Given some minimal level of secure habitat, wild prey and human tolerance, viable wolf populations will persist. The changing social values that led to wolf restoration in the NRM, under the ESA, will likely enable additional state and tribal-led wolf recovery efforts elsewhere.

Reference List

Bangs, E. E., S. H. Fritts, D. A. Harms, J. A. Fontaine, M. D. Jimenez, W. G. Brewster, and C. C. Niemeyer. 1995. Control of endangered gray wolves in Montana. In *Ecology and conservation of wolves in a changing world*, eds. L. N. Carbyn, S. H. Fritts, and D. R. Seip, 127–34. Edmonton, Alberta: Canadian Circumpolar Institute.

Bangs, E. E., and S. H. Fritts. 1996. Reintroducing the gray wolf to central Idaho and YellowstoneNational Park. *Wildlife Society Bulletin*. 24:402–13.

Bangs, E. E., S. H. Fritts, J. A. Fontaine, D. W. Smith, K. M. Murphy, C. M. Mack, and C. C. Niemeyer. 1998. Status of gray wolf restoration in Montana, Idaho, and Wyoming. *Wildlife Society Bulletin*. 26:785–98.

Bangs, E., J. Fontaine, M. Jimenez, T. Meier, C. Niememyer, D. Smith, K. Murphy, D. Guernsey, L. Handegard, M. Collinge, R. Krischke, J. Shivik, C. Mack, I. Babcock, V. Asher, and D. Domenici. 2001. Gray wolf restoration in the northwestern United States. *Endangered Species UPDATE* 18:147–52.

Bangs, E., and J. Shivik. 2001. Managing wolf conflict with livestock in the Northwestern United States. *Carnivore Damage Prevention News* 3:2–5

Bath, A. J. 1992. Identification and documentation of public attitudes toward wolf reintroduction in Yellowstone National Park. In *Wolves for Yellowstone?: A Report to the United States Congress, volume IV* *research and analysis*, eds. J. D. Varley, and W. G. Brewster, 2–5, Yellowstone National Park, Wyoming: National Park Service.

Boyce, M. S. 1995. Anticipating consequences of wolves in Yellowstone: Model validation. In *Ecology and conservation of wolves in a changing world*, eds. L. N. Carbyn, S. H. Fritts, and D. R. Seip, 199–210. Edmonton, Alberta: Canadian Circumpolar Institute.

Boitani, L. 1995. Ecological and cultural diversities in the evolution of wolf-human relationships. In *Ecology and conservation of wolves in a changing world*, eds. L. N. Carbyn, S. H. Fritts, and D. R. Seip, 3–12. Edmonton, Alberta: Canadian Circumpolar Institute.

Bradley, E. H. 2004. *Effects of wolf removal on livestock depredations in Montana, Idaho, and Wyoming*. M. S. thesis, University of Montana. Missoula, Montana.

Fischer, H. 1989. Restoring the wolf: Defenders launches a compensation fund. *Defenders*. 64:9, 36.

Fritts, S. H., and L. N. Carbyn. 1995. Population viability, nature reserves, and the outlook for gray wolf conservation in North America. *Restoration Ecology*. 3:26–38.

Fritts, S. H., E. E. Bangs, J. A. Fontaine, M. R. Johnson, M. K. Phillips, E. D. Koch, and J. R. Gunson. 1997. Planning and implementing a reintroduction of wolves to Yellowstone National Park and central Idaho. *Restoration Ecology*. 5:7–27.

Fritts, S. H., R. O. Stephenson, R. D. Hayes, and L. Boitani. 2003. Wolves and Humans. In *Behavior, ecology and conservation in wolves*, eds. L. D. Mech, and L. Boitani, 289–316. Chicago, Illinois: University of Chicago Press.

Gipson, P. S., W. B. Ballard, and R. M. Nowak 1998. Famous North America wolves and the credibility of early wildlife literature. *Wildlife Society Bulletin*. 26:808–16.

Kellert, S. R., M. Black, C. R. Rush, and A. J. Bath. 1996. Human culture and large carnivore conservation in North America. *Conservation Biology*. 10:977–90.

Kunkel, K. E. 1997. *Predation by wolves and other large carnivores in northwestern Montana and southeastern British Columbia*. Ph.D. dissertation. University of Montana, Missoula, Montana.

Linnell, J. D. C., R. Anderson, Z. Andersone, L. Balciauskas, J. C. Blanco, L. Boitani, S. Branderd, U. Breitenmoser, I. Kojola, O. Liberg, J. Loe, H.

Okarma, H.C. Petersen, C. Promberger, H. S, E. J. Solberg, H. Valdmann, and P. Wabakken. 2002. The fear of wolves: A review of wolf attacks on humans. *NINA Oppdragsmelding*. 731:1–65.

Lopez, B. H. 1978. *Of wolves and men.* New York, New York: Charles Scribner's Sons.

McIntyre, R., ed. 1995. *War against the wolf: America's campaign to exterminate the wolf.* Stillwater, Minnesota: Vouaguer Press.

McNay, M. E. 2002. Wolf-human interactions in Alaska and Canada: A review of the case history. *Wildlife Society Bulletin*. 30:831–43.

Mech, L. D. 1970. *The wolf: Ecology and behavior of an endangered species*. Garden City, New York: Doubleday/Natural History Press.

———. 1995. The challenge and opportunity of recovering wolf populations. *Conservation Biology*. 9:1–9.

Mech, L. D., and R. O. Peterson. 2003. Wolf-prey relations. In *Wolves: Behavior, ecology, and conservation*, eds. L. David Mech, and Lugui Boitani, 131–60. Chicago, Illinois: The University of Chicago Press.

National Agriculture Statistics Service. 2001. *Cattle and calf loss in 2000. Sheep and lamb loss in 2000*. Washington, DC: National Agricultural Statistics Service, Agricultural Board, U.S. Department of Agriculture.

Nowak, R. M. 1995. Another look at wolf taxonomy. In *Ecology and conservation of wolves in a changing world*, eds. L. N. Carbyn, S. H. Fritts, and D. R. Seip, 375–97. Edmonton, Alberta: Canadian Circumpolar Institute.

Oakleaf, J. K. 2002. *Wolf-cattle interactions and habitat selection by recolonizing wolves in the northwestern United States*. M. S. thesis, University of Idaho, Moscow, Idaho.

Oakleaf, J. K., C. Mack, and D. L. Murray. 2003. Effects of wolves on livestock calf survival and movements in central Idaho. *Journal of Wildlife Management.* 67(2):299–306

Paul, W. J., and P. S. Gipson. 1994. Wolves. In *Prevention and control of wildlife damage*, 123–29. Lincoln, Nebraska: University of Nebraska Press, Cooperative Extension Service.

Ream, R. R., M. W. Fairchild, D. K. Boyd, and A. J. Blakesley. 1989. First wolf den in western United States in recent history. *Northwest Naturalist.* 70:39–40.

Ritter, D.G. 1991. Rabies in Alaskan furbearers: A review. *Sixth North American Furbearer Conference*. April:10–1.

Smith, D. W., R. O. Peterson, and D. B. Houston. 2003. Yellowstone after wolves. *BioScience*. 53:330–40.

Smith, D. W., T. D. Drummer, K. M. Murphy, D. S. Guernsey, and S. B. Evans. 2004. Winter prey selection and estimation of wolf kill rates in Yellowstone National Park. *Journal of Wildlife Management*. 68:153–66.

U. S. Fish and Wildlife Service. 1987. *Northern Rocky Mountain Wolf Recovery Plan*. Denver, Colorado: U. S. Fish and Wildlife Service.

———. 1988. *Interim wolf control plan: Northern Rocky Mountains of Montana and Wyoming*. Denver, Colorado: U. S. Fish and Wildlife Service.

———. 1994. *The reintroduction of gray wolves to Yellowstone National Park and central Idaho. Final Environmental Impact Statement*. Helena, Montana: U. S. Fish and Wildlife Service.

———. 2003. Endangered and threatened wildlife and plants: Final rule to reclassify and remove the gray wolf from the list of endangered and threatened wildlife in portions of the conterminous United States; establishment of two special regulations for threatened gray wolves. *Federal Register*. 68 (62):15,804–75.

U. S. Fish and Wildlife Service, Nez Perce Tribe, U. S. National Park Service, and U. S. Department of Agriculture, Wildlife Services. 2004. *Rocky Mountain wolf recovery 2003 annual report*. Helena, Montana: U. S. Fish and Wildlife Service, Nez Perce Tribe, U. S. National Park Service, and U. S. Department of Agriculture, Wildlife Services.

Williams, C. K., G. Ericsson, and T. A. Heberlein. 2002. A quantitative summary of attitudes toward wolves and their reintroduction (1972–2000). *Wildlife Society Bulletin*. 30:575–84.

Young, S. P. 1944. History, life habits, economic status, and control. In *The wolves of North America*. ed. S. P. Young, and E. A. Goldman, 1–385. Washington DC: American Wildlife Institute.

Management of Habituated Grizzly Bears in North America: Report from a Workshop

Kerry A. Gunther
Yellowstone National Park
Yellowstone National Park, Wyoming

Kathy Tonnessen
U. S. National Park Service, Intermountatin Program
Missoula, Montana

Peter Dratch
U. S. National Park Service, Biological Resource Management Division
Fort Collins, Colorado

Chris Servheen
U. S. Fish and Wildlife Service
Missoula, Montana

Introduction

Throughout the early history of the U. S. National Park Service (NPS), many parks with grizzly bear (*Ursus arctos*) habitat experienced significant problems with bear-human conflicts, including bear-inflicted human injuries and bear-caused property damage (Herrero 1985, Gunther 1994, Gniadek and Kendall 1998, Herrero and Higgins 2003). Almost all bears involved in conflicts with people were conditioned to human foods or garbage (Herrero 1970, 1985). Bears involved in conflicts with people were usually removed from the wild in management actions.

From the 1960s to the 1990s, most parks experiencing bear-human conflicts with human food conditioned bears implemented food and garbage storage regulations and sanitation practices that mostly eliminated these types of problems (Gunther 1994, Gniadek and Kendall 1998, Herrero and Higgins 2003). Beginning in the 1990s, some national parks began to experience a new type of bear behavior that raised new concerns for visitor safety. Bears that were habituated to people but were not conditioned to human foods began frequenting roadside corridors foraging for native foods, often with hundreds of park visitors watching and photographing them within 20 to 50 meters (Gunther 1994, Gunther and Biel 1999).

Habituation without food conditioning is not necessarily detrimental to bears or to people (Herrero et al., in press). Habituation enables bears to access high quality habitat adjacent to roads, habitat that is underutilized by bears that are wary of humans (Gunther and Biel 1999, Herrero et al., in press). In addition, habituated bears may actually be less prone to aggression toward people during surprise encounters (Jope 1985), and they also provide excellent bear viewing and educational opportunities (Herrero et al., in press). In areas where human activity is strictly controlled, such as at McNeil River State Game Sanctuary in Alaska, public viewing of habituated bears has been very popular, and management has been very successful at preventing bear-inflicted human injuries (Aumiller and Matt 1994).

Park visitors are less strictly controlled in national parks with road access. Although habituated bears typically exhibit very predictable behavior, park visitors in less controlled situations do not. The unpredictable nature of park visitors in areas with road access increases the potential for habituated bears to conflict with people, and this has lead to different management strategies to promote human safety.

In Yellowstone National Park (YNP), the frequency of habituated grizzly bears using habitat along roadside corridors has been increasing and is now the biggest bear management challenge in the park (Gunther 1994, Gunther and Biel 1999). When habituated bears are present along roadsides during daylight hours, hundreds of visitors may stop along the road to view and photograph the bears; these incidents are referred to as bear-jams. Management of habituated grizzly bears along roadsides in YNP has evolved for over a decade under an informal adaptive management strategy. During the 1980s, YNP discouraged habituation and emphasized modifying bear behavior through hazing, aversive conditioning, capture and translocation. Under this management strategy there were few bear-human conflicts and bear-inflicted human injuries. However, these efforts were unpopular with park visitors that wanted to see bears, and they were only marginally successful at changing bear behavior. Beginning in the 1990s, YNP began tolerating habituated bears and shifted management emphasis to managing park visitors at bear-jams. YNP currently dispatches patrol rangers to monitor visitor behavior at bear-jams and to prevent visitors from feeding habituated bears or approaching them. This approach has been very popular with park visitors that enjoy viewing and photographing bears, but it also requires much more staff time than previous management strategies. Under this strategy, bear-human conflicts and bear-inflicted human injuries have also been very low. In the near future, YNP will be developing a formal management policy to address roadside habituated grizzly bears. A formal policy will help ensure consistent management of habituated bears throughout the park, making human behavior more predictable to bears and making YNP policy clearer to park employees and visitors. Prior to developing a formal plan for YNP, a review of strategies used by other areas to manage habituated grizzly bears was warranted.

The scientific literature, however, contains almost no studies addressing the habituation process or management of habituated grizzly bears and bear viewers. To address this shortfall, we organized a workshop, entitled "Management of Habituated Grizzly Bears," that was held in Missoula, Montana, October 22 to 23, 2003. The workshop was attended by 78 state and federal agency grizzly bear and land managers as well as interested NGOs and the public. The workshop included presentations and discussions on the similarities and differences in management strategies used by different national parks, sanctuaries and preserves to address the bear and visitor safety concerns related to managing habituated grizzly bears, especially those frequenting public lands with high levels of human use. Here, we discuss the ideas and strategies presented at the workshop.

Standardization of Terms

The current scientific literature contains ambiguous terms and definitions for measurements of wildlife responses to humans (Whittaker and Knight 1998, Taylor and Knight 2004). The term habituation has been frequently misused by wildlife biologists and the press (B. Gilbert, personal communication 2003, Whittaker and Knight 1998). The workshop opened with a discussion on standardization of the definitions commonly used in management of habituated bears.

Habituation is a decline in responses, when a stimulus has little or no consequence for an animal (B. Gilbert, personal communication 2003). Many species, including bears, become habituated toward people when not harassed or not killed, as in national parks and refuges (B. Gilbert, personal communication

2003). In the wild, bears can become habituated to other bears, other wildlife, people, vehicles, noise and other stimuli. Habituation of bears to other bears and to people happens because benefits to individual bears exceed perceived risks (S. Herrero, personal communication 2003). Thus habituation is not unnatural and is not considered taming or domestication (B. Gilbert, personal communication 2003). Habituation allows animals to ignore humans while going about obtaining their daily needs.

Food conditioning is the shaping of an animal's behavior by positive reinforcement (food reward) and can lead to a bear's attraction to humans or human developments (Herrero 1985; B. Gilbert, personal communication 2003). Bear-caused property damage and bear-inflicted human injuries are often associated with bears conditioned to human food or garbage (Herrero 1970, 1985).

Grizzly bears can be habituated to people, conditioned to human food or both (Herrero 1985). Allowing bears to become conditioned to human foods or garbage is almost universally accepted as negative for both bears and people, and preventing bears from becoming conditioned to anthropogenic foods is the foundation of most bear management programs in North America. In contrast, habituation may be appropriate in some situations but not desirable in others (S. Herrero, personnel communication 2003; C. Matt, personal communication 2003). To facilitate science-based management of habituated bears, managers should clearly distinguish between habituation and food conditioning when setting management objectives.

Aversive conditioning is a learning process in which the bear changes behavior following one or more stressful or painful stimuli (B. Gilbert, personal communication 2003). Wildlife management agencies commonly use rubber bullets, cracker shells or dogs as aversive conditioning agents to dehabituate bears. However, most agencies are not measuring the long-term success of their aversive conditioning programs.

Managing Habituated Grizzly Bears

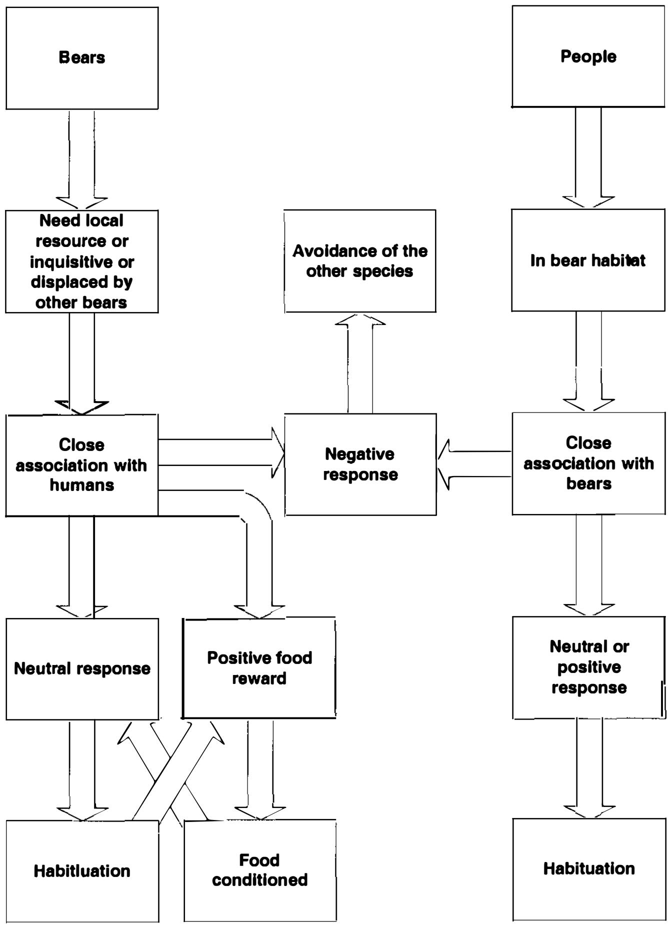
Managing bears that become habituated to people is a challenge to many national parks, preserves and sanctuaries with low levels of human-caused bear mortality. Presentations from land and wildlife managers of habituated grizzly bears indicated that, on a broad scale, there are two general models or strategies currently being used to manage habituated grizzly bears in North America (Table 1). Under one model, habituation is encouraged (such as in McNeil State Game Sanctuary) or tolerated (such as in Katmai, Yellowstone and Grand Teton national parks) to promote bear viewing and to increase habitat effectiveness while maintaining a low rate of bear-inflicted injuries and human-caused bear mortalities. Under the second model, habituation is actively discouraged (such as in Banff, Yoho, Kootenay and Glacier national parks) to minimize human-caused bear mortality and to maximize human safety. The particular management model chosen by different land and wildlife agencies is influenced by many factors, including budgets, types of habitat, bear density, human visitation, bear population objectives, sources of bear mortality and visitor needs. In threatened or endangered bear populations, if habituation increases the risks of human-caused bear mortality, it should be discouraged (Herrero et al., in press). In many other contexts habituation of bears to people has more benefits than risks (Herrero et al. in press). A general model of the process of bear habituation to people, how it can lead to food conditioning, and points where managers may be able to dehabituate bears is displayed in Figure 1.

Table 1. Two general models currently used for managing habituated grizzly bears in North America.

	Model #1	Model #2
Land Management Area	McNeil River State Game Sanctuary[a], Katmai[b], Yellowstone, and Grand Teton[c] National Parks	Banff[d], Yoho[d], Kootenay[d], and Glacier National Parks[e]
Management Objective	Allow habituation	Discourage habituation
Management Goals	Improve habitat effectiveness, promote bear viewing, acceptable level of human-caused bear mortality	Reduce human-caused bear mortality (road-kill, management removal, poaching)
Management Strategy	Emphasis on human control	Emphasis on bear control
Management Cost	High	Low
Human Safety	High (low rate of bear-inflicted human injuries)	High (low rate of bear-inflicted human injuries)
Visitor Satisfaction	Higher due to more bear viewing opportunities[f]	Lower due to fewer bear viewing opportunities[f]

[a]Larry Aumiller, Alaska Department of Fish and Game, personal communication 2003
[b]Tamara Olson, Katmai National Park and Preserve, personal communication 2003
[c]Steve Cain, Grand Teton National Park, personal communication 2003
[d]Mike Gibeau, Parks Canada, Banff National Park, personal communication 2003
[e]John Waller, Glacier National Park, personal communication 2003.
[f]This assumption needs validation through visitor attitude surveys/analysis.

Figure 1. A model of human and bear habituation. Gray lines are times in the habituation process where intervention is possible.



Characteristics of land management areas where grizzly bear habituation to human activity has been acceptable to wildlife managers include:

- areas with high bear density and high human control, such as McNeil River State Game Sanctuary and Katmai National Park
- areas with a moderate bear density and a moderate human control, such as Yellowstone and Grand Teton national parks
- areas with low bear density and low human control, such as the Montana Rocky Mountain Front, where habituated behavior is tolerated during nocturnal time periods, so bears can use quality habitats close to people.

Characteristics of land management areas where habituation is not tolerated and habituated bears are dehabituated include:

- areas with high bear density and low human control, such as Banff, Yoho, Kootenay and Glacier national parks
- areas with low human tolerance for bears, such as some private lands outside of national parks
- areas with low bear density and high bear vulnerability, such as hunted populations or areas close to human settlements.

Measures of Success

In areas where managers promote or tolerate habituation and bear viewing, the success or failure of programs should be measured, so managers can better understand the ramifications of their policies. Potential measures of success in management of habituated bears include:

- few bear or human mortalities
- few bear-human conflicts
- few bear-inflicted human injuries
- few bears killed on highways
- few food-conditioned bears
- correct message to the public
- high quality experience for visitors
- high quality habitat available to bears
- contentment of distance from bears for visitors
- lower costs due to lower management efforts
- approved behaviors exhibited by bears
- realized interpretations of regulations and laws
- cooccupation of important habitats of bears and people.

Research Needs

Managers of grizzly bears and their habitat need to develop science-based strategies for managing habituated bears. However, there is a paucity of information and studies in the scientific literature on the processes of habituation, the management of habituated bears and the benefits and costs of habituation to both bears and people. To address this shortfall, the following research needs were identified at the workshop.

- How long does habituation continue without further neutral response? Is it fairly permanent once established in individual bears?
- Can habituation to human activity be changed by negative response, and, if so, how much is needed and how long does it last?
- Physiological measures and the possible hidden costs of habituation such as heart rate and stress issues, should be measured; implant transmitters and glucocorticoid measurements may be useful in addressing these issues.
- The relationship between habituation and the impacts of roads and trails should be studied; the main impacts of forest roads and trails are displacement and mortality risk; habituation may increase mortality risks and may decrease displacement.
- How does habituation relate to NPS versus U.S. Forest Service (USFS) access management?
- Is it possible to dehabituate individual bears to certain sites and to habituate them to others? Can we have them respond where we want them to respond?
- If we haze bears from roads and trails, does it further fragment populations and habitats?
- What are the best ways to improve our knowledge of and to address human habituation to bears?
- What are the mortality costs of habituation, and how can careful survivorship comparisons between habituated versus nonhabituated bears be provided?
- An analysis of areas with high habituation (i.e. viewing sites, the source) with areas outside where habituation is not beneficial (the sink) is necessary.
- Based on the source-sink analyses, how much of special management areas surrounding habituation locations is required to avoid detrimental impacts of habituation? Should this be based on home ranges?
- Global positioning system (GPS) collars can be used to document habitat use changes as bears become more habituated. How is this a cost or a benefit for these animals?
- The temporal aspects of habituation should be explored, including what are the diurnal and nocturnal differences in expressions of habituation and can GPS collars and hourly duty cycles monitor nighttime

movements of habituated and nonhabituated bears in relation to human presence?

Fundamentals of Design for Studies of Habituation

There is often disparity in methodologies used to study wildlife responses to humans (Taylor and Knight 2004). To address these issues in regards to studies of habituated bears, the following needs were identified:

- standardization of the use of the phrases and terms: food conditioned, habituation, tolerance, aversive conditioning, hazing and other terms describing habituated bears and their management
- use of marked animals to monitor long-term responses
- use of GPS collars to monitor nocturnal responses to human activity
- careful definitions of success of dehabituation because it contributes to long-term definitions of success
- controlled variables
- multiple replicates of treatments.

Park Service Policy Perspectives on Habituation of Grizzly Bears

There are NPS policies at several levels to address the habituation of grizzly bears on NPS lands. They include the Organic Act of 1916 (Organic Act), management policies of 2001, and park unit policies and plans.

The Organic Act, the formative legislation of the NPS states: "the fundamental purpose of the said parks, monuments and reservations is to conserve the scenery and the natural and historical objects and the wild life therein." The two words, wild life, referred to in the Organic Act have a broader meaning than we commonly connote for wildlife, a meaning that derives from the development of the discipline of wildlife management which postdates the Organic Act of 1916 by more than a decade.

Wild life in this context includes both plants and animals, but it does not include organisms that have been domesticated. While domesticated animals in general would not be covered under the act, habituated individuals of wild species might be.

From their first legislation, national parks have different responsibilities than other federal land management agencies, and these differences could effect how we manage grizzly bears and brown bears. In general, national parks are wildlife sanctuaries, subject to different regulation than the surrounding land. The default natural condition of wildlife in national parks is the condition that predates European settlement of the land. Since behavior is highly adaptive, methods of determining what constitutes natural behavior, particularly in a legal framework, are not straightforward.

The Organic Act does make clear that our obligation to wild places and species extends beyond immediate management prerogatives. We are to provide for the enjoyment of the scenery and the wild life in such manner and by such means as to leave them unimpaired by future generations. This could be interpreted to mean that we are to insure that the wildlife in national parks remains wild, in its ecology, physiology and behavior.

The management policies of 2001 leave determinations of impairment largely to individual parks, but they do not permit actions that harm the integrity of park resources or values. To apply these policies to this case, park managers must answer the following question: does allowing bears to come in close proximity to park visitors affect the integrity of that resource? To put it another way, does the reduced flight distance that is occurring between habituated bears and humans constitute an impairment of natural bear behavior.

From information provided by parks before the workshop and presented here, it is clear that parks do not have the same view or the same policy. Response to habituated brown bears and grizzlies in parks varies from encouragement at established viewing areas to harassment by hazing and removal, from closure of areas frequented by bears to tolerance of bear-jams to emphasis on human education and safety.

Such disparate management actions toward the same species could subject the NPS to criticism. Although parks attributed most of their recent grizzly bear attacks to food conditioning or surprise, most also responded that habituation was a potential cause of some attacks. Information gained at the workshop does, however, give us some basis for different management actions towards grizzly bears in different habitats.

When food resources are superabundant, as during the salmon runs at Katmai and McNeil rivers, bears lose their intolerance of conspecifics and of humans (Smith et al. in press). In other areas, such as YNP where there is a high density of people but a low level of human-caused grizzly bear mortality, bears will also habituate to human activity. Bear behavior where resources are much more widely distributed and where densities of people are much lower, such as interior Alaska, will be similar to what we have previously considered the norm, that is, bears wary of people but sometimes aggressive to them (T. Smith, personal communication 2003). The public expectations of the NPS on this issue are threefold:

- to prevent bears from attacking people
- to prevent people from impacting bears
- to provide for public enjoyment of parks, which includes watching bears.

To achieve a policy that is superficially consistent with these three expectations, we would have to harass or remove brown bears on the Alaska coast and grizzly bears in YNP. To have a policy that is consistent with the diversity of bear behavior and does not put the public at risk will require a sophisticated education effort. We must explain why different bears in different parks have different responses to humans and, thus, warrant different management strategies.

Recommendations

People are successfully coexisting with unhunted populations of grizzly bears in national parks, sanctuaries and preserves (S. Herrero, personnel communication 2003). This is being done with acceptable safety for people and for bears, but maintaining such safety standards requires active management of people and bears and the budget to consistently do this (Herrero et al. in press). Innovative strategies for managing people and habituated bears need to be developed to reduce the potential for bear-human conflicts with, and human-caused mortality of, habituated grizzly bears that frequent public land with high levels of human use. In some areas, promotion of habituation and bear viewing may be appropriate if it can be done with few bear-inflicted human injuries and human-caused bear mortalities. In other areas, preventing or discouraging habituation and bear viewing may be the most appropriate strategy for the safety of both bears and people. Management strategies should be based on bear population objectives, bear habitat requirements, public needs and human and bear safety considerations. Regardless of the strategy chosen to manage habituated bears, managers should measure the success or failure of their programs, so they can better understand the ramifications of their management strategies.

Reference List

Aumiller, L. D., and C. A. Matt. 1994. Management of McNeil River State Game Sanctuary for viewing of brown bears. *International Conference on Bear Research and Management*. 9:51–61.

Jope, K. L. 1985. Implications of grizzly bear habituation to hikers. *Wildlife Society Bulletin*. 13:32–7.

Gniadek, S. J., and K. C. Kendall. 1998. A summary of bear management in Glacier National Park, 1960–1994. *Ursus*. 10:155–9.

Gunther, K. A. 1994. Bear management in Yellowstone National Park, 1960–93. *International Conference on Bear Research and Management*. (1):549–60.

Gunther, K. A., and M.J. Biel. 1999. Reducing human-caused black and grizzly bear mortality along roadside corridors in Yellowstone National Park. In *Proceedings of the third international conference on wildlife ecology and transportation*, *FL-ER-73-99*, eds. G. L. Evink, P. Garrett, and D. Zeigler, 25–7. Tallahassee, Florida: Florida Department of Transportation.

Herrero, S. 1970. Human injury inflicted by grizzly bears. *Science*. 170:593–8.

———. 1985. *Bear attack: Their causes and avoidance*. New York, New York: Nick Lyons Books.

Herrero, S., and A. Higgins. 2003. Human injuries inflicted by bears in Alberta: 1960–98. *Ursus*. 14(1):44–54.

Herrero, S., T. Smith, T. D. DeBruyn, and K.A. Gunther. in press. Brown bear habituation to people: Safety, risks, and benefits. *Wildlife Society Bulletin*.

Smith, T. S., S. Herrero, and T. D. DeBruyn. in press. Alaskan brown bears, habituation, and humans. *Ursus*.

Taylor, A. R., and R. L. Knight. 2004. Behavioral responses of wildlife to human activity: Terminology and methods. *Wildlife Society Bulletin*. 31(4):1,263–71.

Whittaker, D., and R.L. Knight. 1998. Understanding wildlife responses to humans. *Wildlife Society Bulletin*. 26(2):312–7.

Collaborative Models for Wildlife Conservation in the Northern Rockies

Sterling D. Miller
National Wildlife Federation
Missoula, Montana

Nina Fascione
Defenders of Wildlife
Washington, DC

Minette Johnson
Defenders of Wildlife
Missoula, Montana

Tom France
National Wildlife Federation
Missoula, Montana

Suzanne Stone
Defenders of Wildlife
Boise, Idaho

Management of large mammalian predators requires more than the technical skills of biologists in state and federal wildlife management agencies. Nongovernmental organizations (NGOs) interested in the conservation and restoration of these species are both potentially important allies in these efforts as well as potentially crippling adversaries to agency efforts. The National Wildlife Federation (NWF) and Defenders of Wildlife (Defenders) are two national conservation groups that generally have been able to work cooperatively with state and federal agencies as well as with private landowners on wolf and grizzly bear restoration efforts in the northern Rocky Mountains. This cooperation over the past few decades illustrates a route where collaboration rather than confrontation with agencies and individuals can produce tangible benefits for wildlife conservation and for wildlife managers. In this paper, we provide some examples of programs where our two NGOs have worked cooperatively with state, tribal and federal agencies and the general public to generate such benefits.

Grizzly Bear Restoration in the Selway-Bitterroots

The NWF and Defenders worked cooperatively with representatives of the timber industry and other stakeholders in Idaho to develop a plan for grizzly bear *(Ursus arctos*) reintroduction in 3.7 million acres (1.5 ha) of designated wilderness in central Idaho. Development of this plan was coordinated with the Grizzly Bear Recovery Coordinator of the U. S. Fish and Wildlife Service (USFWS) and the state, provincial and federal agencies that form the Interagency Grizzly Bear Committee (IGBC). The USFWS adopted the plan negotiated by the conservation groups and the timber industry representatives (Intermountain Forest Association, Resource Organization on Timber Supply) as their preferred alternative in the environmental impact statement (U. S. Fish and Wildlife Service 2000a). This led to a published record of decision (ROD) and regulations in the Federal Register in November 2000 to reintroduce grizzly bears in this area under the terms negotiated by the conservation groups and the timber industry representatives (Schoen and Miller 2002, U. S. Fish and Wildlife Service 2000b).

To date, the tangible benefits from this cooperative effort have not included actual reintroduction of grizzly bears. This is because the current Secretary of the Department of Interior (DOI) has so far refused to implement the 2000 ROD. Nevertheless, this ROD remains in place, pending the election of an administration willing to implement the product of this collaborative effort to recover grizzly bears.

Retaining the 2000 ROD remains a significant accomplishment. When DOI Secretary Gale Norton and other federal officials took office in 2001, negotiations were initiated between the DOI and the Governor of Idaho (who opposed reintroduction). The product of these negotiations was a notice of intent to replace the ROD published in 2000 with a "no action" alternative. The NWF and Defenders strongly opposed this proposal, along with many members of the public and the scientific community. Altogether, 98 percent of the comments received from the public and 100 percent of the comments received from scientific or professional organizations opposed DOI's proposal to abandon the reintroduction program (U. S. Fish and Wildlife Service 2001).

Compensation

Wolves

After an absence of 60 years, wolves (*Canis lupis*) began returning to the northern Rockies in the 1980s. In 1987, a wolf killed several sheep in Montana. This was the first wolf-caused livestock mortality in the region in decades. There was a tremendous antiwolf furor when the news spread about the dead sheep, particularly among the agricultural community. In response, Defenders raised the money within 48 hours to compensate the ranchers for their losses. Recognizing the value and necessity of such compensation to build tolerance for wolves, Defenders worked with The Bailey Wildlife Foundation to establish the first privately funded livestock compensation program of its kind to provide reimbursement for wolf-caused losses. The Bailey Wildlife Foundation Wolf Compensation Trust, named in honor of its largest contributor, has since reimbursed ranchers for verified livestock losses to wolves in the northern Rocky Mountains and southwest United States. Since 1987, Defenders paid $328,930 for 377 cattle, 847 sheep, 48 other types of livestock. Over $200,000 was paid in the northern Rocky Mountains alone and the rest in the southwest United States and Canada. Livestock covered by the program include cattle, sheep, goats, llamas, mules, horses, and guarding and herding dogs.

We believe that the compensation program has been instrumental in ensuring wolf survival by building tolerance within local communities. Preliminary findings indicate nearly all livestock owners in the northern Rocky Mountains seek and receive compensation for their wolf-caused livestock losses. From 2000 to 2002, USFWS reported 124 cattle, 317 sheep, and 9 horses and llamas were killed or injured by wolves in northern Rocky Mountains. During this same time period, Defenders paid livestock owners for 151 cattle, 419 sheep, and 8 horses and llamas; (the differences in numbers are attributable to postinvestigative losses and subsequent mortalities from injuries).

Grizzly Bears

After USFWS protected grizzly bears as threatened species under the Endangered Species Act (ESA), bear populations slowly began to recover and broaden their range. That expansion resulted in increased contact with humans and livestock. In the early 1980s, another NGO—the Great Bear Foundation—began paying compensation for bear kills of livestock around Glacier National

Park. Unfortunately, they ran out of funds, leaving an announced compensation program with no money to address claims. Based on their track record with the wolf compensation fund, bear managers asked Defenders to take over the grizzly bear depredation compensation program, and this was done in 1997. Defenders expanded the geographic scope of the fund to include Glacier National Park and the surrounding national forests and tribal lands.

In 1999, Defenders further broadened the program to include the Montana and Idaho portions of the Greater Yellowstone Ecosystem (GYE) (Wyoming has its own compensation program). In 2003, the tribal lands of the Wind River Indian Reservation, southeast of Yellowstone National Park, were included because the state did not pay for losses there. During 1997 to 2004, in Idaho and Montana, Defenders has paid claims totaling $107,560 for 138 cattle, 151 sheep and 151 chicken, geese, turkeys and other livestock.

The success of the wolf and grizzly bear compensation programs are based on a simple structure. Defenders relies on tribal or state biologists or U. S. Department of Agriculture Wildlife Services agents to investigate the cause of livestock mortalities when predators are suspected. The official investigating the depredation uses evidence at the scene as well as recent radio-locations of radio-collared predators to determine the cause of death. If agency officials verify that a wolf or grizzly bear killed the livestock, they fill out a report and give it to the livestock owner with instruction to send it to Defenders for compensation. Defenders works with the livestock owner to ascertain the value of the livestock and compensate them within several weeks of receiving the incident report.

Defenders pays for the full, fall market value of the livestock and not the value of the animal at the time the depredation occurred. For example, if a young calf is killed in the spring, the compensation payment is determined by the market value based on the average weight and price of calves sold in the fall at auction or through prior sales. The maximum payment per animal is $2,000. Defenders does not compensate for livestock protected through insurance or a governmental compensation program. It also compensates for grizzly or wolf-killed livestock guarding or herding dogs but does not cover pets or property damage.

Over the years, Defenders has modified the compensation program based on conversations with ranchers. Defenders now pays 50 percent of the market value for livestock that was not verified killed by a grizzly bear or wolf but, based on available evidence, was probably killed by one. For example, if a cow was verified killed and her calf was missing, Defenders would pay 100 percent for the cow and 50 percent for the calf. While we believe the compensation funds have built tolerance for grizzly bear and wolf recovery and reduced the chances of people illegally killing wolves and bears, evidence to support this is difficult to obtain. Compensation can occur only after the damage occurs and ideally is accompanied by the proactive projects described below to keep depredations to a minimum.

Prevention of Adverse Interactions between Humans and Bears and Wolves

Grazing Allotment Retirements in Areas with Conflicts between Wildlife and Livestock

Since 2000, the NWF has been engaged in efforts to retire livestock grazing allotments on national forest lands in the GYE, where conflicts with humans and livestock inhibit recovery of wolves, bears or other wildlife. Many other organizations and individuals have been financial contributors to these efforts, including Defenders, the Foundation for North American Wildlife Sheep, the Greater Yellowstone Coalition, the Sierra Club, the Rocky Mountain Elk Foundation and Predator Conservation Alliance. These allotment retirements have primarily targeted grazing allotments where conflicts between sheep or cattle and wolves, grizzly bears or bison are high, but virtually every other species of wildlife in the GYE has benefited as well. These grazing allotments have been retired by willing sellers selling their grazing privileges and moving their livestock elsewhere, thereby permanently solving their problems, saving the lives of many wolves and grizzly bears and, effectively, expanding the size of the secure area for wildlife adjacent to the national parks. The state management plans for grizzlies developed for implementation, should the GYE's grizzly bears be removed from the list of protected species, call for bears to be allowed to expand into habitats that are biologically and socially acceptable. Removal of livestock from these allotments makes it socially acceptable for predators to live there.

The Blackrock/Spread Creek allotment is a key wildlife corridor between Yellowstone National Forest and the Grand Teton National Park on the Bridger-Teton National Forest that, in 2003, was retired from grazing for $250,000. This allotment includes 87,500 acres (35,400 ha) of diverse habitat types from the Snake River to Togwotee Pass. Conflicts between wolves and bears expanding their ranges out of Yellowstone National Park created difficult conditions for the Walton family, who held the allotment for more than 40 years. Between 1992 and 1998, 108 cattle were known to have been killed or injured by grizzly bears and $158,000 in compensation was paid by Wyoming. As many as 25 different bears were documented to use the area, 86 percent of which was designated as being of high value (management situation 1 or 2) in Interagency Grizzly Bear Management Guidelines. In addition, the Grand Teton National Park's wolf pack was resident in the area, creating problems for livestock. There were also conflicts as 300 bison periodically moved from Yellowstone National Park into the area. Some of these bison had brucellosis, and livestock producers were concerned about the potential to transmit this disease from bison to cattle.

Conflicts between bison and cattle based on concerns about brucellosis were the primary motives for the purchase of the allotment on the Horse Butte Peninsula in the Gallatin National Forest in 2003. Bison moving out of Yellowstone National Park into this area were hazed and killed for more than a decade, creating a seemingly endless source of conflict between wildlife advocates, lessees and agency staff. The NWF and the U. S. Forest Service made an agreement with the Horse Butte grazing permit holders to move to a new allotment on the nearby Targhee National Forest where there were no similar conflicts with bison. This settlement resolved a long-standing legal action between grazing permittees, the U. S. Forest Service and several conservation groups. In addition to benefits to bison, the area provides key habitat for an expanding population of grizzly bears as well as bald eagles (*Haliaeetus leucocephalus*), elk (*Cervus elaphus*), waterfowl, sandhill cranes (*Grus canadensis*) and furbearers.

Three sheep grazing allotments were retired in the Island Park area (Canyon/Taylor Creek, Snyder Creek and West Lake) totaling about 12,000 acres (4,860 ha). These allotments are on the southern slope of the Centennial Mountains on the Montana-Idaho border, about 20 miles west of Yellowstone National Park. This area has a history of grizzly bear-sheep conflicts and is in a key area for wildlife movements between Yellowstone National Park and the large wilderness areas to the west where grizzly bears were exterminated more than 50 years ago. Conflicts between sheep and grizzly bears in this area are inhibiting natural movements to the West that may ultimately permit grizzly bears to recolonize the wilderness areas in central Idaho.

The Moose Creek allotment is on the western border of Grand Teton National Park on the Caribou-Targhee National Forest. Much of this sheep grazing allotment is in the Jedediah Smith Wilderness Area. In key habitat for bighorn sheep, moose, mule deer, elk and other wildlife, this allotment is in the path for expanding populations of grizzly bears and wolves from Yellowstone National Park. Retirement of the allotment created 22,500 acres (9,106 ha) of conflict-free habitat for wildlife.

Mortality Reduction Efforts

Increasing populations of grizzly bears create a greater likelihood of conflict between bears and humans and often results in bear mortality. This situation provides an excellent opportunity for conservation groups to collaborate with federal, state and tribal officials and with private landowners to implement programs on the ground to prevent conflicts and reduce bear mortalities. Both Defenders and NWF have been involved in such programs.

To reduce the number of wolves and grizzly bears being legally and illegally killed by humans, Defenders uses the Bailey Wildlife Foundation Proactive Carnivore Conservation Fund, created in 1998, to share costs with private landowners and others to prevent livestock depredations and other conflicts. Human-caused mortality is a major concern for bear recovery. In the GYE, of known grizzly bear mortalities between 1980 and 2002, more than half (56%) of the human-caused mortalities were a result of conflicts at sites where bears were killed when attracted to garbage or other attractants. The second and third most common causes of mortality were illegal killings (n = 37) and livestock depredation (n = 11). Fifty-two percent of the nonhunting, human-caused mortalities between 1980 and 2002 in the Northern Continental Divide Ecosystem (NCDE) were human-site conflicts (55), illegal killing (48) and livestock depredations (22).

During 1998 to 2004, Defenders has cost-shared on 96 proactive projects throughout the northern Rocky Mountains to increase tolerance for grizzly bears and wolves and to reduce the chances of these predators being relocated or removed from the ecosystem. The amount invested by Defenders in these projects is $343,606, and similar projects are ongoing. These projects include 5 sheep fences ($10,825), 2 calving ground fences ($11,927), 40 beeyard fences ($17,875), use of bear dogs for aversive conditioning ($22,000), purchase of 164 bear-resistant dumpsters and containers ($57,723) and 5 allotment retirements ($35,000 contributed). In addition, Defenders has created and distributed educational materials to provide guidance to residents on simple steps they can take to reduce their chances of having problems with bears. Reducing human-caused mortality and building local acceptance of grizzly bears are key to making progress on grizzly bear recovery and to increasing occupation of currently vacant habitats. Conservation groups can foster grizzly bear recovery by building partnerships with agencies, landowners and other groups to prevent bear-human conflicts before they occur.

In a collaborative process with state and federal agencies, the NWF and Defenders have been the primary private contributors to a program that placed $120,000 worth of wildlife-resistant dumpsters and associated educational materials in the Bitterroot Ecosystem during 2002 to 2004. Work in 2004 included the provision of bear-resistant panniers and related equipment to outfitters affiliated with the Idaho Guides and Outfitters Association to demonstrate the utility of the equipment to their colleagues. The NWF is coordinating this work and funding it with matching grants received from the National Forest Foundation. Sanitation of the Bitterroot Ecosystem is an identified task in the recovery plan for grizzly bears in this area, but, because our active reintroduction efforts are currently stalled (discussed above) progress is ongoing on the sanitation component of the long-term recovery effort. Identification of the highest priority sites requiring sanitation is being done cooperatively with state agencies in Idaho and Montana as well as the U. S. Forest Service.

Rewards

Another approach to reduce human-caused bear mortality is to offer rewards for information leading to an arrest when wolves or grizzly bears are illegally killed. Defenders has offered a total of $15,500 for six grizzly bear poaching incidents and $139,500 for 45 illegal killings of wolves since 1997. Unfortunately, USFWS law enforcement agents have yet to catch anyone for these crimes.

Science Projects

Both NWF and Defenders have been involved in cooperative projects designed to ensure that the best scientific information is available to agencies. These projects range from sponsored workshops and conferences to actual collection of field data.

Adopt-a-Lek

Although much of our work has focused on recovery of carnivores, NWF's Adopt-a-Lek Program is a program designed to improve the amount of information available for state agencies to manage a declining species of grouse. This program is designed to collect field data on greater sage grouse (*Centrocercus urophasianus*) populations and habitats. These data are time-consuming and expensive to obtain by agency staff. During 2000 to 2004, the Adopt-a-Lek Program recruited and trained volunteers to survey known breeding leks and to search for new leks in Montana, Wyoming and Nevada; additional states may be added in the future. Data obtained are provided to state wildlife management agencies. During 2004, 93 volunteers contributed 2,465 hours of their time and drove 34,524 miles (55,560 km) to count more than 150 active and historic leks.

These surveys provide critical data needed by managers to evaluate trends in sage grouse numbers and also pinpoint sites where habitat improvements and private landowner incentives can most effectively be applied to address conservation needs. This project receives funding and cooperation from the National Fish and Wildlife Foundation and other foundations, individuals, businesses and government agencies.

Conferences and Workshops

Since 1996, Defenders has sponsored the biennial Carnivore Conference at various locations throughout the United States. This conference provides an opportunity for biologists conducting field work to present and discuss their findings with colleagues, as well as with educators, advocates, landowners and other stakeholders. Only abstracts of the presented papers are distributed, so the Carnivore Conference, like the annual conference of The Wildlife Society, attracts many researchers presenting preliminary results of ongoing work as well as completed work without jeopardizing subsequent publication in technical journals. The last conference, held in Monterey, California in November 2002, had more than 800 attendees. This is one of the few large-scale conferences that attracts both scientists and the general public, and Defenders strives to develop session themes that include both biological and sociological issues.

In December 2002, NWF initiated and cosponsored (with state and federal agencies) the Border Bears Workshop. This international workshop was subtitled "Small Populations of Grizzly Bears in the U. S.-Canada Transborder

Region: How Can We Work Together to Enhance Recovery?" The purpose of this workshop was to present and compile the best available science on the highly endangered bear populations in the North Cascades in northwestern Washington and southwestern British Columbia, as well as the Purcell, Cabinet-Yaak, and Selkirk Mountains in southeastern British Columbia, southwestern Alberta, northeastern Washington, northern Idaho and northwestern Montana. In addition to invited technical papers, this workshop included three separate sessions designed to involve local citizens and political leaders in recovery efforts for these highly endangered populations. The papers presented at this workshop were peer-reviewed and were published in 2004 as a Special Section in Volume 15, Issue 1 of *Ursus* (the journal of the International Association for Bear Research and Management).

Discussion

Collaboration between NGOs and agencies can produce tangible benefits for wildlife as illustrated by the projects described above. Agencies bring assets to this collaboration that are lacking or are in short supply in NGOs including:

1. abundant data on wildlife and wildlife users
2. strong biological capacity
3. relatively stable budgets
4. regulatory authority or high ability to influence regulations
5. dispersed staffs with frequently high credibility in local areas.

NGOs bring different assets to collaborative efforts including the:

1. ability to advocate for unpopular species or positions
2. lobbying capacity, including the ability to lobby for agency budgets
3. rapid-response capability
4. public outreach and opinion survey capabilities
5. ability to bring legal actions
6. budgetary and fiscal-year flexibility
7. lesser constrainment by changes in political leadership or direction
8. different interest group constraints.

Combining these assets can build stronger wildlife conservation programs with a broader base of public support than isolated actions by either agencies or NGOs.

NGO and agency perceptions of the best way to progress in wildlife conservation will not always coincide and will, at times, follow different paths. It is our hope, however, that the foundations built through the kinds of collaborative projects described above will translate into enhanced receptiveness by agency staff to listen to and incorporate our ideas into their programs and priorities. This has happened; we remain hopeful that it will occur more frequently and that, as a result, extreme voices on both sides of controversial issues will become less influential.

Reference List

Clark, J. D., D. Huber, and C. Servheen. 2002. Bear reintroductions: Lessons and challenges. *Ursus*. 13:335–45.

Schoen, J. W., and S. D. Miller. 2002. New strategies for bear conservation: Collaboration between resource agencies and environmental organizations. *Ursus*. 13:361–7.

U. S. Fish and Wildlife Service. 2000a. *Grizzly Bear Recovery in the Bitterroot Ecosystem, Final Environmental Impact Statement*. Missoula, Montana: U. S. Department of the Interior, Fish and Wildlife Service.

———. 2000b. *Record of decision and statement of findings for the environmental impact statement on grizzly bear recovery in the Bitterroot Ecosystem and final rule on establishment of a nonessential experimental population of grizzly bears in the Bitterroot area of Idaho and Montana*. Washington, DC: U. S. Department of Interior, Fish and Wildlife Service.

———. 2001. *Bitterroot notice of intent to reevaluate ROD: Summary of all comments received 6/21/2001–8/21/2001*. Washington DC: U. S. Department of Interior, Fish and Wildlife Service.

Seasonal Reduction of Medium-sized Mammalian Predator Populations to Enhance Waterfowl Production: An Evaluation of Biological Factors and Barriers to Adoption

Frank C. Rohwer
Louisiana State University, School of Renewable Natural Resources/ Delta Waterfowl Foundation
Baton Rouge/Portage la Prairie, Manitoba

Jonathan Scarth
Delta Waterfowl Foundation
Portage la Prairie, Manitoba

Rob Olson
Delta Waterfowl Foundation
Bismarck, North Dakota

Waterfowl have been intensively studied and managed in North America for almost a century, and, as a result, they are the most well known group of birds in the world. This interest reflects the importance of waterfowl hunting and the fact that waterfowl were the primary research focus of the U. S. Fish and Wildlife Service for most of the 20th century. Our extensive knowledge of waterfowl has allowed us to better understand the efficacy of efforts to manage populations than is the case for most other wildlife groups.

However, waterfowl science faces many challenges, resulting from the migratory nature of waterfowl and from the multiple political jurisdictions they occupy during their life cycle. In spite of the sophistication of our knowledge of waterfowl, some basic management questions have not been clearly answered. For example, waterfowl management in the winter is centered on two issues. How many waterfowl can we sustainably harvest, and what habitat management most improves winter survival and condition of northern migrants? Waterfowl biologists have only begun to understand winter ecology, and our efforts to relate winter habitat to survival or body condition have been limited to broad scale studies (Heitmeyer and Fredrickson 1981, Kaminski and Gluesing 1987). The question of the impacts of hunting of populations has been a topic of interest for at least 4 decades, yet a definitive understanding on harvest effects for mallards (*Anas platyrhynchos*), surely the most studied wild bird in the world, remains elusive (Nichols et al. 1995, Pöysä et al. 2004).

Management of Breeding Waterfowl

Most waterfowl research has focused on breeding waterfowl, especially in the prairies of the United States and southern Canada. For managers, the central question is simple—what management actions will most enhance waterfowl production? That question is far easier to answer than questions about harvest effects simply because local management should affect local recruitment, and actions far from the breeding grounds might be expected to have little impact on breeding processes (but see, Heitmeyer and Fredrickson 1981, Kaminski and Gluesing 1987). In spite of extensive research on nesting waterfowl, the most effective management for nesting waterfowl is not as apparent as expected (Williams et al. 1999). In fact, examining recent decades of waterfowl management shows some dramatic directional turns as we learned more about the biology of waterfowl and the efficacy of management actions.

As an example, we note that it was only after a couple of decades of intensive waterfowl management that biologists began to appreciate the value of small wetlands (Hochbaum 1944) and their drought and flooding cycles to the production of ducks on the prairies (Kadlec and Smith 1992, Kaminski and Prince 1981). Early waterfowl biologists recognized the importance of water on the breeding grounds, but they incorrectly assumed that conserving and creating big and permanent water was an effective way to produce ducks. Untold millions of dollars and several decades were spent on engineering projects that flooded and managed water levels over large acreages of staging marshes, described as "duck factories" and measured in "shoreline miles" in the parlance of the 1940s and 1950s. The realization that most of the life cycle factors limiting waterfowl populations occurred in the uplands and small wetlands of the Prairie Pothole Region (PPR) heralded a significant change in direction for waterfowl management. This transition from work on staging marshes to management of uplands around small potholes to enhance nest success involved overcoming significant institutional and fundraising inertia and—most difficult of all—the tacit admission that the waterfowl management community had made significant investments in large marsh areas that were not, in fact, important limiting factors for duck production. By the late 1980s, it was accepted that managed wetlands made up only a small fraction of the total breeding habitat for waterfowl (Kadlec and Smith 1992).

Management within the Uplands of the PPR

As the focus of waterfowl management moved into the PPR, research began to identify nest success as a critical factor in recruitment. In the 1970s and 1980s there were numerous studies of waterfowl nesting in the intensively farmed prairies that made it clear that poor nest success was more than a localized problem for waterfowl (Cowardin et al. 1983, Greenwood et al. 1987, Klett et al. 1988). Intensified focus on quantification of nest success lead to additional studies that documented that low duck recruitment on the prairies was largely due to the high levels of nest predation (Klett et al. 1988, Sargeant and Raveling 1992). Concurrently, the work of biologists at Northern Prairie Wildlife Research Center showed that three mammalian predators, namely striped skunks (*Mephitis mephitis*), red fox (*Vulpes vulpes*) and raccoons (*Procyon lotor*), were the primary agents of mortality for most upland duck nests (Klett et al. 1988, Sargeant et al. 1993). It soon became apparent that management to enhance duck production should focus on ways to improve nest success.

Management to Elevate Nest Success

Early management for nest success appeared to take two divergent paths. Managers looking at particular pieces of property set aside for duck production (including refuges and waterfowl production areas in the U. S. portion of the PPR) took a direct approach to managing nest success. These managers often adopted lethal predator control as a means to elevate nest success on their management areas. Such work formed the basis for several of the early research reports that documented that controlling predators was a feasible way to substantially improve nest success. Early predator control studies all involved the use of multiple methods of lethal control, but toxicants were the main tool of the predator control agents (Balser et al. 1968, Lynch 1972, Duebbert and Kantrud 1974, Duebbert and Lokemoen 1980).

Biologists looking at waterfowl with a larger view of the prairies seemed to take a different approach to reducing nest predation. As farming became more intensive, upland nesting ducks were being crowded into smaller remnants of habitat (Higgins 1977, Turner et al. 1987). One way to manage predation was to provide enough nesting cover, so nests would be dispersed and much more difficult to locate. Some early evaluations of planted cover suggested that establishing relatively small cover blocks had significant potential to elevate nest success (Duebbert and Lokemoen 1976). After some initial experimentation with fenced dense nesting cover, planting unmanaged nesting cover became a favored option of waterfowl managers, especially under the North American Waterfowl Management Plan (NAWMP) as it was implemented in prairie Canada in the late 1980s (Williams et al. 1999).

The Habitat Threshold

While establishing planted cover had become the tool of choice for NAWMP delivery agencies, research suggested that increases in nest success were modest (Klett et al. 1988, Kantrud 1993, McKinnon and Duncan 1999). Even when managers first started to plant upland cover without predator fences (see below) under the NAWMP, there were suggestions that the benefits of cover would be dependent upon the amount of cover in the landscape (Clark and Nudds 1991, Clark and Diamond 1993). Survey studies in prairie Canada found low nest success in grassland patches in eastern prairie areas where upland cover was scarce and higher nest success in patches in western areas, which were imbedded in a landscape with much more grassland (Greenwood et al. 1987, 1995). However, the definitive study in this regard was the work by Reynolds et al. (2001) that found a strong positive relationship between grassland cover and nest success of upland ducks in 161 samples of landscapes in the upper prairies of the United States. Unfortunately, that work suggests that about 40 percent of a landscape must be in grassland cover before the break-even threshold of 15- to 20-percent nest success is exceeded for most species. Reynolds et al.'s (2001) work was the first to demonstrate that a habitat threshold existed, which described to waterfowl managers how much cover in a landscape area would be required to elevate nest success to the point where breeding waterfowl could maintain their numbers.

Prairie Sociopolitical Barrier

Reynolds et al. (2001) established the target for waterfowl managers focused on establishing habitat to increase nest success. Studies, such as the NAWMP assessment, concurrently demonstrated that, over most of the PPR, waterfowl management had not yet secured adequate amounts of upland cover to positively affect duck recruitment (Ducks Unlimited 2004). The predictable response by many was to suggest that managers should strive to secure more habitat.

The main focus of waterfowl management since the inception of NAWMP in 1986 had been the Canadian PPR. The NAWMP management tool of choice was unfenced, upland cover that typically meant acquiring farmland through purchase and converting it to upland cover. As had happened in the U.S. prairies during the acquisition of easement refuges and waterfowl production areas (WPAs), a sociopolitical barrier emerged in the 1990s that threatened to stop NAWMP partners from acquiring any additional land.

Canadian farmers began to view the acquisition and setting aside of agricultural land by waterfowl conservation groups negatively and took action to curtail it through various means: opposing foreign ownership of land not promoting legislation for habitat purchases using U. S. dollars and passing local planning bylaws requiring local approval for habitat purchases (Rural Municipality of Sifton 2002). This reaction was engendered even though the total amount of land acquired—290,000 acres (117,500 ha) across prairie Canada—was relatively insignificant, and it compares to the negative reaction to habitat acquisition experienced in the U. S. portion of the PPR that led to legislative caps on long-term conservation easements. This reaction implies that the habitat threshold established by Reynolds et al. (2001) is not achievable for wildlife interests, suggesting that these habitat goals will only be met through reforms to agricultural policies, such as the Conservation Reserve Program (CRP).

Predator Exclusion Fences and Nesting Islands

The early indications that planting isolated blocks of cover in intensively farmed areas did not adequately protect duck nests from predators intensified other attempts to enhance nest success. Predator exclusion areas that involved building fences or using expanses of open water to discourage predator emigration to nesting cover showed excellent nest success (Lokemoen et al. 1982, Greenwood et al. 1990). Fences invariably included an initial trapping effort to remove predators that had entered the fenced area during the winter. Managers quickly embraced fenced predator exclusion areas, and, in some areas, the prairie fences became the predominate form of intensive waterfowl management. During the planning stages of delivery for the NAWMP in prairie Canada, it was assumed that much of the incremental duck production would come from fenced dense nesting cover (DNC); for example, in Manitoba it was projected that fenced DNC would account for 29 percent of the incremental mallard recruits—three times more than any other single treatment. However, research in the 1990s showed that fences may not enhance local recruitment due to problems with duckling survival as broods attempted to exit the fenced nesting cover (Pietz and Krapu 1994, Trottier et al. 1994). Currently, managers that build fences typically have one side of the fence open into a wetland. This solves the brood exodus problem, but also makes the fenced area more susceptible to predation by emigrants. In prairie Canada, these problems formed the stated rationale for abandoning fenced DNC as a management technique, effectively giving rise to the emergence of unfenced DNC as the primary NAWMP management tool in prairie Canada.

When biologists discovered the great propensity of mallard and gadwall (*Anas strepera*) to nest at high densities on islands in large lakes (Duebbert et al. 1983, Lokemoen 1984) the idea of creating nesting islands was quick to follow. Human made islands are readily accepted by nesting ducks (Giroux 1981), but, like any management that involves moving dirt, creating islands is very expensive and limited in the number of areas where islands can be created.

Predator Reduction to Enhance Nest Success

Chronically poor nest success in the prairies and the scarcity of effective alternative management options to elevate nest success lead to a reconsideration of the feasibility of lethal predator reduction. In the 1990s the Delta Waterfowl Foundation (Delta WF) began a series of experiments to assess whether predator reduction would be a viable form of intensive waterfowl management. The primary question of concern was about biological effectiveness. Could predator reduction that did not involve the use of toxicants sufficiently reduce predator abundance to cause an increase in duck nest success? There was much doubt that predator removal could elevate nest success because initial investigations on small management units in Minnesota and North Dakota produced only minor effects (Sargeant et al. 1995). In that study the nontrapped sites had a dismal 5.6-percent nest success, but the sites that were trapped during the nesting season achieved only 13.5-percent nest success. Although the difference in nest success was significant, the hatch rate on the trapped sites was below the 15 percent benchmark that is believed to be necessary for population maintenance (Cowardin et al. 1985, Klett et al. 1988).

We altered the Sargeant et al. (1995) trapping plan in four ways: (1) we contracted with a professional trapper, (2) we ensured this trapper could legally use leghold traps, body gripping traps and snares, (3) we greatly enlarged the size of the trapped sites, and (4) we included a small incentive clause if nest success surpassed a preset benchmark (50 percent) of apparent nest success. Our trapped blocks and our untrapped controls were 16-square mile (4,147-ha^2) blocks of habitat in north central North Dakota that had abundant wetlands and that had at least 20 percent of the landscape as upland nesting cover, largely due to CRP acreage (Garrettson and Rohwer 2001). For each block of habitat that was randomly chosen for predator removal, we hired a professional trapper and contracted services from March 1 through July 31. Trappers were to target four main predators: stripped skunk, raccoon, red fox and mink. And, they were to remove as many animals as possible. The trapper also was responsible for securing permission to trap on privately owned land, which was the overwhelming majority of the land.

Effectiveness of Predator Reduction by Trapping

Garrettson's study showed that predator removal could substantially improve nest success (Garrettson and Rohwer 2001). For a mix of upland nesting ducks, the Mayfield nest success went from 23 percent on untrapped blocks to 42 percent on blocks where the predator population was reduced. Success for the diving ducks that nest in emergent cover also increased substantially (Table 1). In a subsequent study, we expanded the size of the blocks of habitat to 36 square miles (9,330 ha^2), yet we still only contracted with one professional trapper for each site. The question of interest was whether diluted trapping intensity on larger blocks with less edge, and perhaps reduced immigration of predators, could also show elevated nest success after predator reduction? In this case the untrapped block had 15 percent Mayfield nest success while the trapped blocks had 36 percent nest success (Table 1).

Delta WF continued to examine the questions of biological efficacy of predator reduction to improve nest success with three additional studies. The above studies that were conducted in areas where the abundance of CRP land (about 20 percent) gave background rates of nest success at or just above the 15 percent threshold needed by mallards to assure population stability. In the third research project, Vance Lester, a University of Saskatchewan graduate student, went to an intensively farmed area in Saskatchewan where the background rates

Table 1. Values for nest success and brood survival for waterfowl nesting in the prairie pothole region with and without lethal predator management from 1994 to 2002.

Comparison of interest	Percent success Trapped	Percent success Untrapped	Sample size[a]	Location	Study block size	Source
Nests of upland nesting ducks[b]	42[c]	23[c]	2,706	North Dakota	16 miles^2	Garrettson & Rohwer 2001
Nests of diving ducks[d]	57	29	167	North Dakota	16 miles^2	Mense 1996
Nests of American Coots[e]	67	75	233	North Dakota	16 miles^2	Mense 1996
Nests of upland nesting ducks[b]	36	15	3,305	North Dakota	36 miles^2	Hoff 1999
Nests of upland nesting ducks[b]	48	19	2,376	Saskatchewan	16 miles^2	Vance Lester, pers. comm.
Nests of upland nesting ducks[b]	53	29	4,240	North Dakota	1 mile^2	Chodachek 2003
Northern Shoveler ducklings	71[f]	50[f]	47	North Dakota	16 miles^2	Zimmer 1996
Mallard ducklings	57[f]	36[f]	78	Saskatchewan	16 miles^2	Pearse & Ratti 2004

[a] Number of nests or broods followed

[b] Species composition was primarily blue-winged teal (*Anas discors*), mallard, gadwall, northern shoveler, and northern pintail (*A. acuta*)

[c] Mayfield nest success, but accounting for effects of date and incubation stage.

[d] Species composition primarily ruddy ducks (*Oxyura jamaicensis*) and redheads (*Aythya americana*)

[e] (*Fulica americana*)

[f] Survival percent from hatch to 30 days-old

of predation were high due to the very sparse cover (Richkus 2002). Again, the trapping produced a striking increase in nest success (Table 1).

Delta WF also reexamined the idea of trapping relatively small blocks of cover. We hired one professional to trap predators on 10 sites in North Dakota that were each one square mile (1.6 km^2), of which half or more was grassland habitat. Once again these trapped sites had nearly double the nest success as the untrapped control sites (Table 1).

The unequivocal take home message from these studies of nesting ducks is that lethal reduction of predator populations can substantially increase nest success. Six studies, funded by Delta WF and conducted by graduate students, all showed substantial increases in nest success when predator populations were trapped (Table 1).

Resistance to Predator Management

When Delta WF staff and student researchers began to communicate their results concerning predator management we discovered great resistance to consideration of predator management. The arguments against predator reduction came in many forms, but we can broadly categorize the concerns into four areas: biological effectiveness, cost effectiveness, philosophical acceptance and social concerns. We address each of these areas as we explore the resistance to managing predator abundance on the prairies.

Some of the early objections to the use of predator reduction asserted that it was not biologically effective in increasing recruitment of nesting waterfowl, or that it engendered undesirable side effects on nonwaterfowl species. These objections evolved through a number of arguments.

Biological Effectiveness

Altered Communities

Upon reporting that predator trapping could greatly increase nest success we noticed that biologists began asking much more sophisticated questions about this potential management technique. One concern was that trapping predators would disrupt the ecological system and perhaps cause trophic cascades as have been seen when coyotes (*Canis latrans*) are trapped from Texas scrubland (Henke and Byrant 1999). A particular concern was that removal of mesopredators would allow rodents to increase and this would increase predation rates on the grassland songbirds that readily use the CRP habitats. Grassland sparrows have shown substantial population declines (Askins 1993), so this question was of particular relevance. Nancy Dion, a University of Saskatchewan graduate student, tested this hypothesis by examining the nest success of grassland sparrows on predator trapped and untrapped control blocks. Nest success for songbirds did not differ for these two treatments (Dion et al. 1999); however, the causes of nest failure did shift such that small mammals accounted for a greater fraction of predation mortality on sites where medium-sized mammals were reduced (Dion et al. 1999). The lack of difference in nest success of sparrows was corroborated by a lack of difference in the predation rate of a much larger sample of artificial nests that contained quail eggs (Dion et al. 1999). During the predator removal study in Saskatchewan, Cameron Jackson, a University of Saskatchewan graduate student, compared Mayfield nest success for three relatively common shorebirds. Nest success on trapped sites when pooled over replicates was greater than control sites for Wilson's phalarope (*Phalaropus tricolor*-25 % control; 42 % trapped), willet (*Catoptrophorus semipalmatus*-63 % control; 100 % trapped), and upland sandpipers (*Bartramia longicauda*-49 % control; 72 % trapped) (Jackson 2003).

Jeremy Adkins, a student at Louisiana State University, (2003) tracked the seasonal abundance of mice and shrews during two years in North Dakota on 10 control sites and 10 trapped sites, each one square mile. On trapped and control sites there was a marked increase in the abundance of mice from May to October, but the trapped sites had a greater increase, which suggests a trophic cascade effect. However, by spring of year two, the difference in abundance had disappeared, suggesting that the harsh winter had a much greater impact on the small mammals than did predators.

Brood Survival

The goal of breeding ground management is to enhance the fall flight, but the typical yardstick for success is whether management actions increase nest success. Partially, this reflects the reality of what is readily measurable. Other components of productivity, such as nesting effort, renesting, hen success and brood survival, are difficult to measure (Johnson et al. 1992). However, analyses of duck productivity suggest that nest success is the single greatest contributor to variation in duck production (Johnson et al. 1992, Hoekman et al. 2002). Thus, the emphasis on nest success appears warranted.

It is feasible, however, that high nest success could lead to such large numbers of broods that there might be density dependence in brood survival. This would mean that nest success overestimates the gains in recruitment caused by predator reduction. Conversely, nest success might underestimate recruitment on predator reduction sites because there is a substantial mortality of ducklings, and predator reduction may reduce this form of mortality in addition to elevating nest success. To resolve this issue two students tracked brood survival during the experimental studies of predator reduction.

John Zimmer (1996) worked in North Dakota on the survival of northern shovelers (*Anas clypeata*) broods during 1995. His estimates of the probability of survival from hatch to 30 days were 71 percent on trapped sites and 50 percent on untrapped sites, but these survival estimates were based on a small sample and were not significantly different. A subsequent study by Pearce and Ratti (2004) in Saskatchewan focused on mallard brood survival during two seasons and had a substantial sample size. Trapped sites had 57 percent brood survival, which was significantly greater than 36 percent survival on untrapped sites (Pearce and Ratti 2004), which supports the idea that predator removal also enhances brood survival. These studies of brood survival cannot be considered conclusive, but the existing evidence suggests that brood survival, a second major component affecting duck recruitment (Hoekman et al. 2002), is also enhanced by predator reduction.

Cost Effectiveness

Inefficient Use of Management Funds

When Delta WF studies began to demonstrate that predator reduction could substantially increase duck production, the waterfowl community began to seriously debate predator management. Proposals to convert intensively farmed land into grassland cover that benefits a variety of wildlife might meet with skepticism among farmers, but they appear to be uniformly embraced by wildlife biologists and managers. Such unanimity of opinion does not apply to any predator reduction effort. Critics of predator trapping often argue that such annual management is unlikely to be an efficient use of relatively scarce management funds. We doubt the validity of this argument, but we applaud the underlying premise that alternative management approaches should be evaluated based on their cost effectiveness.

The first criterion of management efficiency for breeding ducks is that the management must enhance production—or its surrogate, nest success. The failure of many forms of management to achieve this basic requirement may explain why there have been few analyses of cost efficiency of waterfowl management. If a management action, such as predator reduction, can increase production then the logical next step is to evaluate management efficiency by examining net production gains in relation to management costs. This approach to evaluate management seems entirely obvious, but we know of only one such study, namely the classic work of John Lokemoen (1984). Perhaps the lack of replication of Lokemoen's work reflects the general concern that even the most effective intensive management produces frighteningly expensive ducks, which gives pause to advocates of intensive management.

Lokemoen (1984) showed that predator removal was the most efficient form of management then available. We should note that Lokemoen's data for predator reduction would have been based on studies that used toxicants (e. g., Duebbert and Lokemoen 1980). Studies using toxicants produced higher nest success than the more recent predator reduction studies that relied on trapping. Likewise, they may have had lower personnel costs for trapping due to the use of toxicants. We strongly believe that new analyses of cost efficiency would be informative.

Habitat Is Perpetual

A common argument to dismiss predator reduction is that land purchase and conversion to nesting cover secures habitat for the long-term and is, therefore, a much better use of intensive management dollars. Although we agree that conservationists should think about long-term goals, we believe this argument includes assumptions that are often false. First, adding cover to a landscape does little to elevate nest success until the cover occurs at levels that cannot be achieved in many agricultural landscapes (Reynolds et al. 2001), due to the sociopolitical barrier or financial limitations. Simply put, wildlife groups do not have enough money to purchase the land required to greatly increase nesting cover. Moreover, even if the money were available, the farm community makes it clear that they want land ownership to remain in the hands of farmers.

We believe that many biologists and wildlife managers labor under the misguided notion that protecting habitat requires only an up-front cost of purchase and seed for cover. This is far from the case because habitat purchase obligates land stewards to a substantial amount of annual maintenance. Stands of grass senesce and require regular rejuvenation by burning or reseeding. These habitat patches occur in a matrix of agricultural land, and local ordinances almost always require weed management. Additionally, land ownership generates a significant annual tax liability that must be paid to local governments.

Waterfowl managers have substantial experience with purchase and management of land, so future analyses of cost efficiency will be able to use reliable measures of the costs of land purchase, cover establishment and annual management. Our limited experience with habitat management suggests that annual maintenance is not a trivial expense associated with the acquisition and maintenance of nesting cover.

We conclude that a quantitative assessment of the cost of net recruits to the duck population is the metric that should be used to evaluate management efficiency. We recognize that predator trapping is an annual expense. On sites where trapping was repeated, the numbers of predators removed in a second year of trapping did not decline (Garrettson and Rohwer 2001, Chodachek 2003). However, we note that predator trapping is highly effective management to increase nest success and may also substantially enhance brood survival. Early work suggested that lethal predator reduction is one of the most cost effective management techniques available to enhance duck recruitment (Lokemoen 1984). We suspect that finding will be supported once cost evaluations are completed using recent data.

Philosophical Issues

We believe that two philosophical issues cause many in the wildlife field to dislike predator reduction. First, there is concern about disrupting natural communities. Second, many wildlifers believe anything that is not habitat management is only a makeshift solution to a population problem.

In the last few decades there has been a strong backlash to the prior centuries of predator control. Aldo Leopold's (1949) essays on this topic are particularly inspirational. Although they lack scientific rigor, they promote the idea that our meddling in community structure can lead to dysfunctional systems that lack predators to maintain balance. The chronic overabundance of white-tailed deer (*Odocoileus virginianus*) and the ecological problems of that excess provide a constant reminder of Leopold's refrain about tinkering with communities.

We argue that communities in the prairies have already been dramatically altered due to species extirpations and introductions. Raccoons are recent additions to these communities, and the abundance of red fox and skunks has surely been increased by agriculture and the associated changes to the landscape (Cowardin et al. 1983). Thus, we have few concerns that management that directly affects predator abundance in the prairies is upsetting a natural ecosystem.

Second, predator control is not habitat management. Again, it was Aldo Leopold's influential writings that elevated habitat management to the apex of all wildlife management. Wildlife educators have made habitat management the Holy Grail. University professors and the texts they select almost always promote the notion that appropriate habitat management can solve most of our wildlife problems. We have nothing against habitat management, but habitat effects on predation are clearly indirect effects. Predators have a rather direct and obvious impact on duck populations. In contrast, efforts to reduce nest predation by managing habitat, even without knowledge of the difficulty of the task, seem to be surprisingly indirect and inherently difficult management.

Manipulating populations of animals by altering habitat is considered ideal management. However, practically any habitat manipulation will benefit some species and adversely affect other species. It is interesting that negative impacts are acceptable if done by habitat manipulation, but they are far less acceptable if they involve direct manipulation of animal numbers. This apparent inconsistency is even more confusing given the wildlife communities' wide acceptance of sport hunting, which clearly manipulates animal numbers, and its readiness to promote hunting as a management tool in the case of overabundant waterfowl species, such as midcontinent populations of lesser snow geese (*Anser caerulescens*). We suspect the reluctance to directly manipulate predator populations to benefit more preferred species reflects the hesitancy of wildlife managers to proclaim value judgments that favor populations with perceived disregard for the status of individual animals.

Social Considerations

One of the predominant concerns with lethal predator management is that it is a socially unacceptable form of management. Many biologists and managers feel that instituting predator management on a large scale could lead to a very negative public perception about waterfowl management and, by association, waterfowl hunting. Management goals and actions have to fall within the bounds of societal values; however, we believe that waterfowl managers have taken an overly defensive attitude toward their desires to harvest waterfowl and to manage duck populations. Waterfowl hunters have supported a tremendous amount of habitat work through contributions and dedicated tax dollars. Because much of that habitat specifically purchased to help duck populations has been below break-even recruitment levels, it seems that managers could make a very good case for lethal predator management.

This is especially the case considering that the target species—skunks, raccoons and foxes—are not the large charismatic predators—wolves, bears or big cats—that attract much public empathy (Kellert 1985). In fact, reducing the populations of these target species is an objective shared with other public policy objectives, notably animal damage control programs aimed at reducing disease, agricultural losses and property losses.

Unfortunately, there is a surprising shortage of reliable information on the human dimensions aspect of lethal predator management in the prairies. The available data suggest the interested public in the United States is willing to support predator management as a wildlife management tool in order to achieve the goal of enhancing avian recruitment (Messmer et al. 1999). We note that there are many segments of society for which we must gain information about perceptions of predator management. Clearly, some people strongly promote the welfare of individual animals; this group would oppose predator management just as vehemently as they oppose sport hunting (e. g., Rutberg 2001). In contrast, the farm community shows much support for predator management, particularly when compared with the negative reaction to habitat acquisition by wildlife interests (see above). This probably reflects most farmers' acceptance of lethal management of individual animals and their dislike of skunks and raccoons, which farmers perceive as damaging to buildings and crops. Biologists seem most concerned about negatively impacting the opinion of the large uncommitted segment of people that do not hunt but do not currently have any active opposition to hunting.

We reiterate that large amounts of data suggests that direct predator management is an effective way to substantially increase nest success for ducks nesting in the prairie pothole region. Predator management appears to benefit more than just upland nesting waterfowl; the evidence gathered to date suggests improvements in duckling survival and in the nest success for over-water nesting waterfowl. The waterfowl management community, however, has been reticent about endorsing this management tool. Our main contention is that biologists should distinguish between the biological and cost effectiveness of various wildlife management techniques and their personal and philosophical biases. When there are legitimate questions of social attitudes with regard to wildlife management techniques, as is the case with predator management, positions from biologists should be based on attitudinal data gathered from relevant segments of the public. We strongly advocate for more research to gauge the social acceptability of predator management on the prairies.

Reference List

Adkins, J. A. 2003. *Experimental predator removal: A response in small mammal communities and relations to duck nest success*. M. S. thesis, Louisiana State University, Baton Rouge, Louisiana.

Askins, R. A. 1993. Population trends in grassland, shrubland, and forest birds in eastern North America. *Current Ornithology*. 11:1–34.

Balser, D. S., H. H. Dill, and H. K. Nelson. 1968. Effect of predator reduction on waterfowl nesting success. *Journal of Wildlife Management*. 32:669–82.

Chodachek, K. D. 2003. *Evaluation of repeated removal of mammalian predators on waterfowl nest success and density*. M. S. thesis, Louisiana State University, Baton Rouge, Louisiana.

Clark, R. G., and T. D. Nudds. 1991. Habitat patch size and duck nesting success: The crucial experiments have not been performed. *Wildlife Society Bulletin*. 19:534–43.

Clark, R. G., and A. W. Diamond. 1993. Restoring upland habitats in the Canadian prairies: Lost opportunity or management by design? In *Transactions of the 58th North American Wildlife and Natural Resources Conference*, 551–64. Washington, DC: Wildlife Management Institute.

Cowardin, L. M., A. B. Sargeant, and H. F. Duebbert. 1983. Problems and potentials for prairie ducks. *Naturalist*. 34:4–11.

Cowardin, L. M., D. S. Gilmer, and C. W. Shaiffer. 1985. Mallard recruitment in the agricultural environment of North Dakota. *Wildlife Monographs* 92:37.

Dion, N., K. A. Hobson, and S. Lariviere. 1999. Effects of removing duck-nest predators on nesting success of grassland songbirds. *Canadian Journal of Zoology*. 77:801–6.

Ducks Unlimited. 2004. *PHJV assessment study update*. Ducks Unlimited. http://www.ducks.ca/research/upb/phjv/index.html.

Duebbert, H. F., and H. A. Kantrud. 1974. Upland duck nesting related to land use and predator reduction. *Journal of Wildlife Management*. 38:257–65.

Duebbert, H. F., and J. T. Lokemoen. 1976. Duck nesting in fields of undisturbed grass-legume cover. *Journal of Wildlife Management*. 40:39–49.

———. 1980. High duck nesting success in a predator-reduced environment. *Journal of Wildlife Management*. 44:428–37.

Duebbert, H. F., J. T. Lokemoen, and D. E. Sharp. 1983. Concentrated nesting of mallards and gadwalls on Miller Lake Island, North Dakota. *Journal of Wildlife Management*. 47:729–40.

Garrettson, P. R., and F. C. Rohwer. 2001. Effects of mammalian predator removal on upland-nesting duck production in North Dakota. *Journal of Wildlife Management*. 65:445–52.

Giroux, J. F. 1981. Use of artificial islands by nesting waterfowl in southeastern Alberta. *Journal of Wildlife Management*. 45:669–79.

Government of Saskatchewan. 2002. *Report on farmland ownership, standing committee on agriculture, third Session of the 24th Legislature, June 21*. Saskatchewan: Legislative Assembly.

Greenwood, R. J., P. M. Arnold, and B. G. McGuire. 1990. Protecting duck nests from mammalian predators with fences, traps, and a toxicant. *Wildlife Society Bulletin*. 18:75–82.

Greenwood, R. J., A. B. Sargeant, D. H. Johnson, L. M. Cowardin, and T. L. Schaffer. 1987. Mallard nest success and recruitment in prairie Canada. In *Transactions of the 52nd North American Wildlife and Natural Resources Conference*, 298–309. Washington, DC: Wildlife Management Institute.

———. 1995. Factors associated with duck nest success in the prairie pothole region of Canada. *Wildlife Monographs*.128:57.

Heitmeyer, M. E., and L. H. Fredrickson. 1981. Do wetland conditions in the Mississippi Delta hardwoods influence mallard recruitment? In *Transactions of the 46th North American Wildlife and Natural Resources Conference*, 44–57. Washington, DC: Wildlife Management Institute.

Henke, S. E., and F. C. Bryant. 1999. Effects of coyote removal on the faunal community in western Texas. *Journal of Wildlife Management*. 63:1,066–81.

Higgins, K. F. 1977. Duck nesting in intensively farmed areas of North Dakota. *Journal of Wildlife Management.* 41:232–42.

Hochbaum, H. A. 1944. *The canvasback on a prairie marsh.* Harrisburg, Pennsylvania: Stackpole Company.

Hoekman, S. T., L. S. Mills, D. W. Howerter, J. H. Devries, and I. J. Ball. 2002. Sensitivity analysis of the life cycle of midcontinent mallards. *Journal of Wildlife Management.* 66:883–900.

Hoff, M. J. 1999. *Predator trapping on township-sized blocks: Does duck nesting success increase?* M. S. thesis, Louisiana State University, Baton Rouge, Louisiana.

Jackson, C. R. 2003. *Breeding ecology of prairie nesting shorebirds: Relationships with habitat, land use and predators*. M. S. Thesis, University of Saskatchewan, Saskatoon, Canada.

Johnson, D. H., J. D. Nichols, and M. D. Schwartz. 1992. Population dynamics of breeding waterfowl. In *Ecology and management of breeding waterfowl*, eds. B. D. J. Batt, A. D. Afton, M. G. Anderson, C. D. Ankney, D. H. Johnson, J. A. Kadlec, and G. L. Krapu, 446–85. Minneapolis, Minnesota: University of Minnesota Press.

Kadlec, J.A., and L.A. Smith. 1992. Habitat management for breeding areas. In *Ecology and management of breeding waterfowl*, eds. B. D. J. Batt, A. D. Afton, M. G. Anderson, C. D. Ankney, D. H. Johnson, J. A. Kadlec, and G. L. Krapu, 590–610. Minneapolis, Minnesota: University of Minnesota Press.

Kaminski, R. M., and E. A. Gluesing. 1987. Density and habitat-related recruitment in mallards. *Journal of Wildlife Management.* 51:141–8.

Kaminski, R. M., and H. H. Prince. 1981. Dabbling duck and aquatic macroinvertebrate responses to manipulated wetland habitat. *Journal of Wildlife Management.* 45:1–15.

Kanrud, H. A. 1993. Duck nest success on conservation reserve program land in the prairie pothole region. *Journal of Soil and Water Conservation.* 48:238–42.

Kellert, S. R. 1985. Public perceptions of predators, particularly the wolf and coyote. *Biological Conservation.* 31:167–89.

Klett, A. T., T. L. Shaffer, and D. H. Johnson. 1988. Duck nest success in the prairie pothole region. *Journal of Wildlife Management.* 52:431–40.

Leopold, A. 1949. *A sand county almanac, and sketches here and there.* Oxford, England: Oxford University Press.

Lokemoen, J. T. 1984. Examining economic efficiency of management practices that enhance waterfowl production. In *Transactions of the 49th North American Wildlife and Natural Resources Conference,* 584–606. Washington, DC: Wildlife Management Institute.

Lokemoen, J. T., H. A. Doty, D. E. Sharp, and J. E. Neaville. 1982. Electric fences to reduce mammalian predation on waterfowl nests. *Wildlife Society Bulletin.*10:318–23.

Lokemoen, J. T., H. F. Duebbert, and D. E. Sharp. 1984. Nest spacing habitat selection and behavior of waterfowl on Miller Lake Island, North Dakota. *Journal of Wildlife Management.* 48:309–21.

Lynch, G. M. 1972. Effect of strychnine control on nest predators of dabbling ducks. *Journal of Wildlife Management.* 36:436–40.

McKinnon, D. T., and D. C. Duncan. 1999. Effectiveness of dense nesting cover for increasing duck production in Saskatchewan. *Journal of Wildlife Management.* 63:382–9.

Mense, B. J. 1996. *The effects of predator removal and nest-site selection on productivity of over-water nesting birds in North Dakota.* M. S. thesis, Pittsburgh State University, Pittsburgh, Kansas.

Messmer, T. A., M. W. Brunson, D. Reiter, and D. G. Hewitt. 1999. United States public attitudes regarding predators and their management to enhance avian recruitment. *Wildlife Society Bulletin.* 27:75–85.

Nichols, J. D., F. A. Johnson, and B. K. Williams. 1995. Managing North American waterfowl in the face of uncertainty. *Annual Review Ecology and Systematics.* 26:177–99.

North American Waterfowl Management Plan. 1986. *Environment Canada.* Ottawa, Ontario: North American Waterfowl Management Plan.

Pearse, A .T., and J. T. Ratti. 2004. Effects of predator removal on mallard duckling survival. *Journal of Wildlife Management.* 68: 342–50.

Pederson, R., J. Nelson, and B. Verbiwski, eds. 1988. *Manitoba implementation plan for the North American Waterfowl Management Plan Prairie Habitat Joint Venture.* Manitoba, Canada: Manitoba North American Waterfowl Management Plan Technical.

Pöysä, H., J. Elmberg, G. Gunnarsson, P. Nummi, G. G. Sjöberg, and K. Sjöberg. 2004. Ecological basis of sustainable harvesting: Is the prevailing paradigm of compensatory mortality still valid? *Oikos.* 104:612–5.

Pietz, P. J. and G. L. Krapu. 1994. Effects of predator exclosure design on duck brood movements. *Wildlife Society Bulletin*. 22:26–33.

Reynolds, R. E., T. L. Shaffer, R. W. Renner, W. E. Newton, and B. D. J. Batt. 2001. Impact of the conservation reserve program on duck recruitment in the U. S. prairie pothole region. *Journal of Wildlife Management*. 65:765–80.

Richkus, K. D. 2002. *Northern pintail nest-site selection, nest success, renesting ecology, and survival in the intensively farmed prairies of southern Saskatchewan: An evaluation of the ecological trap hypothesis*. Ph.D. dissertation. Louisiana State University, Baton Rouge, Louisiana..

Rural Municipality of Sifton. 2002. *Zoning by-law no. 1325*. Sifton, Manitoba.

Rutberg, A.T. 2001. Why state agencies should not advocate hunting or trapping. *Human Dimensions of Wildlife*. 6:33–7.

Sargeant, A. B., R. J. Greenwood, M. A. Sovada, and T. L. Shaffer. 1993. *Distribution and abundance of predators that affect duck production–Prairie Pothole Region, resource publication 194*. Washington, DC: U. S. Fish and Wildlife Service.

Sargeant, A. B., and D. G. Raveling. 1992. Mortality during the breeding season. In *Ecology and management of breeding waterfowl*, eds. B. D. J. Batt, A. D. Afton, M. G. Anderson, C. D. Ankney, D. H. Johnson, J. A. Kadlec, and G. L. Krapu, 396–422. Minneapolis, Minnesota: University of Minnesota Press.

Sargeant, A. B., M. A. Sovada, and T. L. Shaffer. 1995. Seasonal predator removal relative to hatch rate of duck nests in waterfowl production areas. *Wildlife Society Bulletin*. 23:507–13.

Standing Committee on Agriculture. 2002. *Report on farm land ownership, third session of the 24th Legislature*. Saskatchewan: Legislative Assembly.

Trottier, G. C., D. C. Duncan, and S. C. Lee. 1994. Electric predator fences delay mallard brood movements to water. *Wildlife Society Bulletin*. 22:22–6.

Turner, B.C., G. S. Hochbaum, F. D. Caswell, and D. J. Nieman. 1987. Agricultural impacts on wetland habitats on the Canadian prairies, 1981–1985. In *Transactions of the 52nd North American Wildlife and Natural Resources Conference*, 206–15. Washington, DC: Wildlife Management Institute.

Williams, B. K., M. D. Koneff, and D. A. Smith. 1999. Evaluation of waterfowl conservation under the North American waterfowl management plan. *Journal of Wildlife Management*. 63:417–40.
Zimmer, J. M. 1996. *Effects of predator reduction on the survival and movements of northern shoveler broods*. M. S. thesis, Louisiana State University, Baton Rouge.

Session Two.
Our Water Resources: A Candidate for Listing?

Chair
Jack Capp
U. S. Department of Agriculture, Forest Service
Washington, DC

Cochair
Robert P. Davison
Wildlife Management Institute
Bend, Oregon

The Colorado River—Managing a Critical Resource

William E. Rinne
U. S. Bureau of Reclamation
Washington, DC

Introduction

The Colorado River and its tributaries flow 1,450 miles (2,330 km) through much of the southwest United States, including parts of seven states-Colorado, Wyoming, Utah, New Mexico, Nevada, Arizona and California—before emptying into the Gulf of California in Mexico. One-twelfth of the nation's land, 264,000 square miles (684,000 km^2), drain to the Colorado River. The original name of the river, given by the Spanish explorers, was the Rio Colorado, which means Red River, because of the presence of red desert sediments in the river, especially during periods of higher flows.

The Colorado River has been the most important economic and natural resource in the settlement and development of the southwest and is, indeed, the lifeblood of this region. The importance of the Colorado River to the seven basin states and to Mexico cannot be overstated. The average annual precipitation in the Colorado River Basin is only 4 inches (100 m). It is estimated that this amount of average annual precipitation would supply the needs of 3 million people without development of an infrastructure to regulate and store runoff.

Yet, it irrigates for more than 3.7 million acres (1.5 million ha) of farmland, and it supplies the estimated 30 million people in the United States and Mexico. The Colorado River also generates an annual average of over 10 billion kilowatt hours of power and more than 30 million visitor-days of recreation. The river also provides habitat for native and introduced species of fish and wildlife.

The U. S. Bureau of Reclamation (Reclamation), as the primary manager of the operations on the Colorado River, has always been able to meet the water delivery requirements of users as required by law. However, since 1990 and continuing today, management challenges have increased on the Colorado River because of continual population growth, greater use of basic allocations by basin states, prolonged drought in the Colorado River Basin and conflict among stakeholders over the best use of the river. Reclamation has reached a point that has been described as an era of limits. It seems certain that these demands and the pressure on this finite resource will only continue to increase in the years to come.

Overview of Hydrology, Storage and Basic Allocation of the Colorado River

The Colorado River Basin is divided into two roughly equal geographic sections: the Upper and Lower basins. The states receiving nearly all of the Upper Basin's allocation, the Upper Division states, include Colorado, Wyoming, New Mexico and Utah. The states receiving all of the Lower Basin's allocation, the Lower Division states, include Nevada, Arizona and California. The seasonal flows of the Colorado River have varied historically from a trickle in the late summer and fall months to more than 200,000 cubic feet per second (5,665 m^3/sec) during periods of major spring runoff. Over the last hundred years, annual flows have ranged from 5 million acre-feet (6.2 km^3) to 25 million acre-feet (30.8 km^3).

The average annual, natural flow of the river at Lee Ferry, the river's midpoint that divides the Upper Basin and the Lower Basin, was initially estimated to be 18 million acre-feet (about 23 km^3) in the 1920s. This estimate was subsequently used by state and federal managers to develop the basic compact allocations for the Upper and Lower basins in the early 1920s. That estimate is now considered to have been an overestimation of average annual flow because the data used covered a 16-year period of unusually high annual flows on the Colorado River. After collecting almost 100 years of flow data, today the average annual flow is estimated to be less than that, about 15 million acre-feet (19 km^3). Some scientists feel that, based on studies of tree growth rings that span centuries, the long-term average flow is even less, close to 13.5 million acre-feet (16.7 km^3). During the first two thirds of the 20th century, construction of major dams, creation of reservoirs and completion of water delivery infrastructure increased the certainty of water supply throughout the Colorado River Basin. This provided the catalyst for extensive agricultural and urban development, especially in the Lower Basin. Most dams and associated structures were constructed during the 1920s through the 1960s by Reclamation.

Today, the Colorado River reservoirs can store about 60 million acre-feet (74 km^3) of water, nearly 4 years of the annual estimated flow. Over 80 percent of this storage exists in two large reservoirs—Lake Powell, at the lower end of the Upper Basin, and Lake Mead, at the upper end of the Lower Basin. The federal law requires that the operation of these two reservoirs be closely integrated because they are critical to sustaining the water supply and to meeting annual demands within the basin.

The division of the United States' part of the Colorado River Basin was established by the 1922 Colorado River Compact (Compact). The 1944 Mexican Water Treaty provides the allocation of the Colorado River in Mexico. In years of normal water supply these allocations totaled 16.5 million acre-feet (20.4 km^3) of water, with 7.5 million acre-feet (9.3 km^3) each to the Upper and Lower basins and 1.5 million acre-feet (1.9 km^3) to Mexico. At present about 14.5 million acre-feet (17.9 km^3) of Colorado River water is used each year. Of this, more than 8 million acre-feet (9.9 km^3) is used in the Lower Basin for irrigation of agriculture in Arizona and California and for the large and rapidly growing urban areas of Las Vegas, Phoenix, Tucson, Los Angeles and San Diego. Present use in a more slowly developing Upper Basin is estimated to be a little more than 4 million acre-feet (4.9 km^3) per year. Although current use of Colorado River water is slightly below the long-term estimated annual average of 15 million acre-feet (18.5 km^3), the river is still considered to be overallocated because of the initial compact and treaty allocations, which totaled 16.5 million acre-feet (20.4 km^3). This overallocation will likely further complicate the management of the Colorado River.

Overview of the Law of the River

It is important to understand the legal framework that exists and its influence on how the river system works before discussing specific issues and management challenges on the Colorado River. Most of the rivers in the western United States operate under a combination of various state and federal laws. In contrast, operation of the Colorado River is dominated to a greater extent by requirements of federal laws, compacts, a Supreme Court decree and a treaty that were created specifically for the Colorado River. This extensive and complex body of law has developed over the past 82 years and is commonly known as The Law of the River. The existence of conflict, fear and concern over water and its use has existed among the seven basin states throughout much of the history of development and management of the Colorado River and its tributaries. In particular concern among the other six basin states over development and associated water use in California has existed since the time of the compact. The strong federal presence and associated legal structure is due primarily to the inability of basin states to reach agreement on allocation and management of the river, the need for federal funding to develop and regulate the river, and the sharing of the Colorado River's supply with Mexico. Ironically, the legal framework created by the controversy and conflict among users and uses now provides a sound basis for managing this vital resource.

Some of the fundamental parts of the legal framework that guide operation and management of the Colorado River include: theCompact, the 1928 Boulder Canyon Project Act, the 1929 California Limitation Act and the subsequent Seven Party Agreement of 1931, the 1944 Mexican Water Treaty, the 1948 Upper Colorado River Basin Compact, the 1956 Colorado River Project Storage Act, the 1964 Arizona versus California Supreme Court decree, the 1968 Colorado River Basin Project Act, the 1974 Salinity Control Act, the 2001 interim surplus criteria, and the 2003 Colorado River Water delivery agreement.

On a broader scale there is another body of U. S. law related to environmental matters including key legislation such as the 1969 National Environmental Policy Act, the 1973 Endangered Species Act (ESA), and the 1992 Grand Canyon Protection Act. Assuring appropriate compliance with the requirements of environmental legislation and implementing regulations have been integral parts of management approaches and operations on the Colorado River during the last 30 years of the 20th century. Compliance with the ESA is the basis for several significant environmental programs in the Colorado River Basin.

A review of some of the circumstances relating to development of the Colorado River Basin over the last century provides a glimpse of some key events that led to development of the Law of the River. As the 20th century began, the states in the Upper and Lower basins each wanted to have an assured supply of water from the Colorado River. In the early 1900s, significant irrigation development using Colorado River water began in the Imperial Valley of California. But flooding and drought severely hampered progress, and California began seeking federal help to construct dams that would allow development to proceed. The states upstream of California convinced Congress to block federal assistance, fearing that California would develop and use all the available water supplies before the other basin states could develop what they felt was their fair share. Their concerns were based, in part, on the western legal doctrine of prior appropriation, where the first beneficial use of water retains the highest priority when supplies are limited.

Recognizing the need to promote western economic development, representatives from the other basin states called for negotiations to establish an equitable apportionment of Colorado River water. The federal government was called upon to facilitate the negotiation of a compact among the states, which requires specific congressional authorization and approval. Although this effort was difficult and did not result in allocation of Colorado River water to each state, an agreement was ultimately reached to divide the Colorado River into two basins, with an equal allocation of water to each basin. This agreement, the compact, is the foundation of the Law of the River. It wasn't until the passage of the 1928 Boulder Canyon Project Act, the second fundamental element of the Law of the River, that Congress was able to move forward and authorize the construction of major infrastructure to control and develop the Colorado River. The delay was caused by strong disagreement among the Lower Basin states of Arizona and California over the allocation of the Lower Basin's share of the river. Ultimately, the Lower Basin states could not reach an agreement, and Congress, through the 1928 Boulder Canyon Project Act, apportioned the Lower Basin's share of Colorado River water as follows: 4.4 million acre-feet (5.4 km^3) to California, 2.8 million acre-feet (3.5 km^3) to Arizona and 300,000 acre feet (0.4 km^3) to Nevada. The Supreme Court's opinion of 1963, in Arizona versus California, confirmed that Congress had made such an apportionment by authorizing the Secretary of the U. S. Department of the Interior (Secretary) to accomplish this division. This, in essence, federalized the Lower Basin of the Colorado River System, a situation that is unique in the United States. Although many more laws have been subsequently developed, the compact, the Boulder Canyon Project Act and the decree were key components in the authorization for the construction of Hoover Dam, the limitation on the Lower Basin's (in particular California's) use of water and the establishment of the unique federal role in the Lower Basin of the Colorado River.

Operation of the Colorado River

The Colorado River is operated and managed to satisfy specific core objectives, required by law. From a practical standpoint, river operations are designed to meet all legal requirements while maintaining as much water in storage as possible. This approach is based on the hydrology of the Colorado River, which is highly variable, and on supplies of water from other sources, which are extremely limited within this highly arid region. The primary objectives required by law include: providing treaty water to Mexico, providing flood control and river regulation, storing and delivering water for reclamation of public land and other beneficial uses, and generation of electrical energy. In addition, operations conserve environmental resources and enhance recreational opportunities as much as possible.

Management of operations on the Colorado River differs between the Upper and Lower basins. In the Upper Basin, the individual states, through their state engineers, control the allocation and administration of water rights on the Colorado River and its tributaries. The Upper Basin's share of the Colorado River is divided by percentage of the available supply as agreed to in the 1948 Upper Colorado River Basin Compact. This compact also created an Upper Basin Commission to oversee and coordinate matters among the Upper Basin states, and it allowed development to move forward in the Upper Basin states. Reclamation operates and manages dams and reservoirs on the mainstream and tributaries in the Upper Basin and works closely with the commission to meet required water deliveries and other activities. The Secretary, through Reclamation, does control operation of the Colorado River Storage Project.

In contrast, in the Lower Basin, the Secretary is the water master for the river. Reclamation functions as the Secretary's representative to carry out the responsibilities and authorities of the water master in accordance with those elements of the Law of the River that pertains to the Lower Basin. The main functions performed by Reclamation to operate the Colorado River in the Lower Basin include: (1) flood control, navigation and river regulation, (2) irrigation and domestic uses, including the satisfaction of present perfected rights (water rights that existed prior to June 25, 1929), and (3) generation of power.

In addition, each year the Secretary prepares an annual operating plan for the Colorado River as required by the 1968 Colorado River Basin Project Act, that determines whether the water supply in the Lower Basin will be managed under normal, surplus or shortage conditions. The annual operating plan is used to guide annual operations and is developed based on actual and forecast storage, historic stream flow, projected water orders and probabilities of water supply.

Although operations of the Colorado River are heavily regulated, refinements do occur in response to changes in the legal framework. For example, operation of the Colorado River to meet Lower Basin demands was recently influenced by the development and implementation of new guidelines and agreements. In January of 2001, to facilitate California reducing its long-time dependence on as much as 16 percent more than its share of Colorado River water, the Secretary established interim surplus guidelines (ISGs). The ISGs defined specific conditions for annual water releases from Lake Mead to Arizona, Nevada and California during a 15-year period. These ISG were implemented, in part, to provide California greater certainty and the other basin states more assurance about how surplus water would be allocated during this period. To benefit from the ISGs, California was required to demonstrate sufficient progress in implementing a permanent plan that reduced its annual use of Colorado River water. This plan would result in California limiting its use of Colorado River water to 4.4 million acre-feet (5.4 km^3) in years of normal supply.

Ultimately, California achieved this requirement of the ISG through adoption of the historic Colorado River Water Delivery Agreement of 2003. The Secretary signed this agreement on October 10, 2003, and it quantified the agricultural entitlements of Colorado River water for use in California. It also provides the framework to allow conserved agricultural water (primarily in the Imperial Valley) to be transferred for use in urban centers of California's southern coastal plain. These actions improved the ability to administer California's entitlement of Colorado River water and allowed California to limit its use to its legal entitlement of 4.4 million acre feet (5.4 km^3). These actions by the Secretary, in cooperation with the seven basin states, were major steps in managing the operations of the Colorado River to meet the demands of the 21st century.

Current and Future Challenges in Managing the Colorado River

Much of the history of the Colorado River in the 20th century involved harnessing and controlling the river to protect against floods and drought. This work resulted in an extensive network of dams and reservoirs that can store up to four years of average runoff. This reliable development has supplied the water and has generated power to encourage and sustain development of agricultural, municipal, industrial and recreational resources in the basin. The extensive litigation and legislation between 1922 and 1974 provided the legal framework to protect the rights and interests of users in all seven basin states and Mexico.

At the heart of today's challenges is the development of strategies to address the needs of an exploding population in an area of limited water supply. It is likely that there is no single solution for accomplishing this. However, implementation of actions that stretch and balance the use of the Colorado River's limited water supply offer the best chance for long term success in meeting a variety of needs in the basin.

Today, explosive population growth in urban areas, continued demands for irrigation and a rural lifestyle, increasing demands for water to support recreational and environmental needs, and negotiations on Native American water rights claims are causing conflict between citizens in many parts of the basin. Five of the seven states in the Colorado River Basin have experienced a 20- to 60-percent increase in resident population growth between 1990 and 2000. There is no indication of a decrease in this trend during the first four years of the 21st century. Most of this growth is in large urban areas in Nevada, Arizona, California, Colorado and New Mexico. Much of today's conflict around water use is caused by: location of the supply in relation to the area of demand (both intra- and interstate); competition for water among agriculture, municipal, environmental and recreational uses; and conflict between existing non-Native American water use and Native American water rights claims.

Major transbasin diversions have occurred to move Colorado River water from rural sparsely populated areas to heavily populated, water-short urban centers. Transbasin diversions have been used by basin states to help accommodate intrastate needs outside of the basin including: (1) the transfer of water from west slope of the Rocky Mountains by Colorado to front range areas (like Denver on the east slope), (2) the diversion of water from the Colorado River by Utah to Salt Lake City and other west slope areas of the Wasatch range, (3) the transbasin diversion of water from the San Juan River (a tributary of the Colorado River) by New Mexico into the Rio Grande River system for use in the Albuquerque area, and (4) the diversion of Colorado River water by California to the southern coastal plain area. Even Mexico moves some of its allocation across the Baja California Peninsula to Tijuana.

Despite the development of projects to move water to rapidly growing urban areas, nearly 80 percent of the water used in the Colorado River Basin is used for irrigation of agriculture. The use of water for agricultural irrigation usually has the highest priority because of its beneficial use. Although market-based, agricultural-to-urban transfers provide part of the solution in satisfying urban needs, transfer agreements can be difficult to achieve.

For example, in 1931 the California entities entered into the Seven Party Agreement, an agreement allocating California's share of the river among the seven major water users—both agricultural and urban—within California. Like the previous attempts to divide the Colorado River among the basin states, the entities within California were unable to agree on specific allocations of water. The agreement and subsequent contracts allocated 3.85 million acre-feet (4.75 km^3), the bulk of California's 4.4 million acre-feet (5.4 km^3) share, to irrigation, leaving only 550,000 acre-feet (.068 km^3) for use in urban areas of its southern coastal plain. The agricultural allocation remained as unquantified shares.

The rapidly growing areas of the southern coastal plain caused California to look to other sources of supply, including unused Colorado River water. For many years California was able to utilize water unused by Nevada and Arizona. However, this practice ended as both states now utilize their full allocations.

Nevada's small allocation of 300,000 acre-feet (0.4 km^3) of Colorado River water seemed sufficient in the 1920s when its population was quite small and when Nevada had no arable land to support irrigation development. Today southern Nevada has a population of nearly 1.3 million and has the highest rate of population growth in the United States. Nevada has been using its full share of Colorado River water for several years and recently projected that it may need a 50-percent increase in the next 15 years to accommodate the projected growth.

The completion of the Central Arizona Project, establishment of the Arizona Banking Authority, continual growth of the Phoenix and Tucson urban areas, and subordinate position in the event of shortage of Colorado River water, have all increased Arizona's concern over the Colorado River and caused Arizona to increase pressure on California to limit its use to is apportionment as it had agreed to some 70 years earlier in the 1929 California Limitation Act.

As a result, California has turned its attention to conserving and transferring water from higher priority irrigated agriculture areas of southern California, such as the Imperial Valley, to the southern coastal plain. This effort was initiated in the late 1990s and resulted in the creation of a draft plan in 2000, known as the Colorado River Water Use Plan, commonly referred to as the California 4.4 Plan. The goal of the plan was to implement key actions in California that would reduce California's reliance on the Colorado River to its apportionment of 4.4 million acre-feet (5.4 km^3) in years of normal water supply. The other basin states and the U. S. Department of the Interior supported this initiative and provided an incentive through the previously mentioned 2001 Interim Surplus Guidelines to provide California with a soft landing over a 15-year period as it implemented its plan.

As is typical, of Colorado River management issues, completion of the agreements to implement the plan proved difficult. In December 2002, the Secretary enforced the Law of the River and suspend the ISGs, an action which immediately reduced California's use to 4.4 million acre-feet (5.4 m^3) during 2003 because of lack of progress towards meeting required ISG conditions.

Significant concerns with the proposed transfers were raised by residents in the Imperial Valley about the impacts of the proposed transfer on their local economy, their way of life and the environment of the Salton Sea in the Imperial Valley. Litigation also occurred between the Imperial Irrigation District and the U. S. Department of Interior over Imperial Irrigation District's beneficial use of Colorado Rive water. Concerns by environmentalists within California about the potential impact of the proposed transfer of water from the Imperial Valley on the habitat and fish and wildlife species of the Salton Sea, ultimately led to state legislation and agreements. After protracted negotiation and litigation, the 2003 Colorado River Water Delivery Agreement was signed by four of the major water users within California and the U. S. Department of the Interior; it was considered a major step forward in management of the Colorado River. Under this agreement California agreed to take specific, incremental steps to reduce its overreliance on Colorado River water by 2016. California is also addressing concerns of a variety of interests within California under a complex series of separate agreements that do not involve the federal government. The transfer of water from agriculture to urban areas, although contentious and full of challenges, will likely play an important role in stretching and balancing future water use and management in the Colorado River Basin.

Addressing environmental values in the Colorado River Basin is another key management challenge. Prior to development of reservoirs on the Colorado River, it was erratic and very turbid during flow events because it carried significant silt and sediment loads. As a result fish and wildlife resources native to the river and its associated habitat were highly specialized and uniquely adapted to survive in a harsh environment that experienced extreme variations in seasonal and annual river flows. The makeup of the habitat and fish and wildlife resources of the Colorado River is much different today.

Historically, the Colorado River supported one of the most unique fish communities in the world. In fact, most of the nine species of fish native to the Colorado River occur nowhere else. This situation is in stark contrast to other U. S. river systems, such as the Missouri or Mississippi rivers. These rivers typically support many times the numbers of fish with wide distribution.

This specialization to a harsh environment did not serve the fish native to the Colorado River particularly well as the river was developed to meet human needs. The development of water projects and introduction of exotic species, have changed much of the aquatic environment and have increased competition from nonnative species, causing a decline in the abundance and distribution of native species throughout much of the basin.

The challenge facing current and future management of these and other native species is to develop and implement strategies that will sustain these resources while still meeting water and power demands. Although this has not been and likely will not be easy to do, success will likely involve actions that include both human intervention and natural processes to maintain populations of native species. A primary concern within the basin is the conservation of four species of fish in the river that are listed under the ESA. They include: humpback chub (*Gila cypha*), razorback sucker (*Xyrauchen texanus*), bonytail (*Gila elegans*) and Colorado pikeminnow (*Ptychocheilus lucius*). Several activities are underway in the Colorado River Basin to help manage these listed fish and other wildlife resources. Most of these efforts are aimed at protecting or restoring selected reaches or segments in the Colorado River System in order to maintain native species. These programs are usually tied to management actions that comply with the ESA. Four examples are the Upper Colorado River Recovery Implementation Program, the San Juan Recovery Implementation Program, the Glen Canyon Adaptive Management Program and the Lower Colorado River Multispecies Conservation Program. In the case of the first two programs, the efforts provide the federal government and the states with the ability to continue to develop and consume water from the Upper Colorado River Basin. At the same time, appropriate measures maintain viable populations of native fish and restore or protect their habitats.

In the Lower Basin, Reclamation is working with other federal partners, the three states, and Native American tribes to develop the Lower Colorado River Multispecies Conservation Program. This is a coordinated, comprehensive approach to conserve and recover a multitude of species and projects under one, common umbrella over a 50-year period while continuing current river operations and, where possible, providing for future development of water and power.

All of these programs are ESA driven, involve federal and nonfederal participants, and are composed of diverse interest groups. To successfully plan and implement these efforts requires close cooperation among participants, and it typically involves the establishment of oversight or advisory organizations. Long-term success in management of the fish and wildlife resources of the Colorado River will depend on the ability to maintain the necessary cooperation and commitment of resources from participants, the appropriate use of adaptive management to address new or changing conditions, the integration of management among the programs within the basin when possible and a willingness to use human intervention to sustain populations of fish and wildlife.

Although construction of dams and formation of reservoirs has impacted the distribution and abundance of several of the native species, it has also increased habitat for introduced species. Many of the species now contribute significantly to the economic and recreation base of the basin through fishing, hunting, bird watching and other outdoors activities in the southwest. The demand for this use of fish and wildlife resources may increase in the future because of population growth and associated recreation needs.

The current severe drought in the Colorado River Basin continues to impact operations on the Colorado River. Droughts have been cyclic during the last hundred years on the Colorado River.

- From 1931 to 1935, spring runoff was 76 percent of average.
- From 1953 to 1956, spring runoff was 68 percent of average.

- From 1959 to 1964, spring runoff was 76 percent of average.
- From 1988 to 1992, spring runoff was 70 percent of average.

To day is the fifth year of drought that began in 2000 when the spring runoff was 71 percent of average. The current drought has caused other concerns in the Colorado River Basin. The Upper Basin has raised concerns about the continual decline in water levels in Lake Powell because releases to meet Lower Basin requirements are greater than inflow. Lake Powell provides the storage that the Upper Basin relies on to meet compact requirements to the Lower Basin.

Arizona has increasingly called upon Reclamation to start operation of the Yuma Desalting Plant, authorized under the 1974 Salinity Control Act, to reduce the bypass of water to Mexico. The source of the bypass water is from the Welton Mohawk Irrigation District in the lower part of the Gila River Basin. Arizona is concerned that system contents in the Colorado River may fall so low during this drought that shortages may occur. Because of its low priority, the Central Arizona Project's water supply would receive any required shortage first in the Lower Basin.

The Yuma Desalting Plant issue has, in turn, heightened concerns of environmental interests. The operation of the Yuma Desalting Plant would result in decreases in by-pass water reaching the Cienega de Santa Clara, a large brackish water marsh in the Colorado River Delta area of Mexico. Although created and sustained by the artificial bypass flows, this marsh provides habitat for many migratory and special status species and is the largest remaining wetland in the delta.

In addition, the prolonged drought has caused some stakeholders to suggest that shortage criteria should be developed as soon as possible. The previous examples illustrate the impact of hydrology and demand in influencing the operation and management of the Colorado River.

Closing Thoughts

The challenges in managing the Colorado River, although specific in nature, are in many ways indicative of the problems facing all western U. S. water managers and point out that the worst time to plan for an emergency is in the middle of an emergency. In some areas, water supplies are or will be inadequate to meet the demands for water for people, for farms, for cities and for the environment—even in normal years.

In this regard, future management of the Colorado River must focus attention on the reality that explosive population growth in western urban areas, coupled with competing needs for water to support environment, recreation and agriculture will continue to cause conflict among citizens and users of the river. Successful resolution of these issues over the long-term will require the involvement of states, Native American tribes, stakeholders and local governments to provide the necessary skills, funding and solutions. Although there does not appear to be a "silver bullet" to stretch water supplies, several actions will help, including enhancement of water conservation, improvement of the efficiency, repair of ageing infrastructure, improvement in monitoring, measurement of water supplies, implementation of market base transfers and improved water treatment technologies.

Managing St. Lawrence River Discharge in Times of Climatic Uncertainty: How Water Quantity Affects Wildlife, Recreation and the Economy

Christiane Hudon
St. Lawrence Centre, Environment Canada
Montreal, Quebec

Introduction

Large inland lakes with closed drainage basins are largely dependent upon the climatic regime for their water balance, both of which determine the seasonal and interannual variations in lake level. Climatic conditions and the resulting changes in water availability impact human activities; in turn, human adaptation to climate (especially under drought conditions) can exert significant feedback on water resources. This situation is exemplified by the major changes experienced by two of the largest inland lakes of the world. Increased water supply prompted a 9.4-foot (2.7-m) rise in the Caspian Sea level from 1978 to 1998 (more than 5 inches [0.135 m] per year). In contrast, persistent drought and water diversions for irrigation purposes generated a 66-foot (20-m) drop in the Aral Sea level between 1960 and 2000 (fewer than 20 inches [0.5 m] per year) (Jorgensen et al. 2003). As a consequence, the Aral Sea has lost over 75 percent of its original (prediversion) surface area of 26,255 square miles (68 000 km^2) and split into two basins in 1989 (Jorgensen et al. 2003). These two contrasting situations illustrate the vulnerability of inland lakes to climate and human interventions.

Although the situation in the North American Great Lakes differs in many respects, they are nevertheless subject to the same kinds of interactions between climate, hydrology and human activities—all of which bear consequences for aquatic ecosystems (Mortsch 1998, Schindler 2001). As the St. Lawrence River constitutes the major outlet of the Great Lakes through Lake Ontario, the hydrological regime experienced in the river is largely tied to the climatic conditions and human activities taking place in the upper part of its watershed—at the continental scale.

Growing evidence suggests that the climate of the Great Lakes-St. Lawrence region is already changing; winters are getting shorter, annual average temperatures are rising, the duration of lake ice cover is decreasing as air and water temperature rise, and heavy rainstorms are becoming more common (Kling et al. 2003). In the future, increased air temperature (by about 3.6° Fahrenheit [2° C]) and a longer growing season are expected; the resulting decline in ice cover duration and increased evaporation (12–17%) are expected to decrease Great Lakes levels by about 8 to 28 inches (0.2–0.7 m) (Lofgren et al. 2002) (Table 1). A recurrent water deficit in the Great Lakes could, in turn, reduce Lake Ontario discharge to the St. Lawrence River by up to 40 percent of its long-term average, with a concurrent 4.3 feet (1.3 m) decrease in mean water levels at Montreal (Mortsch and Quinn 1996).

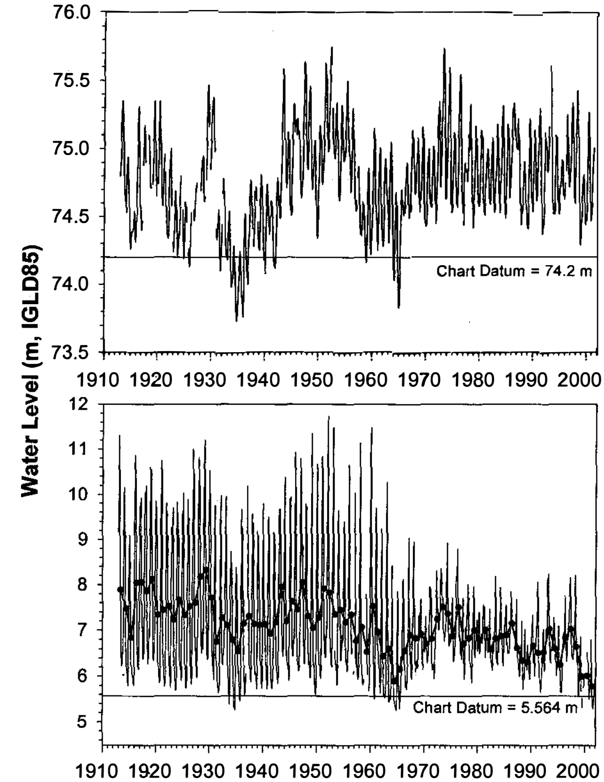
Table 1. Climate and level changes forecast for 2030 in the Laurentian Great Lakes by the Canadian Centre for Climate Modelling and Analysis (model CGCM1). Information adapted from Lofgren et al. (2002).

	Lake Superior	Lakes Michigan -Huron	Lake Erie	Lake Ontario
Difference in air temperature (°C)	+ 1.9	+ 2.2	+ 2.5	+2.1
Precipitation ratio	1.04	1.02	0.97	1.01
Difference in mean annual runoff (%)	- 5	-7 to -12	-23	-10
Difference in mean annual lake evaporation (%)	+ 17	+13 to + 15	+ 12	+ 12
Difference in mean annual lake level (m)	- 0.22	- 0.72	- 0.60	- 0.35

In the past, the Great Lakes level and the St. Lawrence River discharge have fluctuated between low (1930s, the mid-1960s and the late 1990s) and high (1970s and early 1980s) values, following the sequence of water supply to the basin (Figure 1, Changnon 1994). The low levels experienced in 1999 and 2001 in the St. Lawrence River were of the same magnitude as those recorded in the 1930s, which coincided with the Prairie Dust Bowl drought period (Rosenzweig and Hillel 1993). These conditions can be used as analog to identify some of the consequences of low-level conditions on human activities and on the environment in the St. Lawrence River.

Flow management in eastern North America is likely to become severely constrained in the future, as climate change scenarios come into play. In practical terms, understanding the feedback between human and environmental responses to climate change will help identify management practices that will minimize their negative interactions. This paper explores some of these complex feedback mechanisms and describes a study currently underway to conciliate the needs of

Figure 1. Long-term (1913–2002) monthly mean (fine line) water levels in Lake Ontario (Toronto, top panel) and in the St. Lawrence River (Montreal, bottom panel). Mean annual levels (full circles, bold line) are indicated for Montreal. Water levels (meters) are expressed in reference to the International Great Lakes Datum of 1985 (IGLD85); the level of reference of navigation charts (chart datum) is indicated for each location.



different interest groups while addressing the environmental impacts of level regulation in the Great Lakes-St. Lawrence River System.

The Great Lakes-St. Lawrence River System

The Great Lakes-St. Lawrence River system has a total drainage area of 297,500 square miles (770,500 km^2), 32 percent of which is the Great Lakes surface per se (95,000 square miles [246,000 km^2]), with a mean annual discharge from Lake Ontario to the St. Lawrence River of about 247,000 cubic feet per second (7,000 m^3/second)(40-year range from 173,000 to 378,000 cubic feet per second [4,900 to 10,700 m^3/second]) (Carpentier 2003). Given the large size of the lakes, it takes about three years for a significant variation in the discharge of Lake Superior to be observed in Lake Ontario, the last link in the chain of the Great Lakes before the outflow into the St. Lawrence River (Figure 2, Carpentier

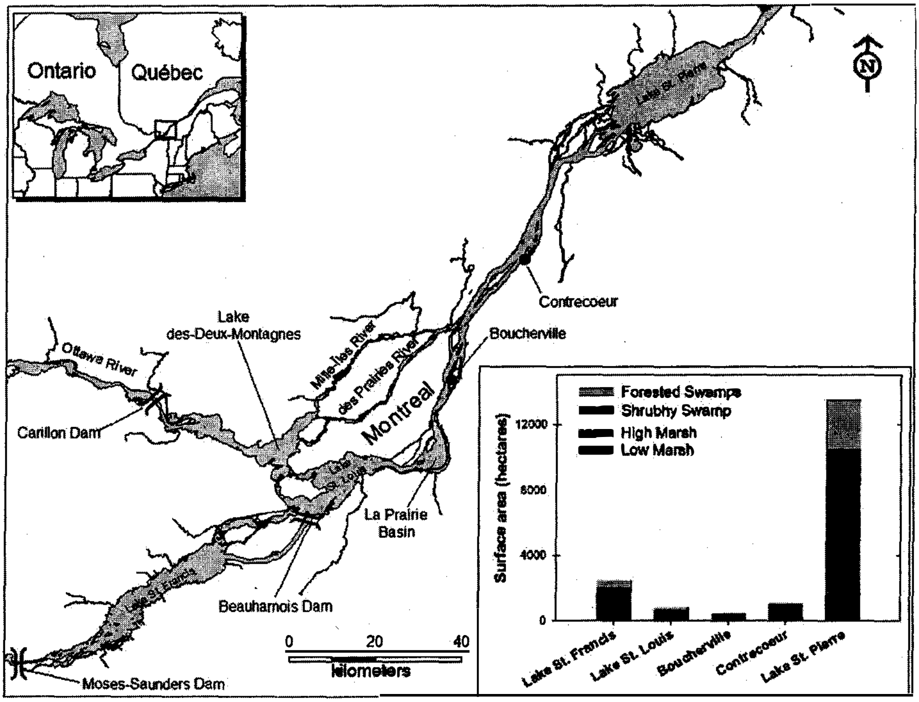


Figure 2. The Lower St. Lawrence River, flowing out of Lake Ontario and the International River section (upper left inset). River discharge is controlled at the Moses-Saunders Dam. The Montreal area is located at the confluence of the Ottawa and St. Lawrence rivers, both of which determine the hydrologic regime downstream to the outlet of Lake Saint-Pierre. The surface area (hectares) of swamps and marshes in five sectors along the longitudinal river axis (moving downstream) is shown (lower right inset)(Jean et al. 2002).

2003). Currently, Lake Ontario outflows are regulated by Plan 1958D (with diviations), based on criteria set by the International Joint Commission (IJC), a binational (Canada-United States) organization created in 1909 that is in charge of applying the Boundary Waters Treaty (International Joint Commission 1909). The criteria aim at maintaining hydroelectric power generation and commercial navigation while protecting the drinking water supply and shoreline properties from flooding (International Joint Commission 2004). The discharge of Lake Ontario into the St. Lawrence River at Cornwall, Ontario has been regulated since 1958 by the Moses-Saunders hydroelectric dam (Figure 2), which is jointly operated by the New York Power Authority and the Ontario Power Generation.

The portion of the St. Lawrence River located downstream of the control structure (hereafter designated the Lower St. Lawrence [LSL] study area) covers a 125-mile (200-km) section of the St. Lawrence River between Lake Saint-François and the outlet of Lake Saint-Pierre (Figure 2). In the Montreal area, the confluence of the Ottawa River (mean 1963–2002 discharge 67,000 cubic feet per second [1,900 m^3/second], ranging monthly from 24,000 to 230,000 cubic feet per second [670–6,500 m^3/second]) with the St. Lawrence River increases seasonal water-level variations in the downstream reaches of the river. Water-level variations in Lake des Deux Montagnes (mean annual range of about 6.6 feet [2 m]) reflect the Ottawa River discharge regime, which is regulated through the operation of several upstream reservoirs. Annual range in level is maximal (about 4 feet [1.4 m]) at Lake Saint-Pierre, owing to the additional influences of intervening tributaries and a small (less than 1 foot [0.3 m]) tidal effect.

Lower St. Lawrence Shoreline Morphology and Wetlands

The LSL alternates between wide (more than 3 miles [5 km]) and fairly shallow (mean depth less than 16 feet [5 m]) fluvial lakes and narrow (less than 2.5 miles [4 km]) corridors (Figure 2). The shores of Lake Saint-Louis, La Prairie basin and Montreal sectors are heavily urbanized (population of about 3 million people) and have been considerably altered by human activities, such as dredging and channelling for ship traffic, the creation of islands for the 1967 World Exhibition (better known as Expo '67) and the deepening of the Port of Montreal. The sector downstream of Montreal, including the islands of Boucherville and Contrecoeur to Lake Saint-Pierre, is more rural. Agricultural activities and shoreline erosion are probably the main factors responsible for the more recent loss of wetland habitats downstream of Montreal. Overall, about eighty percent of wetlands in the Greater Montreal area have been destroyed since the arrival of the first Europeans in the 16th century; what remains of the LSL wetlands is mostly concentrated along the shores of Lake Saint-Pierre (Figure 2, Jean et al. 2002).

The Lake Saint-Pierre Area: Paradise under Stress

With over 30,000 acres (12,000 ha) of swamps and marshes, Lake Saint-Pierre accounts for nearly 80 percent of LSL wetlands (Jean et al. 2002). Lake Saint-Pierre supports a large population of nesting blue herons (more than 1,300 nests), a major staging area for migratory wildfowl (more than 800,000 ducks and geese annually) and 167 species of nesting birds (St. Lawrence Centre 1996).

Permanently submerged areas, wetlands and the spring floodplain are home to 13 amphibian and 79 fish species, many of which are exploited by sports and commercial fisheries alike. The ecological value of Lake Saint-Pierre has been recognized by its status as a Ramsar Wetland, a UNESCO Biosphere Reserve and its inclusion as a protected site under the Eastern Habitat Joint Venture. Additional habitat protection is indirectly provided by its past (1952–2000) use as ballistic testing grounds for the Canadian Department of National Defence; access to the portion of the lake located south of the navigation channel (62 square miles [160 km^2]) is restricted, and activities involving contact with the lake bed are discouraged for safety reasons. The highly valuable character of Lake Saint-Pierre is thus a product of its morphological features, its abundant marshlands, its use by a highly diversified and abundant fauna, and its protected status.

Interactions Between Human Activities and the Environment under Climate Change Conditions: How Climate Change May Exacerbate Human Disturbances

Climate change brings a new challenge to the integrated, sustainable management of natural resources. Humans adapt to climate variations in many ways and, in so doing, exert (albeit unwittingly) important cumulative impacts on ecosystems. Aquatic ecosystems destabilized by human-induced stress show reduced resistance and resilience to the additional burden of climate variability. The following are some examples of the multiple interactions that already occur or may occur in the future between human adaptations to climate change and aquatic ecosystems, primarily resulting from low summer discharge, mild winter temperatures and a change in the timing of hydrological events (Figure 3). These examples are derived from observations in the LSL but are also relevant to many other large river systems from temperate regions.

Alteration of Summer Conditions

- Under persistent, low-level conditions, demands to dredge the navigation channel as well as the access channels to and from marinas and public launch sites will intensify. By concentrating the passage of water through the main channel, dredging reduces the over-bank flow and modifies the current and sedimentary regime in the littoral areas in which wetlands are found. Between 1854 and 1998, LSL channel depth was more than

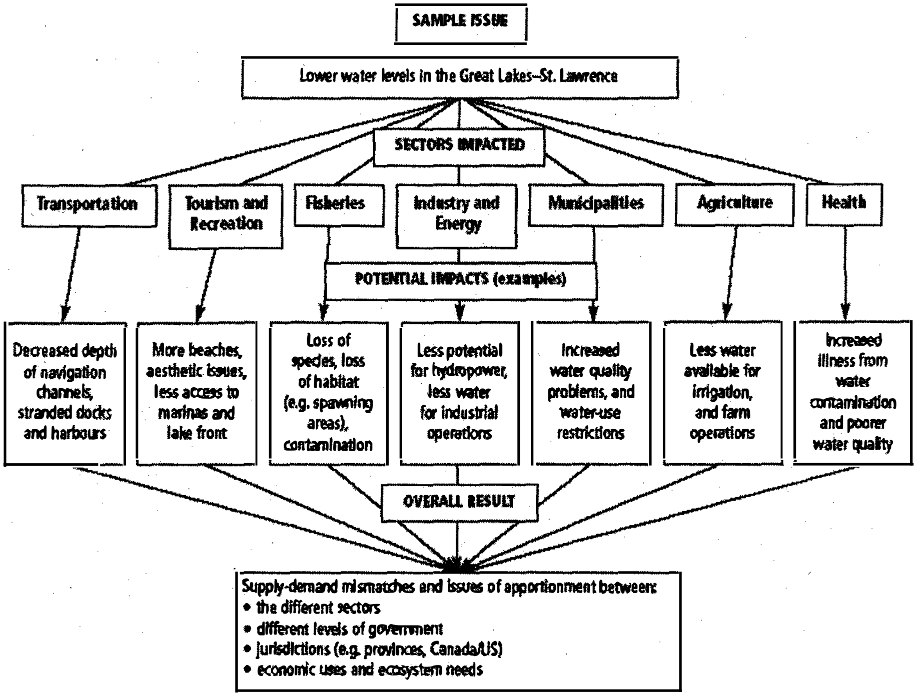


Figure 3. Some examples of the potential impacts of low water levels on different sectors and water users in the Great Lakes-St. Lawrence-Basin (from Natural Resources Canada 2002).

doubled (from 16 to 37 feet [4.9–11.3 m]) and width was tripled (from 246 to 804 feet [75–245 m]), resulting in a 750-percent increase in the cross-section of the navigation channel linking Montreal and Quebec. The sensitivity of St. Lawrence wetlands and the historical and seasonal water-level variations are well documented (Hudon 1997, 2004; Jean et al. 2002).

- Predictions of drier summers suggest that the pressure to increase water extraction for irrigation, drinking water and other human uses will grow within the basin, amplifying the already contentious debate over water withdrawals from the Great Lakes (Kling et al. 2003).
- The anticipated reduction in level and discharge dries out shallow littoral areas, which invites all encroachment, human and otherwise, on the newly (albeit temporarily) available river bed. Progressive shoreline development results in significant cumulative loss of wetland area over time.

- In 1999 and 2001, low-level conditions in LSL were somewhat alleviated by retaining water in Lake Ontario in the early part of the year (thus reducing the LSL spring discharge) to release it later (thus raising the level of the LSL in late summer and fall) to maintain commercial navigation and drinking water supply (Carpentier 2003). The practice of raising river levels for a few days was also used more frequently to allow the passage of high-tonnage ships under critically low-level conditions.
- Discharge reduction also raises the question of dilution of pollutants from domestic and industrial sources, especially with respect to the ecosystems and water uses (drinking water, pleasure boating, swimming) in areas located downstream of point sources.
- Historical (1970–2002) records show an increase in water temperature in the LSL, with significantly higher monthly average temperatures over years of low levels (1995, 1999, 2001) (Hudon et al. 2003). Warmer water temperatures will modify species composition and overall ecosystem productivity by favouring warm-water species at the expense of cold-water ones.
- Low level and discharge conditions can also favor the proliferation of certain invasive species. For example, increased propagation of purple loosestrife (*Lythrum salicaria*), reed canarygrass (*Phalaris arundinacea*) and common reed (*Phragmites australis*) was observed under low-level conditions (1995, 1999) (Hudon 2004). Low river discharge between June and August also coincided with a higher zebra mussel (*Dreissenia polymorpha*) colonization rate (de Lafontaine 2002).

Alteration of Winter Conditions

- Milder winters increase the probability that plant and animal species that are currently found in more southern regions will establish themselves in the LSL basin. These species may be brought there through activities, such as commercial navigation (ballast waters), aquaculture (production for sports and food fisheries) and recreation (gardening, live fish bait, pleasure boats, aquarium fish). Some of these species may proliferate and exert major effects on aquatic ecosystems that are already under climatic stress (Kling et al. 2003).

- Although summer stream flows are generally expected to decline, many researchers project a corresponding increase in winter flows. This is because warmer winters would reduce the duration of ice cover (Magnuson et al. 2000) and increase the frequency of midwinter thaws and rain-on-snow events. This, in turn, would increase the risk of winter flooding in many regions as a result of the production of slush ice, high flows and severe ice jams (Prowse and Beltaos 2002).
- Since snow accumulation will likely be reduced by frequent, small melt events throughout the winter, the magnitude of spring flooding will likely decline (Natural Resources Canada 2002). These events will likely alter the timing of seasonal flow variations, to which aquatic organisms may or may not be able to adapt their life cycle. For example, water level and temperature conditions were shown to modify the availability of suitable spawning grounds (Casselman and Lewis 1996) and to determine year-class strength (Casselman 2002) for various fish species.
- Ice jams in LSL have been partially controlled since about 1910 to reduce flooding. Since 1958, an open-water channel has opened the port of Montreal to winter shipping traffic. Early studies of LSL wetlands identified winter floods and ice scouring of the bottom as major factors in maintaining patchiness of emergent vegetation (Marie-Victorin 1934), also contributing to the long-range dispersal of wetland plants (Dansereau 1945).

Alteration of the Timing of Hydrological Events

- Earlier ice break-up and earlier peaks in spring runoff will change the timing of river flow, and increases in heavy rainstorms may cause more frequent flooding and erosion (Kling et al. 2003). Periods of high flow return the river to its original bed, flooding and eroding properties located in areas borrowed earlier (occasionally decades ago) from the floodplain. Such events, in turn, generate a public outcry for more stringent discharge control and stimulate measures to further modify the shoreline (additional landfill, artificial shorelines, rock, blocks, protection walls), in the hope of preventing future damages. Eighty percent of the LSL shorelines in the Greater Montreal area have been modified and urbanized, with the concurrent elimination of riparian wetlands (Figure 2)(Jean et al. 2002).

- Discharge regulation modifies river flow patterns by reducing the amplitude and duration of spring runoff and raising low water levels in late summer, thereby decreasing the vertical range of seasonal variations in LSL (Hudon 1997). Stabilization of the mean annual level of Lake Ontario has eliminated the decadal-scale variations in water level, which have also reduced its overall vertical range (Figure 1). Maintenance of diversified wetlands requires seasonal and interannual water-level variations (Toner and Keddy 1997).
- Modification of the flow regime of tributaries by periodic heavy rain that generates flash flood events could alter water quality, sedimentation rates and biological production (eutrophication) downstream of their confluence with the LSL. This is currently the case with the Yamaska, Saint-François and Richelieu rivers, all of which drain heavily farmed areas directly into Lake Saint-Pierre. The severity of problems will likely vary locally, depending upon future changes in water quality (suspended solids, nutrients, pesticides) in tributaries draining farmlands and upon the dilution or mixing capacity in the LSL.
- Change in the patterns of tributary discharge (alternating between low flow and abrupt rise in discharge following storm events) will likely modify the pattern of particle deposition in wetlands. Decreased runoff from the land, particularly in summer, decreases the transport of material from uplands to wetlands. The material that does enter wetlands is retained longer before high-water pulses flush it downstream (Kling et al. 2003), contributing to the infilling of shallow areas, which become progressively more terrestrial. Since the low-water-level episodes of 1999 and 2001, willow swamps have colonized previously submerged areas of Lake Saint-Pierre, immediately downstream of the Richelieu, Yamaska and Saint-François rivers.

Since its formation about 10,000 years ago, the LSL has withstood large variations in climate and hydrology, which have resulted in the current assemblage of plants and animals. Over the last five centuries, European colonization and population growth have increasingly modified land use in the watershed, river shoreline and riverbed. In the next decades, climatic variability will impose even bigger changes upon organisms whose life cycles have evolved over millennia. The above examples reveal that human intervention plays an important role in cumulative anthropogenic effects on natural ecosystems. Management of human adaptation can either amplify or reduce the effects of climate change on ecosystems and can be used to prevent or to alleviate some of the concurrent anthropogenic impacts.

Managing Water Flow Under Climatic Uncertainties: Looking for Solutions

Drinking water supply, wastewater flushing and epuration, hydroelectric production, maritime shipping and recreation are only a few of the numerous services the St. Lawrence River provides to its population. Until recently, development was carried out without regard for its effects on the environment, with consequences on water quality, wetland habitats and aquatic fauna. Binational initiatives, such as the Great Lakes Water Quality Agreement (1971), put a strong focus on water quality and the reduction of toxics, implemented through the Great Lakes Action Plan (1971–2005). While water quality was also at the center stage of the first two phases (1988–1997) of the St. Lawrence Action Plan, some of the effects of water-level variations on St. Lawrence River ecosystem components were investigated during its third phase (1998–2003).

The International Joint Commission and the Lake Ontario-St. Lawrence Water Level Study

The Lake Ontario-St. Lawrence Water Level Study was set in motion in 2000 by the IJC to identify flow regulation criteria that best serve the wide range of user groups and are widely accepted, while accounting for future changes in water supply resulting from climate change (International Joint Commission 2004). In particular, the new regulation plan must be environmentally sustainable and respect the integrity of the Lake Ontario-St. Lawrence River ecosystem, in addition to the satisfaction of various interest groups (recreational boating, hydroelectric production, commercial navigation, shoreline property and drinking water supply). The five-year study (2000–2005) aims to identify and assess how changes to current Lake Ontario water-level regulation will affect the interests of various users, while ensuring that any suggested changes are consistent with relevant treaties and agreements between Canada and the United States. The study does not, however, examine structural changes to the existing authorized control works that make Lake Ontario outflow regulation possible. Rather, priority is on the identification of other measures to alleviate the adverse impacts of water level and flow fluctuations.

How the Study Is Organized

The study team comprises experts and decision makers from government, academia, aboriginal communities and other groups from the United States and Canada. Scientific and technical work is carried out by nine technical working groups (TWGs), whose work plans are approved by a binational study board that reports to the IJC (Figure 4). In addition, the Public Interest Advisory Group (PIAG) is a consultative body, which represents public concerns and interacts with the IJC, the study board and TWGs. All groups are cochaired jointly and include members from the United States and Canada.

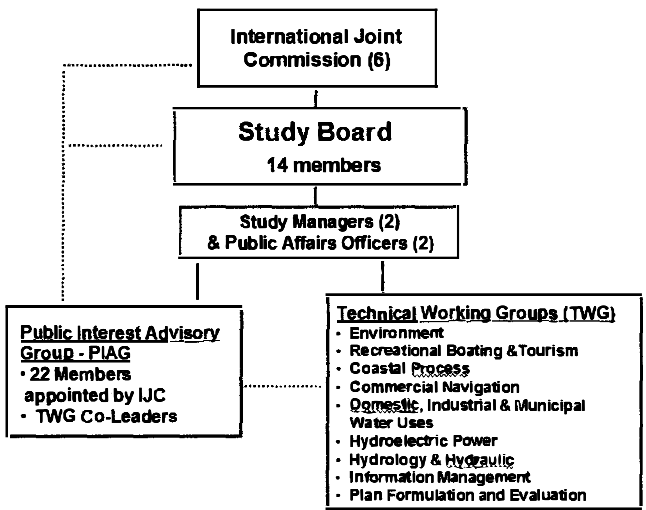


Figure 4. Organization of the Lake Ontario-St. Lawrence Water Level Study. Operational (full lines) and consultative (dotted lines) functions between the different groups are identified. Membership of each group (number of persons) is divided equally between Canadian and U. S. representatives.

- Environmental Training Working Group (ETWG) investigates the impacts of water level variations on fish, birds, rare and endangered species, and other wildlife in the system, with a particular focus on the ecological effects on wetlands (see below).
- Recreational Boating and Tourism investigates the impacts of water levels on individual boaters, marinas and tourism.
- Coastal Zone investigates the impacts of water level fluctuations on shoreline property, with particular attention to erosion and flood processes.

- Commercial Navigation investigates the impacts of water levels on cargo shipping, cruise and tour operations, tug and barge operations, ship construction, and government vessel operations.
- Water Uses investigates impacts of water level variations on industrial, municipal and domestic water intakes and treatment facilities.
- Hydroelectric Power Generation evaluates how different regulation plans affect power generation.
- Common Data Needs collects and updates information on depths and elevations (bathymetric and topographic data) in critical areas of the system and shares findings with other work groups.
- Hydrology and Hydraulics Modeling develops models to predict water levels and flows in the system based on different regulation plans and climate change scenarios.
- Plan Formulation and Evaluation Group (PFEG) is in charge of integrating the results (performance indicators documenting the set of most desirable hydrological variables for each user group) originating from all TWGs into criteria and alternative regulation plans.

Advice to the board and to the TWGs is provided by the PIAG, which is not a TWG, but a group of 24 volunteers working to ensure effective communication between the study team and the public. PIAG is developing and implementing a public awareness program and is liaising with TWGs to ensure that input from the public is heard and that the study's goals and activities are publicized. In addition to the technical issues faced by the study, effective communication and exchange of information between the study board, the nine TWGs, the PIAG and the various interest groups remain major challenges.

The Environment Technical Working Group

The members of ETWG originate from Canadian and United States federal and provincial or state government agencies as well as universities and private consulting firms. They carry out research projects dealing with various components of the environment in Lake Ontario and the upper and lower St. Lawrence River. The first three years of the study (2000–2003) focused primarily on data gathering in order to quantify relationships between hydrological components, wetlands and various faunal groups (fish, amphibians, reptiles, wildfowl, palustrine birds, muskrat). A small number of rare and endangered species of plants and animals specifically affected by the hydrological regime were also investigated. Depending on the particular life history phase (nesting, spawning, juvenile rearing, adult feeding, overwintering, etc.) of the organism under study, different aspects of the hydrological cycle (level or discharge) may be critical (magnitude, duration, timing, frequency, rate of change, interannual sequence), as previously described by Poff et al. (1997).

Future Steps

This year (2004), results from the ETWG as well as those from all other TWGs will be integrated into the shared vision model, developed by the PFEG, to develop new regulation criteria. Different sets of criteria will be used to develop alternatives to the current regulation (1958D with deviations), which, in turn, will be assessed on the basis of how well they ensure environmental integrity and fulfill the needs of all interest groups, upstream and downstream, of the Moses-Saunders Dam. During the final year of the study (2005), the study board will submit a small number of potentially acceptable regulation plans to the IJC commissioners, who will carry out additional consultations prior to recommending a preferred plan to their respective governments. The new regulation plan will likely involve some degree of compromise and difficult choices from all user groups, but it will not result in disproportionate loss to any interest group or any region.

Conclusion

Since 1960, human interventions in the LSL, such as water-level regulation, reduction of ice scouring, control of ice jams and reduction of spring floods, have markedly decreased the within-year range of water levels but increased their between-year variability. Whereas high water levels in Lake Ontario caused concern in the 1970s and 1980s, the 2000s may bring more frequent periods of low water levels in the Great Lakes. Although managing extremes is always a challenge, managing drought conditions may prove to be the biggest challenge of all, since drought exacerbates human pressures, thus imposing the need to weigh economic interests against other concerns. The strong response of St. Lawrence River wetlands to water-level changes and the effects on the quality of faunal habitats make them highly vulnerable to the cumulative impacts of future climate change and human interventions.

Reference List

Carpentier, André. 2003. La régularisation du Saint-Laurent. *Le Naturaliste Canadien*. 127:102–13.

Casselman, John M., and C. A. Lewis. 1996. Habitat requirements of northern pike (*Esox lucius*). *Canadian Journal of Fisheries and Aquatic Science*. 53 (supplement 1):161–74.

Casselman, John M. 2002. Effects of temperature, global extremes, and climate change on year-class production of warm water, cool water, and coldwater fishes in the Great Lakes Basin. *American Fisheries Society Symposium*. 32:39–60.

Changnon, Stanley A., ed. 1994. *Lake Michigan diversion at Chicago and urban drought: Past, present and future regional impacts and responses to global cimate change, final report NOAA contract No. 50WCNR306047*. Ann Arbor, Michigan: Great Lakes Environmental Research Laboratory.

Dansereau, Pierre. 1945. Essai de corr élation sociologique entre les plantes supé rieures et les poissons de la beine du Lac Saint-Louis. *Revue Canadienne de Biologie*. 4:369–417.

de Lafontaine, Yves. 2002. *Impacts des fluctuations de débit d'eau douce sur la colonisation annuelle par les moules zébrées dans le fleuve Saint-Laurent. Report submitted to the International Joint Commission*. Ottawa, Ontario: Lake Ontario-St. Lawrence Water Level Study.

Environment Canada. 2004. *Our Great Lakes*. http://www.on.ec.gc.ca/water/greatlakes/action-plan-e.html.

Hudon, Christiane. 1997. Impact of water level fluctuations on St. Lawrence River aquatic vegetation. *Canadian Journal of Fisheries and Aquatic Science*. 54:2,853–65.

———. 2004. Shift in wetland composition and biomass following low-level episodes in the St. Lawrence River: Looking into the future. *Canadian Journal of Fisheries and Aquatic Science*. 61:603–17

Hudon, Christiane, A. Patoine, and A. Armellin. 2003. *Water temperature variability in the St. Lawrence River near Montreal, report submitted to the International Joint Commission*. Ottawa, Ontario: Lake Ontario-St. Lawrence Water Level Study.

International Joint Commission. 1909. *Boundry waters treaty: Treaty between the United States and Great Britain relating to boundary states, and questions arising between the United States and Canada.* http://www.ijc.org/rel/agree/water.html.

———. 2004. *International Lake Ontario-St. Lawrence River Water Level Study.* http://www.losl.org/about/about-e.html.

Jean, Martin, G. Létourneau, C. Lavoie, and F. Delisle. 2002. *Freshwater wetlands and exotic plant species.* Sainte-Foy, Canada. Bureau de coordination de Saint-Laurent. http://www.slv2000.qe.ca/plan_action/phase3/biodiversite/suivi_ecosysteme/fiches/Milieux_humides_a.pdf.

Jorgensen, S. E., R. de Bernardi, T. J. Ballatore, and V. S. Muhandiki. 2003. *Lake watch 2003: The changing state of the world's lakes.* Kusatsu, Japan: International Lake Environment Committee Foundation.

Kling, G. W., K. Hayhoe, L. B. Johnson, J. J. Magnuson, S. Polasky, S. K. Robinson, B. J. Shuter, M. W. Wander, D. J. Wuebbles, D. R. Zak, R. L. Lindroth, S. C. Moser, and M. L. Wilson. 2003. *Confronting climate change in the Great Lakes region, impacts on our communities and ecosystems.* Cambridge, Massachusetts/Washington, DC: Union of Concerned Scientists/ Ecological Society of America.

Lofgren, Brent M., F. H. Quinn, A. H., R. A. Assel, A. J. Eberhardt, and C. L. Luukkonen. 2002. Evaluation of potential impacts on Great Lakes water resources based on climate scenarios of two GCMs. *Journal of Great Lakes Research.* 28:537–54.

Magnuson, J. J., K. E. Webster, A. H. Clites, R. A. Assel, C. J. Bowser, J. G. Eaton, H. E. Evans, E. J. Fee, R. I. Hall, L. R. Mortsch, D. W. Schindler, and F. H. Quinn. 1997. Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian shield region. *Hydrological Processes.* 11:825–71.

Magnuson, J. J., D. M. Robertson, B. J. Benson, R. H. Wynne, D. M. Livingstone, T. Arai, R. A. Assel, R. G. Barry, V. Card, E. Kuusisto, N. G. Granin, T. D. Prowse, K. M. Stewart, and V. S. Vuglinski. 2000. Historical trends in lake and river ice cover in the northern hemisphere. *Science.* 289:1,743–6.

Marie-Victorin, Fr. 1934. Le fleuve Saint-Laurent, milieu biologique pour les plantes vasculaires. *Transactions of the Royal Society of Canada.* V(III):28.

Mortsch, Linda D. 1998. Assessing the impact of climate change on the Great Lakes shoreline wetlands. *Climate Change*. 40:391–416.

Mortsch, Linda D., and F. H. Quinn. 1996. Climate change scenarios for Great Lakes Basin ecosystem studies. *Limnology and Oceanography*. 41:903–11.

Natural Resources Canada. 2002. *Climate change impacts and adaptation: A Canadian perspective*. Water Resources, Climate Change Impacts and Adaptations Directorate. http://adaptation.nrcan.gc.ca/perspective.asp

Poff, N. Leroy, J. D. Allan, M. Bain, J. Karr, K. Prestegaard, B. Richter, R. Sparks, and J. Stromberg. 1997. The natural flow regime. *Bioscience*. 47:769–84.

Prowse, T., and S. Beltaos. 2002. Climatic control of river-ice hydrology: A review. *Hydrological Processes*. 16:805–22.

Ramsar Convention on Wetlands, The. 2004. *Ramsar convention on wetlands*. http://www.ramsar.org.

Rosenzweig, C., and D. Hillel. 1993. The dust bowl of the 1930s: Analog of greenhouse effect in the Great Plains? *Journal of Environmental Quality*. 22:9–22.

Schindler, David W. 2001. The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. *Canadian Journal of Fisheries and Aquatic Science*. 58:18–29.

St. Lawrence Centre. 1996. *State of the environment report on the St. Lawrence River, volumes 1 and 2, St. Lawrence update series*. Quebec, Montreal: Environment Canada-Conservation Branch and ditionsMultiMondes.

Saint Laurent Vision 2000. 2004. *Saint Laurent Vision 2000*. http://www.slv2000.qc.ca

Toner, Maureen, and P. A. Keddy. 1997. River hydrology and riparian wetlands: A predictive model for ecological assembly. *Ecological Application*. 7:236–46.

United Nations Educational, Scientific and Cultural Organization. 2004. *Programme on man and the biosphere*. http://www.unesco/mab.

Our Water Resources: A Candidate for Listing? The Blackfoot Experience

Stan Bradshaw
Trout Unlimited, Montana Water Project
Helena, Montana

Introduction

The Blackfoot River arises near the continental divide and runs west for 132 miles (212 km) to its confluence with the Clark Fork River near Missoula, Montana. It was part of the route home for Meriwether Lewis and William Clark in 1805. For much of its modern history, it was known as a scenic river with great fishing (Paul Roos, personal communication 2003). It maintained this reputation until the early 1970s, when its fortunes began to change.

The Blackfoot watershed has 1,900 miles (3,056 km) of perennial streams, (Neudecker 1999) Landownership in the valley is 44 percent U. S. Forest Service (USFS), 5 percent U. S. Bureau of Land Management (BLM), 7 percent Montana, 20 percent Plum Creek timberland and 24 percent private ownership (Neudecker 1999). The individual ownership is mostly large cattle ranches.

The Decline of the Blackfoot

During the 1970s, two things, seemingly unrelated, happened that marked a significant change the fortunes of the river's fishery. First, in 1975, a tailings dam (which withholds waste from industrial projects) burst in the headwaters (Stiller 2000). Second, in 1979, the Montana Department of Fish and Game Commission formally adopted a policy of wild trout management in Montana's rivers and streams; no longer would catchable-sized hatchery fish be dumped into rivers or streams in Montana.

The Failure of a Tailings Dam

On June 20, 1975, after a three-day rainstorm, a tailings dam on Little Beartrap Creek in the headwaters of the Blackfoot River failed, flushing approximately 100,000 tons (90,718,474 kg) of toxic, metal-laden tailings from the

Mike Horse Mine into the Blackfoot River (Stiller 2000). For at least 10 miles (16 km) downstream, trout populations declined by 80 percent (Montana Department of Fish and Game 1997). Macroivertebrates declined by 65 to 85 percent. And, habitat for many miles downstream was embedded with fine sediment, choking off habitat for macroinvertebrates and destroying spawning habitat (Stiller 2000). The dam failure largely destroyed what had been a decent cutthroat fishery above Landers Fork (Paul Roos, personal communication 2003).

Montana's Wild Trout Management

In 1974, after a multiyear experiment on the Madison River, the Montana Department of Fish, Wildlife and Parks embarked on a policy not to plant hatchery-reared fish in Montana's rivers and streams (Tom Palmer, personal communication 2004). Under the auspices of this policy, trout in Montana's rivers and streams would be allowed to naturally reproduce without interference from the annual invasion of catchable-size hatchery fish that had for decades been the mainstay of the fishery. A number of things resulted from the decision to use wild trout management. For example, instead of broad-brush, generous bag limits, the 2002 to 2003 Montana fishing regulations are much more detailed and site-specific.

But, the most important result of the wild trout management policy is that fisheries managers became much more sensitive to habitat. Without the crutch of put-and-take fisheries, fish populations are a strong barometer of habitat health.

In the Blackfoot River, the stocking of hatchery fish ceased in 1979 (Peters and Spoon 1989). Within just a few years, there was a growing chorus from anglers and outfitters that the fishing in the middle and lower reaches had severely declined. But, there had been little fisheries census work done on the Blackfoot River since the early 1970s. So, beyond the anecdotal evidence offered by disgruntled anglers, there was nothing to document the problem. In the minds of the public, at least, the Blackfoot River was in trouble.

The Road to Restoration

Ironically, it was another mining-related event that served as a catalyst to the work necessary to identify and address the problem. In 1986, the Sunshine Mining Company proposed the construction of an open-pit gold mine, within a few hundred yards of the Blackfoot River that was a few miles downstream from

Lincoln, Montana. People up and down the valley, with vivid memories of the 1975 disaster, were alarmed at the prospect of a large mining operation so close to the river (Paul Roos, personal communication 2003). The concerns came from a diverse collection of valley residents—anglers, outfitters, ranchers, merchants whose livelihood depended on a robust tourist trade—and from and unusual collection of environmental activists. In 1986, there was no advocacy group that had the Blackfoot as its focus, and a number of the active opponents of the mine felt that they needed a group identity or an affiliation to most effectively address the mining proposal. (Becky Garland, personal communication 2003). After some deliberation, people decided to form a Trout Unlimited chapter.

In late 1987, the Big Blackfoot Chapter of Trout Unlimited (BBCTU) had its charter meeting at a residence close to the site of the proposed new mine. The U. S. Fish and Wildlife Service's (USFWS's) Fish and Wildlife Program (FWP) regional fisheries manager responsible for the Blackfoot River attended the meeting. When questioned on the state of the fishery, he answered that FWP had done no recent population work on the Blackfoot River and, therefore, had no information on the state of the fishery (Dennis Workman, personal communication 2003). Worse, his office had limited funding for fieldwork and was putting its effort into the Bitterroot and Rock creeks. In short, the agency wasn't planning to do anything anytime soon. When questioned as to how much money was needed to initiate the fieldwork, he indicated that approximately $15,000 would cover a season of inventory work. Within a few weeks, the chapter raised the money and presented FWP with a check and a request to get started in 1988 (Paul Roos, personal communication 2004).

In 1988, FWP completed the research and, in 1989, published the report of its findings. The findings largely vindicated the apprehensions of the public (Peters and Spoon 1989). The report concluded as follows: "Trout populations were below expected levels in virtually all reaches sampled . . . . Populations of native trout species, cutthroat and bull trout, of the Blackfoot River appear to be particularly threatened" (Peters and Spoon 1989:44). The report also concluded that, when compared to nearby waters of similar stature, Rock Creek and the Bitterroot rivers, the Blackfoot River fared poorly. To its credit, FWP, in the face of this information, committed to a two-year investigation of the Blackfoot River to refine the cause of this decline. BBCTU assisted with funding of that effort (Neudecker 1999). Over the next two years, FWP initiated extensive surveys and habitat assessments from the north fork of the Blackfoot River to the mouth that pinpointed the cause and location of habitat impairment within the basin.

The survey identified the usual suspects of fisheries habitat damage in the West—impacts from historic mining activities (acid mine drainage), logging (sedimentation from road construction, removal of key streamside vegetation, fish migration blocked by culverts), livestock grazing (riparian damage from overgrazing) and agricultural activities (streamflow depletion from irrigation and stock watering, fish migration blocked by irrigation diversions and entrainment of fish into irrigation ditches) (Neudecker 1999).

This work also identified the tributaries to the Blackfoot River as the areas most severely impaired. Although the main stem of the Blackfoot River had habitat problems, in most cases, those problems had a tributary link and habitat loss to fine sediment from damaged tributaries.

Restoration of the Fishery

In the face of this information, the BBCTU undertook, as its primary mission, the restoration of the Blackfoot River Trout Fishery. (Neudecker 1999) One of the chapter's first acts was to develop a cooperative agreement with the USFWS, through its Partners for Fish and Wildlife Program, to work on the restoration of the Blackfoot River Trout Fishery. In addition, it developed a close working arrangement with FWP. Working with these partners, BBCTU developed a two-pronged restoration strategy: (1) protect the two native trout species—specifically bull trout (*Salvilinus confluentus*) and westslope cutthroat trout (*Onchorhynchus clarki lewisi*) with catch-and-release fishing regulations throughout the drainage—and (2) work with private landowners to restore habitat in the impaired tributary streams (Neudecker 1999). The first prong was simple, and, in 1990, the Montana Fish and Game Commission promulgated a catch-and-release regulation for bull trout and cutthroat trout.

The second part of the strategy was problematic. One of the fundamental challenges of cooperative stream restoration work is to develop enough trust with the landowners to even get started on restoration. Developing that relationship can be a slow, uncertain process, especially when a conservation group initiates the project. Too often, rural landowners, schooled to distrust outside groups that want to tell them how to do their business, are notoriously difficult to approach about modifying their behavior (Greg Neudecker, personal communication 2004). But, unlike many Trout Unlimited chapters, the membership of which is almost exclusively made up of anglers, the BBCTU had an advantage on the trust issue. At the time of its creation, nearly half of the board members didn't even fish. Likewise, nearly half the members came from a ranching background (Paul Roos, personal communication 2003). As a result, there were people on the BBCTU board who, by dint of their agricultural background, could dispel distrust with their neighbors.

Most importantly, the rancher members of BBCTU's board were widely respected in the valley. For example, Land Lindbergh, who ranched in the valley since the early 1960s, was widely regarded for his ability to mediate between diverse interests in the valley and for his commitment to maintaining the rural, agricultural character of the valley (Coughlin et al. 1999). Likewise, Jim Stone—another early board member, a lifetime resident of the valley and a rancher—is the chairman of the Blackfoot Challenge.

In addition, the two agency partners, USFWS and FWP, had people with the fortuitous mix of both good biological talent and exceptional people skills. Don Peters, the author of the 1989 inventory report and the biologist assigned to the Blackfoot River, realized that he would have to swallow his indignation and learn how to work with the landowners; as a biologist, he deplored many of the practices that caused fishing problems. He had to reinvent himself, from being simply a biologist to being a biologist/diplomat (Don Peters, personal communication 2003). Likewise, Greg Neudecker of USFWS's Partners in Wildlife Program, recognized early that he would get a lot more done in the long-run if he talked less and listened more at the front end of the process (Greg Neudecker, personal communication 2003).

Neudecker's experience in the Blackfoot River now informs USFWS training of new employees into the Partners in Wildlife Program in Montana. Now, new employees can expect about two years of cooperator contact in which the USFWS employees will largely be listening and learning; they won't be making suggestions for how landowners should behave or what they should do to their property (a common complaint about federal employees in general and USFWS employees in particular) (Greg Neudecker, personal communication 2004). Neudecker takes his restoration approach a step further by characterizing any successful restoration effort as equally based in science and art. On the science side of the equation, it is important to gather enough data to identify the proper candidate site for restoration—one that is susceptible to rehabilitation. On the art side, it is important to hire or involve people who have more than simply scientific competence; they must also know how to engage people whose land will host the project. Another part of the art is to assure that, as much as possible, the effort is community-based. To that end, Neudecker worked hard in the Blackfoot River to identify key people in the community and to engage them in his work (Coughlin et al. 1999).

With this lucky combination of personalities, talents and approach, the partnership was ready to proceed. In 1990, FWP prioritized tributary streams in the lower basin for restoration, based on their importance to native trout and on their potential for contributing to the fishery of the main stem of the Blackfoot River (Neudecker 1999). That same year, BBCTU embarked upon its first restoration projects. These projects focused on four areas-restoring instream habitat restoration, enhancing instream flows, addressing fish passage barriers and reducing the entrainment of fish into irrigation ditches.

In order to gain some traction and trust, BBCTU and its partners started with small, relatively simple projects that had a high likelihood of success-willow plantings and riparian fences, for example (Neudecker, personal communication 2003). But, BBCTU recognized the importance of finding a showcase project that could expand local interest in the chapter's restoration efforts. In 1992, it found one in the lower reaches of Rock Creek, a small tributary to the north fork of the Blackfoot River that had been severely degraded by decades of livestock use. The lower 1.5 miles (2.4 km) is effectively a spring creek, receiving most of its flow from groundwater discharge (Pierce and Podner 2000). Before the restoration, it was wide, shallow, warm and supportive of few fish. The restoration effort included the removal of six barriers to fish passage, the installation of more efficient diversion structures, the conversion from flood to sprinkler irrigation, and restoration of the stream habitat by significantly narrowing the channel by increasing woody debris, and planting riparian shrubs over the entire 1.8 miles (2.9 km) of the reach (Pierce et al. 1997). In addition, once the channel work was done, extensive planting of riparian vegetation was necessary to maintain the channel configuration. By 1994, within two years of the completion of the restoration work, young brown trout populations had increased almost sevenfold. Because this project was within sight of Highway 200, it received considerable scrutiny, and it became a frequently visited demonstration site (Don Peters, personal communication 2003).

In the wake of this successful project, interest in BBCTU's restoration efforts grew, and it proliferated to the extent that, by 2001, fish screens had been installed on diversions in 12 streams, fish passage structures had been erected on

26 streams, grazing management improvements had been installed on 23 streams, restoration of riparian vegetation had been completed on 27 streams, streamflow improvements had been completed on 25 streams and feedlots had been removed from 12 streams.

Some of the most straightforward projects have had the most dramatic results. In August 1994, the highest catch per unit effort of bull trout in the entire Blackfoot River drainage occurred in the highest upstream irrigation canal on the north fork of the Blackfoot River (Pierce et al. 1997). In 1989, biologists counting bull trout redds (redds) on the north fork found seven. Between 1994 and 1996, screens were installed on the five canals on the north fork. By 2000, redd counts on the north fork had risen to 140 (Pierce and Podner 2002).

The effect of the overall restoration effort on the native fish populations has been dramatic. In two core bull trout spawning and rearing streams, Monture Creek and the north fork of the Blackfoot River, combined redd counts have increased from a low of 18 in 1989 to over 200 in 2001 (Pierce and Podner 2002). Westslope cutthroat densities on two reference reaches of the main stem increased 923 percent and 758 percent, respectively, between 1989 and 2000 (Pierce and Podner 2001). In addition, westslope cutthroat populations have significantly increased on several tributaries (Pierce and Podner 2000).

There have been other positive spinoffs from this work as well. In the mid-1990s, BBCTU realized that many of the state and federal agency people with some understanding to work within stream environments had little understanding of stream morphology. BBCTU arranged an intensive, week-long field seminar in geomorphology with Dave Rosgen, a highly regarded geomorphologist. (Ron Pierce, personal communication 2001). Forty people attended that seminar, and the seminars are now an annual event (Hinson 2002).

With the turn of a new century, BBCTU and its partners have turned their attention to the upper Blackfoot River watershed. To be sure, the work is not finished in the lower basin. Restoration efforts continue, and an aggressive monitoring effort is underway to track the results of the restoration efforts to date (Pierce and Podner 2001).

In 2000, to direct its work, Department of Fish, Wildlife and Parks (DFWP), in collaboration with BBCTU and the USFWS, established restoration priorities for the 88 tributary streams in the Blackfoot River Basin. Of these 88 streams, baseline work has indicated that 83 streams suffer from some kind of habitat impairment. DFWP has ranked the tributary streams in priority, based on biological and resource benefits (150 possible points) and based on social and financial considerations (50 possible points) (Hess 2003).

The Role of Monitoring in Restoration Effort

From its inception, the collection of population and habitat data, both baseline and post-project monitoring, has been paramount to the success of the fisheries restoration efforts. The data gathering process, beginning with the 1988 inventory, has grown more intensive and more focused each year (Ron Pierce, personal communication 2004). From the time of the earliest habitat work, there has been a parallel track of habitat and biological data collection. As restoration projects are completed, monitoring of habitat and biological response to the effort begins (Ron Pierce, personal communication 2004). And, as restoration work progressed in the lower basin throughout the late 1900s, baseline population and habitat research began in the upper basin (Pierce and Podner 2002).

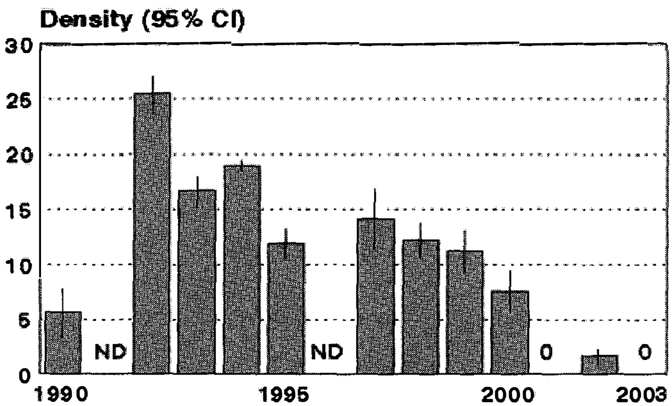
As the work has expanded, so has the cost of the work. Funding the research and the ongoing monitoring has been one of the greatest challenges facing FWP, which is responsible for most of the work (Pierce, personal communication 2004). Funding the work, like many other parts of this restoration effort, has been a collaborative undertaking. In addition to FWP, BBCTU, the North Powell Conservation District, the USFWS, the Blackfoot Challenge, and a number of other public and private entities have contributed funding to data gathering effort (Pierce et al. 2004).

In 1986, Kai Lee, a member of the Northwest Power Planning Council (NWPPC), enunciated a direction for the Columbia Basin Fish and Wildlife Program of the NWPPC grounded in the principals of adaptive management (Lee et al. 1986). Lee described adaptive management at its essence as learning by doing: "Adaptive management, as a strategy for implementation, provides a framework within which measures can be evaluated systematically as they are carried out." (Lee and Lawrence 1986). While Lee's strategy was focused on anadromous fish, it may have found its clearest expression in practice in the Blackfoot River. Pierce, responsible for most of the monitoring effort in the Blackfoot River, characterizes the restoration effort as iterative restoration, "Restoration is also iterative and relies on continued habitat and population monitoring, expanding the scope and modifying methods of restoration based on monitoring results" (Pierce et al. 2004).

Typically, there appears to be an institutional bias among many natural resource agencies against funding monitoring ongoing research (Ron Pierce, personal communication 2004). Yet, those involved in the restoration effort are emphatic that the data gathering, intensive as it is, is crucial to the success of the overall restoration effort (Greg Neudecker, personal communication 2004). In order to assess the efficacy of restoration efforts, it is important to: (1) know what was there when the effort started and (2) know the result. Stream restoration is still a relatively new discipline, with a relatively small body of empirical work testing various restoration hypotheses. In the Blackfoot River, the monitoring effort has been aggressive and unflinching. It could well stand as the poster child for Lee's strategy of adaptive management. Monitoring lies at the heart of that effort.

The Blackfoot River effort, as documented by the monitoring, has experienced a full range of result. One case in particular underscores the importance of monitoring in the restoration efforts. Blanchard Creek is a rainbow trout spawning tributary on Clearwater River that, prior to 1991, regularly experienced dewatering from an irrigation diversion. FWP first informally worked with the irrigator to increase flows in 1991. Rainbow trout densities responded quickly (Pierce and Podner 2000). In 1993 FWP entered into a water lease with the irrigator. After a couple of years of higher densities, the populations started to decline (Figure 1). Monitoring showed that, even as flows remained in the reach, intensified livestock grazing in the reach protected by the lease resulted in both riparian and instream habitat damage, largely offsetting the benefits of the lease (Pierce, personal communication 2003). The lease ended after the 2000 irrigation season, and in 2001, the lower 1.1 miles (1.8 km) of Blanchard Creek were completely dewatered (Pierce and Podner 2002).

Figure 1. Estimated rainbow trout (greater than 4.0) for Blanchard Creek at mile 0.1, 1990 to 2003.



The Blanchard Creek experience is instructive in a number of aspects. First, it underscores the importance of monitoring for the efficacy of the restoration project. In this case, even though FWP was unable to reach resolution with the landowner over the livestock grazing, it was able to isolate the cause of the decline. Second, it highlighted the importance of approaching habitat problems holistically. Dealing with one habitat problem to the exclusion of others on a stream may doom the one project.

Restoration Beyond the Ordinary High Water Marks—Building Community

In 1992, BBCTU organized a symposium, the results of which would extend far beyond the organizers' highest expectations. A recurring frustration for BBCTU in its first years was the persistent failure of the many government agencies within the basin to talk to each other. In one case, this resulted in one agency developing a river plan limiting the placement of additional access sites in one reach of the river, while another agency was actively seeking funding to pursue additional access in that very reach (Mark Gerlach, personal communication 1993).

In an effort to stem this kind of behavior, BBCTU held a symposium in the fall of 1992 to which it invited all the government agencies—local, state and federal—active within the basin. The primary, if not sole, purpose of the meeting was to develop a communication network among the agencies, so they did not work at cross-purposes. The symposium led to other meetings, and, over the course of a year, a broad-based watershed group emerged from the effort that became known as the Blackfoot Challenge (Challenge). The Challenge formally organized in 1993. The series of meetings that led to the formation of the challenge also coincided with BBCTU's growing recognition that restoration needs in the Blackfoot Valley exceeded the focus of its mission. At the same time, it understood the importance of a broader approach (Coughlin et al. 1999).

The objectives of the organization are to, "coordinate efforts that will enhance, conserve, and protect the natural resources and rural lifestyle of Montana's Blackfoot River Valley for present and future generations," (McDonald 2003) and to provide a forum for interested parties to discuss projects and issues in a nonadversarial setting (Stanley 2003). The Challenge has a much broader focus than does the effort initiated by BBCTU. The focus of the

Challenge has been characterized as ridgetop-to-ridgetop (Mark Gerlach, personal communication 2002). The Challenge has chosen to address the entire watershed (McDonald 2003).

The Challenge has five primary areas of emphasis: (1) communication and coordination, (2) information, education and outreach, (3) partnering and facilitation, (4) financial and technical assistance, and (5) administration, planning and program development (Hess 2003). The Challenge implements its mission through the activities of its committees. In addition to an executive committee made up of board officers, there are a number of issue-specific committees: weed steering committee; education committee; conservation strategies committee; drought and water conservation committee; the habitat, water quality and restoration committee; and the wildlife committee. The committees are comprised of a mix of public agency representatives, private groups, members and individuals. A common feature of all these committees is that they facilitate information exchange between groups working in the valley, between groups and individuals, and between the Challenge and the interested public.

While the Challenge does not actually initiate most of the conservation projects in the basin, it has become an effective clearing house, mediator and information source for the myriad conservation efforts ongoing in the valley. In addition, the Challenge has become an effective fundraising force, and it has been able to significantly assist its various conservation partners financially by pursuing diverse sources of funding (Tina Bernd-Cohen, personal communication 2004). The results of the Challenge's efforts have been impressive. They include: more than 350,000 acres (141,640 ha) of weeds mapped with 120,000 acres (48,562 ha) under active management, 2,100 acres (850 ha) of wetland restoration, 2,300 acres (931 ha) of native grasslands restored, 46,000 acres (18,616 ha) of grazing management improvements and 84,600 acres (34,236 ha) in conservation easements on private lands (24% of the private land in the Blackfoot Valley).

Perhaps the most dramatic example of the potential of the Challenge's committee approach can be seen in the work of the drought and water conservation committee. This committee had its genesis in the perilously low snow pack of the winter of 2000. DFWP has a relatively junior water right for fisheries on the main stem of the Blackfoot River below the confluence of the north fork of the Blackfoot River. Rather than make a call for their water from junior users, FWP, working with Trout Unlimited's Montana Water Project and the Blackfoot Challenge, agreed to try a voluntary, cooperative approach to streamflow maintenance, with the Challenge implementing the effort (Laura Ziemer, personal communication 2001). Within two months in the late spring of 2000, the newly formed emergency drought response committee, managed to get over 70 irrigators involved in a voluntary conservation effort that was based on the concept of shared sacrifice—a recognition that everybody in the community gets hurt by drought. When flows on the Blackfoot River reached critical levels, various parts of the drought plan would go into effect. When flows gotto 700 cubic feet per second (cfs), those who had joined the response would reduce their diversions as described in their plans. When flows hit 600 cfs, DFWP would issue a fishing advisory that would request anglers to not fish at certain times or on certain waters. (U. S. Fish and Wildlife Service 2004). In three of the last 4 years, the drought has been sufficiently severe to invoke the plan. In every year that the plan has gone into effect, this voluntary effort succeeded in securing more water in the river than would be been possible had DFWP invoked its water right in a traditional adversarial approach (Mike McLane, personal communication 2003).

The Challenge continues to expand the scope of its activities. Recently, the Challenge has taken on the ambitious task of doing the baseline work to establish the total maximum daily load (TMDL) of pollutants under section 319 of the Clean Water Act for the entire Blackfoot River (Tina Bernd-Cohen, personal communication 2004). As with its other efforts, the Challenge has been successful at marshaling the available expertise of the diverse government agencies and private groups.

Finally, over the last 3 years, the Challenge has led a coalition of public agencies, private conservation groups, and valley residents to the purchase of 89,000 acres (36,017 ha) of land in the Blackfoot River watershed (Nature Conservancy 2003). Perhaps as importantly, the Challenge led a valley-wide planning effort to assure that the purchase would complement the traditional uses in the valley-ranching, forestry, public access and wildlife habitat. The Challenge began its planning effort almost 2 years before the lands became available for sale.

The success of the challenge rests heavily on a few things. First, landowners and other stakeholders have bought into the projects. Second, the restoration effort has been fortunate in securing the necessary funding to complete the projects it has. Third, the projects have focused on key species that serve as indicator species. Fourth, government agencies have not attempted to direct the process, but rather to assist it when requested by other partners in the process (Hess 2003).

Others have noted the importance of identifying the opinion leaders in the valley and getting them ownership in the effort (Coughlin et al. 1999). To that end, the agency people active in the Blackfoot River Projects are notable for their longevity. Pierce and Peters, of DFWP, and Neudecker, of USFWS, have all been working in the valley for more than a decade (Greg Neudecker, personal communication 2004). They are now widely perceived as an integral part of the community (Jim Stone, personal communication 2004). Their agencies could not have achieved what they have in the Blackfoot River if they had adhered to the revolving-door policy that is common in some government agencies.

Finally, there is a widespread perception that the presence of neutral ground, where people can meet in a social setting without the pressures of defending their position, is a valuable part of the cultural mix necessary to make a large-scale collaboration work. In the area around the Blackfoot River that place is Trixi's Saloon (Coughlin et al. 1999). Trixi's has been the site of innumerable discussions, good-natured debates and socializing, all of which have been a substantial factor in forging a sense of community among the diverse interests in the valley.

In its essence, the Challenge is about building a community's interests in the valley, and it operates on the premise that each stakeholder has a legitimate interest and is an active part of the valley culture. This approach has fostered mutual respect among groups and individuals who have traditionally thought of themselves as adversaries (Jim Stone, personal communication 2004). Out of that respect has grown the ability of the larger community to act in concert to the benefit of all the interests in the valley.

Conclusion

The Blackfoot experience of collaborative restoration efforts can be explained by the science underlying the restoration effort and by the cultural and social context in which the effort must occur. Careful attention to each of these details is essential. To proceed on one front while neglecting the other is to doom a project.

On the scientific front, the Blackfoot experience underscores the importance of gathering baseline data before restoration begins, of monitoring after the restoration work is complete (to record the response of both habitat and biology to the restoration effort) and of responding to confirm, adjust or discard a restoration approach. In short, thoughtful adherence to the principals of adaptive (or iterative) management are key to long-term success of restoration.

Another key aspect of the information gathering effort is to approach it on a watershed-wide basis. In the Blackfoot River Valley, as the effort intensified, it became clear that simply examining the narrow band of land along rivers and streams wasn't telling the whole story. In order to understand the role of wetland, forest cover and larger land management efforts on the watershed, it is necessary to look from ridgetop-to-ridgetop.

Any restoration effort has to achieve local acceptance, either through specific activities on private land or through its impact on the local economy and culture. As a practical matter, restoration experts have to learn the art of patience; there is no instant gratification in the business of natural resource restoration. To that end, the qualities of the people taking on the task of restoration work are crucial to its success. As the USFWS and DFWP experience teaches, a little respect for the people on the land—even when their practices may offend—can go a long way.

In addition, in an effort as expansive as that in the Blackfoot River Valley, it is important to engage the right people in the effort. Identifying widely respected community leaders should be an early task. The courtship of those leaders can be long and, sometimes, arduous. It takes time to develop trust. Finding people who will stay the course over years and decades is invaluable.

Finally, while the Challenge is the beneficiary of a particularly lucky collision of personalities and events that may not be easily replicated elsewhere, the lessons of that experience, grounded in straightforward approaches to science and in some basic precepts of civil society (respect for the position of others and commitment to protect cultural and community values) can encourage restoration efforts in other watersheds, even in the face of widely divergent circumstances.

Reference List

Coughlin, C., M. Hoben, D. Manskopf, and S. Quesada, 1999. *Blackfoot challenge in a systematic assessment of collaborative resource management partnerships*. M. S. thesis. School of Natural Resource and Environment, University of Michigan.

Hess, Susan. 2003. *Case studies of collaborative planning and adaptive management in the western United States*. Tucson Arizona: Sonoran Institute.

Hinson, Dustin. 2002. *A critical review of the impacts, restoration, and monitoring in the Blackfoot River Watershed, Montana*. Unpublished report. University of Idaho, Aquatic Restoration Ecology.

Lee, Kai N., and Jody Lawrence. 1986. Adaptive management: Learning from the Columbia River Basin fish and wildlife program. *Environmental Law*. 16:43.

McDonald, B., 2003. *The blackfoot watershed.* unpublished paper. Missoula, Montana: University of Montana.

Montana Department of Fish and Game. 1997. *Effects of the June 1975 Mike Horse tailings dam failure on water quality and aquatic resources of the upper Blackfoot River*. Helena, Montana: Montana Department of Fish and Game.

Nature Conservancy, The. 2003. *The blackfoot challenge announces the conservation of important forest lands*. http://nature.org/wherewework/northamerica/states/montana/press/press1201.html

Neudecker, Greg. 1999 *The big blackfoot river: A watershed initiative to restore native fish populations*. Great Falls, Montana: U. S. Fish and Wildlife Service.

Peters, Don, and Ron Spoon. 1989. *Preliminary fisheries inventory of the big Blackfoot River*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

Pierce, Ron, Don Peters, and Tim Swanberg. 1997. *Blackfoot River restoration project progress report*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

Pierce, Ron, and Craig Podner. 2000. *Blackfoot River fisheries inventory, monitoring, and restoration report 2000*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

———. 2001. *Blackfoot River fisheries inventory, monitoring, and restoration report 2001*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

———. 2002. *Blackfoot River fisheries inventory, monitoring, and restoration progress report 2002*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

Pierce, Ron, Ryen Anderson, and Craig Podner. 2004. *Blackfoot River fisheries inventory, monitoring, and restoration report 2003–2004.* Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

Stanley, Forest R. 2003. *The blackfoot challenge: A model for collaborative and restoration efforts*. unpublished report.

Stiller, David. 2000. *Wounding the West*. Lincoln, Nebraska: University of Nebraska Press.

U. S. Fish and Wildlife Service. 2004. *The Blackfoot Challenge*. http://mountain-prairie.fws.gov/pfw/montana/mt6.htm.

U. S. Fish and Wildlife Service. 2001. B*lackfoot challenge: Blackfoot drought response plan*. city, state: publisher.

Session Three.
Producing Energy and Conserving Wildlife

Chair
Jim Mosher
Izzak Walton League of America
Gaithersburg, Maryland

Contrasting the Potential Effects of Biomass Fuel, Soy-based Fuel, Ethanol and Wind Energy Developments on Northern Great Plains Wildlife

Kenneth F. Higgins
U. S. Geological Survey, South Dakota Cooperative Fish and Wildlife Research Unit, South Dakota State University
Brookings

Robert G. Osborn
Minnesota Department of Natural Resources, Farmland Research Group
Madelia

David E. Naugle
University of Montana, Wildlife Biology Program
Missoula

Kristel K. Bakker
Dakota State University, College of Arts and Sciences
Madison, South Dakota

Between 1870 and 1980, energy developments in the northern Great Plains focused on petroleum exploration, petroleum extraction and replacement of windmill powered electricity with hydro-, coal, gas or nuclear powered electric generators. Networks of electric transmission lines and stations transfer these energy sources to communities or regions with higher energy needs. Concern about declining world-wide petroleum reserves, limited potential for hydropowered energy, and environmental and health impacts related to coal and nuclear power developments has spurred an interest in the development and evaluation of alternative, renewable energy sources. Four large-scale renewable energy sources that are being developed in the northern Great Plains region include wind-powered turbines, fuel from grass biomass products, ethanol from corn (*Zea mays*) products and soy fuels from soybean (*Glycine max*) products. The purpose of this paper is to compare information-based scenarios addressing the potential positive and negative effects of each of the four energy development types on wildlife or their habitats in the northern Great Plains Region. Our goal is to inform decision-makers for natural resources about the potential local and large-scale effects of these recent alternative energy developments on wildlife and its habitats, so they will be better able to implement plans to offset potential negative impacts.

Wind Power Developments

In the Great Plains of North America, wind energy was the basic source of power used to run windmills to pump water for livestock for domestic use and for generating electricity between 1870 and 1940. Beginning in the 1940s, wind-powered energy sources were almost totally replaced by gasoline and diesel powered water pumps and by electrical generators. Within a decade after the 1940s nearly all of the older wind-powered electric sources were replaced with hydro-, coal, gas or nuclear powered electric generators and with networks of electric power transmission lines and stations. Recently, concern for environmental quality has brought wind-powered electric energy back to the northern Great Plains.

The Buffalo Ridge Windplant was the first windplant facility to be developed in the midwestern United States (Higgins et al. 1995, Nelson and Curry 1995, Johnson et al. 2000). This 354-turbine windplant is located southeast of Lake Benton, Minnesota (44–15'45' N latitude and 96–17'33' W longitude) and extends 34 miles (54 km) from the northwest to the southeast corners. Since the construction of the Buffalo Ridge Windplant, several other windplants have been developed or are in various stages of development in the northern Great Plains. A 41-turbine windplant has been developed on the Coteau Hills between the towns of Edgeley and Kulm in southcentral North Dakota (Dart 2003). Another windplant, containing 17 turbines, is being developed near the town of Miller in central South Dakota (Hildebrandt 2003). Three windplants are operating in northcentral Iowa, one with 257 turbines in Beuna Vista County, one with 55 turbines near the town of Clear Lake and one with 148 turbines near the town of Duncan. The number and size of windplants in the northern Great Plains will likely increase, with as many as 1,000 to 2,000 turbines on some sites (Dart 2003).

Biomass Fuel Developments

The use of biomass fuel crops, wood chips and herbaceous products, to facilitate development of alternative sources of electricity generation has the potential to reduce amounts of air pollution (e. g., sulfur dioxide and carbon dioxide) currently emanating from coal-fired power plants (Boman and Turnbull 1997). Switchgrass (*Panicum virgatum*) has been identified as a potential biomass fuel crop for much of the United States (Volgel 1996, Vogel et al. 2002, De La Torre Ugarte et al. 2003) and eastern Canada (Madakadze et al. 1996). Switchgrass and coastal panic grass (*Panicum amarum*) also have been evaluated as biomass fuel crops in the United Kingdom (Christian et al. 2002). Major costs associated with production and use of switchgrass biomass as fuel include nitrogen fertilization, harvest techniques, storage costs and transportation to final processing facilities (Keeney and DeLuca 1992). Switchgrass biomass has replaced coal at the Ottumwa, Iowa Electrical Generating Plant, where an estimated 50,000 acres (20,234 ha) of cropland would be needed to produce 35,000 tons (31,751 metric tons) of switchgrass fuel, which in turn would reduce coal fuel use by up to 5 percent. Switchgrass biomass is also being tested as a base product from which to produce ethanol (Teel 1998). Regardless of its use as an alternative energy source, fields of switchgrass grown as biomass fuel has the potential to reduce soil erosion, improve water quality and provide additional habitat acreages for numerous species of wildlife, including grassland nesting birds (McLaughlin and Walsh 1998, Best and Murray 2003).

Ethanol Fuel Developments

Approximately 6 percent of the United States corn crop was used to produce up to 2 billion gallons (7.57 billion liters) of ethanol in 2003. In the United States, there are 73 ethanol-producing plants in operation (Tonneson 2003) and

13 more are under construction. Higgins et al. (2002) reported that 23 ethanol processing plants occurred in South Dakota and parts of four adjoining states that occur in the Prairie Pothole Region of the United States. The Archer Midland Company alone accounts for about half of the United States' ethanol production. A strong incentive to develop more ethanol lies with the potential to reduce air pollutants from combustion-engine exhaust emissions, even though ethanol is currently used only as a blend additive in petroleum-based fuels. Of the three alternative fuels being produced from farmer-grown crops, research, development and production of ethanol is greater than for soy-based diesel or switchgrass biomass fuels.

Soy Biodiesel Developments

Biodiesel is typically produced through the reaction of vegetable oil or animal fat with methanol in the presence of a catalyst to yield glycerin or biodiesel (chemically called methyl esters). In Europe, most biodiesel is made from rapeseed (*Brassica napus*) oil, whereas in the Unites States it is made primarily from soybean oil. Biodiesel is an alternative fuel, which can be used in neat form or blended with petroleum diesel for use in compression ignition (diesel) engines. Use of biodiesel fuels will reduce emissions of unburned hydrocarbons, carbon dioxide, carbon monoxide and particulate matter into the environment. Soybean oil-based biodiesel also increases diesel fuel lubricity, which extends engine longevity whether used in total or as an additive or blending component (e. g., B20 is a 20% blend of biodiesel in petrodiesel).

When spilled, biodiesels biodegrade up to three times faster than petrodiesel fuels. Of the 11 biodiesel manufacturers in the United States, three are in Iowa, and there are plans are to develop one or more manufacturing plants in Minnesota where, in 2002, the Minnesota lawmakers passed a statewide mandate requiring biodiesel use. Biodiesel production continues to be one of the American Soybean Association's top legislative priorities.

Positive Energy Development Scenarios

Wind Power

One of the most positive aspects of windplants for northern Great Plains citizens lies with economic benefits. Landowners receive annual lease payments per turbine or a dividend for a percentage of the electricity generated per turbine. This income could ease the financial burden on farmers and lessen the need to remove marginally productive land from state and federal conservation programs. Additionally, as more windplants are developed in the region, a surplus of electricity will be generated, and, like agricultural crops, it will be sold to other states as another export crop. Another way to benefit from the export of wind-generated electricity to other states or provinces would be to apply a mitigation surcharge tax for exported electricity. States would collect these tax dollars and distribute them to their citizens. People in several states are already voluntarily opting to pay a premium price for "green power" electricity, sometimes identified as "green-e" that is generated from renewable natural resources, such as windpower, hydropower or geothermal power (Farhar 1999). As illustrated in 2001, Californians would have paid a superpremium for green power from any source. Premium rates paid by consumers have ranged from \$0.04 to \$0.20 per kilowatt hour with a median value of \$0.25 per kilowatt hour (Swezey and Bird 2000). Instead of a rate charge, customers in some states are assessed a customer surcharge of \$3.00 to \$6.00 per month for kilowatt blocks of green-e (Swezey and Bird 2000). Numerous examples of green-e participant pricing schedules are available in the United States (see review by Swezey and Bird 2000).

Regardless of how the proceeds from the sales of electricity are collected, a portion of the revenue should be returned to the state and the local communities from which the electricity was produced. To benefit all future generations, a portion of the green-e revenue could be allocated to all levels of education; that is one possibility. A second use would be to preserve more natural areas, areas without major cultural inclusions, including wind turbines and transmission lines, since we already know that the presence of wind turbines and transmission lines causes some mortality to migrating birds and bats, and the turbines also diminish the overall naturalness of the landscape. Finally, green-e revenue could support future research that would potentially lessen bird and bat mortality related to wind resource areas.

Biomass Fuels

Taller, high-yielding stands of monoculture or low-diversity multiple-species biomass crops that have genotypes that are environmentally adapted for an area are essential to successful biofuel production. Stands of tall perennial grasses composed of either single or multiple grass species or their cultivars provide more suitable habitat for most species of prairie wildlife than do fields of annual crops, such as corn or soybeans (Best et al. 1995, 1997). In Iowa, Murray (2002) found that switchgrass fields grown for biomass fuel attracted grassland birds, which successfully produced enough young birds to stablize bird populations. His results showed that the average abundance of 45 bird species varied between total- and strip-harvested fields or nonharvested fields. He also found that, during winter, American tree (*Spizella arborea*) and song sparrows (*Melospiza melodia*) occurred more frequently in strip-harvested fields than in total-harvested or nonharvested switchgrass fields, whereas ring-necked pheasants (*Phasianus colchicus*) occurred only in non-harvested and uncut portions of strip-harvested fields. In eastern South Dakota and in western Minnesota, Bakker and Higgins (2004) found lower densities for six of eight grassland bird species in monotypic switchgrass fields, compared to stands seeded to multiple warm-season grass species ($3 < N < 6$) or native-sod prairies containing from 28 to 137 species of grasses and forbs (Higgins et al. 2001). However, all eight grassland obligates analyzed were detected in switchgrass fields.

The Conservation Reserve Program (CRP) currently provides thousands of acres of grassland habitat in the northern Great Plains, but, as contracts expire, most fields will be reverted to cropland. Best and Murray (2003) suggest that one alternative to returning CRP fields to row crops is to produce switchgrass for biomass fuel. Because biomass is harvested in fall and winter, breeding birds would not be directly affected by mowing fields but might be influenced by changes in vegetation structure resulting from harvest. Best and Murray (2003) evaluated bird abundances and nest success in totally-harvested, partially-harvested (i. e., alternating cut and uncut strips), and nonharvested CRP switchgrass fields in southern Iowa. They found that abundances of most grassland bird species (16 of 18) were not affected by harvesting fields, and that strip width did not affect bird numbers in strip-harvested fields (Best and Murray 2003). Grasshopper sparrows (*Ammodramus savannarum*) were more abundant in harvested portions of fields, and more sedge wrens (*Cistothorus platensis*) were recorded in nonharvested areas. Residual vegetation in nonharvested areas provided nesting cover for northern harriers (*Circus cyaneus*) and ring-necked pheasants (Best and Murray 2003) who also reported that rates of nest success for grasshopper sparrows and common yellowthroats

(*Geothlypis trichas*) were probably sufficient to maintain populations. Furthermore, findings from Best and Murray (2003) indicate that a mixture of harvested and nonharvested fields would be more beneficial to grassland birds than totally harvesting or partially harvesting all switchgrass fields.

In brief, grass biomass crop fields are better for birds than row crop fields, but monotypic stands may not be as good as multiple-species stands or native sod prairies.

Ethanol and Soy Biodiesel

The only positive aspect to wildlife of more acres of corn and soybeans in the northern Great Plains may be as food items (Krapu et al. 2004). Deer (*Odocoileus* spp.) and tree squirrels (*Sciurus* spp.) can forage on standing corn whereas many game and nongame birds can forage on waste grains, especially during postharvest activities during fall and winter. For decades, corn has been recognized as an excellent winter food-plot species along with sunflowers (*Helianthus* spp.) and some sorghums (*Sorghum* spp.). Soybeans are also ingested by some wild game, most often by ring-necked pheasants; however, due to poor digestibility factors, they constitute a less than adequate forage item during harsh winters. Some wildlife also graze on newly-emerged corn and soybean plants; however, farmers see this as animal damage and game and fish departments have to find ways to offset (e. g., lure crops) or reduce depredation damages.

Negative Energy Development Scenarios

Wind Power

Although renewable energy resources, such as wind power, have received strong public support (Hansen et al. 1992), the effects of wind turbines on avian and bat communities have not been adequately researched, especially in grassland landscapes with numerous wetlands. Avian mortality from collisions with wind turbines varies from little or no mortality (Byrne 1983; Winkelman 1985, 1990; Higgins et al. 1995) to substantial mortality (McCrary et al. 1983, Howell and Noone 1992, Orloff and Flannery 1992). However, some species, such as raptors, appear to be disproportionately susceptible to collisions with wind turbines (Orloff and Flannery 1992). To date, most studies were designed to monitor the biological impacts of windplants upon bird communities. We now know that bat communities are affected as well (Howell and DiDonato 1991, Osborn et al. 1996, Johnson et al. 2000).

Several factors that may contribute to bird and bat collisions with wind turbines have been identified (Nelson and Curry 1995). Many of these factors, such as storm fronts (which increase rate of travel) and fog (which reduces visibility) are uncontrollable events. Other factors, such as topography, surrounding land use and the presence of dense breeding or wintering avian populations are partially controllable through habitat manipulation or presite reconnaissance. However, even a well-sited facility may be the occasion of some bird or bat mortality (Osborn et al. 1996, Johnson et al. 2000, Osborn et al. 2000). As a result, most research has focused on finding ways to decrease the likelihood of bird collisions with turbines.

Initial efforts to reduce mortality were hampered by some birds' habituation to wind turbines, and they would use them as perching sites (Nelson and Curry 1995). Raptors were of special concern because they used wind turbines as perching sites and as observational platforms from which to hunt (Nelson and Curry 1995); this behavior greatly increases the likelihood of a collision. Therefore, wind power and utility companies and state and federal natural resource agencies have modified structural characteristics of wind turbines to reduce their attractiveness to birds, to increase their visibility to birds and to make them more difficult for perching or nesting. For example, replacing lattice frame towers with tubular towers has successfully reduced perching and nesting sites (Nelson and Curry 1995). Currently, tests are being conducted to determine if painting turbine blades with different colors or patterns might reduce the frequency of bird collisions with turbines.

In addition to direct mortality from collisions, several researchers have reported that birds tend to avoid using areas in which turbines are located. For example, waterfowl, wading bird and raptor densities near turbines were lower compared to densities in similar habitats away from turbines (Winkelman 1990, Pedersen and Poulsen 1991, Usgaard et al. 1997). Athough the effects of wind turbines on grassland passerine species have received less attention, Leddy et al. (1999) provided evidence that several passerine species also avoided otherwise suitable habitats in which turbines were located.

Results from the Buffalo Ridge studies in Minnesota (Osborn et al. 1996, 2000; Johnson et al. 2000) indicated that bird and bat mortality due to collisions with wind turbines in a grassland/cropland landscape is small (less than 10 birds or bats per turbine per year), and wind turbines do not appear to kill more birds than other human-made structures (see the annotated bibliography of Hebert and Reese 1995). However, even a well-sited facility will kill some animals. Therefore, resource professionals need to ask, "what number of mortalities is acceptable?" Acceptable loss is likely to vary from area to area, agency to agency, and it will depend greatly upon the species at risk, especially bird and bat species of threatened or endangered status. Until further research is conducted, it would seem advisable to avoid building windplants near breeding, staging or wintering areas, migration corridors, refuges, or other areas with large concentrations of birds or bats.

Preconstruction site reconnaissance can reduce subsequent bird mortality rates. Consideration of potential effects on avian communities before designing and siting a facility may be the best first step to reduce mortality at windpower projects (Nelson and Curry 1995). Baseline data can establish initial abundance, migration patterns and can identify species of special concern at a proposed site. Additionally, baseline data are essential for evaluating postconstruction effects of wind turbines on bird and bat populations. Specific locations should be evaluated on a site by site basis before future development of windplants in the northern Great Plains.

We suggest that even low mortality rates per turbine should not be taken lightly, especially in relation to the number of windplants being proposed in future construction plans. For example, even at a low average mortality rate of five birds and five bats per turbine, the addition of 2,000 turbines in the northern Great Plains could equate to a loss of 10,000 birds and 10,000 bats annually. Such losses would be significant. Of greater concern, however, is Johnson et al.'s (2000) results showing that a significant relationship exists between increasing avian and bat mortalities and decreasing distances between turbines and wetlands at the Buffalo Ridge Windplant. The importance of their findings lies with the fact that most areas of eastern North and South Dakota and of the southern provinces of Canada have wetland and bird densities greatly exceeding those of western Minnesota.

We believe one of the most negative aspects of having several large windplants located in the northern Great Plains would be the need for additional development of large, above-ground, electrical transmission lines. At present, most of the northern Great Plains is lacking the transmission line infrastructure to export a large volume of electrical power. As several researchers have pointed out, overhead electrical transmission lines can cause significant avian mortality

(Malcolm 1982, Faanes 1987, Bevanger 1994), especially in areas with a high density of wetlands. Also, from an aesthetic perspective, the presence of wind turbines and transmission power lines will lessen the quality of many landscape vistas.

Negative Energy Development Scenarios—Ethanol and Soy Biodiesel

Prior to human settlement, wetlands of the northern Great Plains were embedded within extensive landscapes of perennial grasslands that provided ideal cover for upland nesting waterbirds. By the turn of the century, however, cropland had replaced most of the native grassland in the tallgrass prairie region of Minnesota and Iowa, where today less than 5 percent of the original tallgrass prairie is still intact (Samson and Knopf 1996). Until recently, the loss of mixed-grass prairie had occurred slower than the rate of loss for tallgrass prairie. A common public misconception is that mixed-grass prairie is not at high risk of tillage because growing conditions would not support tillage agriculture in the West. In reality, approximately 60 percent of all native mixed-grass prairie in South Dakota, North Dakota and Montana has already been converted to cropland (United States Department of Agriculture 2000a). A surge in row crop acreages for the production of ethanol and soy biodiesel will accelerate the conversion of wetland and native grassland habitats throughout the northern Great Plains.

The native rangeland losses and the decrease in number of farm families have coincided with changes in farm equipment that now enable fewer workers to more efficiently till , plant and harvest cropland acres. Farmers today can till and plant two to three times as many corn and wheat (*Triticum* spp.) acres using tractors with horsepower ratings that have quadrupled since 1960 (Higgins et al. 2002). Changes in equipment have also led to cleaner farming practices, meaning operators remove grass margins along fields and drain small wetland areas that once served as important wildlife habitat but are now perceived as problem areas that impede the movements of large machinery. Grassland losses will likely increase in relation to the capability to farm more acres with larger equipment and as operators change from diversified grain and livestock operations to monoculture grain farming operations.

Changes in crop types that coincided with the movement towards monoculture tillage have decreased the quality of farmland wildlife habitat. Areas of suitable habitat types (e. g., wheat and barley [*Hordeum vulgare*]) that provide nesting cover for birds in spring have decreased (16.4%) while unsuitable habitat types (e. g., soybeans and corn) that provide little wildlife cover increased (25.1%) from 1992 to 1997 (Higgins et al. 2002, United States Department of Agriculture 2000b). The most evident change in crop types is the northern and western expansion of soybeans into the northern Great Plains where it was considered too dry to grow soybeans just 60 years ago. Development of drought-resistant, genetically modified soybeans has accelerated the conversion of native grassland to cropland. Soybean acreages now exceed that of corn and some other small grain crops. Agricultural interests in Minnesota and Iowa, where approximately 1 percent of tallgrass prairie remains, have built 32 soybean and corn (i. e., ethanol) industrial plants to process crops because of low commodity prices and because of crop surpluses. The increasing number of processing plants is a mechanism to utilize the surpluses while providing some income to farmers and investors. Recent construction of 13 more processing plants suggests that rangeland losses will likely continue as new row and oil-seed crop varieties to develop that entice producers to farm shallow, drought-prone soils that are common throughout much of the northern Great Plains. Ethanol and soybean processing plants also contribute to reduced diversity in crop choices and may invite greater risk of disease infestations, such as occurred in potatoes in Ireland in the 19th century.

The Big Picture—A Comparison of Renewable Energy Source Development Effects

Our current assessment indicates that, of the four major renewable energy developments in the northern Great Plains, the only one with potentially major benefits for prairie wildlife is biomass fuels. But, this is true only if annually-tilled croplands are replaced with biomass crops. If biomass crops replace CRP grassland acreages or if native prairies are plowed to establish biomass crops, the benefits are less apparent, possibly negative. The major detrimental aspects related to wind turbines, soybean-oil based biodiesel and corn-based ethanol developments exceed most of the positive benefits that can be attributed to them in respect to wildlife or their habitats. However, if electricity generated from wind farms resulted in fewer coal burning plants, wildlife and people alike might receive some indirect positive benefits (e. g., cleaner air). If wind farms resulted in less power being generated from hydroelectric plants, perhaps managers of river systems would have greater flexibility in flow cycle and flood spike management, which, in turn, might benefit wildlife that are dependent on sand bars and other riverine habitats (e. g., species listed as endangered or threatened such as piping plovers [*Charadrius melodus*], interior least terns [*Sterna antillarum athalossos*] and pallid sturgeon [*Scaphirhynchus albus*]).

Wildlife conservation efforts should focus on the best administrative or management choices to reduce or to eliminate the greatest or longest-term impacts due to wind energy, ethanol or soy biodiesel developments. Relative to windpower, initial efforts should be aimed at preconstruction site reconnaissance to ensure windplants are not built on or adjacent to important breeding grounds, staging areas or migration routes of birds and bats. Relative to biomass crops (e. g., switchgrass), considerations should be made to favor retention of grasslands for biomass fuel production or to use biomass crops to extend contracts or to replace CRP stands when present contracts expire. Relative to corn and soybean developments, conservation efforts should initially focus on restricting or directing cropping to soils with capabilities of producing profitable yields in at least 8 out of 10 years, despite weather and water conditions.

Wildlife conservation efforts in the northern Great Plains are more integrated with energy needs of the nation and the world than ever before. To meet the future energy needs of ever-increasing U. S. and global populations, energy providers will eventually need to convert from a high dependence on fossil-based energy sources to alternative renewable energy sources. To keep pace, conservation agencies will need to provide current research and evaluation results on all facets of energy development and use both positive and negative, in order for conservation administrators and managers to be able to plan for the future of wildlife and its habitat.

To ensure a reasonable balance between future energy development needs and future wildlife resource needs, there must be some visionary and integrated planning that is aimed at all levels of technical development, economic administration and citizen use of energy resources. Hopefully, our review of the state-of-the-art of four major renewable energy developments in the northern Great Plains will aid conservationists in their decision processes toward achievable goals for the benefit of tomorrow's wildlife populations.

Reference List

Bakker, K. K., and K. F. Higgins. 2004. *Avian response to single and multiple plant species grasslands managed for biomass in the northern*

Great Plains, unpublished report. Madison, South Dakota: Dakota State University.

Best, L. B., H. Campa III, K. E. Kemps, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks Jr., and S. R. Winterstein. 1997. Bird abundance and nesting in CRP fields and cropland in the Midwest: A regional approach. *Wildlife Society Bulletin*. 25:864–77.

Best, L. B., K. E. Freemark, J. J. Dinsmore, and M. Camp. 1995. A review and synthesis of habitat use by breeding birds in agricultural landscapes of Iowa. *American Midland Naturalist*. 134:1–29.

Best, L. B., and L. D. Murray. 2003. Short-term bird response to harvesting switchgrass for biomass in Iowa. *Journal of Wildlife Management*. 67:611–21.

Bevanger, K. 1994. Bird interactions with utility structures: Collision, electrocution, causes and mitigating measures. *Ibis*. 136:412–25.

Boman, U. R., and J. H. Turnbull. 1997. Integrated biomass energy systems and emissions of carbon dioxide. *Biomass and Bioenergy*. 13:333–43.

Byrne, S. 1983. Bird movements and collision mortality at a large horizontal axis wind turbine. *California-Nevada Wildlife Transactions*. 1983:76–83.

Christian, D. G., A. B. Riche, and N. E. Yates. 2002. The yield and composition of switchgrass and coastal panic grass grown as biofuel in southern England. *Biosource Technology*. 83:115–24.

Dart, L. 2003. Prairie power: There's a new wind blowing in North Dakota. *North Dakota Living*. 50:10–2.

De La Torre Ugarte, D. G., M. E. Walsh, H. Shapouri, and S. P. Slinsky. 2003. *The economic impacts of bioenergy crop production on U. S. Agriculture. U. S. Department of Agriculture, Agriculture Economic Report Number 816*. Washington, DC: U. S. Department of Agriculture.

Faanes, C. A. 1987. *Bird behavior and mortality in relation to power lines in prairie habitats, U. S. Fish and Wildlife Service, Fish and Wildlife, technical report number 7*. Washington, DC: U. S. Department of the Interior.

Farhar, B. 1999. *Willingness to pay for electricity from renewable resources: A review of utility market research, report NREL/TP.550.26148*. Golden, Colorado: National Renewable Energy Laboratory.

Hansen, P., B. Grant and N. Lange. 1992. *Power to spare in the Upper Midwest*. Minneapolis, Minnesota: Izaak Walton League of America.

Hebert, E., and E. Reese. 1995. *Avian collision and electrocution: An annotated bibliography. California Energy Commission Report P700-95-001*. Sacremento, California: California Energy Commission.

Higgins, J. J., G. E. Larson, and K. F. Higgins. 2001. Floristic comparisons of tallgrass prairie remnants managed by different land stewardships in eastern South Dakota. *Proceedings of the North American Prairie Conference*. 17:21–31.

Higgins, K. F., D. E. Naugle, and K. J. Korman. 2002. A case study of changing land use practices in the northern Great Plains, U.S.A.: An uncertain future for waterbird conservation. *Waterbirds*. 25:2:42–50.

Higgins, K. F., R. E. Usgaard, and C. D. Dieter. 1995. *Monitoring seasonal bird activity and mortality at the Buffalo Ridge Windplant, Minnesota*. Brookings, South Dakota: South Dakota State University.

Hildebrandt, K. 2003. South Dakota: Potential energy exporter. *Argus Leader Newspaper*. October 19.

Howell, J., and J. DiDonato. 1991. *Assessment of avian use and mortality related to wind turbine operations: Altamont Pass, Alameda and Conta Costa counties, California, September 1988 through August 1989, final report*. Livermore, California: U. S. Windpower.

Howell, J., and J. Noone. 1992. *Examination of avian use and mortality at a U. S. Windpower wind energy development site, Solano County, California*. Fairfield, California: Solano County Department of Environmental Management.

Johnson, G. D., W. P. Erickson, M. D. Strickland, M. F. Shepherd, and D. A. Shepherd. 2000. *Avian monitoring studies at the Buffalo Ridge, Minnesota wind resource area: Results of a 4-year study, final report*. Cheyenne, Wyoming: Western Ecosystems Technology, Inc.

Keeney, D. R., and T. H. DeLuca. 1992. Biomass as an energy source for the Midwestern United States. *American Journal of Alternative Agriculture*. 7:137–44.

Krapu, G. L., D. A. Brandt, and R. R. Cox, Jr. 2004. Less waste corn, more land in soybeans, and the switch to genetically modified crops: Trends with important implications for wildlife management. *Wildlife Society Bulletin*. 32:127–36.

Leddy, K. L., K. F. Higgins, and D. E. Naugle. 1999. Effects of wind turbines on upland nesting birds in Conservation Reserve Program grasslands. *Wilson Bulletin*. 111:100–4.

Madakadze, I. C., D. L. Samson, D. L. Smith, and B. E. Coulman. 1996. The phenological development and yield of warm season grasses in SW Quebec. *Canadian Journal of Plant Science*. 76:354.

Malcolm, J. M. 1982. Bird collisions with a power transmission line and their relation to botulism at a Montana wetland. *Wildlife Society Bulletin*. 10:29–304.

McCrary, M. D., R. L. McKernan, R. E. Landry, W. D. Wagner, and R. W. Schreiber. 1983. *Nocturnal avian migration assessment of the San Gorgonio wind resource study area, spring 1982*. Rosemead, California: Southern California Edison Co.

McLaughlin, S. B. and M. E. Walsh. 1998. Evaluating environmental consequences of producing herbaceous crops for bioenergy. *Biomass and Bioenergy*. 14:317–24.

Murray, L. D. 2002. *Avian response to harvesting switchgrass for biomass in southern Iowa*. M. S. thesis, Iowa State University, Ames.

Nelson, H. K., and R. C. Curry. 1995. Assessing avian interactions with windplant development and operation. *Transactions of the 60th North American Wildlife and Natural Resources Conference*, 266–87. Washington, DC: Wildlife Management Institute.

Orloff, S. G., and A. W. Flannery. 1992. Wind turbine effects on avian activity, habitat use, and mortality in Altamont Pass and Solano County Wind Resource Areas, 1989–1991. Sacramento, California: California Energy Commission.

Osborn, R. G., K. F. Higgins, C. D. Dieter, and R. E. Usgaard. 1996. Bat collisions with wind turbines in southwestern Minnesota. *Bat Research News*. 37:105–8.

Osborn, R. G., K. F. Higgins, R. E. Usgaard, C. D. Dieter, and R. D. Neiger. 2000. Bird mortality associated with wind turbines at the Buffalo Ridge Wind Resource Area, Minnesota. *American Midland Naturalist*. 143:41–52.

Pedersen, M. B., and E. Poulsen. 1991. Avian responses to the implementation of the Tjaereborg Wind Turbine at the Danish Wadden Sea. *Danske Vildtundersogelser*. 47:1–44

Samson, F. B., and F. L. Knopf, eds. 1996. Prairie conservation: Preserving North America's most endangered ecosystem. Washington, DC: Island Press.

Swezey, B., and L. Bird. 2000. *Green power marketing in the United States: A status report, technical report, fifth edition, NREL/TP-620-28738*. Golden, Colorado: National Renewable Energy Laboratory.

Teel, A. 1998. Management guide for the production of switchgrass for biomass fuel production in southern Iowa. Ins *Proceedings of 1998 Bioenergy Conference*. Madison, Wisconsin: Great Lakes Biomass Energy Project.

Tonneson, L. 2003. Investing in ethanol: How many more plants can we build? *Dakota Farmer*. 121:20.

U. S. Department of Agriculture. 2000a. *Summary report 1997 National Resources Inventory (revised December 2000)*. Ames, Iowa: U. S. Department of Agriculture, Natural Resources Conservation Service, Iowa State University.

———. 2000b. *100 years of South Dakota agriculture 1900–1999*. Sioux Falls, South Dakota: South Dakota Agricultural Statistics Service.

Usgaard, R. E., D. E. Naugle, K. F. Higgins, and R. G. Osborn. 1997. Effects of wind turbines on nesting raptors at Buffalo Ridge in southwestern Minnesota. *Proceedings of the South Dakota Academy of Science*. 76:113–7.

Vogel, K. P. 1996. Energy production from forages (or American agriculture-back to the future). *Journal or Soil and Water Conservation*. 51:137–9.

Vogel, K. P., J. U. Brejda, D. T. Walters, and D. R. Buxton. 2002. Switchgrass biomass production in the Midwest USA: Harvest and nitrogen management. *Agronomic Journal*. 94:413–20.

Winkelman, J. E. 1985. Bird impact by middle-sized wind turbines-on flight behavior, victims, and disturbance. *Limosa*. 58:117–21.

———. 1990. Impact of the wind park near Urk, Netherlands, on birds: Bird collision victims and disturbance of wintering fowl. *International Ornithological Congress*. 20:402–3.

Hydropower Relicensing and Hatchery Reform

Jeff Curtis
Trout Unlimited
Portland, Oregon

In the past century, thousands of hydroelectric dams were built throughout the nation. The dams include very primitive, almost "Rube Goldberg" dams that were constructed since the early part of the century to provide power. And, they include the massive dams on major rivers of the West, near cities, that were built during the 1950s and 1960s. Dams forever changed the nature of North America's rivers.

Many of the hydroelectric dams are now up for relicensing. The licenses that allowed the power companies to put in these dams are expiring and the companies are applying to the Federal Energy Regulatory Commission (FERC) for new licenses. This issuance of new licenses gives the public, particularly state and federal agencies, the opportunity to have a say in what the new licenses will allow. Trout Unlimited (TU), along with a number of other conservation organizations, most notably American Rivers, is involved in a number of these relicensing procedures.

Relicensing has led to a number of remarkable victories for conservation and restoration of rivers. In some instances—such as Condit Dam on the White Salmon River in Washington state, the Marmot and Little Sandy dams on the Sandy River in Oregon, and the Edwards Dam in Maine—the energy companies that owned the dams have agreed to remove them, opening up miles of habitat for salmon and steelhead. In many other instances, major improvements to flows, fish passage and the general health of the rivers have been achieved through the FERC relicensing process. The Pit River in California provides a good example of both the complexity of the process and the habitat gains that can be achieved through relicensing.

The pit 3, 4 and 5 projects are 317-megawatt hydroelectric projects, owned and operated by Pacific Gas and Electric (PG&E) that are spread across the Pit River just outside Burney, California. The projects largely control the flow of water in the river, and the Pit River and its tributary Hat Creek are home to one of California's most storied and treasured wild trout fisheries. Recreational anglers across California and the West consider the Pit River hallowed fly-fishing waters. TU and its fish conservation partners California Trout and the Northern California Council of the Federation of Fly Fishers worked for three years in this relicensing process. In November of 2003, TU joined with California Trout, California's largest utility and state and federal agencies to reach an agreement that will ultimately lead to better conditions for wild trout and other aquatic species in the Pit River in northeastern California. Stakeholders, including the U. S. Forest Service, the U. S. Fish and Wildlife Service, the U. S. National Park Service, the California Department of Fish and Game, the California Department of Parks, South Fork Irrigation District, Modoc County and others, created a consensus-based package of natural resource conditions for PG&E's new hydro license for its Pit 3, 4 and 5 projects. The conditions were sent to the FERC, along with a formal request that FERC adopts these natural resource conditions as part of PG&E's new license for the Pit River facilities. Those recommended consensus conditions will be subject to further public input and environmental review before FERC makes a final decision. The agreement was the result of 3 years of meetings, technical studies and heated negotiations.

In the end, the parties agreed upon the following resource conditions, which illustrate the positive changes that can come from FERC's process: (1) an overall increase of minimum base streamflows, (2) a guarantee of appropriate spring freshet flows designed to mimic historical flows that cleanse spawning gravels and maintain fish habitat, (3) a complete reoperation of the Lake Britton reservoir to more closely mimic run-of-the-river flow fluctuations, resulting in a more natural flow pattern during the winter and spring months, (4) a limit on ramping rates for power fluctuations, (5) an enhancement of recreational opportunities, (6) a long-term monitoring program with a fund of up to $500,000 a year for specific years, and (7) a commitment for maintenance responsibilities at the Hat Creek Fish Barrier Dam, below the wild trout waters on Hat Creek, and a dedicated maintenance and restoration fund for those waters up to $500,000. TU, along with California Trout, will sit on the Hat Creek Technical Advisory Committee, which is charged with implementing restoration projects.

As with all complex negotiations, one group will not get everything it wants. Some anglers, particularly the older ones, were concerned that the higher flows would make wading more difficult. Fishing the Pit River will be a very different experience than what it is now, but the new flows will result in more holding water for more fish. Recreational boaters wanted whitewater flows for the three reaches of the river for several weekends each year; they got two weekends on only one stretch and a $250,000 study. PG&E will benefit from the dams for 30 years, but it will be much more constrained in the operation of the facilities. More importantly—looking at the entire forest and not just one tree—the recommended license conditions to increase baseflows, to provide dynamic winter and spring flows, to implement and fund biological monitoring, and to provide money for restoration and enhancement of Hat Creek will protect and enhance the Pit River fishery throughout the 30-year term of the new license.

TU and other conservation organizations had similar success in negotiating the relicensing of a hydroelectric and water supply project on the south fork of the American River. This project involved a 21-megawatt hydroelectric and water supply project spread across the south fork of the American River, tributaries to this south fork and several high mountain lakes. By working collaboratively to understand the goals of the El Dorado Irrigation District (EID), which owned the project as well as the fish, the wildlife and the recreation needs in the area, the parties produced a comprehensive settlement that will meet the needs of both the irrigation district and the ecological resources. The agreement provides: (1) instream flow regimes to mimic natural river processes and to create healthy fish and amphibian species habitat and populations, (2) fish screens to prevent entrainment of trout, (3) enhancement of recreational opportunities, and (4) long-term monitoring and an adaptive management approach to project operations.

Much of what was accomplished on the Pit and the American rivers was due to a provision in the Federal Power Act, referred to as 10(j). This provision requires that each license issued shall include conditions for protection, mitigation and enhancement of fish and wildlife resources. The conditions are based on recommendations of the National Oceanic and Atmospheric Administration Fisheries Service, the U. S. Fish and Wildlife Service, and state fish and game agencies. Although the 10(j) recommendations can be overridden by FERC if they are deemed inconsistent with other provisions of the Federal Power Act or with other laws, the recommendations carry great weight. In most of the relicensing processes, the federal and state agencies have used their authority to argue for habitat protections, and the conservation community has worked closely with the agencies to achieve those goals. The exceptions to this rule, from TU's perspective, have involved hatcheries. This is a problem that goes back a long way.

Seventy years ago, in the midst of the great depression, a group of conservationists became concerned about the loss of the migratory bird population. Although they were concerned to some degree about unregulated hunting of waterfowl, their real concern was the loss of habitat. They understood that the loss of habitat would ultimately destroy the migratory bird resource. From their advocacy emerged a series of laws designed to protect the habitat of migratory birds, particularly migratory waterfowl. The Duck Stamp Act and the Pittman-Robinson Federal Aid in Wildlife Management Act came out of this movement, as did the National Wildlife Refuge System. That movement also spawned several organizations, including the Wildlife Management Institute and the National Wildlife Federation.

At about the same time, Pacific salmon, a highly migratory fish species, were threatened by a number of causes, including over-fishing and the siltation of streams brought on by logging in the watersheds they occupied. The major threat, however, came from the construction of dams that blocked their migration corridors and inundated their spawning habitat. From the 1930s through the 1960s, the construction of many new dams, both federal and private, brought major changes to the river systems of the Pacific Northwest and California. However, rather than adopt the habitat model used to conserve migratory bird species, managers in this case developed a different model. Hatcheries became the almost universal response to the loss of salmon and of salmon habitat. Jim Lichatowich documents the history of salmon propagation in the Pacific Northwest and California in his book, *Salmon without Rivers: A history of the Pacific salmon crisis*.

Over the past decade, evolving science has confirmed that, in many instances, hatcheries pose a significant threat to the health of wild and naturally spawning salmon populations. Wild salmon represent the primordial link between the Pacific Ocean and the Pacific Northwest, Californian and Alaskan watersheds; they are a resource that should be protected. Hatchery fish are fundamentally different from wild fish; they are spawned in plastic buckets, incubated in trays and raised in concrete raceways. With some exceptions, hatchery fish are more likely to be eaten by predators, more likely to get lost in migration and less likely to be successful in spawning. Hatchery fish spread disease and diminish the genetic integrity of wild fish through interbreeding. Finally, hatcheries promulgate a sense of false abundance by artificially inflating salmon runs, which often leads to the overfishing of wild stocks and the relaxation of legal protections for Endangered Species Act-listed stocks. Although TU believes that hatcheries can play an integral part in restoration of salmon, it is essential to examine that role of hatcheries and to begin to rethink the trade-offs between habitat and hatcheries that were made over the last century.

During the past decade, TU has become involved in relicensing procedures for dams and hatcheries that mitigated for the habitat loss when the dams were built. It is finding that, in more than a few instances, the federal and state fishery agencies are still operating under the old hatchery model. State fish and game agencies, to some extent the federal agencies, have looked at the relicensing process as a means to update and augment their hatchery programs as opposed to improving habitat conditions. This has put TU, and its allies in the relicensing process, in the awkward position of having to often oppose the recommendations of the fish and wildlife agencies.

Nowhere was this problem more pronounced than on the Sandy River in Oregon when PG&E agreed to the removal of their dams on that river system. As the negotiation process unfolded and it became clear that there was an opportunity to remove the dams that had hindered, in some cases blocked, salmon and steelhead migration into the upper basin of the Sandy River, TU and several other conservation organizations suggested that, since the dams were being removed, there was no longer any rationale for maintaining the hatchery production that had been put in place to mitigate for the effects of the dams. TU recommended that the Sandy River and its tributaries be designated a wild salmon sanctuary and that the hatchery program be discontinued. There was an immediate and strong reaction to TU's proposal from anglers who fished for the hatchery steelhead and coho that originated at hatcheries that were funded by PG&E as mitigation for the damage to fisheries caused by their dams. Their concerns were echoed by the Oregon Department of Fish and Wildlife (ODFW) biologists who operated a trap at the ladder of one of the dams. The trap allowed ODFW to pass nonhatchery coho and steelhead above the dam while returning hatchery fish, which are marked with an adipose fin clip, to the river below the dam to run the gauntlet of anglers yet again. The loss of the dam would eliminate this ability to sort fish and manipulate the fishery.

For a time, the dispute over the continued operation of the hatchery threatened the dam removal plans. Finally, a solution was reached that continued hatchery production at different locations but at approximately the same level; therefore, ODFW and the steelhead anglers finally agreed to the removal of the dams. The ODFW press release stated that the settlement that led to dam removal could not have been accomplished without agreement on continued hatchery production. There is a great deal of irony in this conflict. If the dams were the reasons that mitigation, in the form of hatchery production, were necessary, why should the hatcheries be necessary after the habitat is restored? The answer is that the natural capacity of the river to produce fish was not sufficient to meet the perceived needs of the people who wanted to catch those fish. Those opposed to the removal of the hatcheries did not value the Sandy River as one that produced steelhead; they viewed it, instead, as providing a location for a hatchery that produced fish.

TU and American Rivers faced similar challenges on the Cowlitz River in Washington. Historically, the Cowlitz River produced some of Washington's largest anadromous fish runs. Since the creation of the Cowlitz River Hydroelectric Project (Mayfield and Mossyrock dams) in 1948, Tacoma Power funded and Washington authorized the stocking of millions of juvenile chinook, coho, steelhead and cutthroat trout annually in the Cowlitz River Basin. Throughout the FERC negotiation process TU argued that agency participants and watershed stakeholders needed to look at anadromous fish production in the basin and needed new license terms to guarantee a link between the ecological health and the restoration of the remaining wild stocks in the basin and the hatchery program. To say that TU's approach was greeted with skepticism by the Washington Department of Fish and Wildlife would be an understatement.

Although much of the discussion and negotiation process centered on overall hatchery production, consensus was finally reached on important long-term innovations, such as the upgrading of existing facilities and the development of satellite rearing facilities to raise fish that were more acclimated to the wild. Further, it was agreed that hatchery production during the new license term would be managed via an adaptive management approach, and the plan would be updated in six-year intervals with stock-specific rearing strategies and the ability to ramp-down production to assure natural stocks are not being adversely affected. The agreement also funded long-term monitoring and established a sliding-scale credit mechanism to reduce Tacoma Power's production obligations based on the success of the natural stock reintroduction efforts. Perhaps most importantly, the settlement explicitly demanded that future Cowlitz River fisheries management strategies—developed pursuant to the new license order and serving as the basis for hatchery production numbers—maximize natural production of wild indigenous fish species and stocks in the basin.

Although TU considers the agreement a success much of the real benefit, in terms of wild fish production in the Cowlitz River Basin, will only be realized if the overall fish management priorities and objectives that are related to wild fish production and reintroduction are achieved. Until that time, the Cowlitz River will still be one of the biggest producers of anadromous fish in the state, and genetic and other concerns will remain. Further, Although the settlement agreement and new license certainly represent a major step forward in balancing river ecological needs with hatchery production, there will still be millions of dollars dedicated to production-oriented hatchery operations over the next license term that could have been dedicated to streamflow improvement, habitat restoration or small dam removal efforts in the Cowlitz River Basin that could have benefited wild fish production.

As TU became increasingly involved in hatchery issues through relicensing processes and in other forums, it has become clear that there is no conceptual framework for dealing with hatchery issues. Throughout the Pacific Northwest and California, there is a general agreement that hatchery reform is necessary. The alphabet soup of ongoing hatchery reform efforts include: the Hatchery Scientific Review Group (HSRG), the Hatchery and Genetic Management Plan (HGMP), and, the Artificial Production Review and Evaluation (APRE). Many of these processes overlap; for example, hatcheries that are being reviewed by the HSRG will still have to go through the HGMP process if their production affects a species listed under the Endangered Species Act.

To address the lack of an overall vision for hatchery reform, TU contracted with two of the most respected salmon scientists in the Columbia Basin, Jim Lichatowich and Dr. Rick Williams, to look at hatcheries from a broader ecological perspective. The result of their collaboration is a paper, entitled *Integrating Artificial Production with Salmonid Life History, Genetic and Ecosystem Diversity: A Landscape Perspective.*

Landscape Perspective

The thrust of the paper refers to the dichotomy of the different approaches to the conservation and management of highly migratory avian species as opposed to highly migratory fish. For migratory birds, a habitat model was adopted, and, since the institution of the Duck Stamp in the 1930s, hundreds of thousands of acres of wetlands have been protected by a series of funding mechanisms as well as regulatory approaches. For migratory fish species, specifically salmon, a factory model was adopted that allowed the despoliation of rivers while maintaining the promise of continued fisheries.

Hatcheries should be viewed as integral parts of the river systems in which they are located. The region should adopt a landscape approach to hatcheries and hatchery management. A landscape approach to hatcheries would link artificial and natural production with the landscape of the basin and the subbasin in which the hatchery is located. It would require the integration of the hatchery with the ecology, geology and climate of the basin. It would also require that the salmon produced at the hatchery fit within the ecology of the basin in terms of number of fish released as well as genetics and life history.

Although TU believes Lickatowich and Williams' paper is valuable from a scientific perspective, perhaps the real value will be its contribution to the debate about hatcheries. Those authors strongly believe that we need to change the metaphors for hatcheries. For over a century, society has regarded hatcheries essentially as factories that are connected to the watersheds in which they are located only as a matter of production efficiency, much like a paper mill might be located near a river for water supply and transportation needs. They argue that, rather than thinking of hatcheries as factories, a more appropriate metaphor would be to think of them as tributaries to the larger watersheds in which they are located. This adoption of a new metaphor would shape the process of hatchery reform. If a hatchery is thought of as a tributary within a watershed, then it would follow that the genetics of the fish produced there would match the genetics of the fish in the watershed. The number of fish produced there would not overwhelm the ecological capacity of the watershed, including the estuaries and, ultimately, the ocean. The hatchery production would also fit within the nutrient cycle of the watershed. Finally, many hatchery reform efforts currently being undertaken are directed at improvements inside the hatchery fence, such as improving water supplies or mimicking more natural rearing conditions by adding evergreen trees to the raceways. The real effort of hatchery reform should take place outside the hatchery fence, linking the hatchery production to the natural production of the watershed.

The connection between hatchery reform and hydrorelicensing is obvious. For hydrorelicensings that involve hatcheries, TU will advocate that those hatcheries be modified to fit the landscape model. For state and federal agencies with 10(j) authority, I would urge them to resist the temptation to use the relicensing process to "gold plate" hatcheries—to increase hatchery production—or indeed, to protect hatchery production that is inconsistent with a watershed approach. In the long-term, TU believes that approach is essential to integrate hatchery production with the natural production and the ecology of the watersheds in which the hatcheries are located. Hydrorelicensing can assist in achieving that goal.

Reference List

Hatchery Genetic Management Plans. 2000. *65 Federal register*. 42421–81. July 10.

Hatchery Scientific Review Group 2003. Hatchery reform recommendations for the Puget Sound and coastal Washington hatchery reform project. In *Long live the kings*, eds. Lars Mobrand, John Barr, Lee Blankenship, Don Campton, Trevor Evelyn, Conrad Mahnken, Paul Seidel, Lisa Seeb and Bill Smoker. Seattle, Washington: Hatchery Scientific Review Group.

Lichatowich, Jim. 1999. *Salmon without rivers: A history of the Pacific salmon crisis*. Washington, DC: Island Press.

Northwest Power and Conservation Council. 2003. *Artificial Production Review and Evaluation (APRE): Basin-Level Report Draft. Document 2003–17*. http://www.nwcouncil.org/library/2003/2003-17.htm

Williams, Richard N., James A. Lichatowich, Phillip R. Mundy, and Madison Powell. 2003. *Integrating artificial production with salmonid life history, genetic and ecosystem diversity: A landscape perspective*. Portland, Oregon: Trout Unlimited, West Coast Conservation Office.

Bird Responses to Harvesting Switchgrass Fields for Biomass

Louis B. Best
Iowa State University, Department of Natural Resource Ecology and Management
Ames

Les D. Murray
Iowa State University, Department of Natural Resource Ecology and Management
Ames

Introduction

The Conservation Reserve Program (CRP) was established in 1985 to reduce soil erosion and to improve water quality by reimbursing farmers for removing highly erodible cropland from agricultural production and for planting it to perennial cover, commonly grasses in the Midwest (Heard et al. 2000). Many bird species are more abundant and more frequently nest in CRP fields than in rowcrop fields (Johnson and Schwartz 1993, Best et al. 1997), but as CRP contracts expire, some CRP fields will be returned to rowcrop production (Kurzejeski et al. 1992). One proposed alternative to returning CRP land to rowcrops is to use it to produce switchgrass for use as a biomass fuel.

The use of switchgrass as a biomass fuel would provide a domestic energy source, maintain the ecological benefits of CRP and create habitat for grassland birds (Paine et al. 1996). Switchgrass has been chosen for use as a biomass fuel because it produces more biomass per area than other herbaceous plant species (Brower et al. 1993), and it requires minimal maintenance once established in a field. Because the biomass is harvested during the fall and winter, breeding birds would not be directly affected by mowing the fields as is the case with hay fields (e. g., Warner et al.1989, Frawley and Best 1991). Changes in vegetation structure resulting from harvest, however, might influence breeding bird abundances and nest success (e.g., Dwernychuk and Boag 1972, Frawley and Best 1991, Horn et al. 2000) and winter bird abundances.

In southern Iowa, studies were undertaken to determine the feasibility of using switchgrass as a biomass fuel. Our research focused on the effects such a program could have on grassland birds. There were three aspects to our study.

1. We compared avian abundance during the breeding season and nest success among total-harvested, strip-harvested (alternating cut and uncut strips) and nonharvested CRP switchgrass fields (Murray and Best 2003). We predicted that strip-harvested fields would provide habitat for more bird species because the presence of cut and uncut areas within each field would provide greater habitat heterogeneity. We expected fields with wider strips to support higher abundances of some bird species because of minimum area (Herkert 1994) or territory size requirements. Thus, we compared strips of different widths.
2. We evaluated the effects of biomass harvest on winter bird use of switchgrass fields, by comparing bird abundances among total-harvested, strip-harvested and nonharvested switchgrass fields (Murray 2002). In the winter, CRP fields provide more protective cover from predators and adverse weather conditions than rowcrop fields, and several bird species are more abundant or more widely distributed in CRP fields than in rowcrop fields (Best et al. 1998). We expected that harvesting switchgrass fields would remove most of the protective cover and could limit bird use of biomass fields to those with cover available in adjacent habitats.
3. We modeled the regional effects of converting rowcrop (corn, soybeans) and CRP switchgrass fields to biomass production (Murray et al. 2003). Land-use coverage was classified for the Rathbun Lake Watershed in southern Iowa, where tests were underway to evaluate the use of switchgrass as a biomass fuel. Bird abundance values for each habitat were used to model bird abundances in the watershed before and after the conversions. Two scenarios were evaluated, one with conversion to total-harvested biomass fields and one with conversion to strip-harvested biomass fields.

Study Area and Methods

Details of the study area and research methods are described elsewhere (Murray 2002, Murray and Best 2003, Murray et al. 2003). Only those aspects necessary to understand the general study design are presented herein.

Study Area

To evaluate the use of switchgrass as a biomass fuel, Chariton Valley Resource Conservation and Development, Inc., received permission from the U. S. Department of Agriculture to harvest 4,000 acres (16,019 ha) of CRP switchgrass fields in southern Iowa. Our study sites were located in the rolling hills of the Southern Iowa Drift Plain (Prior 1991), where the primary land cover is grassland (pasture, hay) interspersed with rowcrop and woodland.

We used 21 CRP fields that had been planted to switchgrass 2 to 12 years before our study and that ranged from 9.4 to 32.1 acres (3.8–13.0 ha). In 1999 and 2000, we evaluated three treatments: (1) total-harvested fields that were completely harvested, (2) strip-harvested fields that consisted of alternating cut and uncut strips of different widths (197 feet [60 m] cut, 131 feet [40 m] uncut or 98 feet [30 m] cut, 66 feet [20 m] uncut) with 60 percent of the field being harvested, and (3) nonharvested fields that were not manipulated and served as controls. During November through March the switchgrass was cut to a height of about 3.5 inches (9 cm), baled and removed from the harvested fields. Fertilizer and herbicides were applied to most harvested fields in June or July of both years.

Vegetation

The vegetation on fields was characterized by measuring vegetation density, height, coverage and litter depth. To characterize conditions during the breeding season, vegetation density and height were measured once every 2 weeks from mid-May to late July; plant percent coverages and litter depth were measured in mid-June and mid-July. In the winter, vegetation density and height and litter depth were measured once in January or February.

Bird Abundances during the Breeding Season and Nest Success

We surveyed birds weekly on each field from May 15 to July 15 in 1999 and 2000 by using fixed-width, nonoverlapping transects that covered each field entirely. The strip type (cut, uncut) in which each bird was first seen also was recorded. We systematically searched five fields of each treatment for nests three times between May 15 to July 15. Nests found during other activities also were recorded for all 21 fields. We checked nests every 2 to 4 days to determine their outcome.

Bird Abundances during the Winter

We surveyed birds once in January and once in February in 2000 by using fixed-width, nonoverlapping transects that covered each field entirely. Total-harvested and strip-harvested fields were surveyed after they were harvested.

Modeling Effects of Land-use Conversion

A geographic information system (GIS) land-use coverage was created for the Rathbun Lake Watershed in southern Iowa. Bird abundance values for each habitat in the watershed were obtained from previous studies. We selected 13 bird species associated with grassland that are management priority species (Fitzgerald and Pashley 2000), game species, or abundant in switchgrass or rowcrop fields in Iowa or Missouri (Best et al. 1997, McCoy et al. 2001, Murray and Best 2003). We then modeled the regional effects on bird abundance if marginal rowcrop fields (with greater erosion and leaching potential) and CRP switchgrass fields wer converted to biomass production. Two scenarios, one with conversion to total-harvested biomass fields and one with conversion to strip-harvested biomass fields, were modeled.

Results and Discussion

Vegetation

Breeding Season. Averaged over the season, the vegetation height and density were similar among treatments. The litter layer was deeper and the coverage of standing dead vegetation greater in nonharvested fields than in total-harvested fields. Strip-harvested fields were intermediate. Switchgrass coverage was greater in total-harvested and strip-harvested fields than in nonharvested fields. Cut and uncut strips in strip-harvested fields were similar in vegetation structure to total-harvested and nonharvested fields, respectively.

The phenology of vegetation growth differed among treatments. Early in the growing season, the vegetation density was greatest in nonharvested fields, least in total-harvested fields and intermediate in strip-harvested fields. Later in the season the vegetation density of all three treatments converged, at least initially (see Murray and Best 2003).

Winter. Residual vegetation in total-harvested fields was significantly shorter and sparser than that in the other two treatments. Vegetation structure in uncut strips of strip-harvested fields was similar to that in nonharvested fields, and cut strips were similar to total-harvested fields. Mean litter depth was similar among treatments.

Bird Abundances during the Breeding Season and Nest Success

The mean number of bird species observed per field was similar among treatments. No major shifts in species richness resulted from the harvest because, in harvested areas, the absence of species that are common in tall, dense vegetation was offset by the presence of species that prefer shorter, sparser vegetation. Bird abundances did not differ in fields with different strip widths. Abundances of only two of the 18 bird species abundant enough to test for treatment differences were significantly different among treatments. Both species have specific habitat requirements that were affected by the harvest of the fields for biomass. Grasshopper sparrows (*Ammodramus savannarum*) are more abundant in grassland with short, sparse vegetation and a shallow litter layer (Herkert et al. 1993). In our study this species was observed more often in total-harvested fields and cut strips of strip-harvested fields than in nonharvested fields and uncut strips. Sedge wrens (*Cistothorus platensis*) showed the opposite trend in abundance from grasshopper sparrows and were more abundant in nonharvested fields than total-harvested fields.

Within strip-harvested fields, the rate of use of uncut strips was greater than that of cut strips for four species (red-winged blackbird [*Agelaius phoeniceus*], song sparrow [*Melospiza melodia*], common yellowthroat [*Geothlypis trichas*] and sedge wren). The grasshopper sparrow was the only species that preferred cut strips to uncut strips.

Nests of 20 species were found in switchgrass fields. Red-winged blackbirds and common yellowthroats accounted for 56 and 28 percent of the nests found, respectively. Red-winged blackbirds nested in fields of all three treatments, but in 1999 the most nests were found in nonharvested fields whereas in 2000 the greatest proportion of nests was found in total-harvested fields. Common yellowthroat nests were most abundant in nonharvested fields and least numerous in total-harvested fields. In strip-harvested fields, more than 85 percent of redwing and yellowthroat nests were found in uncut strips.

Residual vegetation in nonharvested fields and uncut strips provided nesting habitat for other species. Northern harriers (*Circus cyaneus*) and ring-necked pheasants (*Phasianus colchicus*) are ground-nesting species that often begin nesting in April (MacWhirter and Bildstein 1996, Clark and Bogenschutz

1999). The residual vegetation in nonharvested areas provided nest cover for these species.

The removal of residual vegetation in harvested fields seemed to have a negative effect on nest success, nest predation and brood parasitism in biomass fields. For all species combined, the proportion of nests that were successful was greater in nonharvested fields (59%) than in total-harvested (40%) or strip-harvested (33%) fields. Red-winged blackbird daily nest survival rates were greatest in nonharvested fields, but common yellowthroat daily nest survival rates did not vary among treatments.

Predation accounted for 78 percent and brown-headed cowbird (*Molothrus ater*) brood parasitism for 9 percent of nest failures; a lower proportion of nests failed because of predation in nonharvested fields (35%) than in total- (48%) or strip-harvested (47%) fields. The incidence of cowbird brood parasitism was similar among treatments for red-winged blackbirds, but common yellowthroat nests in total-harvested and strip-harvested fields were more likely to be parasitized than those in nonharvested fields. Removal of the residual vegetation in harvested fields may allow easier movement and may increase the searching efficiency of nest predators (e. g., Crabtree et al. 1989) and may facilitate locating nests of some species (e. g., common yellowthroat) by cowbirds by making observation of adults near the nest easier (Clotfelter 1998).

Bird Abundances during the Winter

Mean total bird abundance in strip-harvested fields was more than twice that in fields of the other two treatments. Ring-necked pheasants were observed only in nonharvested fields and uncut portions of strip-harvested fields. American tree sparrows (*Spizella arborea*) were observed most frequently in strip-harvested fields, and most of the observations were in the uncut strips. Song sparrows (*Melospiza melodia*) were only seen in strip-harvested fields, and all observations were in uncut strips.

Availability of both food and protective cover affects bird abundance (Beck and Watts 1997). The greater residual vegetation in nonharvested fields and uncut portions of strip-harvested fields might provide more protection from predators (Watts 1990), greater thermal benefits or a better food source than harvested areas. But, the removal of vegetation in harvested areas might have made fallen seeds more accessible to the sparrows because they commonly forage in open areas (West 1967, Whalen and Watts 2000). In contrast to total-harvested fields, strip-harvested fields provided sparrows with foraging sites and protective cover in close proximity. Ring-necked pheasants probably spend more time foraging in nearby rowcrop fields than in switchgrass fields (Bogenschutz et al. 1995), but protective cover is important for pheasant survival in the winter (Gabbert et al. 1999). Thus, pheasants probably used switchgrass fields primarily for escape and roosting cover, not as foraging sites.

Modeling Effects of Land-use Conversion

Total abundance of grassland bird species of management priority (e. g., grasshopper sparrow, sedge wren) increased in both biomass scenarios compared with the current land use. The same was true for the ring-necked pheasant, an important upland game species. The abundances of three species that commonly nest in rowcrop fields (horned lark [*Eremophila alpestris*], killdeer [*Charadrius vociferous*], vesper sparrow [*Pooecetes gramineus*]) and brown-headed cowbird abundance decreased in each of the biomass scenarios.

Because grassland bird species differ in their habitat requirements (Herkert et al. 1993), maintaining both harvested and nonharvested switchgrass fields in a region would benefit the greatest number of species. Conversion of rowcrop fields to fields used for biomass production could affect aspects of avian biology other than availability of breeding and winter habitat. Replacing rowcrops with switchgrass would reduce the availability of waste grain. Corn is an important food source for pheasants (Boghenschutz et al. 1995) and other wildlife (Martin et al. 1961), particularly during the winter. The establishment of food plots (areas of rowcrop grown to produce food for wildlife) in biomass fields is a feasible management option that would provide food for pheasants and other species.

Conclusion

Switchgrass biomass production could provide benefits similar to those of CRP in that it removes land from rowcrop production, reduces soil erosion compared with rowcrops and provides habitat for wildlife (Johnson and Schwartz 1993, Best et al. 1997). In particular, some grassland birds would benefit from switchgrass production because grassland habitat would be added to the landscape, and the fall and winter harvest does not cause the high rates of nest loss associated with spring and summer mowing. Converting rowcrop land to switchgrass biomass production would decrease habitat for some bird species common to cropland, but none of theses species are management priorities, and, in the Midwest, there is an abundance of rowcrop habitat.

Because habitat requirements differ among bird species (Herkert et al. 1993), a mixture of harvested and nonharvested switchgrass fields in a landscape would likely benefit more grassland bird species than totally harvesting or strip harvesting all biomass fields. Nonharvested areas provide habitat for species that prefer tall vegetation (e. g., sedge wren, northern harrier), whereas those that prefer short, sparse vegetation (e. g., grasshopper sparrow) are more abundant in harvested areas. This mixture of habitats could be accomplished by selection of some fields to remain idle or through a rotational harvest regime.

There are benefits to both strip-harvesting and total-harvesting switchgrass fields. Strip-harvested biomass fields provided nesting habitat for all three species of management concern (grasshopper sparrow, sedge wren, northern harrier) because of the presence of both tall, dense vegetation and of short, sparse vegetation. But the probability of occurrence and density of grasshopper sparrows are lower in small habitat patches than in large ones (Herkert 1994), and, in our study, these birds were much more abundant in total-harvested fields than in strip-harvested fields. A mixture of total-harvested and nonharvested fields might be a better option than strip-harvested fields to provide habitat for all three species during the breeding season. In contrast, providing both food and cover in the same habitat patch would benefit birds during winter. Nonharvested switchgrass fields provide protective winter cover for American tree and song sparrows and ring-necked pheasants, but harvesting such fields for biomass drastically reduces this cover. Strip-harvested biomass fields provide foraging areas for song and American tree sparrows while retaining protective cover in close proximity. If switchgrass fields are to be harvested, harvesting them in alternating cut and uncut strips is more beneficial to the winter bird community than harvesting them completely.

If switchgrass biomass fields are to be managed for bird use, options to minimize negative effects on the bird community should be considered. Although the birds that used switchgrass fields are predominantly habitat generalists, planning efforts should focus on species of management concern because they have the most to gain or lose from land-use changes. The taller, denser growth resulting from fertilizer use might severely limit the benefits of switchgrass fields for species preferring shorter vegetation (e. g., grasshopper sparrows), and reduction of forb abundance through herbicide use would create less attractive habitat for species that nest in forbs (e. g., dickcissels [*Spiza americana*]). The number of species using biomass fields might also be reduced because of less variation in vegetation structure created by the use of herbicides and fertilizer. In addition, application of fertilizer during the breeding season causes some nest failures. Use of multiple species plantings in biomass fields would create habitat heterogeneity not found in switchgrass monocultures and would provide habitat for a more diverse bird community.

Future Research Needs

There are future research needs if switchgrass fields (or other herbaceous plantings) continue to be used for biomass production. We did not evaluate the effects of field size and shape and the juxtaposition of habitats on bird abundance in switchgrass fields. These are known to influence bird habitat use (Herkert 1994, Helzer and Jelinski 1999). An understanding of how these factors affect bird occurrence could be used in the selection of potential fields for biomass production. Landscape-level effects on bird abundance should be evaluated because abundance of some bird species is related to the habitat composition in the surrounding landscape (e. g., Ribic and Sample 2001, Fletcher and Koford 2002). This has important implications on biomass production planning at the regional level. More detailed studies of nest success and long-term population dynamics of species that nest in switchgrass fields, particularly those of management priority, should be conducted to determine if biomass fields will support stable populations of grassland birds. In our study, nest success was lower in harvested fields than in nonharvested fields for red-winged blackbirds because of increased predation and human disturbance; common yellowthroats were not affected. Future research also should examine the effects of long-term management on bird use of biomass fields, particularly if switchgrass biomass production is increased. This will help ensure that species not currently of management concern do not become so.

Reference List

Beck, C. W., and B. D. Watts. 1997. The effect of cover and food on space use by wintering song and field sparrows. *Canadian Journal of Zoology*. 75:1,636-41.

Best, L. B., H. Campa, III, K. E. Kemp, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks, Jr., and S. R. Winterstein. 1997. Bird abundance and nesting in CRP fields and cropland in the Midwest: A regional approach. *Wildlife Society Bulletin*. 25:864–77.

Best, L. B., H. Campa, III, K. E. Kemp, R. J. Robel, M. R. Ryan, J. A. Savidge, H. P. Weeks, Jr., and S. R. Winterstein. 1998. Avian abundance in CRP and crop fields during winter in the Midwest. *American Midland Naturalist*. 139:311–24.

Bogenschutz, T. R., D. E. Hubbard, and A. P. Leif. 1995. Corn and sorghum as a winter food source for ring-necked pheasants. *Journal of Wildlife Management*. 59:776–84.

Brower, M. C., M. W. Tennis, E. W. Denzler, and M. M. Kaplan. 1993. *Powering the Midwest: Renewable electricity for the economy and the environment*. Washington, DC: Union of Concerned Scientists.

Clark, W. R., and T. R. Bogenschutz. 1999. Grassland habitat and reproductive success of ring-necked pheasants in northern Iowa. *Journal of Field Ornithology*. 70:380–92.

Clotfelter, E. D. 1998. What cues do brown-headed cowbirds use to locate red-winged blackbird host nests? *Animal Behaviour*. 55:1,181–9.

Crabtree, R. L., L. S. Broome, and M. L. Wolfe. 1989. Effects of habitat characteristics on gadwall nest predation and nest-site selection. *Journal of Wildlife Management*. 53:129–37.

Dwernychuk, L. W., and D. A. Boag. 1972. How vegetative cover protects duck nests from egg-eating birds. *Journal of Wildlife Management*. 36:955–8.

Fitzgerald, J.A., and D. N. Pashley. 2000. *Partners in Flight bird conservation plan for the Dissected Till Plains (physiographic area 32)*. Partners in Flight. http://www.blm.gov/wildlife/pl_32sum.htm.

Fletcher, R. J., Jr., and R. R. Koford. 2002. Habitat and landscape associations of breeding birds in native and restored grasslands. *Journal of Wildlife Management*. 66:1,011–22.

Frawley, B. J., and L. B. Best. 1991. Effects of mowing on breeding bird abundance and species composition in alfalfa fields. *Wildlife Society Bulletin*. 19:135–42.

Gabbert, A. E., A. P. Leif, J. R. Purvis, and L. D. Flake. 1999. Survival and habitat use by ring-necked pheasants during two disparate winters in South Dakota. *Journal of Wildlife Management*. 63:711–22.

Heard, L. P., A. W. Allen, L. B. Best, S. J. Brady, W. Burger, A. J. Esser, E. Hackett, D. H. Johnson, R. L. Pederson, R. E. Reynolds, C. Rewa, M. R. Ryan, R. T. Molleur, and P. Buck. 2000. *A comprehensive review of Farm Bill contributions to wildlife conservation, 1985–2000: Technical report USDA/NRCS/WHMI-2000*. http://www.whmi.nrcs.usda.gov/technical/comprehensivereview.html

Helzer. C. J., and D. E. Jelinski. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications*. 9:1,448–58.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. *Ecological Applications*. 43:461–71.

Herkert, J. R., R. E. Szafoni, V. M. Kleen, and J. E. Schwegman. 1993. *Habitat establishment, enhancement and management for forest and grassland birds in Illinois*. Springfield, Illinois: Division of Natural Heritage, Illinois Department of Conservation.

Horn, D. J., and R. R. Koford. 2000. Relation of grassland bird abundance to mowing of Conservation Reserve Program fields in North Dakota. *Wildlife Society Bulletin*. 28:653–9.

Johnson, D. H., and M. D. Schwartz. 1993. The Conservation Reserve Program and grassland birds. *Conservation Biology*. 7:934–7.

Kurzejeski, E. W., L. W. Burger, Jr., M. J. Monson, and R. Lenker. 1992. Wildlife conservation attitudes and land use intentions of Conservation Reserve Program participants in Missouri. *Wildlife Society Bulletin*. 20:253–9.

MacWhirter, R. B., and K. L. Bildstein. 1996. Northern harrier (*Circus cyaneus*). In *The birds of North America, number 210*, eds. A. Poole, and F. Gill., 31. Washington, DC: The Academy of Natural Sciences, Philadelphia, Pennsylvania, and the American Ornithologists' Union.

Martin, A.C., H. S. Zim, and A. L. Nelson. 1961. *American wildlife and plants: A guide to wildlife food habits*. New York, New York: Dover Publications.

McCoy, T. D., M. R. Ryan, L. W. Burger, Jr., and E. W. Kurzejeski. 2001. Grassland bird conservation: CP1 vs. CP2 plantings in Conservation Reserve Program fields in Missouri. *American Midland Naturalist*. 145:1–17.

Murray, L. D. 2002. *Avian response to harvesting switchgrass for biomass in southern Iowa*. M. S. Thesis, Department of Natural Resource Ecology and Management, Iowa State University, Ames.

Murray, L. D., and L. B. Best. 2003. Short-term bird response to harvesting switchgrass for biomass in Iowa. *Journal of Wildlife Management*. 67:611–21.

Murray, L. D., L. B. Best, T. J. Jacobsen, and M. L. Braster. 2003. Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. *Biomass and Bioenergy*. 25:167–75.

Paine, L. K., T. L. Peterson, D. J. Undersander, K. C. Rineer, G. A. Bartelt, S. A. Temple, D. W. Sample, and R. M. Klemme. 1996. Some ecological and socio-economic considerations for biomass energy crop production. *Biomass and Bioenergy*. 10:231–42.

Prior, J. C. 1991. Landforms of Iowa. Iowa City, Iowa: University of Iowa Press.

Ribic, C. A., and D. W. Sample. 2001. *Associations of grassland birds with landscape factors in southern Wisconsin. American Midland Naturalist*. 146:105–21.

Warner, R. E., and S. L. Etter. 1989. Hay cutting and the survival of pheasants: A long-term perspective. *Journal of Wildlife Management*. 53:455–61.

Watts, B. D. 1990. Cover use and predator-related mortality in song and savannah sparrows. *Auk*. 107:775–8.

West, G. C. 1967. Nutrition of American tree sparrows during winter in Illinois. *Ecology*. 48:58–67.

Whalen, D. M., and B. D. Watts. 2000. Interspecific variation in extraction of buried seeds within an assemblage of sparrows. *Oikos*. 88:574–84.

Surface Mining and Wildlife Resources: Addition and Subtraction on the Cumberland Plateau

John J. Cox
University of Kentucky, Department of Forestry
Lexington

David S. Maehr
University of Kentucky, Department of Forestry
Lexington

Humans have used fossil fuels for various purposes since at least the beginning of written history. The Babylonians used native asphalt, Persian kings had their food cooked in caves over natural gas seepages and both the Chinese and the Europeans burned coal to generate heat for over a thousand years (Ayres 1956). However, the industrial age of the late 19th century, the advent of electrical power and the invention of the internal combustion engine catapulted the extraction, use and dependence on fossil fuels to the forefront of human society. As such, fossil fuels continue to be the dominant resource used to satisfy the appetite of an increasingly energy-hungry world.

Coal is one of the most abundant and economically important fossil fuels in the United States. It is used to fuel over half of all power plants in the United States, including 95 percent of those in Kentucky (Kentucky Coal Association 2000), and 99 percent of those in West Virginia (West Virginia Office of Miners Health Safety and Training 2004). Since the mid-1980s production of this nonrenewable resource rose 23 percent and nationwide consumption increased by 16 percent (Kentucky Coal Association 2000). With a reserve base estimated at 375 billion short tons (340 metric tons), geologists have forecast that there is enough coal remaining to meet projected energy demands for the next two centuries based on current consumption rates (U. S. Geological Survey 2004). Thus, barring a rapid switch to alternative energy sources, it appears that King Coal will dominate the energy market for the immediate future.

One of the largest coal fields in the eastern United States lies beneath the Cumberland Plateau physiographic region. This deeply eroded, ancient, forested tableland comprises the southwestern portion of the Appalachian Mountains and extends northeast to southwest through parts of six states, including Kentucky, West Virginia, Virginia, Tennessee, Alabama and Georgia. Historically, the Cumberland Plateau was covered by a nearly contiguous deciduous forest composed of nearly 30 hardwoods in some areas (Braun 1950). Today, this ecosystem is recognized for its high biodiversity and as a globally outstanding ecological resource.

As early as the 1740s, explorers of the Cumberland Plateau noted exposed coal deposits that were occasionally found during their travels (Eller 1982). Only as a result of late 19th century industrialization and the arrival of railroads did widespread extraction of coal begin in this region (Eller 1982). This combination of economic and technological events facilitated resource exploitation in areas, such as eastern Kentucky, and transformed North America's largest contiguous human subculture (Shackleford and Weinberg 1977) from a largely agrarian, self-sufficient society to one dependent upon hourly wages and the provisions of the coal company town (Eller 1982).

It is doubtful that the buyers and sellers of mineral rights a century ago envisioned the types and scale of modern coal mining practiced today in the Cumberland Plateau. The impacts of surface mining on local biota are a function of extraction method, spatial extent of operations and choice of postmining land use that often dictates the method and species used for reclamation. Two types of surface mining that commonly occur in this region are contour mining and mountaintop-removal mining. Contour mining removes the overburden along topographic gradients to expose underlying coal seams. Successive cuts that follow the coal seam around the side of the hill are made until the overburden becomes too thick to make further exposure of the coal economically feasible.

Mountaintop removal (MTR) is the latest and most controversial method of surface mining. This method consists of the removal of the highest elevations of a mountain to expose and remove successive layers of overburden and coal. Frequently, the overburden is placed into adjacent valleys to create what are known as hollowfills, a process that has resulted in the burial of over 1,200 miles (1448 km) of intermittent and perennial streams in Appalachia (Ohio River Valley Environmental Coalition 2004). Large-scale mines utilizing MTR have created contiguous areas of flattened terrain that can exceed 10,000 acres (4,047 ha). During the past 20 years, 612,000 acres (247,669 ha)—an area slightly less than the federal holdings of the entire Daniel Boone National Forest in eastern Kentucky—of mined land have been reclaimed in Kentucky (Kentucky Environmental Quality Commission 2001). Reclamation typically results in the creation of extensive grassland that perforate an otherwise forested landscape.

The regional conflict between resource extraction and ecosystem preservation is exemplified by recent events surrounding Black Mountain, the highest peak in the Cumberland Plateau. Straddling the Kentucky-Virginia border, Black Mountain reaches an elevation of 4,145 feet (1,263 m) and rises several hundred feet above neighboring mountains. The top of the mountain supports a unique biota that is rare or nonexistent in other portions of the Cumberland Plateau. Although Black Mountain has been logged at least twice in the past 200 years and, although it has been partially stripped for coal along some midelevation seams, most of it was still intact in the late 1990s when landowners were on the verge of using MTR methods to obtain coal found in its high-elevation seams. The threat of decapitating the state's tallest peak, however, unleashed a public outcry that caused the state of Kentucky to purchase the mineral rights to the mountain at elevations above 3,800 feet (1,158 m).

Initially, Cumberland Plateau topography constrained human settlement primarily to coves, hollows and gaps. But, the premining impact of European settlers on their environment was far from benign. Before the Civil War, bison (*Bison bison*), elk (*Cervus elaphus*), mountain lion (*Puma concolor*), wolf (*Canis* spp.), beaver (*Castor canadensis*) and otter (*Lutra canadensis*) were extirpated from much of the region. Forests were logged for home construction and commercial use, understory communities were denuded and soils were compacted by herds of livestock. The forest ecosystem of the region proved remarkably resilient to these and other changes that accompanied more intensive logging. However, the methods used to extract coal profoundly altered the landscape and its biodiversity to an extent that may exceed the last glaciation. This paper discusses some of the ecological effects of coal mining as manifested in changes to and implications for regional fauna. Although we focus our discussion largely on surface mining impacts in eastern Kentucky, the patterns of change that have occurred there are typical of those found elsewhere in the Cumberland Plateau.

Impacts of Surface Mining on Fauna

Among 14 energy sources, the use of coal from surface mining has the greatest potential to adversely affect fish and wildlife (Hatcher 1978). Surface mining can impact fauna in a variety of negative and positive ways. Physical processes associated with mining, such as blasting, earth-moving, and vehicular traffic, are obvious and immediate causes of wildlife mortality and habitat destruction. Mine waste can alter the chemical and physical qualities of aquatic systems in ways that extirpate sensitive species or entire communities. Loss of habitat due to mining can be a significant source of mortality for those species that have specialized life requirements, and fragmentation of habitat can isolate organisms by creating physical barriers that impede dispersal. Finally, organisms in surrounding landscapes are not immune to the effects of surface mining.

Disturbances, such as blasting, may cause some species to abandon traditional home ranges and breeding grounds. Edge effects at the mine-forest interface can negatively impact species within adjacent forest by altering climatic conditions and hydrologic regimes, or by increasing predation or parasitism. Conversely, reclaimed surface mines can create conditions favorable to grassland, early successional or wetland species that may or may not have been historically present.

Aquatic Systems

Acid mine drainage (AMD) produced by surface and deep mining is characterized by a low pH and often a high concentration of iron and other metals, such as aluminum and copper (Haigh 1993). These conditions can cause direct toxicological effects to aquatic biota, such as diatoms, benthic invertebrates and fish (Matter et al. 1978, Winger 1978, Short et al. 1990, Verb and Vis 2000) that lower biodiversity by shifting community composition towards more metal-tolerant taxa (Short et al. 1990). Girts and Kleinmann (1986) estimated that as much as 12,430 miles (20,000 km) of Appalachian streams were affected by AMD.

Surface mining also causes extensive sedimentation of streams that negatively impacts taxa, such as crayfish (Holl et al. 2001) and mussels (Houpe 1993). The Cumberland River system of Kentucky and Tennessee, for example, has been severely impacted by upstream surface mining activities; nearly one-quarter of the mussel species once found there are extinct or listed as federally endangered (Layzer et al. 1993). The blackside dace (*Phoxinus cumberlandensis*), a federally threatened fish found in only 30 separate streams in the upper Cumberland River watershed (United States Fish and Wildlife Service 1987), is a well-publicized example of a fish species that has been harmed due to habitat degradation related to surface mining (Starnes and Starnes 1981, O'Bara 1985). Indeed, a number of freshwater impoundments in this region are being filled at a rapid rate by high sediment loads from surface mines (Mueller 2003).

Surface mine wetlands constructed for water quality improvement or wildlife habitat can benefit some groups of organisms. Amphibians and semiaquatic snakes (Lacki et al. 1992, Keck 1998) as well as waterfowl, wading and passerine birds (Perkins and Lawrence 1985, Urbanek and Klimstra 1986) utilize these wetlands. In addition, constructed wetlands serve as habitat for prey (e. g., raccoon [*Procyon lotor*]) of some mammals, while providing others (e. g., elk) with a source of water for drinking and cooling during summer. Further, some ponds on surface mines are stocked with fish that are consumed by wildlife and humans.

Terrestrial Systems

Soils constitute the foundation for terrestrial communities. Boettcher and Kalisz (1990) observed that eastern deciduous forest soils were composed of a mosaic of site-specific properties determined by the occurrence and composition of different plant species. Surface mining dramatically alters soils in these communities by causing the loss of vital microbes (Mummey et al. 2002) and invertebrate components, such as earthworms (Abdul-Marashi and Scullion 2003). Further, the homogenization of soil profiles disrupts nutrient cycling. In many parts of the Cumberland Plateau, modern reclamation practices allow revegetation on mineral overburden absent of stockpiled soils. Thus, natural colonization rates and diversification can be impeded by artificially xeric conditions with little to no organic matter.

Reclamation is used to reduce the environmental impacts caused by the extraction of coal by preparing the disturbed area for designated postmining uses mandated by state and federal laws, such as the 1998 Kentucky Surface Mining Law and the 1977 Surface Mining Control and Reclamation Act, respectively. Coal operators are required to post performance bonds to assure fulfillment of reclamation requirements—typically, a certain percentage of bare ground must be successfully revegetated. The cheapest and least labor-intensive method to reclaim surface mines is to grade the soils and establish a fast-growing and hardy herbaceous groundcover via hydroseeding to minimize immediate subsidence and erosion. Unfortunately, grading often compacts the soils to a degree that makes root penetration and establishment of many plants difficult to impossible, thus limiting the choice of species that can be successfully used for reclamation.

Nearly 6,600 exotic species have been introduced to North America since European colonization, and they now comprise approximately 5 percent of the continental biota (Cox 1999). Their cheap cost and the ability of some exotic plant species to quickly establish on minespoil make them an attractive and thus frequently used component to meet reclamation objectives. Exotic plants commonly used for reclamation, such as Kentucky-31 tall fescue (*Festuca arundinacea*), lespedeza (*Lespedeza* spp.), crown vetch (*Coronilla varia*) and autumn olive (*Elaeagnus umbellata*), are highly invasive and frequently found on natural resource managers least wanted plant lists (e. g., Kentucky Exotic Pest Plant Council). Although fast growing cover species may initially reduce erosion, they can form dense layers or thickets that prevent natural recolonization of reclaimed areas by native species or impede reforestation efforts (Torbert et al. 2000). In addition, exotic plants can invade and outcompete native plants through direct competition or by acting synergistically with other forms of stress, such as disease or disturbance (Cox 1999). Although some exotic species on reclaimed mines may benefit species, such as the coyote (*Canis latrans*) (Cox 2003) and several bird species (J. Cox, personal observation 2000), these same species can accelerate the invasion of adjacent forest by exotic plants through seed dispersal mechanisms. The use of invasive exotic plants to reclaim surface mines should be strongly discouraged because this practice compounds the process of fragmentation that furthers loss of biodiversity.

Surface mining causes structural changes to the Cumberland Plateau landscape that result in reduced topographical gradients at the highest elevations and in a vegetative cover dominated by grasses and other herbaceous species. The response by wildlife to these landscape modifications is observed in the change of forest birds to open grassland and to early successional forest (Table 1). Surface mining fragments or eliminates deciduous forest and, thus, greatly reduces or eliminates forest obligates, such as the cerulean warbler (*Dendroica cerulea*) (Hamel 2000), ovenbird (*Seiurus aurocapillus*) and Acadian flycatcher (*Empidonax virescens*). Allaire (1978a) suggested that disturbance from mining also affected bird density and diversity at the forest-mine interface.

Conversely, the artificial openings created by surface mining have attracted historically uncommon bird species that favor open habitat, such as the eastern meadowlark (*Sturnella magna*), northern harrier (*Circus cyaneus*), short-eared owl (*Asio flammeus*), grasshopper sparrow (*Ammodramus savannarum*), and horned lark (*Eremophia alpestris*) (Allaire 1978b, Whitmore

Table 1. Bird species characteristic of forest interior and reclaimed grassland habitat in eastern Kentucky, and that exhibit preferences for one cover type over the other. Species that benefit from natural or artificial forest fragmentation, that have very broad habitat requirements or that are very uncommon are excluded.

Forest interior species that are unlikely to benefit from expanding surface mining and associated reclaimed habitat	Species that are expected to benefit from expanding reclaimed grasslands and associated early successional habitats
Sharp-shinned hawk	Northern harrier
Broad-winged hawk	Red-tailed hawk
Yellow-billed cuckoo	Northern bobwhite
Barred owl	Killdeer
Hairy woodpecker	Short-eared owl
Pileated woodpecker	Common nighthawk
Acadian flycatcher	Northern flicker
Blue jay	Eastern kingbird
Carolina chickadee	Horned lark
White-breasted nuthatch	Purple martin
Blue-gray gnatcatcher	Northern rough-winged swallow
Wood thrush	Barn swallow
Solitary vireo	House wren
Yellow-throated vireo	Sedge wren
Red-eyed vireo	Eastern bluebird
Northern parula	Northern mockingbird
Black-throated blue warbler	Brown thrasher
Black-throated green warbler	Loggerhead shrike
Yellow-throated warbler	European starling
Pine warbler	White-eyed vireo
Cerulean warbler	Blue-winged warbler
Black-and-white warbler	Golden-winged warbler
American redstart	Yellow warbler
Worm-eating warbler	Chestnut-sided warbler
Ovenbird	Prairie warbler
Louisiana waterthrush	Common yellowthroat
Kentucky warbler	Yellow-breasted chat
Hooded warbler	Northern cardinal
Summer tanager	Blue grosbeak
Scarlet tanager	Indigo bunting
	Dickcissel
	Eastern towhee
	Field sparrow
	Grasshopper sparrow
	Henslow's sparrow
	Song sparrow
	Red-winged blackbird
	Eastern meadowlark
	Common grackle
	Brown-headed cowbird
	Orchard oriole
	American goldfinch
Total species = 30	Total species = 42

and Hall 1978, Bajema et al. 2001). Ravens (*Corvus corax*) have been found to use cliff habitat created by surface mining (Cox et al. 2003), and birds, such as the Canada goose (*Branta canadensis*), red-winged blackbird *(Agelaius phoeniceus*) and great blue heron (*Ardea herodias*), are found in or near water impoundments on reclaimed areas.

Careful observations of biota that inhabit surface mines will also reveal post-mining changes in the mammal community. Although some species such as the white-footed mouse (*Peromyscus leucopus*) may be found in greater abundance on surface mines when compared to forest (Krupa and Haskins 1996), McShea et al. (2003) found that mesic forest patches contain the bulk of small mammal species in the southern Appalachian ecotype. Large expanses of surface mines that harbor an abundance of patrolling grassland predators and a depauperate plant community may be formidable obstacles to small mammal recolonization of degraded land and may limit exchange among populations that persist in forest remnants.

The mixture of surface mine and forest habitat appears to provide some benefits to white-tailed deer (*Odocoileus virginianus*) (Cox 2003). Surface mining creates edge habitat used by white-tailed deer, while older deciduous forest provides white-tailed deer with mast, shade-tolerant understory species, saprophytes and lichens (Wentworth et al. 1990, Johnson et al. 1995). In addition, forests provide cover that deer use for security and thermoregulation (Crawford 1984). Large surface mines, however, may discourage white-tailed deer from using interior sections where they are more vulnerable to coursing predators, such as the coyote and feral dog (*Canis familiaris*) (J. Cox, personal observation 2001).

Another ungulate that has benefited from surface mines is the elk. Although most early attempts to restore elk to parts of the eastern United States failed (Witmer 1990), renewed public interest in repatriating the species to this area emerged during the late 1990s (Maehr et al. 1999). In response, wildlife agencies in Kentucky and Tennessee initiated reintroduction efforts designed to establish free-ranging populations in designated zones within the sparsely human-populated Cumberland Plateau region. To decrease the likelihood of human-elk conflict, more than 1,500 elk were translocated to Kentucky, and nearly 150 were brought to Tennessee; they were released primarily on reclaimed surface mines that provided elk with high elevation grassland habitat.

Although surface mined areas comprised only 10 percent of the elk restoration zone in Kentucky, reintroduced elk were found to select mined habitat over other types at both landscape and home range scales (Cox 2003). Although not a required habitat, the expanses of grasslands found on reclaimed surface mines provided elk with high elevation foraging opportunities that likely reduced depredation and collisions with automobiles by attracting elk away from low elevation human communities and roads, respectively. Additionally, the open grasslands provided by surface mines may have facilitated elk movement and social contact (Cox 2003). Further benefits to elk in these areas may be gained if more palatable species were established on reclaimed mines. We recommend that native plant species (e. g., warm season grasses) replace the exotic and often invasive species commonly planted in these areas. These plantings would provide needed forage to a growing elk population and would create habitat for other grassland species.

Although one might debate the degree to which restoration has occurred as a result of returning elk to the central Appalachians, there is no doubt that this large ungulate and new arrivals, such as the coyote, complement the new landscape. Whether additional large mammals will become a part of this remembered fauna will depend largely on future reclamation practices and the way the land is managed. Although surface mining benefits species that favor grassland, such as the coyote and elk, they may limit movements of forest-dwelling carnivores, such as the black bear (*Ursus americanus*), which in Kentucky appears to be using contiguous forests as corridors to recolonize the eastern portion of the state.

Conclusions

Surface mining profoundly disrupts ecological processes, alters landscape structure, and reduces or extirpates species that are forest-interior obligates. Therefore, natural resource managers should develop strategies and implement efforts that encourage human use of available resources that do not compromise the ecological and evolutionary integrity of the globally unique and threatened ecosystem of the Cumberland Plateau.

Given the high rates of global forest loss and decline, is it appropriate to celebrate the benefits of grassland ecosystems in what was once a sea of forest? Since the advent of agriculture some 8,000 years ago, nearly 4 million square miles (10,360,000 km^2) of forest have been lost; more than 100,000 acres (40,469 ha) disappear each day (Hunter 2002), and rates of forest decline can exceed 70 percent in some countries (Cox 1997). It might appear that the creation of additional grasslands contradicts the need to restore forests, especially in light of the fact that 70 percent of the earth's terrestrial ecosystems are rangeland (Noss and Cooperrider 1994). However, temperate grasslands have, "been almost completely destroyed by human activity" (Primack 1998:211), and, in the United States, the "long-grass prairie" is gone (Leopold 1949:189). Is it possible that the new landscapes associated with surface mining in the Cumberland Plateau could serve a dual conservation role that provides habitat for both forest obligates and for grassland species? Although the avifauna exhibits a dramatic shift in species and guild composition following mining (Table 1), the colonization of reclaimed mines by sensitive early successional species suggests the potential for some of these lands to replace degraded rangelands elsewhere.

Perhaps the best hope for biodiversity conservation on mined land is the recognition of the management potential that exists there. Success in promoting the restoration of native forests and in creating grasslands composed of noninvasive, nonexotic plants and animals will require major changes in current reclamation regulations, philosophies and practices. That the United States lags behind Europe and India with respect to mandated ecosystem restoration on mined land (Haigh 1993, Singh and Sinha 1993) should be no surprise, however. With millennia of human population growth and resource exploitation in the Old World, proportionally less native habitats remain in these highly denatured countries. When the virtual elimination of native habitats and landscapes confronts us, vigorous efforts are often made to save the remnants of once intact systems. As Leopold, (1949:189) observed, "It is the last call" in our efforts to protect some of these vestiges. We challenge the managers of reclaimed land and the authorities that dictate the trajectories of future landscapes on the Cumberland Plateau to work toward an integrated system of functional, interconnected forests and native grasslands that promote in situ biodiversity and new habitats that can serve as replacements for vanishing open land. Specific steps in this direction could include: (1) prevent hollowfills from further reductions in streams and other aquatic ecosystems, (2) avoid using exotic, invasive plant species for reclamation, (3) make forest and native grassland restoration economically feasible, (4) avoid loss of forest habitat whenever possible, (5) avoid soil compaction on reclaimed surfaces, (6) promote education that underscores the value of native biodiversity and its many ecological and cultural benefits to humanity, and (7) make the reclamation process planned biodiversity restoration rather than an artifact of convenience and tradition.

The renewed attraction to coal mining sparked by clean-coal technologies, coupled with a large coal reserve indicates a future in which surface-mined land increases and contiguous forests decrease. In this scenario, large blocks of intact deciduous forest will become the limiting factor in the survival of forest obligates and area-sensitive species. Thus, we believe that natural resource managers should view the current state of the eastern deciduous forest as a dynamic matrix of mixed-age forest patches that transcends geopolitical boundaries. As such, managers must consider the impacts of activities on the forest that occur outside agency jurisdictions and incorporate these changes in their management strategies. Securing and protecting core areas of intact deciduous forest will be vital to stemming the loss of biodiversity and should be of high management priority to resource agencies and organizations interested in maintaining the ecological and evolutionary processes of this globally threatened ecosystem.

Reference List

Abdul-Marashi, R. A., and J. Scullion. 2003. Earthworm casts form stable aggregates in physically degraded soils. *Biology and Fertility of Soils*. 37:375–80.

Allaire, P. N. 1978a. Effects on avian populations adjacent to an active strip-mine site. In *Surface mining and fish/wildlife needs in the eastern United States*, eds. D. E. Samuel, J. R. Stauffer, C. H. Hocutt, and W. T. Mason, 232–40. Morgantown, West Virginia: United States Fish and Wildlife Service, Biological Services Program.

Allaire, P. N. 1978b. Reclaimed surface mines: New potential for some North American birds. *American Birds*. 32:3–5.

Ayers, E. 1956. The age of fossil fuels. In *Man's role in changing the face of the Earth*, eds. W. L. Thomas, Jr., C. O. Sauer, M. Bates, and L. Mumford, 367–81. Chicago, Illinois: The University of Chicago Press.

Bajema, R. A., T. L. DeVault, P. E. Scott, and S. L. Lima. 2001. Reclaimed coal mine grasslands and their significance for Henslow's sparrows in the American Midwest. *The Auk*. 118:422–31.

Boettcher, S. E., and P. J. Kalisz. 1990. Single-tree influence on soil properties in the mountains of Eastern Kentucky. *Ecology*. 71:1365–72.

Branson, B. A., D. L. Batch, and W. R. Curtis. 1984. Small stream recovery following surface mining in east central Kentucky. *Transactions of the Kentucky Academy of Science*. 45:55–72.

Braun, E. L. 1950. Deciduous forests of eastern North America. New York: Hafner.

Cox, G. W. 1997. *Conservation biology: Concepts and applications*. Dubuque, Iowa: W. C. Brown.

Cox, G. W. 1999. *Alien species in North America and Hawaii: Impacts on natural ecosystems*. Washington, DC: Island Press.

Cox, J. J. 2003. *Community dynamics among reintroduced elk, white-tailed deer, and coyote in southeastern Kentucky*. Ph.D. dissertation, Lexington, Kentucky: Departments of Forestry and Animal Science, University of Kentucky.

Cox, J. J., N. W. Seward, J. L. Larkin, and D. S. Maehr. 2003. Common raven nests in southeastern Kentucky. *Southeastern Naturalist*. 2:99–104.

Crawford, H. S. 1984. Habitat management. In *White-tailed deer ecology and management*, ed. L. K. Halls, 629–47. Harrisburg, Pennsylvania: Stackpole Books.

Eller, R. D. 1982. *Miners, millhands, and mountaineers: Industrialization of the Appalachian South*, 1880–1930. Knoxville, Tennessee: University of Tennessee Press.

Girts, M. A., and R. L. P. Kleinmann. 1986. Constructed wetlands for treatment of acid mine drainage: A preliminary review. In *National symposium on mining, hydrology, sedimentology, and reclamation*, 165–71. Lexington, Kentucky: University of Kentucky.

Haigh, M. J. 1993. Surface mining and the environment in Europe. *International Journal of Surface Mining, Reclamation, and the Environment*. 7:91–104.

Hamel, P. B. 2000. *Cerulean warbler status assessment*. Washington, DC: U.S. Fish and Wildlife Service/United States Department of Interior.

Hatcher, R. M. 1978. Wildlife impacts from surface mining in perspective with other energy sources. In *Surface mining and fish/wildlife needs in the eastern United States*, eds. D. E. Samuel, J. R. Stauffer, C. H. Hocutt, and W. T. Mason, 76–84. United States Fish and Wildlife Service, Biological Services Program.

Holl, K. D., C. E. Zipper, and J. A. Burger. 2001. *Recovery of native plant communities after mining, Virginia Cooperative Extension Publication,* 460–140. Blacksburg, Virginia: Virginia Polytechnic Institute and State University.

Houp, R. E. 1993. Observations of long-term effects of sedimentation on freshwater mussels (*Mollusca: Unionidae*) in the North Fork of Red River, Kentucky. *Transactions of the Kentucky Academy of Scienc*e. 53:94–7.

Hunter, M. L., Jr. 2002. *Fundamentals of conservation biology*. Malden, Massachusetts: Blackwell Science.

Johnson, A. S., P. E. Hale, W. M. Ford, J. M. Wentworth, J. R. French, O. Anderson, and G. B. Pullen. 1995. White-tailed deer foraging in relation to successional stage, overstory type and management of Southern Appalachian forests. *American Midland Naturalist*. 133:18–35.

Keck, M.B. 1998. Habitat use by semi-aquatic snakes at ponds on a reclaimed strip mine. S*outhwestern Naturalis*t. 43:13–9.

Kentucky Coal Association. 2000. *Kentucky coal facts 1999–2000.* Lexington, Kentucky: Kentucky Coal Association.

Kentucky Environmental Quality Commission. 2001. *State of Kentucky's environment 2000–2001: A report on environmental trends and conditions*. Frankfort, Kentucky: Kentucky Environmental Quality Commission.

Krupa, J. J., and K. E. Haskins. 1996. Invasion of the meadow vole (*Microtus pennsylvanicus*) in southeastern Kentucky and its possible impact on the southern bog lemming (*Synaptomys cooperi*). *American Midland Naturalist*. 135:14–22.

Lacki, M. J., J. W. Hummer, and H. J. Webster. 1992. Mine drainage treatment wetland as habitat for herpetofaunal wildlife. *Environmental Management*. 16:513-20.

Layzer, J. B., M. E. Gordon, and R. M. Anderson. 1993. Mussels: The forgotten fauna of regulated rivers. A case study of the Caney Fork River. *Regulated Rivers Research and Management*. 8:63–71.

Leopold, A. 1949. *A Sand County almanac*. New York, New York: Oxford University Press.

Maehr, D. S., R. Grimes, and J. L. Larkin. 1999. Initiating elk restoration: The Kentucky case study. *Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies*. 53:350–63.

Matter, W. J., J. J. Ney, and O. E. Maughan. 1978. Sustained impact of abandoned surface mines on fish and benthic invertebrate populations in headwater streams of southwestern Virginia. In *Surface mining and fish/wildlife needs in the eastern United States*, eds. D. E. Samuel, J. R. Stauffer, C. H. Hocutt, and W. T. Mason, 203–16. United States Fish and Wildlife Service, Biological Services Program.

McShea, W. J., J. Pagels, J. Orrock, E. Harper, and K. Koy. 2003. Mesic deciduous forest as patches of small-mammal richness within an Appalachian mountain forest. *Journal of Mammalogy*. 84:627–43.

Mueller, L. 2003. Proposed state park threatened. *Lexington-Herald Leader*. December 13:B1.

Mummey, D. L., P. D. Stahl, and J. S. Buyer. 2002. Soil microbial properties 20 years after surface mine reclamation: Spatial analysis of reclaimed and undisturbed sites. *Soil Biology and Biogeochemistry*. 34:1717–25.

Noss, R. F., and A. Y. Cooper rider. 1994. *Saving nature's legacy*. Washington, DC: Island Press.

O'Bara, C. J. 1985. *Status survey of the blackside dace (*Phoxinus cumberlandensis*)*. Asheville, North Carolina: United States Department of the Interior, Fish and Wildlife Service.

Ohio River Valley Environmental Coalition. 2004. *Mountaintop removal mining fact sheet*. http://www.ohvec.org/issues/mountaintop_removal/articles/mtr_fact_sheet.pdf

Perkins, G. A., and J. S. Lawrence. 1985. Bird use of wetlands created by surface mining. *Transactions of the Illinois Academy of Science*. 78:87–96.

Primack, R. B. 1998. *Essentials of conservation biology*. Sunderland, Massachusetts: Sinauer and Associates.

Shakleford, L., and B. Weinberg. 1977. *Our Appalachia*. New York, New York: Hill and Wang.

Short, T. M., J. A. Black, and W. J. Birge. 1990. Effects of acid-mine drainage on the chemical and biological character of an alkaline headwater stream. *Archives of Environmental Contamination and Toxicology*. 19:241–8.

Singh, G., and D. K. Sinha. 1993. Land reclamation and restoration of natural ecosystem: A case study from opencast mines of north eastern coalfields of India. *International Journal of Surface Mining, Reclamation, and the Environment*. 7:171–6.

Starnes, L. B., and W. C. Starnes. 1981. Biology of the blackside dace Phoxinus cumberlandensis. *American Midland Naturalist*. 106:360–71.

Torbert, J. L., J. A. Burger, S. H. Schoenholtz, and R. E. Kreh. 2000. Growth of three pine species after eleven years on reclaimed minesoils in Virginia. *Northern Journal of Applied Forestry*. 17:95–9.

United States Fish and Wildlife Service. 1987. *Endangered and threatened wildlife and plants: Determination of threatened species status for the blackside dace, federal register 52:22580–22585*. Washington, DC: United States Department of Interior.

United States Geological Survey. 2004. Coal availability studies: A new look at resource estimates. In *Energy resource surveys program fact sheet*. Washington, DC: United States Department of Interior.

Urbanek, R. P., and W. D. Klimstra. 1986. Vertebrates and vegetation on a surface-mined area in southern Illinois. *Transactions of the Illinois Academy of Science*. 79:175–87.

Verb, R. G., and M. L. Vis. 2000. Comparison of benthic diatom assemblages from streams draining abandoned reclaimed coal mines and nonimpacted sites. *Journal of the North American Benthological Society*. 19:274–88.

Wentworth, J. M., A. S. Johnson, P. E. Hale, and K. E. Kammermeyer. 1990. Seasonal use of clearcuts and food plots by white-tailed deer in the Southern Appalachians. *Proceedings of the Annual Conference of Southeastern Fish and Wildlife Agencies*. 44:215–23.

West Virginia Office of Miners Health Safety and Training. 2004. *West Virginia coal mining facts*. http://www.wvminesafety.org/wvcoalfacts.htm.

Whitmore, R. C., and G. A. Hall. 1978. The response of passerine species to a new resource: Reclaimed surface mines in West Virginia. *American Birds*. 32:6–9.

Winger. P. V. 1978. Fish and benthic populations of the New River, Tennessee. In *Surface mining and fish/wildlife needs in the eastern United States*, eds. D. E. Samuel, J. R. Stauffer, C. H. Hocutt, and W. T. Mason, 190–216. United States Fish and Wildlife Service, Biological Services Program.

Witmer, G. 1990. Re-introduction of elk in the United States. *Journal of the Pennsylvania Academy of Science*. 64:131–5.

Effect of Energy Development and Human Activity on the Use of Sand Sagebrush Habitat by Lesser Prairie Chickens in Southwestern Kansas

Robert J. Robel
Kansas State University, Division of Biology
Manhattan

John A. Harrington, Jr.
Kansas State University, Department of Geography
Manhattan

Christian A. Hagen
Kansas State University, Division of Biology
Manhattan

James C. Pitman
Kansas State University, Division of Biology
Manhattan

Ryan R. Reker
Kansas State University, Department of Geography
Manhattan

Lesser prairie chickens (*Tympanuchus pallidicinctus*) occupy xeric grasslands dominated primarily by sand sagebrush (*Artemisia filifolia*) or shinnery oak (*Quercus harvardii*) in portions of southwestern Kansas, southeastern Colorado, western Oklahoma, northern Texas and eastern New Mexico (Giesen 1998), and their populations have declined rangewide since the 1800s (Braun et al. 1994). In southwestern Kansas, lesser prairie-chickens are most abundant in mixed- and short-grass prairies dominated by sand sagebrush south of the Arkansas River. Population indices (lek counts) suggest lesser prairie-chicken numbers have declined since the 1970s (Jensen et al. 2000). Generally the decline has been attributed to the deterioration of the sandsage habitat and the conversion of suitable habitat to intensive agriculture, primarily center-pivot irrigated corn. Even though most of the large-scale conversion of sand sagebrush prairie to intensive agriculture ceased in the mid-1980s, lek indices to lesser prairie chicken populations continued to decline in southwestern Kansas (Jensen et al. 2000).

A 6-year study initiated in 1997 examined factors that may have contributed to the 1980 to 2000 decline in numbers of lesser prairie chickens in southwestern Kansas. Low nest success and poor chick survival were determined to be the most important factors contributing to the decline (Hagen 2003, Pitman 2003). The research was conducted in Finney County, an area of southwestern Kansas that historically supported a viable lesser prairie chicken population. Lek survey indices to prairie chicken populations in that county averaged 12.1 birds per square mile (4.7 birds/km^2) during the late 1960s (Church 1987). Between 1960 and 1975, approximately 60 percent of the native sand sagebrush prairie in Finney County was converted to intensive agriculture (Sexson 1980). That loss of habitat originally was thought to be the sole cause of the 33-percent decline (from 12.1 to 8.1 birds per square mile [4.7–3.1 birds/km^2]) in the lesser prairie chicken lek survey indices in Finney County during the 1980s and the 50 percent decrease (from 8.1 to 4.1 birds per square mile [3.1 to 1.6 birds/km^2]) in the 1990s. However, these declines occurred even though large expanses of sand sagebrush prairie existed in the county through the 1980s and 1990s. During Hagen's (2003) and Pitman's (2003) studies, radio telemetry data disclosed avoidance by lesser prairie chickens of what appeared to be suitable sand sagebrush habitat near anthropogenic features, e. g., roads, buildings, oil and gas wellheads, electric transmission lines and center-pivot irrigation fields.

The human population of Finney County increased by over 25 percent between 1980 and 2000 (U. S. Census Bureau 2003), coincidental with the construction of a coal-fired electric generating station and associated transmission lines, road improvements and an increased number of houses in rural settings. Petroleum exploration and production also increased in the county, and compressor stations were constructed to move natural gas through underground pipelines. These anthropogenic changes in Finney County coincided with declines in lek survey indices to lesser prairie chicken populations in the 1980s and 1990s.

We conducted this study to assess the magnitude of the impacts of anthropogenic factors on use of sand sagebrush habitat by lesser prairie chickens. We focused our efforts on the remaining sand sagebrush habitat in Finney, Kearny and Hamilton counties of southwestern Kansas, the three counties supporting 25 to 50 percent of the lesser prairie chicken population in Kansas during the early 2000s (assuming lek survey data are a realistic reflection of lesser prairie chicken numbers).

Methods and Procedures

Data used to determine use of sand sagebrush habitat by lesser prairie chickens were obtained from transmitter-equipped birds on two 12,500 acre (5,070 ha) study sites in Finney County during a 1997 to 2003 field study. Lesser prairie chickens were captured on leks using walk-in funnel traps (Haukos et al. 1990) primarily during March and April. Captured birds were fitted with less-than-0.4-ounce (11-g) transmitters (less than 2% of each bird's body mass) and released within 15 minutes at capture sites. Birds were located daily by triangulation using a truck-mounted, null-peak, twin-Yagi telemetry system. The influence of anthropogenic features on the use of sand sagebrush habitat was estimated from these data, and its impact was extrapolated to the remaining sand sagebrush habitat in Finney, Kearny and Hamilton counties during 2003 to 2004.

Study Area

The sand sagebrush prairies of Finney, Kearny and Hamilton counties exist primarily on undulating sand dunes south of the Arkansas River (Kuchler 1974). Two soil types are typical across the sand sagebrush vegetation complexes: Tivoli fine sand and Tivoli-Vona loamy fine sands (Harner et al. 1965). The long-term average annual precipitation for the area was 19 inches (48 cm) with 75 percent of it occurring between March and August; the mean annual temperature was 55° Fahrenheit (13°C), ranging from 21° Fahrenheit (-6°C) for January to 79° Fahrenheit (26°C) for July (U. S. Department of Commerce 2003).

Sand sagebrush dominated the vegetative community and was interspersed with grasses, including blue grama (*Bouteloua gracilis*), sand dropseed (*Sporobolus cryptandrus*), prairie sandreed (*Calamovilfa longifolia*), sand bluestem (*Andropogon halii*) and little bluestem (*Schizachyrium scoparium*). Other plants common on the area included western ragweed (*Ambrosia psilostachya*), annual erigonum (*Erigonum annum*), sunflowers (*Helianthus* spp.), plains yucca (*Yucca glauca*), prickly pear (*Opuntia polyacantha*) and Russian thistle (*Salsola kali*). Kuchler (1974) presents a detailed description of the vegetation of the sand sagebrush prairie of the three counties. Over 90 percent of the sand sagebrush rangeland was grazed annually by cattle at various intensities resulting in highly variable vegetation structure across the study area.

Determining Coverage of Sand Sagebrush Prairie

The historical distribution of sand sagebrush habitat in Finney, Kearny and Hamilton counties was primarily defined by the extent of the Tivoli association soil complex and vegetation based landcover maps by Kuchler (1974). Defining natural habitat based on soil types has been successfully used in similar studies (Johnson et al. 1995). Two Landsat 1 multispectral scanner images (pixel resolution of 66 yards [60 m]) were used to identify sand sagebrush acreage in the three counties for 1973 whereas two Landsat 7 Enhanced Thematic Mapper (ETM+) images (pixel resolution of 36 yards [33 m]) and ground truthing were used for 2001 determinations.

Inventory of Anthropogenic Features in the Three Counties

Locations of anthropogenic features in the sand sagebrush habitat were entered into a GIS system for display and analysis. Road center lines from the U. S. Census Bureau were downloaded from the Kansas Data Access and Support Center. Point data for oil and oil/gas wellheads were downloaded from the Kansas Geological Survey Database. Locations of buildings (large houses, feedlots, ranch steads, compressor stations and the power plant) were identified on U. S. Geological Survey 1:24,000 topographic maps and Landsat 7 satellite imagery and a polygon layer of building sites was created by digitizing feature boundaries. Paper maps of electric transmission line routes were provided by the Sunflower Electric Corporation, georeferenced to Landsat 7 ETM+ satellite imagery, then digitized and uploaded into ArcInfo 8.1 to create a transmission line layer. Center-pivot fields were identified by their distinctive spectral and textural properties and classified in ERDAS Imagine 8.7 (Leica Geosystems 2003).

The scale of satellite images and course pixel resolution limited our ability to identify small anthropogenic features (minor roads and trails, individual houses, trailers and small outbuildings). Therefore, our assessment of the impacts of anthropogenic features on lesser prairie chickens in the sand sagebrush habitat of the three-county area must be interpreted as a conservative estimate.

Determining Areas Avoided by Nesting Lesser Prairie chickens

The movements of transmitter-equipped, female lesser prairie chickens were monitored daily during April through June. When a bird's location was unchanged for more than three days it was assumed to be nesting. The nesting bird was approached and the location of its nest recorded with a global positioning system. Nesting females were monitored daily by telemetry, but nest sites were not visited again until the female departed the site with a brood or the nest was depredated or abandoned. Vegetation structure was quantified at each nest site and at a paired, random point within 200 yards of the nest, within three days of hatching, depredation or abandonment. Vegetation measurements included height, visual obstruction readings and percent canopy cover of grass, sagebrush and forbs. Details of vegetation measurements are presented in Pitman (2003).

Locations of nests were incorporated into a geographic information system of the two study areas created in ArcView 3.1 (Environmental Systems Research Institute 1998) along with locations of wellheads, buildings, transmission lines, improved roads and unimproved roads and center-pivot irrigated fields (hereafter center-pivot field). Distances from each nest to the nearest wellhead, building, transmission line, roads and center-pivot field edge were calculated for each nest.

Wellheads were oil and oil/gas wells with pumping units powered primarily by diesel fuel. Buildings consisted of houses, gas compressor stations, and a 380 megawatts coal-fired electric generating station. Transmission lines primarily were 125, 138 and 345 kilovolts, double-circuit conductors distributing electricity from the generating station. Improved roads were graveled or paved and carried up to 486 vehicles per day (vpd) whereas unimproved roads were 2-lane pasture trails and ungraded service roads to wellheads with traffic less than 3 vpd. Center-pivot fields covered 160 acres (65 ha) with a water pump in the center and a 13- to 16-foot (4–5-m) high sprinkler boom extending from the center to the edge of the field. When in operation (generally from late April or early May through summer), the sprinkler boom irrigated the field by rotating circularly across the field on self-powered wheels.

We used Monte Carlo simulations (modified from Manly 1998) to determine if any of the six anthropogenic features were related to distances to locations of lesser prairie chicken nests. Because features far from nest sites were unlikely to impact nesting birds, we used only nests close to each feature (closest 10% of the nests) to assess the impacts of the six anthropogenic features.

Distances from each nest and the anthropogenic features were compared to distances of random points created by 1,000 draws in Monte Carlo simulations (details in Pitman 2003). This was done for each of the 10 percent closest nests to each of the anthropogenic features. Probability distributions were used to determine if nests were significantly ($P = 0.05$) farther than expected from a particular feature. If nests were significantly farther than expected from a feature, that feature was determined to negatively effect lesser prairie-chicken nest location. The mean distance of the closest 10 percent of the nests to a specific anthropogenic feature was determined, and that distance was used as the avoidance distance of nesting lesser prairie chickens for that feature.

Determining Areas Avoided by Adult Lesser Prairie chickens

We quantified use and nonuse areas of sand sagebrush habitat from telemetry locations of lesser prairie chickens. Use areas were defined using a 95 percent fixed kernel home range (Worton 1989) of bird locations. Because multiple locations at nest or lek sites may have underestimated the size of the kernel, we used only one lek or nest location per bird for kernel home range calculations. Sand sagebrush habitat not within the 95 percent fixed kernel home range was considered the nonuse area. Although we cannot be absolutely certain that lesser prairie chickens never used the nonuse areas, we never recorded transmitter-equipped or unmarked birds, or signs (droppings or feathers) of the birds in those areas during the six years of our field study.

Random location points were generated within use and nonuse areas in ArcView 3.1 (Environmental Systems Research Institute 1998) to serve as sampling points to characterize vegetation structure in those areas. At each point, diameters of individual sand sagebrush plants were measured, and a sampling quadrate was used to estimate the percent canopy cover of sand sagebrush, grass, forbs and litter. The vegetation structure of use and nonuse areas was compared using a fixed-model analysis of variance. Details of analytical procedures are in Hagen (2003).

Lesser prairie chicken location data from April to September, 2000 to 2002, were analyzed for impacts of four anthropogenic features (roads, buildings, wellheads and transmission lines) on their distribution on the two study areas. Locations of individual birds were stratified by month and year and were imported into ArcView 3.1 containing locations of the four anthropogenic features. Monthly home ranges (95% fixed kernel [Worton 1989]) were estimated for birds with more than 19 locations per month. A modified Monte Carlo simulation (Manly 1998) was used to test if the centroids of monthly home ranges were farther from the four anthropogenic features than expected at random. See Hagen (2003) for details of analytical procedures.

Quantifying the Acreage Impacted by Anthropogenic Features

Distance to anthropogenic features avoided by 90 percent of nesting and 95 percent of adult lesser prairie chickens were entered into an avoidance buffer database. Avoidance buffers around or along individual anthropogenic features were created in ArcInfo 8.1 (Environmental Systems Research Institute 2001) with the width of avoidance determined from previously described field data. Buffers around anthropogenic features often overlapped, e. g., transmission lines running adjacent to roads, wellheads lying within center-pivot fields. This overlap could result in overestimation of avoidance area. To eliminate bias associated with overlap, the merge and dissolve by attribute functions in ArcInfo 8.1 were used to create one comprehensive avoidance buffer that apportioned duplicated areas of impact among overlapping features.

Results and Discussion

Extent of Sand Sagebrush Prairie Habitat

Historically, an estimated 339,645 acres (137,556 ha) of sand sagebrush habitat existed in Finney, Kearny and Hamilton counties. By 1973, this acreage had been reduced to 298,806 acres (121,016 ha), primarily due to conversion of sand sagebrush habitat to center-pivot agriculture. Another 81,994 acres (33,208 ha) of sand sagebrush habitat was lost to center-pivot agriculture between 1973 and 2001. That loss and a commitment of 3,277 acres (1,327 ha) to urban development (residences, golf courses, etc.) left only 214,183 acres (86,744 ha) of sand sagebrush habitat in the three counties by 2001, approximately 63 percent of the historical acreage. Most of the loss of sand sagebrush habitat occurred in Finney County (74,154 of 142,132 acres [30,032 of 57,564 ha]; 52%) and in Kearny County (48,625 of 67,321 acres [19,693 of 27,265 ha]; 72%), and least in Hamilton County (2,004 of 130,192 acres [812 of 52,728 ha]; 2%).

Vegetation Structure and Nest Location

Vegetation structure around 174 nests of lesser prairie chickens were characterized during this study. Vegetation structure at nest sites differed significantly ($P < 0.05$) from that at paired random sites in the sand sagebrush prairie. Visual obstruction, sagebrush cover, sagebrush height, forb height and grass height were greater at nest sites than at paired random points, whereas forb cover and bare ground were less at nest sites than at paired random points. Vegetation structure within 219 yards (200 m) of anthropogenic features differed little from that at random points from those features; visual obstruction did not differ ($P = 0.492$), nor did sagebrush cover ($P = 0.382$) or bare ground ($P = 0.068$). Forb cover was less ($P = 0.014$) near anthropogenic features and grass cover greater ($P = 0.028$), both probably reflecting the use of herbicides along road rights-of-way and on center-pivot fields.

Nest Locations Relative to Anthropogenic Features

Lesser prairie chicken nests were located farther from five of six anthropogenic features than would be expected at random. The presence of unimproved roads had the least effect on placement of lesser prairie chicken nests, and buildings had the greatest (Table 1). Essentially, the presence of anthropogenic features reduced the suitability for nesting of the surrounding sand sagebrush habitat.

Table 1. Mean distance (yards ± standard error [SE]) to anthropogenic features avoided by 90 percent of 187 nesting lesser prairie chickens during 1997–2002 in sand sagebrush prairie habitat of southwestern Kansas.

Anthropogenic feature	Distance to feature
Oil or gas wellheads	93 ± 25
Center-pivot fields	336 ± 51
Unimproved roads	32 ± 15[a]
Improved roads	859 ± 44
Buildings	1,371 ± 65
Electric transmission lines	397 ± 70

[a] Not significantly different ($P > 0.05$) from zero; the distance was not used in estimate of less suitable sand sagebrush habitat for nesting by lesser prairie chickens in the three-county area.

We did not attempt to determine factors causing nesting lesser prairie chickens to avoid anthropogenic features; however, it appears that movement and noise might be implicated. Wellheads had pump jacks with weighted extensions that moved up and down when pumping, with larger units on deeper wells. The sprinkler booms of center-pivot irrigation systems moved across fields when operating, and water nozzles on the ends sprayed back and forth. Vehicles ranging from small sedans to large grain and tanker trucks moved along roads, and more vehicle movement occurred on improved than unimproved roads. All of the anthropogenic features were sources of noise. On calm nights, pumps of center-pivots could be heard by humans at a distance of approximately 0.6 mile. Similarly, gas compressor stations were audible at over 2 miles, and noise from the power plant could be heard from at least 1 mile. Low frequency sounds were audible from the electric transmission lines when high voltage charges were being moved through the lines. Truck traffic on improved roads was commonly heard at 1.5 miles and farther when drivers geared the trucks down for curves or inclines.

Additional research is needed to determine why the anthropogenic features we examined deterred lesser prairie chickens from nesting near them. Such answers may allow modification of those features to reduce their negative impacts on nest placement by lesser prairie chickens and possibly other avian species sensitive to human activity.

Vegetation Structure of Use and Nonuse Areas

Vegetation measurements from 42 random locations in areas of sand sagebrush habitat used by lesser prairie chickens (use area) and 42 in areas not used by the birds (nonuse area) were the basis of comparing vegetation structure of use and nonuse areas. Sagebrush canopy cover did not differ ($P > 0.05$) between use and nonuse areas, nor did forb or grass canopy cover. Neither sagebrush density nor mean diameter of sagebrush plants differed ($P > 0.05$) between use and nonuse areas. The only vegetation variable that differed between the two areas was litter cover; it was significantly greater ($P < 0.05$) on nonuse than use areas. Thus, based on the vegetation measurements we made, the vegetation structure of the sand sagebrush habitat in areas used and not used by adult lesser prairie chickens was virtually identical.

Adult Bird Locations Relative to Anthropogenic Features

Adult lesser prairie chickens used the sand sagebrush habitat near roads, oil wellheads and oil/gas wellheads, buildings and electric transmission lines less than areas of that habitat far from those features. The negative impacts of roads and wellheads were less than those of buildings and transmission lines (Table 2). Because we combined use of areas near unimproved and improved roads in this analysis (we separated them in our assessment of impacts of roads on nest locations), the magnitude for the negative impact of roads was not statistically

Table 2. Mean distance (yards ± SE) to anthropogenic features across which 95 percent of 18,866 telemetry locations of lesser prairie chickens were absent in sand sagebrush prairie habitat of southwestern Kansas.

Anthropogenic feature	Distance to feature
Improved and unimproved roads	51 ± 4[a]
Oil or gas wellheads	79 ± 5
Buildings	659 ± 46
Electric transmission lines	693 ± 44

[a] Not significantly different ($P > 0.05$) from zero, but distance was used in estimate of less suitable sand sagebrush habitat for use by lesser prairie chickens in the three-county area.

significant ($P > 0.05$). However, we used the impacted distance (51 yards [47 m]) in our estimate of sand sagebrush habitat area made less suitable for lesser prairie chickens because of the impacts we detected for improved roads on nesting behavior of the birds. Had we analyzed improved roads separately from unimproved roads, we likely would have found significant impacts of improved roads on use of nearby sand sagebrush habitat and feel justified in including roads as a negative factor in the landscape of lesser prairie chickens.

We did not attempt to determine why adult lesser prairie chickens avoided using the sand sagebrush habitat near anthropogenic features. However, noise and movement might be implicated as we speculated earlier for avoidance of anthropogenic features by nesting female lesser prairie chickens.

Acreage of Avoidance Buffers around Anthropogenic Features

The acreage of sand sagebrush habitat made less suitable for lesser prairie-chicken nesting and for use by the presence of anthropogenic features is substantial. The area approaches the acreage of sand sagebrush habitat converted to center-pivot agriculture in Finney, Kearny and Hamilton counties.

Our research, and that of Peterson and Silvy (1996) and Wisdom and Mills (1997), determined that nest success was one of the most critical demographic parameters regulating prairie grouse populations. Therefore, any negative impacts of anthropogenic features on nesting of lesser prairie-chickens is of great concern.

The presence of improved roads reduced nesting by lesser prairie-chickens in the 859-yard (785-m) buffer on either side of the roads. In 1973, that buffer included 141,312 acres (57,231 ha) of sand sagebrush habitat (Table 3). That estimate may be high because not all of the roads identified on the Landsat 7 imagery carried high volumes of traffic. However, even if only 50 percent of

Table 3. Acreage of sand sagebrush prairie habitat converted to intensive agriculture or made less suitable as lesser prairie chicken nesting areas by the presence of anthropogenic features in Finney, Kearny and Hamilton counties, Kansas, in 1973 and 2001.

	1973	2001
Historical acreage of sagebrush habitat	339,645	339,645
Converted to intensive agriculture	40,191	122,185
Converted to other uses (urban, recreation, etc.)	649	3,277
Sagebrush habitat remaining	298,806	214,183
Remaining sagebrush habitat made less suitable by presence of:		
Improved roads	141,312	89,297
Oil and oil/gas wellheads	8,564	17,562
Buildings	8,974	15,774
Electric transmission lines	0	16,803
Center-pivot fields	30,795	53,694
Total acreage made less suitable by anthropogenic features	157,296	125,962
Remaining sagebrush habitat suitable for nesting by lesser prairie-chickens	41,570	88,221

the roads carried high levels of traffic, the acreage negatively impacted would exceed 70,000 acres (28,000 ha). The amount of sand sagebrush habitat negatively impacted by roads in 2001 was less than in 1973 because the roads crossed areas that were sagebrush in 1973 but had been converted to center-pivot fields by 2001. Even so—and assuming that only 50 percent of the roads carried high traffic volumes—the negative impacts of roads in the three counties extended to over 40,000 acres (16,200 ha) of sand sagebrush habitat in 2001.

Oil and oil/gas wellheads negatively impacted lesser prairie-chicken nesting on an estimated 8,564 acres (3,468 ha) of sandsage habitat in 1973. This area increased to 17,562 acres (7,113 ha) by 2001.

The presence of buildings negatively impacted 8,974 acres (3,634 ha) of lesser prairie chicken nesting habitat in 1973, and increased the presence of it 15,774 acres (6,388 ha) by 2001. Buildings included a coal-fired power electric generating station, at least three compressor stations and numerous large houses and cattle feedlots.

There were few to no major electric transmission lines crossing the sand sagebrush habitat of Finney, Kearny and Hamilton counties in 1973, but several lines were present in 2001. These transmission lines were distributing electricity primarily to the west and southwest of the power station 7 miles (11 km) southwest of Garden City, in Finney County. Lesser prairie-chickens seldom nested within 400 yards (366 m) of the electric transmission lines. The avoidance buffer encompassed by the transmission lines in the early 2000s approximated 16,803 acres (6,805 ha) of sand sagebrush habitat (the estimate includes the impacts of a transmission line under construction in 2003).

The avoidance buffers on the edges of center-pivot fields negatively affected approximately 31,795 acres (12,472 ha) of sand sagebrush habitat, in 1973, and 53,694 acres (21,746 ha), by 2001. We do not know if nesting female lesser prairie chickens were avoiding the center-pivot fields themselves, the noise from the irrigation pump in the center of the fields or the large sprinkler booms that rotated across the fields. Whatever the reason, lesser prairie chickens seldom nested within 336 yards (307 m) of the edges of center-pivot fields.

Combined, the total avoidance buffer areas around anthropogenic features in the three counties reduced the suitability of 157,296 acres (63,705 ha) and 125,962 acres (51,015 ha) of sand sagebrush habitat for lesser prairie chicken nesting in 1973 and 2001, respectively. These areas represent 52 percent of the sand sagebrush habitat remaining in the three counties, in 1973, and 58 percent of that remaining, in 2001.

The area of sand sagebrush habitat made less suitable for general use by adult lesser prairie chickens was less than that made unsuitable for nesting. Avoidance buffers along roads encompassed 12,320 acres (4,990 ha), in 1973, and 9,739 acres (3,944 ha), in 2001. Oil and oil/gas wellheads negatively impacted 1,440 acres (583 ha), in 1973, and 3,183 acres (1,289 ha), in 2001. Avoidance buffers around buildings contained 3,034 acres (1,229 ha), in 1973, increasing to 7,399 acres (2,997 ha), by 2001. Adult lesser prairie chickens seldom used sand sagebrush habitat within 693 yards (633 m) of electric transmission lines, and that avoidance buffer area amounted to 6,615 acres (2,679 ha) in 2001. Although some of these totals appear large, their impacts on the lesser prairie chicken population are minor, compared to the impacts of the presence of anthropogenic features on nesting habitat.

Summary and Management Implications

The decline of lesser prairie chicken numbers in southwestern Kansas has been attributed to the loss of suitable habitat during the 1960s through the 1980s, primarily sand sagebrush prairie. However, population declines continued after large-scale losses of sand sagebrush habitat ceased and large tracts of that habitat remained. A 6-year study disclosed that low nest success was one of the most important factors in the decline of lesser prairie chicken numbers in southwestern Kansas and that the birds avoided otherwise suitable habitat near human activity, especially for nesting. Human activity (population numbers, oil and gas development, construction of a coal-fired electric generation station, etc.) increased in southwestern Kansas coincidental with the decline of lesser prairie chicken numbers. To gain a better understanding of the relationship between human activity and the viability of lesser prairie chicken populations, we assessed the impacts of the presence of anthropogenic features (roads, buildings, transmission lines, etc.) on the use of sand sagebrush habitat by the birds.

The distance to anthropogenic features avoided by nesting lesser prairie chicken females were used to create 'avoidance buffers' around anthropogenic features in sand sagebrush habitat. Sand sagebrush habitat within the avoidance buffer was less suitable for nest locations of 90 percent of nesting females and for use by 95 percent of adult birds. The area contained in these buffer areas was then estimated for the sand sagebrush habitat remaining in Finney, Kearny and Hamilton counties of southwestern Kansas. These three counties were thought to support a significant portion of the Kansas lesser prairie chicken population.

By 2001, approximately 125,462 acres (50,812 ha) of the historic 339,645 acres (137,556 ha) of sand sagebrush prairie in the three counties had been converted to intensive agriculture or used for urban development. Avoidance buffers around improved roads, oil wellheads and oil/gas wellheads, buildings, electric transmission lines, and center-pivot fields reduced the nesting suitability of an additional 125,962 acres (51,015 ha) of sand sagebrush habitat. Thus, the amount of sand sagebrush habitat suitable for nesting by lesser prairie chickens was only 88,221 acres (35,729 ha) in 2001, approximately 26 percent of the historic amount.

The impact of the avoidance buffers is depicted in Figure 1. The historic range of sand sagebrush habitat in Finney County (A) was reduced to 214,183 acres (86,744 ha) by 2001 (B), and that area was further reduced to only 88,221 acres (35,729 ha) of suitable nesting habitat (C) by the presence of anthropogenic features in the sand sagebrush prairie landscape. We suspect that the poor nest success and low chick survival experienced by lesser prairie chickens in Finney County (Hagen 2003, Pitman 2003) were the result of a shortage of suitable nesting habitat. The remaining patches of sand sagebrush habitat suitable for nesting in that county were small and isolated, potentially increasing the predator pressure on lesser prairie-chicken nests, the nesting birds and any fledgling chicks.

In the future, the negative impacts of anthropogenic features should be considered when assessing the suitability of habitat for lesser prairie chickens, purchasing or leasing habitat for those birds, or implementing management actions for the benefit of those populations. The avoidance buffers around oil and oil/gas wellheads, electric transmission lines, and buildings must be recognized and assessed for the development of petroleum resources and the construction of industrial wind energy facilities. The results of our research most likely apply to other prairie grouse, but specific studies are needed to determine the magnitude of the impacts on individual species in various landscapes.

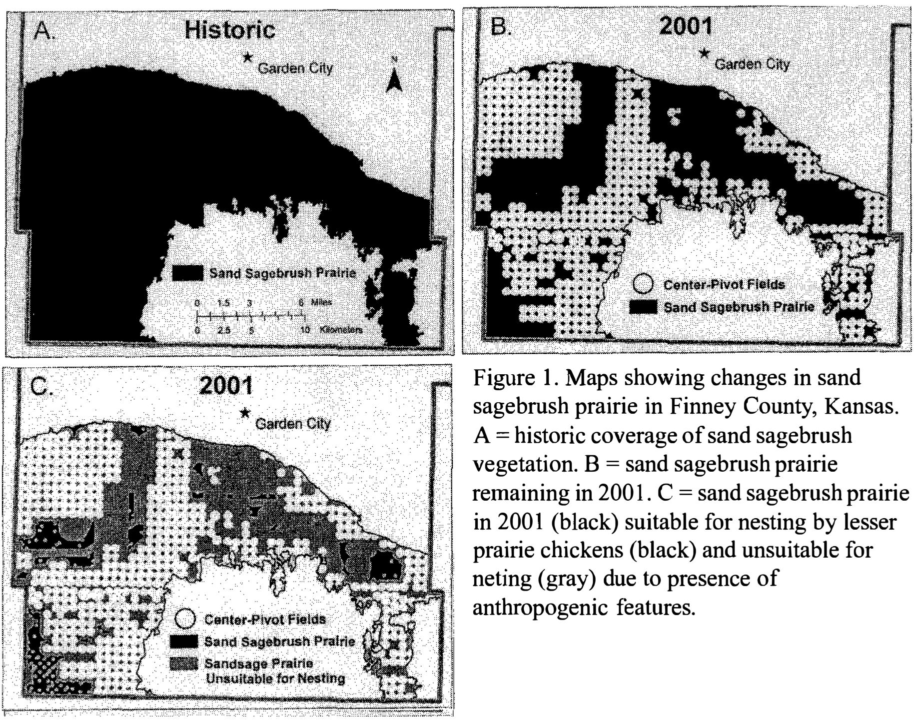


Figure 1. Maps showing changes in sand sagebrush prairie in Finney County, Kansas. A = historic coverage of sand sagebrush vegetation. B = sand sagebrush prairie remaining in 2001. C = sand sagebrush prairie in 2001 (black) suitable for nesting by lesser prairie chickens (black) and unsuitable for neting (gray) due to presence of anthropogenic features.

References

Braun, C. E., K. Martin, T. E. Remington, and J. R. Young. 1994. North American grouse: Issues and strategies for the 21st century. In *Transactions of the 59th North American Wildlife and Natural Resources Conference*, 428–37. Washington, DC: Wildlife Management Institute.

Church, K. E. 1987. *Prairie grouse and ruffed grous population and harvest summary, federal aid project no. FW-9-P-5.* Pratt, Kansas: Kansas Department of Wildlife and Parks.

Environmental Systems Research Institute. 1998. *ArcView GIS version 3.1 user's guide.* Redlands, California: Environmental Systems Research Institute, Inc.

Environmental Systems Research Institute. 2001. *ArcInfo GIS version 8.1 user's guide.* Redlands, California: Environmental Systems Research Institute, Inc.

Giesen, K. M. 1998. Lesser prairie-chicken (*Tympanuchus pallidicinctus*). In *The birds of North America, no. 364*, eds. F. Gill, and A. Poole. Philadelphia, Pennsylvania: The Birds of North America, Inc.

Hagen, C. A. 2003. *A demographic analysis of lesser prairie-chicken populations in southwestern Kansas: Survival, population viability, and habitat use.* Ph.D. dissertation, Kansas State University, Manhattan, Kansas.

Harner, R. F., R. C. Angell, M. A. Lobmeyer, and D. R. Jantz. 1965. *Soil surveys of Finney County, Kansas, series 1961, number 30.* Washington, DC: U. S. Department of Agriculture,

Haukos, D. A., L. M. Smith, and G. S. Broda. 1990. Spring trapping of lesser prairie chickens. *Journal of Field Ornithology.* 61: 20–5.

Jensen, W. E., D. A. Robinson, Jr., and R. D. Applegate. 2000. Distribution and population trend of lesser prairie-chicken in Kansas. *Prairie Naturalist.* 32: 169–75.

Johnson, R. R., K. F. Higgins, and D. E. Hubbard. 1995. Using soils to delineate South Dakota physiographic regions. *Great Plains Research.* 5:309–22.

Kuchler, A. W. 1974. A new vegetation map of Kansas. *Ecology.* 55: 586–604.

Leica Geosystems. 2003. *ERDAS imagine version 8.7 field guide.* Atlanta, Georgia: Leica geosystems GIS and mapping, LLC.

Manly, B. F. J. 1998. *Randomization, bootstrap and Monte Carlo methods in biology.* London, United Kingdom: Chapman and Hall,

Peterson, M. J., and N. J. Silvy. 1996. Reproductive stages limiting productivity of the endangered Attwater's prairie chicken. *Conservation Biology.* 10: 1,264–76.

Pitman, J. C. 2003. *Lesser prairie-chicken nest site selection and nest success, juvenile gender determination and growth, and juvenile* *survival and dispersal in southwestern Kansas*. M. S. thesis, Kansas State University, Manhattan, Kansas.

Sexson, M. L. 1980. Destruction of sandsage prairie in southwest Kansas. In *Proceedings of the seventh North American prairie conference*, ed. C. L. Kucera. Springfield, Missouri: Southwestern Missouri State University.

U. S. Census Bureau. 2003. *State and county quick facts. Finney County, Kansas*. http://quickfacts.census.gov/gfd/state/20/2055.html.

U. S. Department of Commerce. 2003. *National Oceanic and Atmospheric Administration, national climatic data center*.http:// www.ncdc.noaa.gov/.

Wisdom, M. L., and L. S. Mills. 1997. Sensitivity analysis to guide population recovery: Prairie-chickens as an example. *Journal of Wildlife Management*. 61: 302–12.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. *Ecology*. 70: 164–8.

Assessing Impacts of Oil and Gas Development on Mule Deer

Hall Sawyer
Western Ecosystems Technology, Inc.
Cheyenne, Wyoming

Frederick Lindzey
U. S. Geological Survey, Wyoming Cooperative Fish and Wildlife Research Unit
Laramie

Introduction

Increased levels of energy exploration and development on public lands, particularly for natural gas, have generated much concern over the welfare of wildlife populations that reside on these lands. Changes to the habitats these animals rely on are often obvious, such as the replacement of native vegetation with well pads, roads and pipelines. Added to these direct losses may be the subtle, indirect habitat losses caused by avoidance of areas in and around structures associated with development. Behavioral changes may result from increased levels of traffic, noise, pollution or human activity. Unfortunately, we know little about the impacts, either direct or indirect, of natural gas development on wildlife populations. There is a need among land management and wildlife agencies to better understand potential impacts of oil and gas development on wildlife to ensure that informed land-use decisions are made, reasonable and effective mitigation measures are identified, and appropriate monitoring programs are implemented.

We suggested at the 67th North American Wildlife and Natural Resources Conference (Sawyer et al. 2001) that population-level effects predicted from results of earlier, often disparate studies would face stiff challenges. Ultimately, we would need long-term, experimental research to document the degree to which oil and gas development may affect mule deer (*Odocoileus hemionus*) and other ungulate populations. It is not surprising that little data exist to demonstrate this assumed cause and effect relationship; many variables must be accounted for over a relatively long time, and someone must fund these efforts. While we are aware of some National Environmental Policy Act- (NEPA-) related monitoring efforts conducted for oil and gas development projects over the last 20 years, these efforts are not of the quality needed to contribute to our understanding of the relationship between mule deer populations and alterations to their habitat resulting from oil and gas development projects. To the best of our knowledge, no NEPA-related monitoring program involving oil and gas development, or involving potential impacts to ungulates in the Intermountain West, has ever been published in a peer-reviewed journal. This information gap makes it difficult for agencies and industry to improve their planning process, and it may limit their ability to develop energy resources in ways that are environmentally sensitive to ungulate species. The typical pattern has been to direct most of our efforts at simply mitigating the impacts gas development is assumed to have on mule deer populations and to be satisfied when our efforts result in these unverified concessions for wildlife in development plans. The goal should be to learn while mitigating, if we expect to improve our ability to protect wildlife in areas where energy resources will be developed.

We propose a three-step approach that should be included in future monitoring efforts, if we wish to learn about the response of wildlife populations in habitats modified by oil and gas development. The preferred design for monitoring wildlife response to development is the Before/After Control/Impact (BACI) method (Morrison et al. 2001); however, this may not always be possible, and the absence of predevelopment data, should not be used as a reason to not carefully design and conduct monitoring over the life of the project. Minimally, research or monitoring designs should include the following three steps to test the interrelated predictions that disturbance can be identified and quantified, that animals will respond to this disturbance and that there will be a measurable effect at the population level.

1. Document and quantify direct habitat loss that results from development activities (i. e., well pads, road networks, pipelines).
2. Document dispersion and behavior of species of interest during development. Specifically, do behaviors, dispersion or habitat-use patterns change through time as the project area is developed (indirect habitat loss)?
3. Measure population characteristics (e. g., survival, density, reproduction) during development.

Step 1: Documenting and Quantifying Direct Habitat Loss

Because most oil and gas development projects on federal lands require some level of NEPA analysis, direct impacts are generally estimated for the entire project area over the life of the project. For example, the U. S. Bureau of Land Management (BLM) Record of Decision (ROD) for the Pinedale Anticline environmental impact statement (EIS) approved 700 producing well pads, 276 miles (444 km) of roads and 400 miles (644 km) of pipeline for the 10 to 15-year project (U. S. Bureau of Land Management 2000). Although such figures are required in the NEPA process and provide a measure of the potential impacts of the project, land and wildlife managers are hindered in their development of monitoring or structured mitigation programs because they have little idea of when, where and to what degree development will occur. Additionally, project area planning boundaries tend to be much larger than the area actually developed, and development is rarely distributed evenly across project areas. Documenting and quantifying direct habitat loss is a critical element in any research or monitoring program for many reasons.

1. The complexity of leases, geologic uncertainties and gas price fluctuations require that industry be given flexibility in their development of project areas. Placement of wells and supporting structures and timing of development are, as a result, somewhat dynamic. Nonetheless, it is necessary to document both the temporal and spatial characteristics of development to investigate the influence that development may have on resident wildlife.
2. Project plans on federal lands reflect input from natural resource agencies and interest groups, with specified well density, miles of roads and pipelines often viewed as the primary input they have in mitigating potential impacts on wildlife populations. Accordingly, documenting adherence to these specifications is necessary.
3. Only concurrent monitoring of development and wildlife populations will allow documentation of the level of disturbance at which a response is elicited in the wildlife population.
4. Only when direct habitat loss is quantified can we begin to assess indirect losses (Step 2).

Documenting and quantifying habitat loss is often overlooked in project designs because we assume it is easily done because it can be borrowed from others that are doing it. In practice however, documenting direct losses may be difficult for several reasons.

1. Project areas are often large, making them difficult to monitor and to accurately map new structures as the project develops, particularly with traditional ground-based methods. Additionally, coal-bed methane and traditional deep-well natural gas development are now occurring so rapidly that ground-based monitoring is becoming impractical.
2. Land ownership patterns are often mixed, with private and public lands developed differently and regulated by different agencies, but they all contribute to potential habitat changes in the area of concern. Even if the various agencies administering development on public lands and operators developing private lands freely shared their records, records likely would differ in resolution, timing and quality, making it difficult to combine them into a single document.
3. Wildlife monitoring or inventory responsibilities for NEPA-related projects are often not well defined.

Objectively documenting development activities for compliance, monitoring and research needs is important, yet we have oversimplified or underestimated the effort needed to accomplish this. We suggest using satellite imagery and geographic information system (GIS) technology to document and quantify direct impacts by measuring habitat losses resulting from construction of roads and well pads. Satellite imagery provides objective, cost-effective data from large areas and can be easily acquired on an annual basis. Enhanced thematic mapper (ETM) images cover an area approximately 100 miles by 100 miles (160 km x 160 km) and can be purchased for about $600 from the U. S. Geological Survey (USGS) Earth Resources Observation Systems (EROS) Data Center (2004). Original ETM data are provided in a binary raster format with detached headers, so creating color graphics of the image and converting it to a user-friendly GIS format requires additional image processing. However, following processing, the image can be used to accurately map and digitize road networks and well pads associated with oil and gas development. Additionally, the image can be used as a base layer to map animal distribution data and generate other project area maps. Satellite images of the Pinedale Anticline Project Area

(PAPA), in western Wyoming, illustrate the progression of development in the first 3 years of this project, 2001 to 2003 (Figure 1).

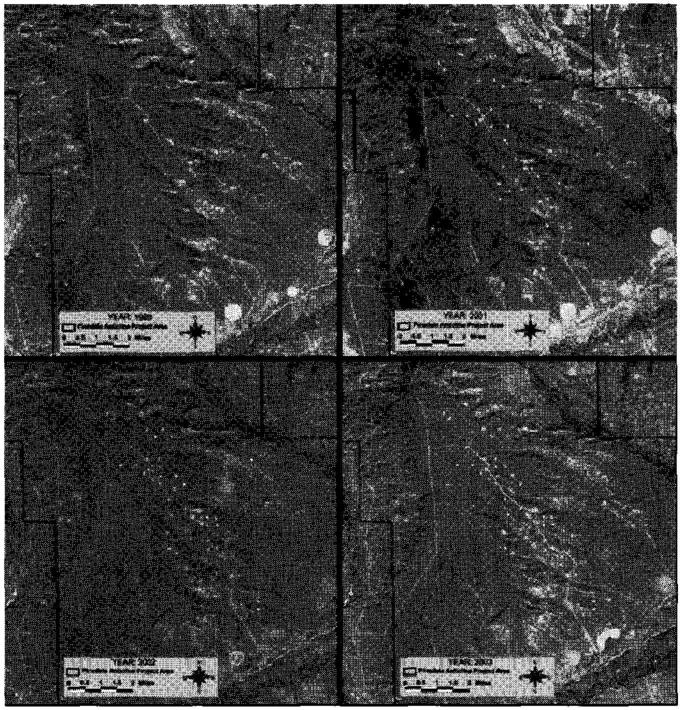


Figure 1. Satellite images depicting progression of natural gas development in Pinedale Anticline Project Area, located in western Wyoming, 1999 to 2003.

Step 2: Documenting a Response and Calculating Indirect Effects or Habitat Loss

It is relatively easy to quantify habitat change that results from conversion of native vegetation to well pads or roads. Less easy will be any subtle reduction in the effective value of habitats to mule deer because of the habitat's proximity to structures or human activities associated with energy development. Although effective habitat loss or indirect impacts have been documented in elk (*Cervus elaphus*) (Lyon 1983, Wisdom et al. 1986, Czech 1991, Morrison et al. 1995, Rowland et al. 2000, Powell 2003), data that suggest similar behavior in mule deer (Rost and Bailey 1979, Yarmaloy et al. 1988, Easterly et al. 1992, Merrill et al. 1994) are limited and largely observational in nature. Documenting movement and activity patterns of mule deer as development proceeds is the necessary first step in identifying indirect losses, if they occur. Collaring and monitoring deer with either conventional very high frequency (VHF) radio transmitters or global positioning system (GPS) receivers will provide these data. GPS data are preferred and lack the inherent bias often associated with traditional telemetry methods because they are systematically collected, irrespective of human error, habitat type, poor weather conditions or time of day. Forest density and canopy cover may reduce the fixed rates and positional accuracy of GPS collars (Rempel et al. 1995, Moen et al. 1996, D'Eon et al. 2002, Di Orio et al. 2003). However, most BLM lands across the Intermountain West are characterized by open shrublands and gentle topography that allow for optimal satellite coverage and GPS performance. Since the discontinuation of selective availability, three-dimensional GPS locations typically have less than 66 feet (20 m) error (Di Orio et al. 2003).

Regardless of how distribution data are collected, statistical analyses should be used to examine distribution patterns in relation to development features. Visual comparison of distribution patterns over time may be a useful first step, but analyses should be taken further, so changes in distribution can be statistically detected and quantified. Probability density functions (Marzluff et al. 2001) and resource selection functions (Manly et al. 2002) are two promising methods that account for the relative amount of use and can estimate relationships between animal distribution and habitat variables of interest. Resource selection functions and similar approaches can identify significant trends in data sets that are not readily detected with visual plots. Quantified relationships between deer and development-related habitat features will be needed to demonstrate response and thus support inferences that may be drawn about population-level effects. Further, statistically sound relationships resulting from these analyses will provide the basis for defensible mitigation strategies aimed at reducing the possibility that development activities will negatively influence dynamics of mule deer populations.

Step 3: Monitoring Populations to Document Population Level Changes

Plans to develop energy resources on mule deer ranges are commonly confronted with concerns that exploration and development activities will result in harm to the mule deer population. As noted earlier, although the logical combination of individual studies of mule deer ecology, nutrition and physiology offer compelling reason for concern, population level effects conclusively related to energy development have not been demonstrated. For this reason alone, populations should be properly monitored during energy development projects on public lands. Consistent with monitoring recommendations outlined by White and Bartmann (1998a), we suggest adult female survival, overwinter fawn survival, reproduction and density are parameters that should be estimated at least every other year, through the life of the project. Adult female survival can be estimated by monitoring an adequate and representative sample of radio-collared animals (White and Bartmann 1998a, Powell et al. 2000, Winterstein et al. 2001), perhaps the same animals used to document distribution patterns. It should be remembered that, although the number of radio-collared animals may refine the quality of the measurement (i. e., survival), sample size (n) for determining the treatment effect is the number of years of study. Over-winter fawn survival can be estimated indirectly using change in ratio estimators with data obtained from fall and spring composition surveys adjusted for adult female survival (White et al. 1996). Although more expensive and labor-intensive, overwinter fawn survival can be directly estimated by radio-tagging fawns (White and Bartmann 1998b). State wildlife agencies often conduct annual composition surveys to estimate sex and age classes in populations, with doe:fawn ratios obtained from these surveys used as an index to reproduction (Czaplewski et al. 1983). It is likely these data are already available for most areas across the Intermountain West.

Density may be estimated in a number of ways, including capture-recapture estimators using the radio-collared deer as the marked sample or block count approaches that may or may not use radio-collared animals to correct for missed individuals (Thompson et al. 1998, White and Shenk 2001). We recognize changes in population characteristics may be subtle and that variation in parameter estimates high. However, monitoring four population parameters, rather than one, allows us to build a preponderance of evidence argument for or against population-level impacts.

Conclusion

Prior to 1983, the Mineral Management Service (MMS) was responsible for oil and gas leasing on federal lands. In 1983, however, the BLM initiated the

Resource Management Plan (RMP); with it, came the responsibility of administering oil and gas leases on federal lands. The BLM manages all federally owned minerals, whether they are under private surface or a surface administered by another federal agency. Most BLM field office RMP and NEPA documents (e. g., EIS, EA) include a wildlife monitoring component. Unfortunately, after 20 years of implementation of the BLM's RMP, in conjunction with NEPA, we have virtually no information on the potential impacts of oil and gas development on mule deer populations across the Intermountain West. And, we have no measure of success of any mitigation measures (e. g., timing restrictions, well density, surface occupancy) that may have been included in plans. This lack of information has sometimes led to acrimonious debates over the potential impacts of past and future projects. While compliance standards are generally met, there is certainly room for improvement in monitoring of wildlife during oil and gas exploration and development. Implementing this three-step approach to impact assessment will result in monitoring programs that document direct and indirect habitat losses, as well as determine if measurable population-level effects have occurred. Ultimately, these programs will provide a better understanding of impacts oil and gas development may have on ungulate populations, thereby improving management opportunities and ensuring development plans are sensitive to mule deer.

Our experiences in western Wyoming suggest that some in industry are willing to support efforts to better understand the relationship between the dynamics of mule deer populations and habitat disturbances that are an inevitable result of energy development activities. Ultra Petroleum and Questar Exploration and Production Company have provided the majority of funding for wildlife research associated with oil and gas development in western Wyoming (Sawyer and Lindzey 2000, Sawyer et al. 2002, Lyon and Anderson 2003). The Wyoming Game and Fish Department and the BLM contributed monies as they could and provided invaluable logistic support; although, both agencies had many other pressing wildlife issues and responsibilities. The numerous nongovernmental organizations (NGOs) interested in this part of Wyoming have made extensive use of data generated from these studies and have sponsored workshops to synthesize the state of knowledge and inform local, state and national decision makers.

Yet, long-term financial support has not materialized and the vagaries of year-to-year funding do little to foster long-term efforts. It is unfortunate but, at the same time, not surprising that we have little in the way of long-term work; traditional funding sources, such as agencies and NGOs often do not have the luxury of such commitments when limited resources need to be continually shifted to address more immediate needs. On the other hand, the framework already exists within the NEPA process to accommodate long-term efforts. Past failure of this framework to yield new information, even that sufficient to evaluate success of imposed mitigation measures, may be due in part to the perceived absence of techniques and approaches needed to appropriately monitor disturbance and populations. Recent development of new techniques and analysis tools (e. g., GIS, GPS, satellite imagery) make it possible to incorporate the three steps we have outlined above into the NEPA wildlife-monitoring framework. We encourage industry, agencies and NGOs to work together to ensure proper monitoring programs are designed, implemented and adequately funded.

Reference List

Czaplewski, R. L., D. M. Crowe, and L. L. McDonald. 1983. Sample sizes and confidence intervals for wildlife population ratios. *Wildlife Society Bulletin*. 11:12–8.

Czech, B. 1991. Elk behavior in response to human disturbance at Mount St. Helens National Volcanic Monument. *Applied Animal Behaviour Science*. 29:269–77.

D'Eon, R. G., R. Serrouya, G. Smith, and C. O. Kochanny. 2002. GPS radiotelemetry error and bias in mountainous terrain. *Wildlife Society Bulletin*. 30:430–9.

Di Orio, A. P., R. Callas, and R. J. Schaefer. 2003. Performance of two GPS telemetry collars under different habitat conditions. *Wildlife Society Bulletin*. 31:372–9.

Easterly, T., A. Wood, and T. Litchfield. 1992. *Responses of pronghorn and mule deer to petroleum development on crucial winter range in the Rattlesnake Hills*. Cheyenne, Wyoming: Wyoming Game and Fish Department.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. *Wildlife Society Bulletin*. 31:486–91.

Lyon, J. L. 1983. Road density models describing habitat effectiveness for elk. *Journal of Forestry*. 81:592–4.

Manly, B. F., L. McDonald, D. Thomas, T. McDonald, and W. Erickson. 2002. *Resource selection by animals: Statistical design and analysis for field studies, second edition*. Boston, Massachusetts: Kluwer Academic Publishers.

Marzluff, J. M., S. T. Knick, and J. J. Millspaugh. 2001. High-tech behavioral ecology: Modeling the distribution of animal activities to better understand wildlife space use and resource selection. In *Radio tracking and animal populations*, 309–26. San Diego, California: Academic Press.

Merrill, E. H., T. P. Hemker, K. P. Woodruff, and L. Kuck. 1994. Impacts of mining facilities on fall migration of mule deer. *Wildlife Society Bulletin*. 22:68–73.

Moen, R., J. Pastor, Y. Cohen, and C. Schwartz. 1996. Effects of moose movement and habitat use on GPS collar performance. *Journal of Wildlife Management*. 60:659–68.

Morrison, J. R., W. J. deVergie, A. W. Alldredge, A. . Byrne, and W. W. Andree. 1995. The effects of ski area expansion on elk. *Wildlife Society Bulletin*. 23:481–9.

Morrison, M. L., W. M. Block, M. D. Strickland, and W. L. Kendall. 2001. *Wildlife study design*. New York, New York: Springer.

Powell, J. 2003. *Distribution, habitat use patterns, and elk response to human disturbance in the Jack Morrow Hills, Wyoming*. M. S. thesis, Laramie: Zoology and Physiology Department, University of Wyoming.

Powell, L. A., M. J. Conroy, J. E. Hines, J. D. Nichols, and D. G. Krementz. 2000. Simultaneous use of mark-recapture and radiotelemetry to estimate survival, movement, and capture rates. *Journal of Wildlife Management*. 64:302–13.

Rempel, R. S., A. R. Rodgers, and K. F. Abraham. 1995. Performance of a GPS animal location system under boreal forest canopy. *Journal of Wildlife Management*. 59:543–51.

Rost, G. R. and J. A. Bailey. 1979. Distribution of mule deer and elk in relation to roads. *Journal of Wildlife Management*. 43:634-41.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Sawyer, H., and F. Lindzey. 2000. *The Jackson Hole pronghorn study*. Laramie, Wyoming: Wyoming Cooperative Fish and Wildlife Research Unit.

Sawyer, H., F. Lindzey, D. McWhirter, and K. Andrews. 2001. Potential effects of oil and gas development on mule deer and pronghorn populations in western Wyoming. In *Transactions of the 67th North American wildlife and natural resources conference*, 350–65. Washington, DC: Wildlife Management Institute.

Sawyer, H., D. Strickland, and L. McDonald. 2002. *Summary of 2002 project results and scope of long-term monitoring plan to assess potential impacts of energy development on mule deer in the Pinedale Anticline Project Area.* Cheyenne, Wyoming: Western Ecosystems Technology, Inc.

Thompson, W. L., G. C. White, and C. Gowan. 1998. *Monitoring vertebrate populations*. San Diego, California: Academic Press.

U. S. Bureau of Land Management. 2000. *Record of decision: Environmental impact statement for the Pinedale Anticline natural gas field exploration and development project.* Pinedale, Wyoming: Pinedale Field Office.

U. S. Geological Survey, Earth Resources Observation Systems. 2004. *USGS-EROS data center*. http://edc.usgs.gov/.

White, G. C., A. F. Reeve, F. G. Lindzey, and K. P. Burnham. 1996. Estimation of mule deer winter mortality from age ratios. *Journal of Wildlife Management*. 60:37–44.

White, G. C. and R.M. Bartmann. 1998a. Mule deer management-What should be monitored? In *Proceeding from the 1997 western states and provinces deer and elk workshop*, 104–18. Phoenix, Arizona: Arizona Game and Fish Department.

White, G. C. and R. M. Bartmann. 1998b. Effect of density reduction on overwinter survival of free-ranging mule deer fawns. *Journal of Wildlife Management.* 62:214–25.

White, G. C. and T. M. Shenk. 2001. Population estimation with radio-marked animals. In *Radio tracking and animal populations*, 329–50. San Diego, California: Academic Press.

Winterstein, S. R., K. H. Pollock, and C. M. Bunck. 2001. Analysis of survival data from radiotelemetry studies. In *Radio tracking and animal populations*, 351–80. San Diego, California: Academic Press.

Wisdom, M. J., L. R. Bright, C. G. Carey, W. W. Hines, R. J. Pedersen, D. A. Smithey, J. W. Thomas, and G. W. Whitmer. 1986. *A model to evaluate elk habitat in western Oregon, publication R6-F&WL-216–1986.* Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Yarmaloy, C., M. Bayer, and V. Geist. 1988. Behavior responses and reproduction of mule deer does following experimental harassment with all-terrain vehicle. *Canadian Field Naturalist.* 102:425–9.

Public Lands, Energy and Wildlife—Can We Have It All?

Rebecca W. Watson
U. S. Department of the Interior
Washington, DC

We must work to build a new harmony between energy needs and our environmental concerns. The truth is energy production and environmental protection are not competing priorities. They are dual aspects of a single purpose, to live well and wisely upon the earth.

President George Bush, National Energy Policy

The quality of life we North Americans take for granted is the envy of the rest of the world. Whether we want to admit it or not, our quality of life is totally dependent upon a reliable source of abundant and inexpensive energy. Our enviable quality of life also includes healthy populations of fish and wildlife for fishing and hunting or passive enjoyment by North Americans of all walks of life. Public and private energy experts are warning us of a difficult energy future if we do not take major steps to change our energy situation. In this context, I am pleased to join you today to discuss our, i. e., the United States', energy situation and your role in assisting this Administration's efforts to promote both the conservation of our fish and wildlife resources and the environmentally responsible development of our domestic energy resources.

North America's Energy Situation

For the last 30 years, North America has been grappling with the consequences of the imbalance between its growing demand for energy and its ability to produce that energy domestically. The most memorable consequences of that imbalance were the oil crises of 1973 and 1978. During the 1990s, North America enjoyed a period of relative energy stability. Although both our energy demand and our dependence on foreign oil increased, energy was readily available. World politics did not seriously threaten our energy security. The United States did, however, make an energy and quality-of-life decision that is having an impact today. That decision was to promote the use of clean-burning natural gas for the generation of electricity. The electricity industry responded, and more than 90 percent of all new generating capacity is now fueled by natural gas. This new demand for natural gas competes with home heating and cooling and other industrial uses of gas for our limited supply. That increased demand is responsible for the high natural gas prices U. S. citizens are experiencing today. Energy experts agree that unless we make some substantial changes in our energy policies, The United States is headed for a prolonged and serious tightening of our natural gas supply.

There are some people who would argue that there is no energy problem in North America today. After all, we don't have the gasoline lines of the 1970s. We are, however, seeing clear evidence of emerging energy difficulties that may only get worse. Today, the United States imports more than 54 percent of the oil we use and that number will rise to 70 percent by 2025 (U. S. Energy Information Administration 2004). In 2002, we imported crude oil and refined products at an average rate of 443 million gallons (1. 68 billion 1) every day. That very large number equates to more than 307,000 gallons (1. 2 million 1) every minute. In 2002, we paid more than $90 billion for our imported petroleum. That very large number equates to $172,000 every minute. Consider that in the 15 minutes allotted for my presentation, we will spend nearly $2. 6 million on imported petroleum. And, that is only the direct monetary cost of our oil dependence. That is U. S. money flowing overseas for a commodity we have here. That is U. S. money that could be paying for U. S. jobs and could be providing benefits to local communities. Instead, it produces fewer primary or secondary jobs; it generates fewer taxes, it does not pay for local schools, police or social services. By 2025, our payments for imported petroleum will increase to more than $200 billion or $381,000 every minute (U. S. Energy Information Administration 2004).

This is important because the United States is now looking at a similar situation with natural gas. Today, we produce nearly 86 percent of the gas we use from our domestic resources (U. S. Energy Information Administration 2004). Most of our imported gas comes from Canada. Our use of gas is projected to increase by 38 percent during the next 20 years, but that production will not increase as rapidly. In addition, Canada's ability to provide gas will decrease in the future. By 2025, we will import 23 percent of the natural gas we use and will pay an average rate of slightly more than $64,000 every minute. Barring some energy miracle, the cost for this natural gas will be added to our increasing cost for imported oil. The good part of this story is that the United States has vast resources of natural gas and doesn't have to be a major gas importer.

Let's look at one other side of the energy situation, and then I will move on. On February 17, 2003, *The Wall Street Journal* published an article about the effects of high natural gas prices on North American companies; they were companies we all know—Unilever, Owens Coming and Dow Chemical—and they have had to adjust to record-high gas prices. While it is easy for some people to dismiss the problems of large corporations, one number stays with me. According to *The Wall Street Journal,* since the rise in natural gas prices in 2000, the U. S. chemical industry has lost 78,000 jobs. In one industry, 78,000 people faced, at a minimum, a major disruption in their lives. How many families are faced with the prospect of a lower standard of living or an unemployed parent as a result of this? Experts, including Federal Reserve Chairman Alan Greenspan, are very concerned that high natural gas prices will harm our economy and that many more U. S. workers will suffer the same fate as the chemical workers. Again, the sad part of this story is that we have large natural gas resources that, if produced, can alleviate some of the demand, which creates the higher prices, which, in turn, leads to the loss of jobs.

Role of U. S. Department of the Interior's Land in North America's Energy Situation

Federal land, both onshore and offshore, play a crucial role in U. S. domestic energy production. Offshore federal land provides 25 percent of our domestically produced gas and 30 percent of our oil (Minerals Management Service 2004). Onshore federal land provides 11 percent of our domestically produced gas and 5 percent of our oil (U. S. Bureau of Land Management 2004a). Onshore federal land also provides 38 percent of our coal. Except for oil, the contribution of public land to our domestic energy production is expected to grow significantly by 2025.

In addition to the traditional fossil fuels, the U. S. Department of the Interior (DOI) is encouraging the production of renewable energy, such as wind, biomass, geothermal and solar, on the land it manages. Today, federal land and facilities provide 17 percent of our hydropower, 10 percent of our wind energy and 48 percent of our geothermal energy. The U. S. Bureau of Reclamation's 58 hydropower plants generate enough electricity to supply the annual needs of more than 9 million people. Without energy resources on federal land, we would be substantially more dependent on foreign countries for the energy we rely upon to support our lifestyle. Moreover, in our western states, where federal land is 30 to 80 percent of a state's total land area, energy development helps rural economies by providing new and well-paying jobs and by increased tax revenues to support local schools and public services.

With this information in mind, I would like to dispose of a myth that seems to accompany most discussions of energy and public-land management. President George W. Bush's National Energy Policy does not (I repeat does not) give the oil and gas industry *carte blanche* on U. S. public land. As important as energy is to our economy, military security and lifestyle, the policy requires that energy resources be produced in a manner that minimizes the environmental consequences of their production. Our energy needs do not trump our fish and wildlife resources. We need them both; they are two sides of the quality-of-life coin.

Because we are here today to discuss concerns regarding activities of the U. S. Bureau of Land Management (BLM) related to the national energy policy, I will focus my remarks on those efforts.

The BLM is a multiple-use agency. It manages 261 million surface acres (106 million ha) of public land and nearly 700 million acres (284 million ha) of subsurface federal mineral estate. It has a responsibility to supply energy that the public demands while protecting the natural resources that same public loves to enjoy. The BLM manages spectacular landscapes and historic resources, critical habitat for endangered species, wetlands and wildlife resources, and untrammeled-by-human-wilderness and wild-and-scenic river corridors. Public land supports a myriad of activities, including most forms of outdoor recreation, transportation, grazing, timber harvesting, mining, and oil and gas development.

Taking Care of the Land and Its Living Resources

The BLM manages public land in a variety of ways to protect wildlife resources. In 2000, the BLM combined the land it manages under various protective prescriptions into its National Landscape Conservation System. (U. S. Bureau of Land Management 2003) The system includes national monuments, national conservation areas, wilderness, wilderness study areas and other congressional and administrative designations. With a few exceptions, these areas are off-limits to oil and gas activities. In 2000, this system included more than 39 million acres (16 million ha) of land, about 15 percent of the total land area managed by the BLM. By the end of 2002, that area had increased by an additional 4. 5 million acres (1. 8 million ha).

Beyond the National Landscape Conservation System, BLM designates areas of critical environmental concern (ACECs) to protect fish and wildlife and other natural and cultural resources from irreparable effects of human activity. At the end of 1998, BLM had designated 13.1 million acres (5.3 million ha) of ACECs. That area increased by nearly 900,000 acres (365,000 ha) by the end of 2002 (U. S. Bureau of Land Management 2004b). ACECs occupy 5 percent of BLM's total land area.

In addition to setting land aside for long periods of time, the BLM also restricts activities for shorter periods to protect fish and wildlife resources at critical periods of their life cycles. In 2003, the BLM closed approximately 106,000 acres (43,000 ha) to protect a variety of wildlife species because existing conditions warranted preventing their disturbance by human activities. Just recently, BLM's Kemmerer Field Office in Wyoming restricted motorized travel on nearly 350,000 acres (142,000 ha) of big game habitat to minimize stress on wintering big game. The BLM also closes areas that are recovering from the effects of wildfire to promote the recovery of healthy ecosystems that can support abundant and diverse fish and wildlife populations. For our oil and gas program, leases may include no-surface-occupancy or scheduling restrictions that prevent or minimize the intrusion of oil and gas activities on fish and wildlife during sensitive periods of their lives.

In addition to setting land aside to protect fish and wildlife resources, the BLM also manages habitat, in cooperation with states, to maintain healthy fish and wildlife populations. A census of big game populations on public land indicates that these populations are relatively stable (U. S. Bureau of Land Management 2004b). Populations of antelope, big horn sheep, caribou, elk and turkey on public land all increased by 5 percent or more between 1998 and 2002. The population of elk on public land increased by 16 percent during that same period. Wildlife management continues to be a high priority for the BLM. Our goal is to maintain healthy populations of fish and wildlife with which our land is blessed.

In recognition that private citizens are often the Unites States' most effective conservationists, President Bush developed the Challenge Cost Share Initiative to promote citizen stewardship. Under this initiative, the Administration provides matching financial support to conservation efforts conceived and conducted at the local level. These efforts achieve results that could not be achieved by the government alone. In 2003, the initiative supported more than 700 partners. The BLM was a major contributor to this initiative. In 2003, BLM provided nearly $5 million to 279 partners involved in 88 separate projects. Among those projects, BLM partnered with the Audubon Research Ranch in Arizona to restore native grasslands, the Agua Callente Indians to improve riparian habitat in California, the Rocky Mountain Elk Foundation on habitat improvement projects in six western states, and 15 local groups to restore wildlife habitat near Glenwood Springs, Colorado. In addition to the environmental benefits provided by this initiative, it also establishes new relationships and strengthens those that already exist. It provides common ground from which understandings of other issues, such as oil and gas leasing, can be developed.

Public Land Management Decisions Involve the Public

Because of our stewardship responsibilities, we in the U. S. Department of the Interior (DOI) (and the BLM in particular) must constantly seek to ensure the protection of some resources and the production of others. How do we manage public resources in a balanced manner that reflects the best interests of the U. S. people? As President Bush and Secretary Norton, of the DOI, have said repeatedly, we take our direction from the people of the United States. That direction is first expressed through Congress. Article 11 of the Constitution gives Congress broad plenary authority over our public land. DOI can manage this land only to the extent that Congress has delegated its authority. Congress has said in the Federal Land Policy and Management Act (FLPMA) that land management decisions must be made through a public, land-use planning process. Congress has also directed that agency actions are subject to the public disclosure of environmental and socioeconomic impacts that are required by the National Environmental Policy Act (NEPA). And, our decisions are guided by other statutes to protect the environment (Clean Air Act, Clean Water Act); by other historic resources (National Historic Preservation Act) and by other fish and wildlife litigation (Endangered Species Act, Lacey Act, Migratory Bird Treaty Act).

Land Use Planning Is the Foundation

The primary methods used by the BLM to establish its priorities for managing public land at the local level are its 162 resource management plans.

These plans are the basis for every action and use of public land approved by BLM. These plans are prepared through a public process, which includes considerable collaboration with affected federal, state and local agencies, Native American tribes and the public. Their preparation normally includes the development of an environmental impact statement with several opportunities for public review and comment.

At DOI, Secretary Norton has emphasized a new way of doing business—the four Cs: consultation, cooperation and communication, in the service of conservation. That means moving away from the formal "the federal government knows best" style of land management to a style in which DOI agencies work in partnership with communities of place and interest. Examples of these partnerships include Challenge Cost Share Initiative grants, Take Pride in America, BLM's Shared Community Stewardship, and U. S. Fish and Wildlife Service private and Native American land grants for fish and wildlife restoration and enhancement. Many of you in this room are our partners on wildlife enhancement projects from Florida to Alaska. We work together to go beyond the requirements of law, to do more than the minimum.

We have found that the most successful planning efforts occur when communities agree upon desired outcomes and the means to achieve those conditions. The Council on Environmental Quality (CEQ) has provided a significant impetus to citizen involvement in its NEPA guidance of 2003. This guidance directs federal agencies to afford the opportunity to states, counties and local communities to become cooperating agencies in the NEPA analysis of federal activities in their vicinity. This takes communities from a passive role of simply commenting to that of an equal partner with other parties in the analysis of the alternatives considered in the NEPA process. These two approaches, the four Cs and cooperating agency status, will improve public participation, enhance citizen stewardship and result in decisions that better reflect the values of our citizens.

The key to good land-use planning is good information. This is an area in which this community can be of significant assistance to the BLM. In many areas, wildlife groups and state wildlife agencies have developed valuable information but have not provided that information to BLM in the planning process, in the belief that BLM already has it. The BLM must communicate more effectively with the public in soliciting available information, but state wildlife agencies, private wildlife groups and communities must also be assertive in bringing information they believe is important to the planning process. BLM cannot make good land use decisions based on poor or inadequate information. We need and welcome your input.

Most of BLM's resource management plans were completed in the early 1980s and 1990s. As a result of changing social, economic and environmental conditions, of increased demands for energy and other minerals and of technological advances, these plans are becoming increasingly obsolete. As a consequence, many of these plans may no longer provide an appropriate management framework for making decisions about land-use authorizations.

In response to these general deficiencies, BLM committed to updating all of its resource management plans. In 2001, the BLM identified 21 time-sensitive plans that merited immediate attention. Although some people have asserted that these plans were designated only to promote energy development, that is not the case. These plans also address such pressing issues as managing the interface between growing urban areas and public land, increased demand for recreation, wildland fire management, newly designated endangered and threatened species, increased demands for mineral development, need for rights-of-way, and new land designations, particularly monuments. In fact, less than half of the 21 time-sensitive plans have energy development as a primary cause for revision.

The Energy Policy and Conservation Act and Land-use Planning

Recently, public attention has been focused on a report published by the DOI, the U. S. Department of Agriculture and the U. S. Department of Energy on the availability of federal land in five western oil and gas basins for energy development (U. S. Department of the Interior, U. S. Department of Agriculture, U. S. Department of Energy 2003). The report was required by the Energy Policy and Conservation Act Amendments of 2000 (EPCA). The five basins are important because they include 59 million acres (24 million ha) of federal land and overlie most of the onshore natural gas and much of the oil in public ownership within the contiguous 48 states. These basins hold our second largest domestic natural gas resource, exceeded only by the Gulf of Mexico.

The public focused on the information about the availability of land for leasing, but, from a purely management perspective, the review revealed some interesting information. The BLM and the U. S. Forest Service have about 1,000 separate restrictions on oil and gas operations in these five basins. Unfortunately, these restrictions are not being evaluated and applied in a systematic and scientific manner. In some cases, these restrictions change at political boundaries even though the environment and wildlife on each side of the boundary are identical. In other cases, antiquated restrictions are being applied when more recent, scientific information indicates that they are either ineffective or unnecessary. This situation serves neither the public nor the environment.

Based on these findings, the BLM recently directed its field offices to review its restrictions on oil and gas operations to ensure that they remained effective in protecting fish and wildlife without being overly burdensome on industry. Neither interest, wildlife or energy, should trump the others. Differences between field offices on similar resources in similar situations were to be reconciled based upon the best available information. That process is occurring as we speak.

Even the Best Plans Change

The EPCA study demonstrated that the BLM has a long way to go to institute adaptive management into its land-management processes. Adaptive management requires using the best available information to make decisions, monitoring the consequences of those decisions and modifying decisions when monitoring demonstrates that desired outcomes are not occurring or that different approaches can provide similar or better results.

Adaptive management is often reflected in decisions related to lease stipulations and other postlease conditions of approval. Lease stipulations are conditions that BLM places on a lessee to protect resources, such as fish and wildlife or recreational, historic or archaeological sites. Lease stipulations can delay operations, change operation locations, or deny operations within the terms of the lease. These authorities are vested in the BLM-authorized officer in every standard oil and gas lease.

When BLM determines that a lease stipulation is no longer necessary or is ineffective, it must either change the stipulation or consider granting waivers, exceptions or modifications. The processes and circumstances for granting waivers, exceptions or modifications are documented in most existing resource management plans and are required in all new plans. A manager may waive a stipulation by granting a permanent exception to its conditions. A manager may grant a one-time exception to a stipulation based on a case-by-case analysis of the situation. A manager may also modify a lease stipulation to change its provisions either temporarily or for the term of the lease. Making significant changes to a lease stipulation requires revising the resource management plans with the accompanying environmental analysis and opportunity for public review and comment. BLM managers, however, may make minimal changes to stipulations and other conditions without public review and comment.

When processing an application for permit to drill (APD) or related-use authorization, stipulations associated with the lease must be reviewed in a site-specific NEPA analysis prior to any decision authorizing use. If the lessee or operator requests a change in a stipulation, BLM must consider that request. If BLM determines that fulfilling the lessee's request will not lead to irreparable environmental damage, it may grant a waiver, exception or modification of the stipulation. The BLM has received numerous requests for exceptions to stipulations and other conditions of approval. Industry normally consults with the BLM prior to formally submitting a request for an exception. As a result, a preliminary examination of the circumstances surrounding the request has occurred prior to any formal action. If based on this informal process, BLM concludes that the request may be approved, and formal consideration may commence. Such consideration may or may not include public review and comment, depending upon the significance of the decision. The informal consultation process between the lessee and BLM decreases the number of requests for exceptions that would most likely be denied formally. Denials of requests for exceptions are generally informal and are not documented. As a result, the public may believe that no requests for exceptions are denied, when in fact, many more requests may be denied than are approved.

I have heard concerns from the wildlife community that a large percentage of lease stipulations are waived with the accompanying implication that industry can easily change these conditions. This perception may be the result of the BLM failing to communicate effectively with you about this practice. This is especially true where a large number of environmental waivers are being requested. For those of you interested in tracking the exceptions being granted by local BLM offices, I suggest that you discuss your interest with the field office manager. The manager can arrange to provide you the information in which you are interested and discuss with you the basis for granting specific waivers. Secretary Norton's four Cs require no less.

The BLM planning and use-authorization processes are continuous. They never really end. As a result, the BLM can accept information at any time. Depending upon its significance, the information may be immediately woven into the decision-making process, or it may be maintained and incorporated later. You do not have to wait for a formal comment period to provide information you believe would improve BLM decisions.

Summary

The U. S. lifestyle depends upon energy. U. S. citizens consume more energy than any other country in the world, and our consumption is projected to increase. We are now facing the consequences of our increasing demand for natural gas, which is outstripping our ability to produce it domestically or to import it. High prices for natural gas are affecting our businesses. Experts are concerned that high natural gas prices will harm our economy and our workers.

North America has a large natural gas resource base, and much of that gas is beneath public land. Development of our domestic natural gas in an environmentally responsible manner is a national imperative. The national energy policy calls for increased gas production but does not give industry carte blanche on public land. Gas must be produced in a manner that protects the environment from the consequences of its production.

Our public land is also home to world-class fish and wildlife resources. The BLM must carefully manage the use of public land and the protection and enhancement of fish and wildlife resources. This may entail prohibiting uses in certain areas or placing restrictions on uses to avoid conflicts with fish and wildlife, especially in critical phases of their life cycles.

Public land is managed adaptively and with considerable consultation and communication with the public of places of interest. New information on the effectiveness of use restrictions is necessary to determine whether or not they remain effective without being overly burdensome.

As our energy demands increase, so will the conflicts between and among users of our public land. Americans show no signs of decreasing their demand for energy and appear to believe that inexpensive and abundant energy is a right rather than good fortune. As the U. S. energy situation tightens in the coming years, we will be more prone to support energy development activities that we might not support today. Maintaining our life styles will require these changes of opinion.

We need to ensure that we have established a means to effectively communicate our interests in protecting and enhancing our environment in the face of the impending tightening of our energy supply. We cannot be unprepared for ill-conceived responses to energy problems that forsake our environment for energy production because of political expediency. Our time is running out; the tightening of the world energy supply has just begun. Let us be prepared to make the right decisions when we are called.

Reference List

Gold, R. 2004. Natural-gas costs hurt U. S. firms. *The Wall Street Journal.* February 17.

Minerals Management Service. 2004. *Offshore minerals management.* http://www.mms.gov/offshore/.

U. S. Bureau of Land Management. 2003. *National landscape conservation association.* http://www.blm.gov/nlcs/.

———. 2004a. *National Energy Office: Energy facts.* http://www.blm.gov/energy/facts.html.

———. 2004b. *Bureau of Land Management information.* http://www.blm.gov/nhp/info/index.htm.

U. S. Departments of the Interior, Agriculture and Energy. 2003. *Scientific inventory of onshore federal lands oil and gas resources and reserves and the extent and nature of restrictions or impediments to Their Development.* http://www. blm. gov.

U. S. Energy Information Administration. 2004. *Annual energy outlook 2004 with projections to 2025.* http://www.eia.doe.gov.

Drilling in the Rocky Mountains: How Much and at What Cost?

Pete Morton
The Wilderness Society
Denver, Colorado

Chris Weller
The Wilderness Society
Seattle, Washington

Janice Thomson
The Wilderness Society
Seattle, Washington

Michelle Haefele
The Wilderness Society
Denver, Colorado

Nada Culver
The Wilderness Society
Denver, Colorado

Introduction

The Wilderness Society is a 200,000-member national conservation group that focuses on public land management issues. The Wilderness Society scientists conduct energy research and regularly present the results at congressional hearings and other forums. Our research is guided by the need to examine the explicit and implicit assumptions that underlie the national energy plan unveiled by the George W. Bush Administration in May 2001. Among other things, the plan called for the opening of additional public land in western states to gas and oil drilling, and it required a review of lease stipulations that protect fish and wildlife. Executive Order 13212 required federal land management agencies to expedite their review of gas and oil drilling permits, and a new White House task force was established to oversee agency efforts to speed up the permitting process.

In this paper, we first define terms to establish economically recoverable energy resources as the policy-relevant measure in evaluating the energy potential of public lands. Next, we provide estimates of the amount of gas and oil in western wildlands, focusing on roadless areas in national forests and in national monuments administered by the U. S. Bureau of Land Management (BLM). We then explore (1) the costs to wildlife from energy development, specifically habitat fragmentation, (2) the failure to enforce lease stipulations, (3) the damage to water quality and aquatic species associated with development of coal bed methane gas, (4) the hidden costs to the regional economy and (5) the high risks of accelerated large-scale energy development in the absence of sufficient data and cumulative impact analyses. We conclude with recommendations for federal agencies to heed the risks to wildlife, regional economies and the public that are posed by the administration's plans for large-scale drilling of public land in the Rocky Mountains (Rockies).

Terminology

The debate over energy development on western public lands centers on drilling for methane (natural) gas, also the primary focus of this paper. Scientists at the U. S. Geological Survey (USGS) classify natural gas as conventional or unconventional, partially based on the technology used during extraction. Unconventional gas typically has higher production costs because it requires a significant degree of stimulation—hydraulic fracturing, for example—to attain sufficient levels for economically profitable production (Energy Information Administration 2001a).

The two main unconventional gases are coal bed methane and continuous-type gas, commonly called tight sands gas. Coal bed methane is a form of natural gas trapped within coal formations, while tight sands gas is trapped in low permeability sandstone. In the Rockies, 92 percent of the undiscovered, technically recoverable, gas on federal land is unconventional gas, primarily tight sands gas (U. S. Department of the Interior and U. S. Department of Energy 2003). There is a clear distinction between discovered gas reserves—known to be both technically and economically recoverable—and undiscovered gas resources that are not yet proven to be either technically or economically recoverable. This distinction is important, given the current pressure to develop the higher risk, undiscovered resources on public wildlands.

To estimate quantities of undiscovered resources, USGS makes a distinction between gas in place, technically recoverable gas, and economically recoverable gas (Figure 1). Gas that is estimated to exist in sufficient quantities for recovery with current technology but without regard to profit or extraction costs is called technically recoverable gas. Technically recoverable gas that is estimated to be profitable to extract is called economically recoverable gas. The costs that USGS uses to assess economically recoverable gas and oil include the direct costs of exploration, development and production at the wellhead, plus a profit margin (Root et al. 1997, Attanasi 1998). To account for the uncertainty inherent in price forecasts, USGS uses a range of prices, rather than a single-point estimate. USGS estimates do not include infrastructure costs, the costs of transporting the gas to market, nonmarket costs (such as loss of local economic benefits from lower quality hunting, fishing and camping experiences) or off-site mitigation costs (like increased water treatment costs). If USGS included these hidden costs, the estimate of economically recoverable gas in the Rockies would be lower (Figure 1).

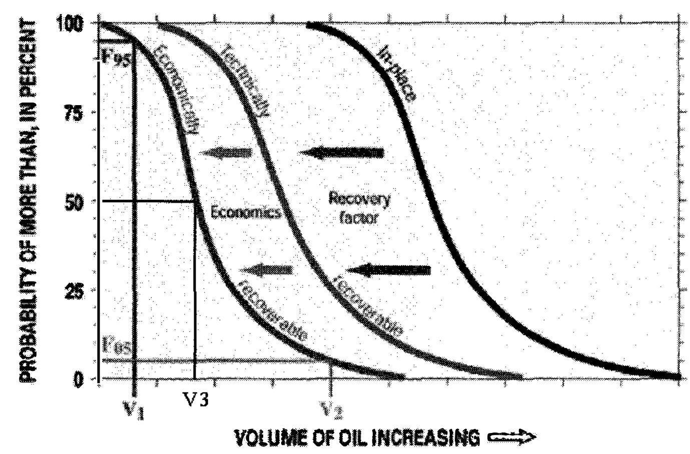


Figure 1. Gas volumes and probabilities for estimating undiscovered quantities. There is a 95-percent chance of at least V1 of economically recoverable gas, and a 5-percent chance of at least V2 of economically recoverable gas. Adapted from the U. S. Geological Survey 2001.

Estimating the Opportunity Costs of Protecting Wildlife Habitat

The opportunity cost of a policy or action that protects wildlife habitat equals the net benefits that are foregone as a consequence of that policy or action. With respect to energy policy, the opportunity cost to protect critical wildlife habitat or native fisheries is the amount of economically recoverable gas that is foregone as a result of such actions—*not* the amount of gas that is technically recoverable. As recommended by the Congressional Research Service (Corn et al. 2001), economically recoverable resources should be the basis of policy analysis. If economic constraints on gas production are ignored, resource assessments will overestimate the quantity of gas that is potentially off-limits because of its location in a migratory corridor or roadless area.

USGS estimates that less than 20 percent of technically recoverable gas in the Rockies is economically recoverable when prices (adjusted to 2001 dollars) are between \$2.30 and \$3.90 per thousand cubic feet (mcf) (Table 1). Before recent price spikes, \$2.00 per mcf was viewed as the long-term price for natural gas (Energy Information Administration 2001b). Current projections suggest that natural gas wellhead prices will decline from the high levels of 2003 (around \$5.00/mcf) to \$3.40 per mcf (2002 dollars) in 2010, then rise to \$4.40 per mcf in 2025 (Energy Information Administration 2004). The price projected in 2010 is slightly lower and the price projected in 2025 is slightly higher, than the USGS high price scenario. As with any long-term price forecast, uncertainty is large. If actual prices are higher or lower than the USGS high price scenario, the amount of gas economically recoverable is likely to increase or decrease, respectively, from our estimates cited here (Table 1).

Table 1. Economic recovery rates for technically recoverable gas in the United States based on prices of \$2.30 and \$3.90 (2001 dollars) per thousand cubic feet (mcf).

Region	USGS Economic recovery rates[a]
United States	38–46%
Rockies and Northern Plains	13–18%
Southwestern Wyoming	1–5%

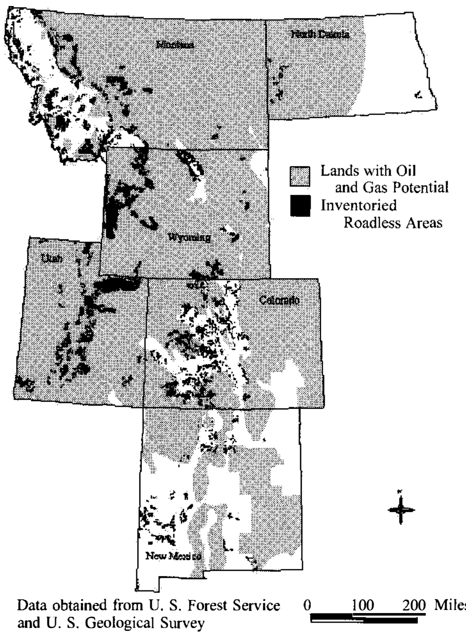
[a] Percent of technically recoverable gas in reserves and gas left undiscovered that is profitable to extract (before accounting for environment-related costs). This excludes recovery rates for offshore gas. Sources: Root et al. 1997, Attanasi 1998 and LaTourrette et al. 2002.

Drilling the Rocky Mountains: How Much Gas?

In January 2001, The Wilderness Society assessed the energy potential on western federal lands (Morton 2002a). We used government data (U. S. Geological Survey 1996a, Attanasi 1998) to complete a geographic information system (GIS) analysis of the overlap between the boundaries of 200 gas and oil plays and the boundaries of western wildlands (Figure 2). We focused our analysis on national forest roadless areas in six Rockies states (Montana, North

Figure 2. Potential gas and oil resources and roadless areas in six Rocky Mountain states. National forest roadless areas account for less than 4 percent of the land that has gas and oil potential.

National Forest Roadless Areas and Lands with Oil and Gas Potential in Six Rocky Mountain Sites



Dakota, Wyoming, Utah, Colorado and New Mexico) and in 15 national monuments managed by the BLM in Oregon, California, Idaho, Utah, Montana, Colorado, New Mexico and Arizona. We used USGS (1996a) mean estimates of technically recoverable gas and oil because we believe that USGS estimates represent the best, unbiased estimate available. We developed economic recovery rates based on cost functions for gas-oil provinces (Root et al. 1997, Attanasi 1998) and reported our results using both a high and low price scenario. For details, see Morton et al. (2002a).

Using the USGS low-end and high-end prices, we found that national forest roadless areas in the Rockies contain approximately 3.9 trillion cubic feet to 4.9 trillion cubic feet of economically recoverable gas (Table 2), or 48 percent to 59 percent of the technically recoverable gas in the roadless areas. (Figure 2, Table 2)

The roadless areas contain approximately 410 million to 478 million barrels of economically recoverable oil, or 69 percent to 81 percent of the technically recoverable oil. National forest roadless areas in Wyoming and Colorado contain the majority of economically recoverable gas and oil, much of

Table 2. Estimates of gas and oil in national forest roadless areas in six Rockies states[a]

Resource	Economically recoverable quantity	Economically recoverable as percent of technically recoverable in percentage	Economically recoverable in relation to total U. S. consumption
Conventional gas	3,223–3,665[b]	74–84	52–59 days
Tight sands gas	199–285[b]	8–11	3–5 days
Coal bed methane gas	500–943[b]	41–77	8–15 days
Total Gas	3,922–4,893[b]	48–59	63–79 days
Oil and natural gas liquids	410–478[c]	69–81	21–24 days

[a] Montana, North Dakota, Wyoming, Utah, Colorado and New Mexico
[b] in billion cubic feet
[c] in million barres

which is located in Bridger-Teton National Forest, south of Jackson Hole, Wyoming, and in San Juan National Forest near Durango, Colorado. Based on total demand for gas and oil in the United States and based on current energy consumption rates, economically recoverable gas in the roadless areas would meet total U. S. gas consumption for 2 to 2.5 months. Economically recoverable oil in the roadless areas would meet total U. S. oil consumption for 21 to 24 days. Obviously, the gas would be produced over a much longer period of time, but these estimates provide an indication of the relatively small amount of economically recoverable gas and oil in national forest roadless areas.

The 15 national monuments contain less than 15 days of oil and 6 days of gas. Even this small amount is overestimated, however, because spatial inaccuracies in the GIS layer make it impossible to separate energy resources under ocean waters adjacent to the California Coastal National Monument. If a more accurate boundary were used, the amount of gas and oil in the California Coastal National Monument as well as the total for all monuments would drop dramatically.

Additional analysis of government data indicate that across the country, development of undiscovered gas and oil resources on federal lands—national parks, national forests, lands managed by BLM and national wildlife refuges (including the Arctic National Wildlife Refuge)—would satisfy U. S. demand for gas and oil for less than 2 years (Attanasi 1998, Minerals Management Service 2001, Morton et al. 2002a). In contrast, the gas and oil supply already discovered in proven reserves, along with the expected growth of those reserves, is projected to meet U.S. demand for oil for 15 years and gas demand for 21 years (Attanasi 1998, Minerals Management Service 2000). If we make strategic investments in energy conservation, energy efficiency and alternative energy sources, the gas and oil in our proven reserves will last even longer.

Drilling the Rocky Mountains: At What Cost?

An economic analysis of recoverable gas must include a full accounting of the nonmarket costs in addition to those more readily observed and measured in market prices (Loomis 1993). Nonmarket costs include erosion, loss of wildlife and fish habitat, decline in quality of recreational experiences, proliferation of noxious weeds, and increased air and water pollution. Although difficult to value in traditional monetary terms through standard cost-benefit analyses, these costs are nonetheless very real. Here we will focus on the costs associated with the loss and fragmentation of habitat associated with energy development.

Habitat Fragmentation from Drilling: The View from Above

Amos (2003) used historical Landsat satellite imagery to show the temporal development of the ecological footprint from gas drilling in the Jonah Field in Wyoming. Figure 3 shows undisturbed sagebrush and grassland habitat prior to drilling in 1986. In 1998, the BLM approved full field development of 497 wells to be drilled over 10 to 15 years, with a maximum drilling density of 1 well per 80 acres (8 pads/mile^2). Figure 4 shows the same area in 1999 with 100 gas wells drilled and embedded in a web of access roads, waste pits and pipelines that were clearly visible from space. In 2000, two years after the management plan was completed, BLM approved spacing of 1 well per 40 acres (16 pads/mile^2). By 2002, nearly 400 wells had been drilled (Figure 5), approaching the maximum number projected in the 1998 management plan. By 2003 more than 500 wells had been drilled, and industry requested a plan revision allowing 1,250 additional wells from 850 new well pads, with well spacing of just 16 acres (6.5 ha) and a drilling density of 40 well pads per square mile (Amos 2003) (Figures 3, 4 and 5).

Habitat Fragmentation from Drilling: Quantifying the Landscape Impacts

The satellite images illustrate the loss and fragmentation of habitat associated with drilling. Fragmentation of habitat is widely acknowledged as detrimental to many plant and wildlife species, including birds, but there are few studies that examine the exact size and extent of the ecological footprint of energy development. Spatial analysis can help fill this information gap.

In 2002, scientists at The Wilderness Society completed a habitat fragmentation analysis of the Big Piney-LaBarge gas field in the Upper Green

Figure 3. The area of the Jonah Gas Field in Wyoming, showing the undisturbed sagebrush and grassland habitat prior to drilling in 1986 (Amos 2003).

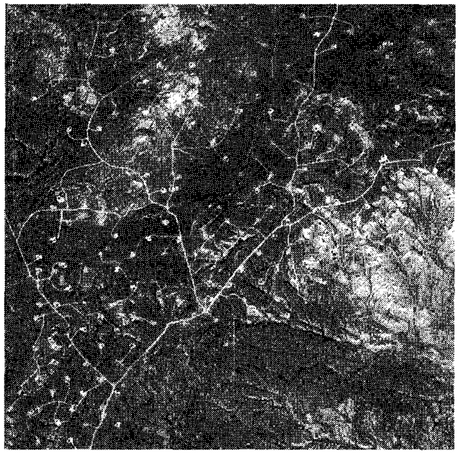


Figure 4. The Jonah Gas Field in 1999 after one year of full-field development at 80-acre (32.4-ha) spacing (8 well pads per square mile [2.6 km]). The web of wells pads, access roads, compressor stations and waste pits is clearly visible. (Amos 2003).

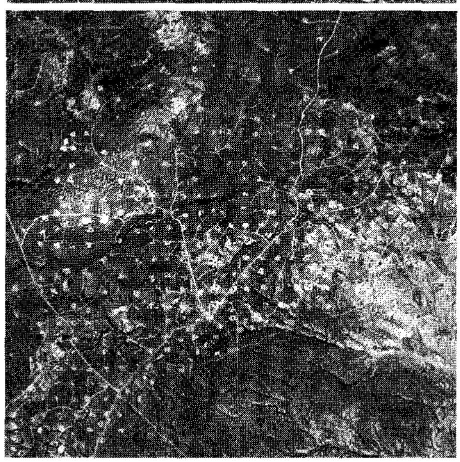


Figure 5. The Jonah Gas Field in 2002, after nearly 400 wells have been drilled at 40-acre (16.2-ha) spacing (16 well pads per square mile [2.6 km]), approaching the maximum number allowed by the 1998 management plan. Industry is now asking for 1,250 additional wells from 850 new well pads, resulting in 16-acre (6.5-ha) well spacing (40 well pads per square mile [2.6 km]) (Amos 2003).

River Valley of Wyoming (Weller et al. 2002). The valley is home to at least 25 species listed as threatened or endangered, including the bald eagle (*Haliaeetus leucocephalus*), mountain plover (*Charadrius montanus*), northern goshawk (*Accipiter gentilis*) and peregrine falcon (*Falco peregrinus*), and it is wintering ground for elk (*Cervus elaphus*), pronghorn antelope (*Antilocapra americano*) and mule deer (*Odocoileus hemionus*) (U. S. Bureau of Land Management 2001).

As of 1990, the Big Piney-LaBarge gas field had a total of 1,864 drilled wells, of which 1,080 were still active (U. S. Bureau of Land Management 1990). The field has produced oil and gas in the past, but current production is primarily tight sands gas. To complete our study, we generated infrastructure data through on-screen digitizing of 12 digital orthophoto quads, then quantified the degree of habitat fragmentation using three landscape metrics: linear feature density (primarily roads and pipelines), habitat in the infrastructure effect zone, and the amount of habitat in core areas (interior habitat that is remote from infrastructure) (see Weller et al. 2002).

Linear Feature Density. Linear feature density was calculated both as an average for the entire study area and as a series of 1 square-mile and 4 square-mile sampling windows across the landscape. Measuring density in sampling windows of different sizes provides an understanding of the variability of density across scales, which is important to gauge the effects on different species (Urban et al. 1987, Wiens and Milne 1989, Turner et al. 1994). For example, differences in dispersal distances among species cause them to respond to habitat features at different scales.

The overall area of oil and gas infrastructure (roads, pipelines, pads, waste pits, etc.) at Big Piney-LaBarge gas field covers 7 square miles (18.13 km^2) of habitat, or 4 percent of the study area (Figure 6). Our results indicate a direct physical footprint of 1,400 miles (2,253 km)of linear features and 3.8 square miles (9.2 km^2) of polygon features, resulting in an overall density of 8.43 miles (5.2 km) of roads and pipelines per square mile. This density is at least three times greater than road densities on national forests in Wyoming, South Dakota and Colorado and is extremely high, based on ratings in the Interior Columbia Basin Ecosystem Management Plan (U. S. Forest Service 1996) (Figure 6).

Linear feature density estimates are scale dependent and vary across the study area, ranging from 17.1 miles (10.6 km^2) per square mile to 0.9 miles (0.5 km) per square mile (Figure 7). At all scales analyzed, most grid cells have a

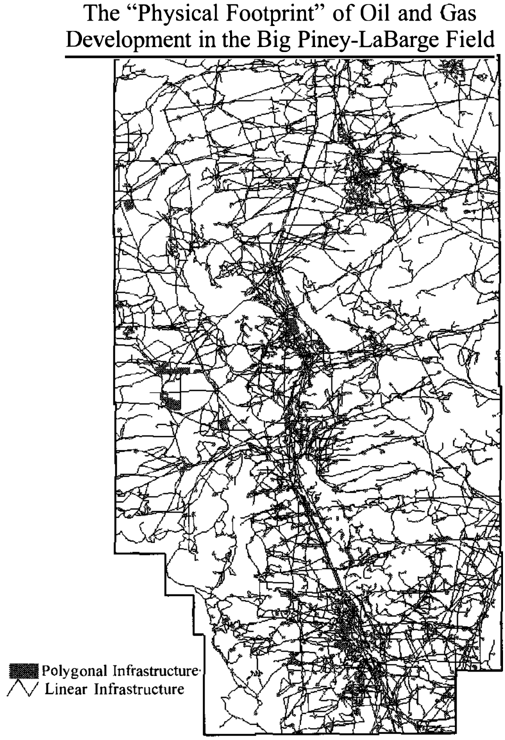


Figure 6. The digitized physical footprint from oil and gas development in the Big Piney-LaBarge field, Wyoming. The footprint includes both linear infrastructure features, such as roads and pipelines, and polygonal features, such as well pads, pumping stations and waste pits.

density between three and six per square mile. Twenty-nine percent of the landscape in the 1 square-mile scenario, and 24 percent of the landscape in the 4 square-mile scenario, have linear densities of more than 6 miles (3.7 km) per square mile. (Figure 7).

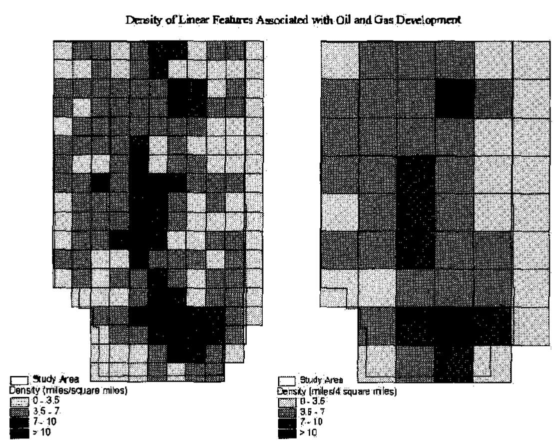


Figure 7. Density of linear features, Big Piney-LaBarge gas field, Wyoming. The density of linear infrastructure features was calculated using both a 1-square mile grid and a 4-square mile grid. Linear feature density ranges from 17.1 to 0.9 miles per square mile (10.6 to 0.5 km/km^2). The darker the shading, the higher the linear feature density.

Infrastructure Effect Zone. The ecological effects of infrastructure features extend across the landscape beyond physical structures of the oil or gas field. Forman (1999) calls the influence on edge environments parallel to roads the "road effect zone." We extended this zone of influence to all forms of infrastructure and completed the road effect zone analyses using widths of one mile, one-half mile, one-quarter mile, 500 feet, 250 feet and 100 feet. Results of the infrastructure effect zone analyses show that the entire 166 square-mile (430 km^2) study area is within one-half mile of a road, well head, pipeline, compressor station, waste pit or other component of the infrastructure involved in oil and gas drilling. One-hundred sixty square miles (414.4 km^2)—97 percent of the landscape—fall within one-quarter mile of the infrastructure.

Core Area Analyses. Another commonly used measure for landscape fragmentation is core area, sometimes referred to as interior habitat. Core areas exist in natural landscapes as contiguous blocks of uniform habitat types away from natural breaks or habitat edges. We examined habitat patches on the landscape outside of the infrastructure effect zones. Our results show that for the entire study area, no core areas exist farther than 1 mile (1.6 km) from the infrastructure. Only 27 percent of the study area is more than 500 feet (152.4 m) from infrastructure, and only 3 percent is more than one-quarter mile away (Figure 8).

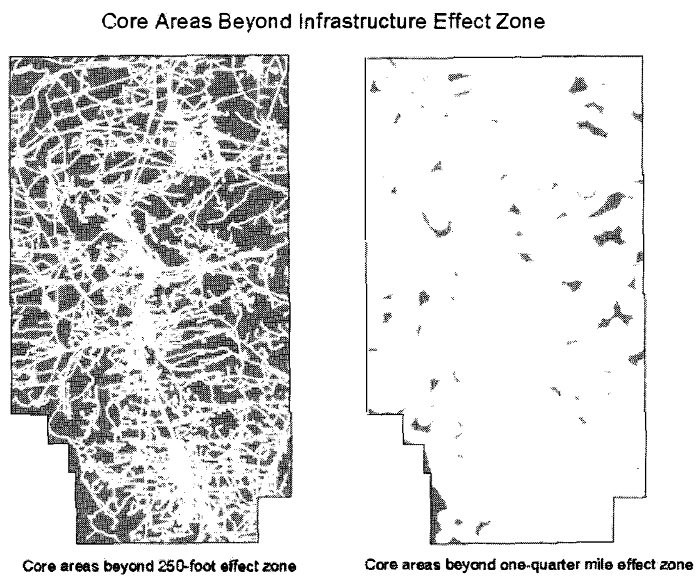


Figure 8. Core areas beyond infrastructure effect zone, Big Piney-LaBarge gas field, Wyoming. Two examples of core area maps based on 250-foot (76.2 m) and one-quarter mile (0.4) infrastructure effect zones. Three percent of the area is more than one-quarter mile away from oil infrastructure. Shading represents the areas beyond relatively narrow and wide infrastructure effect zones.

Habitat Fragmentation from Drilling: The Hidden Costs to Wildlife

Without access to population or habitat data, we examined the habitat costs to wildlife from oil and gas development by connecting the results of our spatial analysis to spatial metrics found in the scientific literature. Numerous studies document that elk avoid roads. Lyon (1983) found that when road densities are 2 miles per square mile (1.2 km/km^2), elk can be displaced from up to 50 percent of their habitat. When road densities exceed 6 miles per square mile (3.7 km/km^2), elk can be displaced from 75 percent of the habitat. Roughly a quarter of our study area falls within the latter category.

Road avoidance by wildlife is evident in open landscapes with little surrounding vegetation (Perry and Overly 1976, Morgantini and Hudson 1979, Rost and Bailey 1979). In areas with little cover, habitat is compromised at a road density of only 0.8 miles per square mile (0.5 km/km^2) of road (Lyon 1979). A study on elk habitat effectiveness in northcentral Wyoming found that fewer elk used areas with road densities higher than 0.5 mile per square mile (0.3 km/km^2) (Sawyer et al. 1997). Our results indicate that most of our study area has linear feature densities much higher than 0.5 mile per square mile (0.3 km/km^2). Another study in western Wyoming indicates that elk avoid a relatively high-density oil and gas field in open habitat (Bock and Lindzey 1999). The lack of physical barriers to screen drilling activities has displaced elk up to 3 miles.

Wyoming has the greatest concentration of pronghorn antelope in any state or provincial authority in North America, and the Green River Valley holds the highest concentration of this animal in Wyoming (U. S. Bureau of Land Management 2000). With respect to potential impacts, antelope in the nearby Whitney Canyon-Carter Lease fields felt the impacts of oil and gas projects with, "nearly one mile of road per every square mile of occupied habitat" (U. S. Bureau of Land Management 1999). Our study area has average linear feature densities more than eight times greater than 1 mile per square mile (0.6 km/km^2).

The bulk of the study area is designated as winter habitat for mule deer (U. S. Bureau of Land Management 1990), an animal that also avoids oil and gas development in their habitat. A study conducted in North Dakota found that mule deer avoided areas within 300 feet (91.4 m) of well sites for feeding and bedding, resulting in a 28 percent reduction in secure bedding areas, and this behavior continued for more than 7 years (Jensen 1991).

Wildlife species other than big game animals also face the impacts of infrastructure. Wyoming is home to the largest and most robust North American population of the greater sage-grouse (*Centrocercus urophasianus*), a bird that is facing severe declines in populations because of habitat loss (Christiansen 2000). Approximately two-thirds of the 150 leks (breeding grounds) for this species in Wyoming are located in Upper Green River Valley (U. S. Bureau of Land Management 1999). A study in Wyoming found that the negative effects of oil and gas development on nest-initiation rates of greater sage-grouse can extend for 2 miles (3.2 km) beyond the infrastructure (Lyon 2000). Given our results, there is no place in Big Piney-LaBarge gas field where the greater sage-grouse would not be affected by natural gas development.

Lease Stipulations Help Protect Wildlife: But Only if They Are Enforced

As part of the oil and gas leasing process, BLM or U. S. Forest Service officials may subject the leases to environmental stipulations that are meant to protect birds and wildlife by stating where, how and when drilling activities may occur. Lease stipulations, designed by agency professionals, may include seasonal closures of critical habitat to benefit wildlife, such as elk, antelope and sage grouse; may not allow surface occupancy provisions to protect sensitive habitats, campgrounds and recreation areas; and may control use to protect endangered species, archaeological and other important cultural sites. Lease stipulations may offset the habitat fragmentation impacts from drilling but only if the stipulations are enforced.

A review of BLM stipulation exception data for the agency's Pinedale District in Wyoming indicates that seasonal wildlife stipulations are waived quite frequently. Between 2001 and 2004, 86 percent of the raptor stipulations, 90 percent of the sage grouse stipulations and 88 percent of the wildlife stipulations protecting winter range were waived (U. S. Bureau of Land Management 2004a). An analysis in southwestern Wyoming indicates that the gas industry's requests to waive wildlife and fish stipulations were granted 97 percent of the time (Trout Unlimited 2004). Many waivers were issued during crucial winter months with no understanding or assessment of the impacts, owing to inadequate data and monitoring by the BLM.

Of equal concern are lease stipulations that only apply during the drilling phase, not during the production phase, providing, at most, a short-term positive effect. According to Noon (2002a: 6–7): "... the nature of the proposed mitigation efforts have only short-term positive effects because they represent 'timing limitations' only .... For example, elk calving areas potentially disturbed by coal bed methane (CBM) projects have a seasonal closure from May 1 to June 30 in a given year. After that time period, development activities at a site near calving habitat may be reinitiated. The end result is that next year there is a high likelihood this site will no longer be suitable. This is not meaningful mitigation. This policy simple delays an inevitable loss of habitat . . . . *In my opinion, it is misleading to refer to these policies as mitigation actions* [emphasis added]."

In addition, a July 28, 2003 BLM policy, in Instruction Memorandum #2003–233, discourages the use of stipulations to protect wildlife resources and will likely mean that even fewer leases will contain special stipulations to protect wildlife and wildlife habitats.

Drilling for Coal Bed Methane: The Hidden Costs to Fish

While we focus here on wildlife impacts, the potential negative impacts to water quality and aquatic species from drilling for coal bed methane may be significant. The Powder River Basin in the semiarid Great Plains region of Wyoming and Montana is a case in point. The river itself, without a major dam, is in relatively healthy shape. With its four tributaries, it supports a rare invertebrate fauna and 25 native fish species that are much less common now than in years past (Allan 2002). According to Hubert (1993:394), "The fish community of the Powder River is unique . . . and . . . probably represents the kind of community that was found in free-flowing Great Plains rivers." Since similar rivers in the region have been altered, there is a special responsibility to ensure that drilling for coal bed methane (CBM) does not eliminate a critical remnant on a once vast and unspoiled ecosystem (Allan 2002).

The Powder River Basin is currently targeted for tens of thousands of CBM wells connected by a network of roads. Roads are the major source of sediment into streams. Clements (2002:5) has the following concerns about the scale of CBM development proposed in the Draft Environmental Impact Statement (DEIS) for the Powder River Basin in Wyoming: "the DEIS has not given sufficient attention to the impacts of increased sedimentation on aquatic ecosystems in the project area. Increased sedimentation resulting from erosion of stream banks, overland flow, and road construction will likely impact aquatic organisms . . . . Input of sediments to aquatic ecosystems is widely regarded as a major source of stream degradation in North America . . . . In particular, fine sediments fill interstitial spaces and reduce available habitat for fish and macroinvertebrates."

Water is the central issue surrounding development of coal bed methane. To release methane gas from coal beds, enormous amounts of saline-sodic water from shallow and deep aquifers must be pumped to the surface. While water quality and quantity vary by region, CBM wells in Wyoming have discharged between 20,000 to 40,000 gallons (75,708–151,416 l) per day per well. The dewatering phase typically lasts 2 to 5 years. Schlesinger (2002:3), after reviewing the Montana Statewide DEIS, concludes that, "Clearly water from CBM wells is likely to reach major regional rivers," raising concerns about pumping too much produced water into streams. The altered water flows from the surface release of the produced water will negatively impact thermal and flow regimes, and it likely will contribute to bank erosion and changes in riparian vegetation (Allan 2002). Gore (2002) warned that the loss of habitat caused by increased water flows from discharged water at CBM projects could eliminate up to 30 aquatic species within 20 years.

The water discharged by CBM production in the Powder River Basin is characterized with very high levels of salinity and total dissolved solids (TDS). Many of the constituents that comprise total dissolved solids are toxic to aquatic organisms and have the potential to negatively impact aquatic resources. In particular, discharge of high salinity and TDS effluents into receiving systems may result in physiologically stressful conditions for some species due to alterations in osmotic conditions (Clements 2002). It is well established that elevated concentrations of major ions can reduce water quality and significantly impact fish and wildlife populations (Goetsch and Palmer 1997, Dickerson and Vinyard 1999, Pillard et al. 1999, Chapman et al. 2000). Unfortunately, the DEIS made no attempt to evaluate potential toxicological impacts of CBM-produced effluents on fish and macroinvertebrates (Clements 2002). Clements concludes his critique with the following opinion: "In summary, the Montana DEIS does not provide sufficient information to evaluate the potential risk... Based on my analysis of information presented in the DEIS and my best professional judgment, I expect that CBM produced effluents and associated sediments released into watersheds in the project area will have deleterious impacts on benthic macroinvertebrates and fish" (6).

Drilling the Rocky Mountains: The Hidden Costs to the Regional Economy

The results of our fragmentation analysis, when combined with scientific concerns over water quantity and quality, indicate that large-scale drilling as currently proposed on public lands will generate substantial costs to fish and wildlife. This will, in turn, result in lost economic benefits for North Americans who enjoy viewing, hunting and fishing wildlife in a pristine environment. More than 82 million North Americans participate in some form of wildlife-related recreation (Fish and Wildlife Service 2003). In the Rockies states of Colorado, Wyoming, Montana, Utah and New Mexico alone, an estimated 3.5 million residents, or 49 percent of the region's entire population, hunt, fish or watch wildlife. Thus, loss and fragmentation of wildlife habitat resulting from proposed, large-scale drilling could negatively effect nearly half of the region's residents.

Development of oil and gas resources can also have negative impacts on communities where revenues from hunters, anglers and wildlife watchers are a significant part of the economy. In the Rockies during 2001, participants in wildlife-viewing activities spent nearly $2.3 billion for license fees, equipment, and other related purchases, while hunters and anglers spent $3.6 billion (U. S. Fish and Wildlife Service 2002). If fragmentation of habitat from proposed oil and gas projects results in, say, declining elk populations and, if the declining elk populations result in a lower-quality hunting experience, fewer hunters and a drop in related spending are the negative economic effects for rural businesses and communities.

The hidden costs associated with development of oil and gas resources can negatively impact other sectors of the economy. Air pollution arising from gas compressors contributes to regional ozone problems and, when combined with dust from roads, creates regional haze and a corresponding decline in visibility (Yuhnke 2002). Loss of, or decline in, the quality of scenic landscapes and viewsheds could hurt the region's billion dollar tourism industry as well as the potential economic growth stemming from engineering firms, business consultants and retirees that chose to locate where the air is cleaner. A growing body of literature suggests that future diversification of rural western economies depends to a large extent on amenity services, such as watershed protection, wildlife habitat and scenic vistas that public land provides (Rasker 1994, Power 1996, Haynes and Horne 1997, Morton 1999). Public land improves the quality of life for retirees and a trained and educated workforce capable of attracting new businesses and capital to communities. Expediting large-scale oil and gas drilling on public land threatens the comparative economic advantage that amenities on public land provides for nearby communities (Morton et al. 2002b).

Drilling the Rocky Mountains: The Environmental Risks are High

In order to protect the West's greatest asset—the environment—we must improve the science behind adaptively managing our public lands, especially our oil and gas resources. Improving the science is vital since expert assessments (Allan 2002, Braun 2002, Clements 2002, Gore 2002, Noon 2002a, Noon 2002b, Schlesinger 2002, Western EcoSystems Technology 2003) reveal that the impacts from proposed oil and gas drilling in the Rockies will be widespread and negative, posing high risks for the environment, wildlife, local economies and our quality of life. We expect the risks to be large, due to the speed and the scale of the proposed drilling, the poor state of scientific knowledge about the environmental impacts from drilling, and the fact that the BLM has inadequate staffing levels, poor baseline data and insufficient budgets to inventory, analyze and monitor resource conditions.

The Environmental Risks Increase with Scale

As the scale and speed of drilling increases, so does the environmental risk, particularly when baseline data are limited or nonexistent. The Administration is currently expediting drilling plans for tens of millions of acres in the West, despite the fact that drilling for oil, especially natural gas, is already at a pretty large scale. In the Rockies, public land managed by the BLM has more than 53,000 producing oil and gas wells (U. S. Bureau of Land Management 2003a). Nationally the BLM has about 33 million acres (13.3 million ha) of federal minerals (public and split estate) under lease to industry (U. S. Bureau of Land Management 2002). The BLM oversees 54,000 oil and gas leases, with only 40 percent of the leases currently producing gas or oil (U. S. Bureau of Land Management 2002, 2004b). In Wyoming, there are over 21,000 federal oil and gas leases, covering approximately 15 million acres (6 million ha) of federal land (Bennett 2003). In 2002, only 3.6 million acres (1.5 million ha) of federal land in Wyoming were in production (U. S. Bureau of Land Management 2003), illustrating the large scale drilling potential (i. e. drilling opportunities) currently available to industry. If leaseholders place the current inventory of nonproducing leases into oil or gas production, the scale of drilling on public lands will increase dramatically—even without any additional leasing. Between 2000 and 2003, more than 46,000 drilling permits were issued for public and private lands in the five Rockies states (Rig Data 2004). It is difficult to understand why, with the large-scale drilling currently occurring, there is a need to speed up the process of approving drilling permits at the expense of a careful examination of the impacts on wildlife and local economies. Note, too, that it appears likely that a substantial backlog exists of surplus permits that have already been approved; although, industry has chosen not to begin drilling.

Environmental Risks Increase when Data Are Limited

The National Environmental Policy Act requires federal agencies to disclose in their environmental impact statements the risks of proposed action and to respond to the adverse opinions held by respected scientists. The Data Quality Act of 2000 (Pub. L. No. 106-554) requires agencies to incorporate high quality, usable, verifiable and objective information. Fulfilling these obligations is especially critical to reduce the risks from large-scale, accelerated drilling plans.

BLM has adopted an adaptive management approach to assess the impacts from oil and gas drilling. A major implication of adaptive management is that acquisition of useful data becomes one of the primary goals of management. Acquiring data is sorely needed as very little wildlife or fish data were used to support the preferred alternatives and conclusions of the fast tracked energy plans. As noted by Schlesinger (2002:1) when reviewing the Montana Statewide DEIS: "In general, I am struck by the lack of data obtained from the existing coal bed methane (CBM) gas wells in Wyoming and Montana. These existing wells, with their associated reservoirs and outflows, represent a large, replicated experiment that should have provided ample opportunity to answer some of the questions that I will pose below." He concludes: "My expert opinion is that the water quality data presented are completely inadequate to assess the impact of waters from additional coal bed methane wells on the regional environment" (9). Allan (2002:9), with respect to the Powder River, echoes these concerns: "The DEIS lacks critical information about the basic ecology of the Powder River Ecosystem, and it lacks critical information about the amount and quality of water that will be discharged onto the land and into surface drainages. Without this information it is an inadequate document on which to assess impacts on aquatic ecosystems."

And, Noon (2002a:2), also with respect to the Powder River Basin, states: "In the DEIS there is a pattern of first asserting a lack of data as a rationale for no quantitative analysis and then concluding no adverse effects. Within the last 10 years a large number of publications have documented adverse effects to wildlife and their habitats as a consequence of habitat fragmentation, human disturbance, roads, and changes in land cover. In the absence of data and high uncertainty, logic would suggest a slow and incremental approach to CBM development coupled with close monitoring to detect possible adverse impacts. The public expects responsible resource managers to implement monitoring and adaptive management in an incremental fashion when irrevocable or irreversible outcomes are possible."

And, Braun (2002:1), with respect to sage grouse, states: "A major deficit is the lack of knowledge about sage-grouse in the areas to be impacted. This includes adequate baseline data on current population levels and trends as well as amount and quality of present habitat.... The present baseline data are totally inadequate to allow an adequate evaluation of the potential impacts on sage-grouse in the area."

And, Noon (2002b:3-40), with respect to wildlife data in the Farmington DEIS, states: "To infer an effect, or lack of an effect, resulting from oil and gas development requires pre-project baseline information. I could find no evidence in the DEIS that baseline data exist for individual species populations or their habitats. In fact, the DEIS openly admits the lack of data. For example, here are some statements from the DEIS: "There have been few surveys for non-game species in the planning area." He goes on to say, "Few non-game mammal studies have been conducted" (p. 3--41). And, there is, "incomplete data on mule deer and elk populations in the planning area" (p. 4–30). Also, there is, "lack of site-specific data on the effects of roads on mule deer and elk" (p. 4--30), In the absence of baseline information, the environmentally responsible course of action would be to collect such information prior to development.

The problem of poor data is not new (Loomis 1993). In 1986, a former BLM planning official stated one of the key ailments in BLM planning: "Lack of solid economic, analytical procedures and hard data continually handicaps planning by failing to portray objectively trade-off values to be gained or lost through managerial decisions" (Crawford 1986:409). Nearly 20 years later the problems, questions and challenges are much more complex, but the data are arguably in worse shape. We can and must do better. A recent survey of BLM staff (U. S. Bureau of Land Management 2003c) affirms our concerns over a data crisis. The issue of inadequate data for fish, wildlife, botany and special status species is particularly critical for the fast tracked energy plans. The authors conclude: "The accelerated time frame for completing time sensitive plans may not provide sufficient time to address FWBSS species conservation issues" (U. S. Bureau of Land Management 2003c).

Recommendations

The Economic Analysis Must Be Improved

As this paper shows, public wildlands in the Rockies contain undiscovered gas and oil resources, the majority of which are not economical to recover. Where economically recoverable gas and oil does exist on those lands, the amount produced would supply the United States at current rates for only a short time. Unconventional gas resources, like tight sands gas and CBM in the Rockies, is subject to higher production costs and substantial uncertainty (LaTourrette et al. 2002). Tight sands gas, for example, are expensive to develop because the gas is often deeply buried, wells have low flow rates, reservoir pathways may be obstructed, concentrations often are more diffuse and costly recovery techniques (such as fracturing) are needed (Cleveland 2003). Failure to recognize these essential elements of low-permeability sandstone reservoirs has led to a misunderstanding of the risks associated with basin-centered gas plays and a significant overestimation of available resource levels (Shanley et al. 2003). Sixty percent of exploratory wells drilled in the United States are either dry or have too little gas to make development economical (Morton 2003), underscoring the high risk and poor economics associated with drilling for undiscovered resources. USGS (Attanasi 1998) estimates that only 18 percent of the technically recoverable tight sands gas is economic to recover.

Focus on Economically Recoverable Gas. Instruction Memorandum #2003–233 of July 28, 2003, requires BLM planners to use estimates of technically recoverable gas in management plans, ensuring that the plans will exaggerate the energy potential, jobs and revenues from proposed drilling projects, as well as the opportunity costs of protecting wildlife habitat or enforcing wildlife stipulations in leases. For reasons documented in this paper, planning documents should not rely on estimates of technically recoverable resources or other measures that ignore economics.

USGS is currently updating its estimates of economically recoverable gas in the Rockies using a range of prices that addresses concerns over price uncertainty and the accuracy of economic forecasting. Until the updates are ready, we recommend that BLM rescind Instruction Memorandum #2003-233 and that the agency and U. S. Forest Service use the USGS high and low price mean estimates of economically viable gas (Attanasi 1998) to evaluate various land management alternatives in upcoming plan revisions.

Include a Full Accounting of Environmental Costs. In addition to market costs, economic analyses of recoverable gas must include a full accounting of nonmarket costs. Because they exclude nonmarket costs, USGS estimates are just the starting point to determine whether undiscovered gas is economically viable to extract. After 35 years of research by academic and federal agency economists (Krutilla 1967, Krutilla and Fisher 1985, Peterson and Sorg 1987, Loomis and Richardson 2000), it is now possible to quantify nonmarket environmental costs that arise from development of natural resources (see Table 3). The BLM and the U. S. Forest Service should include a full account of nonmarket costs in the effects analysis required by the National Environmental Policy Act for leasing and drilling decisions (Table 3).

Account for the Negative Impacts on Local Economies. The BLM and the U. S. Forest Service should assess the potential impacts on the regional economy that may flow from environmental degradation brought about by proposed large-scale oil and gas drilling in the Rockies. Considerations should include the negative impacts on hunting, fishing, ranching, recreation and service jobs, plus the negative impacts on our retirement and investment income. And, it should include our overall quality-of-life-based economy. This recommendation is consistent with that of more than 100 economists, who, in a 2003 letter to President Bush, stated, "The West's natural environment is, arguably, its greatest, long-run economic strength" (Niemi et al. 2003). The economists agreed that protecting the West's natural environment would strengthen the ability of western communities to generate more jobs and more income.

Invest in Baseline Data Collection, Spatial Analysis and U. S. Bureau of Land Management Field Staff

Improve Baseline Data Collection

Data collection and monitoring are prerequisites to cost-effective, science-based adaptive management of public land, but data collection and

Table 3. The economic costs of gas and ail extraction

Cost category	Description of potential cost	Methods for estimating costs
Direct use	Decline in quality of recreation, including hunting, fishing, hiking, biking, horseback riding. Loss of productive land for grazing and farming.	Travel cost and contingent valuation surveys
Community	Air, water and noise pollution negatively impacts quality of life for area residents with potential decline in the number of retirees and households with nonlabor income, loss of educated workforce, and negative impacts on non-recreation businesses. Decline in recreation visits and return visits negatively impact recreation businesses. Socioeconomic costs of boom-bust cycles.	Surveys of residents and businesses; averting expenditure methods for estimating costs of mitigating health and noise impacts; changes in recreation visitation, expenditures and business income; documented migration patterns.
Science	Oil and gas extraction in roadless areas reduces value of area for study of natural ecosystems and as an experimental control for adaptive ecosystem management.	Change in management costs, loss of information from natural studies foregone.
Off site	Air, water and noise pollution decrease quality of life for local residents and decrease quality of recreation experiences for downstream and downwind visitors. Haze and drilling rigs in viewsheds reduce quality of scenic landscapes, driving for pleasure, and other recreation activities and negatively impact adjacent property values. Groundwater discharge can negatively impact adjacent habitat, property and crop yields, while depleting aquifers and wells.	Contingent valuation surveys, hedonic pricing analysis of property values, preventive expenditures, well replacement costs, restoration and environmental mitigation costs, direct impact analysis of the change in crop yields and revenues.
Biodiversity	Air, water and noise pollution can negatively impact fish and wildlife species. Groundwater discharge changes hydrological regimes with negative impacts on riparian areas and species. Road and drill site construction displaces and fragments wildlife habitat.	Replacement costs, restoration and environmental mitigation costs.

Table 3 (continued). The economic costs of gas and oil extraction

Cost category	Description of potential cost	Methods for estimating costs
Ecosystem services	Discharging ground water negatively impacts aquifer recharge and wetland water filtration services. Road and drill site construction increases erosion, causing a decline in watershed protection services.	Change in productivity, replacement costs, increased water treatment costs, preventive expenditures.
Passive use	Roads, drilling rigs and pipelines in roadless areas result in fewer passive use benefits for natural environments.	Contingent valuation surveys; opportunity costs of not utilizing future information about the health, safety and environmental impacts of oil and gas drilling.

monitoring generally take a back seat in BLM and U. S. Forest Service budgets and planning processes. A quick review of the U. S. Forest Service proposed budget for fiscal year 2005 shows that inventory and monitoring, much of which is devoted to the monitoring of visitor use and not resource conditions, represents just 3.7 percent of the total agency budget (U. S. Forest Service 2004). The BLM's fiscal year 2005 budget request is more difficult to decipher, but it appears that funding for monitoring accounts for just 1.6 percent of the total request (U. S. Bureau of Land Management 2004c). In a recent review of the BLM budget, the U. S. Office of Management and Budget identified several weaknesses, including insufficient data and gaps in monitoring resource conditions to support management decisions. This is consistent with opinions of the scientists cited in this paper; the lack of credible data is a fatal flaw with recent BLM decision documents examining the environmental impacts from drilling in the Rockies.

The BLM (2003c:45) acknowledges the pressing need for data collection and monitoring: "The lack of a coordinated, national program for inventory of (wildlife and fish) resources on BLM-managed land is problematic, because it is difficult to manage resources without full knowledge of their status on public land. When inventory is performed, coverage of resources may be inconsistent, and in some instances, current office staff may be unaware of inventory efforts by previous employees."

While the agency is starting to recognize the data crisis, we believe recently developed energy plans fail to comply with the Data Quality Act of 2000, which requires the agency to use data of sufficient quality to make a reasonable analysis. In order to decrease environmental costs and risks, the BLM (and the

U. S. Forest Service) should accelerate efforts to collect baseline data, analyze the data and monitor resource conditions. This information is required to adaptively manage ecosystems and is vital if the public is to fully understand the potentially irreversible, cumulative environmental impacts from large-scale energy development in the Rockies. To their credit, both the U. S. Forest Service and the BLM have increased the budgets for monitoring in the fiscal year 2005 budget. This is a step in the right direction, but a much greater long-term budget commitment to data collection, analysis and monitoring is required to bring the agencies in compliance with the Data Quality Act of 2000. Scientists at Western EcoSystems Technology (2003:35) summarize the risks, uncertainties and data challenges faced by BLM: "there is a paucity of well designed studies that assess the impacts of oil and gas activity on ungulate populations. The Upper Green River Basin contains a variety of ungulate habitats and contains winter ranges for some of the longest migrating ungulate herds in the west. Thus the most effective means for assessing impacts from oil and gas projects on ungulate populations within the area is the implementation of well designed studies of the effects of oil and gas development on ungulate ecology and habitat. Long term monitoring should also be used to verify the efficacy of approved mitigation measures within important big game habitats. The revision of the Pinedale RMP should include requirements for monitoring of ungulate use and movements through radio telemetry to verify the accuracy of existing range designations. Ideally, these studies should be of sufficient duration (e. g., 5–10 years) in order to capture a fairly wide range of winter severity. The studies should be conducted so that inferences can be made to all herd segments within the Pinedale Resource Area potentially impacted by resource development. Additionally, habitat mapping is needed to help identify key areas for ungulates."

Incorporate Spatial Analysis into Evaluation of Proposed Drilling and Monitoring of Actual Drilling

Despite the documented impacts that habitat fragmentation has on wildlife proposed oil and gas projects are moving forward without adequate evaluation of these impacts. The DEIS for the coal bed methane projects in the Powder River Basin led to the following comments on the document's shortcomings by Noon (2002a:2): "The relevance of the fragmentation process affecting wildlife populations rests on the understanding that information on habitat amount alone may be insufficient to predict the status of a species. When habitat is potentially limiting, then information on the spatial pattern of the habitat may be equally or more relevant than information on habitat amount. *The importance of incorporating spatial data into effects analysis cannot be overemphasized.* Knowledge of where on the landscape habitat loss will occur and in what spatial pattern is essential before one can conclude no significant adverse effects" (emphasis added).

The BLM is currently developing best management practices (BMPs) for reducing fragmentation of wildlife habitat. While certainly a step in the right direction, developing BMPs is not a sufficient condition for addressing habitat fragmentation. In addition to developing and enforcing BMPs for existing energy development, spatial analysis should be incorporated into the evaluation of the ecological impacts of proposed oil and gas projects—in addition to monitoring existing energy projects. The significant increase in availability of GIS data and software technology in recent years makes this possible. Prior to exploration and development of new oil and gas fields and before new drilling in existing fields, BLM and the U. S. Forest Service should, at a minimum, complete the same kind of spatial analysis that we used in our study. And, information on habitat quantity and quality, species populations and birth rates, and the frequency of vehicle use of roads should be considered in the analysis. See Weller et al. (2002) and Hartley et al. (2003) for more recommendations.

Invest in Additional U. S. Bureau of Land Management Field Staff

In order to collect and analyze baseline data, complete spatial analysis and monitor cumulative impacts from the proposed drilling, Congress must allocate funds to add field staff to the BLM ranks. Over the past 10 years the number of BLM wildlife biologists decreased nearly 20 percent, while fishery biologists and botany positions increased slightly. Based on an analysis of BLM data (U. S. Bureau of Land Management 2003a, 2003c) the agency has only 12 ecologists, 6 botanists, 9 fishery biologists and 91 wildlife biologist to oversee stewardship of 7 million acres (2.8 million ha) of public land in the Rockies. While the current BLM staff is inadequate to provide oversight of wildlife, fish and plant resources in the Rockies, New Mexico provides a striking case. In 2002, the BLM apparently did not have a single ecologist, botanist or fisheries biologist on field staff in New Mexico (U. S. Bureau of Land Management 2003c). Additional staff is especially needed to address the added workload placed on BLM employees from the executive order requiring fast tracked energy plans. As noted from a survey of BLM employees in Utah (U. S. Bureau of Land Management 2003c:95): "In areas with high demand for energy development there is insufficient time for existing staff to keep up with the workload it creates. In all cases, staffing and funding are insufficient to establish and implement a proactive FWBSSS program . . . . The increased workload generated by energy development, land and realty actions, minerals development and grazing are creating a workforce that is stressed, over-worked, and facing potential burnout." More field ecologists and biologists are needed for stewardship to be successful.

The U. S. Bureau of Land Management Must Complete a Cumulative Impact Analysis

As part of its energy policy, the current administration has essentially directed agencies with jurisdiction over energy development on public lands to prioritize drilling over all other concerns, including protection of wildlife. As one example, in a recent final environmental impact statement, BLM revoked significant protections for wildlife and habitat that were included in the draft, stating that "BLM is required to impose the least restrictive constraints needed to provide adequate protection while allowing fluid minerals leasing and development"(U. S. Bureau of Land Management 2003b).

This nationally mandated approach requires a corresponding national analysis of the potential cumulative impacts to wildlife and other resources. As the National Environmental Policy Act requires, federal agencies must assess the environmental impact of a proposed action, taking a hard look at environmental consequences, and the scope of the analysis, must be appropriate to the action in question.

In determining the appropriate scope of environmental analysis for an action, federal agencies must consider not only the single proposed action, but also three types of related actions: (1) connected actions, (2) cumulative actions and (3) similar actions. Under any of the three classifications, the coordinated actions that federal agencies are directed to take in compliance with the current energy policy trigger a broad assessment of cumulative impacts. Since the government is mandating a program of prioritizing oil and gas development, the resulting agency actions are connected as interdependent parts of a larger action, all of which, depend on the larger action (the government policy) for their justification. Further, the many actions taken by different federal agencies to accelerate drilling will compound species and habitat, so adding together each of the areas of habitat lost to or fragmented by development will have a cumulatively significant impact on species. Finally, because the Administration's energy policy extends to all agencies with responsibilities for oil and gas development and will be concentrated in the Rockies, the reasonably foreseeable actions will have common timing and geography, and will be similar in terms of opening more areas for development and approving more development activities.

The cumulative impact analysis required to accurately evaluate the potential environmental consequences of this policy would necessarily include the entire area that is potentially affected. We interpret this to include the 100 million acres (40 million ha) identified in the administration's recent energy assessment (U. S. Department of the Interior and U. S. Department of Energy 2003) as well as the areas targeted by the administration for expedited energy plans including the Rocky Mountain Front, the Powder River Basin, the Upper Green River Valley, the Roan Plateau, the HD Mountains, the Book Cliffs and Otero Mesa.

Conclusions

Our fragmentation analysis showed the significant fragmentation of wildlife habitat associated with large-scale energy development. Based on our GIS analysis of gas and oil, it is clear that drilling public wildlands in the West will do little to affect the nation's energy future. Therefore, one should not assume that extracting energy resources is always the highest and best use of public lands. As roadless wildlands and critical wildlife habitat become scarce, their economic and ecological values increase. As a result, the marginal benefits from wildland conservation are, in many cases, much greater than the marginal costs in the form of foregone undiscovered, economically recoverable energy resources.

With this in mind, we urge BLM to reduce the environmental risks by slowing and reducing the scale of proposed drilling while taking a more conservative approach that is consistent with adaptive management principles. Stewardship of public lands requires no less.

We urge Congress and the states to reduce the environmental risks by investing in applied research by agency and university scientists to improve the science behind adaptive management decisions. Applied research begets stewardship.

We urge the agencies to address the data crisis by investing in spatially explicit baseline data, based on ground-truthing resource conditions. Investing in accurate baseline data and scientific analysis and monitoring of the data collected is a prerequisite for good stewardship. To accomplish this, we recommend that the agencies add staff while also increasing the budgets for agency biologists, botanists and ecologists in district and regional offices.

We also urge the agencies to stop or slow the outsourcing of environmental analyses to private firms, especially those with consulting connections to the energy industry. The quality of the analysis suffers, the costs to taxpayers are greater and the conflict of interest is obvious. As a result of Enron, we no longer allow accounting firms to be both accountants and consultants to the same clients. We should follow suit with public land management. Rather than outsourcing jobs from rural communities, we recommend increasing the number of BLM field staff. Investing in field staff will help promote economic development because, in many western communities, BLM employees provide an important source of income that supports local jobs.

We urge the BLM and the U. S. Forest Service to consider wilderness a vital component of stewardship. Wilderness epitomizes multiple-use public land in a pure but rugged sense. Wilderness areas protect watersheds and wildlife habitat. Wilderness provides pristine settings for high-quality backcountry hunting, fishing, hiking, bird watching, skiing, horseback riding, climbing and camping. Westerners and visitors alike are not interested in pursuing these activities near drilling rigs, noisy gas compressors, or smelly waste pits.

Although, demand for natural gas has been flat, consumers have suffered from two price spikes. The government has concluded that price manipulation occurred and the current spike in natural gas prices is under investigation for market manipulation (Federal Energy Regulatory Commission 2003, Morton 2003). Rather than using taxpayer dollars to subsidize risky and potentially low-productivity gas wells, federal energy policy should stretch proven gas reserves by reducing waste through investments in low cost, low risk, no regret solutions, like energy conservation and efficiency. We can also reduce our risks by diversifying our energy portfolio with renewable energy sources, such as solar, wind and biofuels energy, at the appropriate locations and scale. Conservation combined with competition from renewable energy will reduce the demand for natural gas and result in lower prices for consumers. Simple efficiency measures can permanently reduce utility bills, while drilling marginal gas wells results in, at most, a temporary cost savings for consumers.

With a ten-year supply of gas in proven reserves, 53,000 producing wells on public land in the Rockies, 46,000 drilling permits issued since 2000 and 32 million acres (13 million ha) of federal minerals already under lease, there is no need to expedite drilling in pristine areas. The public deserves a better understanding of the cumulative environmental impacts that the current drilling boom has on air, water and western landscapes. The public deserves an honest assessment of the economics, including the negative impacts of reducing environmental protection on local tourism, ranching, recreation, hunting, fishing and quality of life. These are not unreasonable requests or concerns.

References

Allan, J. D. 2002. *Expert Comments on Montana statewide draft oil and gas environmental impact statement (EIS) and amendment of the Powder River and Billings resource management plans*, unpublished data.

Amos, J. 2003. *Environmental aspects of modern onshore oil and gas development*. U. S. House of Representatives, subcommittee on energy and mineral resources. http://www.skytruth.org/pdfiles/Amos%20testimony%2017sep03.pdf.

Attanasi, E. D. 1998. Economics and the 1995 national assessment of United States oil and gas resources. In *U. S. Geological Survey circular 1145*. Washington, DC: U. S. Department of the Interior.

Bennett, R. A. 2003. *Testimony before the House resources subcommittee on energy and mineral resources*. Field Hearing; Rawlins, Wyoming. July 12.

Bock, R., and F. Lindzey. 1999. *Progress report, Jack Morrow Hills desert elk study*. Laramie, Wyoming: University of Wyoming Cooperative Fish and Wildlife Research Unit.

Braun, C. 2002. *Expert comments on Montana Statewide Draft Oil and Gas Environmental Impact Statement (EIS) and Amendment of the Powder River and Billings Resource Management Plans (RMPs)*, unpublished data.

Chapman, P. M., H. Bailey, and E. Canaria 2000: Toxicity of total dissolved solids associated with two mine effluents to chironomid larvae and early lifestages of rainbow trout. *Environmental Toxicology and Chemicals*. 19:210–4.

Christiansen, T. 2000. What happened to all of the sage grouse? *Wyoming Wildlife News*. 9(5): 11–14
Clements, W. H. 2002. *Expert comments on Montana statewide draft oil and gas environmental impact statement (EIS) and amendment of the Powder River and Billings resource management plans (RMPs)*, unpublished data.
Cleveland, C. 2003. *Expert comments on EPCA report scientific inventory of onshore federal lands' oil and gas resources and reserves and the extent and nature of restrictions or impediments to their development*, unpublished data.
Corn, M. L., B. A. Gelb, and P. Baldwin. 2001. *The Arctic National Wildlife Refuge: The next chapter.* Washington, DC: Congressional Research Service.
Crawford, J. 1986. Land use planning in the Bureau of Land Management. In *Transactions of 51st North American wildlife and natural resources conference.* Washington, DC: Wildlife Management Institute.
Dickerson, B. R., and G. L. Vinyard 1999. Effects of high levels of total dissolved solids in Walker Lake, Nevada on survival and growth of Lahontan cutthroat trout. *Transactions of the American Fisheries Society*. 128:507–15.
Energy Information Administration, 2001a. *U. S. natural gas markets: Mid-term prospects for natural gas supply, Office of Integrated Analysis and Forecasting*, Washington, DC: U. S. Department of Energy.
———. 2001b. *Energy information administration short-term energy outlook*, Washington, DC: Department of Energy.
———. 2004. *Annual energy outlook 2004 with projections to 2025*. Energy Information Administration. http://www.eia.doe.gov/oiaf/aeo/index.html.
Federal Energy Regulatory Commission. 2003. *Natural gas market assessment, winter 2002–03*. Washington, DC: Federal Energy Regulatory Commission.
Forman, R. T. 1999. Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology*. 14:31–5.
Goetsch, P. A., and C. G. Palmer. 1997. Salinity tolerances of selected macroinvertebrates of the Sabie River, Kruger National Park, South Africa. Arch. *Environmental Contamination Toxicology*. 32:32–41.
Gore, J. A. 2002. *Expert comments on Montana statewide draft oil and gas environmental impact statement (EIS) and amendment of the Powder*

River and Billings resource management plans (RMPs), unpublished data.

Hartley, D., J. Thomson, P. Morton, and E. Schlenker-Goodrich. 2003. *Ecological effects of a transportation network on wildlife: A spatial analysis of the Upper Missouri Breaks National Monument*. The Wilderness Society. http://www.tws.org/Library/Documents/MissouriBreaksTransportationEffects.cfm.

Haynes, R. W., and A. L. Horne. 1997. Economic assessment of the basin. In *An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins*: *Volume IV*, 1715–1870, eds. T. M. Quigley, and S. J. Arbelbide. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station

Hubert, W. A. 1993. The Powder River: A relatively pristine stream on the Great Plains. In *Restoration planning for the rivers of the Mississippi River ecosystem, report 19*, eds. Larry W. Hesse, C. B. Stalnaker, N. G. Benson, and J. R. Zuboy, 387-95. Springfield, Virginia: U. S. Department of the Interior National Biological Survey.

Jensen, W. F. 1991. *Internal memo*. North Dakota Fish and Game Department.

Krutilla, J. V. 1967. Conservation reconsidered. *American Economic Review*. 57:777–86

Krutilla, J. V., and A. C. Fisher. 1985. *The economics of natural environments. Resources for the future*. Washington, DC: Resources for the Future.

LaTourrette, T., M. Bernstein, P. Holtberg, C. Pernin, B. Vollaard, M. Hanson, K. Anderson, and D. Knopman. 2002. *Assessing gas and oil resources in the Intermountain West: Review of methods and framework for a new approach*. Santa Monica, California: RAND Science and Technology.

Loomis, J. 1993. *Integrated public lands management*. New York, New York: Columbia University Press.

Loomis and Richardson, 2000. *The economic benefits of Forest Service roadless areas*. The Wilderness Society. http://www.wilderness.org/Library/reports.cfm.

Lyon, A. G. 2000. *The potential effects of natural gas development on sage grouse* (Centrocercus urophasianus) *near Pinedale, Wyoming*. M. S. thesis. University of Wyoming, Laramie, Wyoming.

Lyon, L. J. 1979. Habitat effectiveness for elk as influenced by roads and cover. *Journal of Forestry*. 77:658–60.

———. 1983. Road density models describing habitat effectiveness for elk. *Journal of Forestry*. 81:592–6.

Minerals Management Service. 2000. *State mineral summaries*. U. S. Department of the Interior, Washington, DC. http://www.mrm.mms.gov/stats/pdfdocs/sms2000.pdf.

———. 2001. *MMS Updates estimates for oil and gas resources for the OCS*. Minerals Management Service. http://www.gomr.mms.gov/homepg/whatsnew/newsreal/010117.html.

Morgantini, L. E., and R. J. Hudson. 1979. Human disturbance and habitat selection in elk. In *North American elk: Ecology and management*, eds. M. S. Boyce, and L. D. Hayden-Wing, 132–9. Laramie, WY: University of Wyoming.

Morton, P. 1999. The economic benefits of wilderness: Theory and practice. *University of Denver Law Review*.76(2):465–518.

Morton, P., C. Weller, and J. Thomson, 2002a. *Energy and western wildlands: A GIS analysis of economically recoverable oil and gas*. The Wilderness Society. http://www.wilderness.org/Library/Documents/Energy_WesternWildlands.cfm.

———. 2002b. Coal bed methane and public lands: How much and at what cost? In *Coalbed methane development in the intermountain West*, 156–75. Boulder, Colorado: Natural Resources Law Center, University of Colorado School of Law.

Morton, P. 2003. *Testimony before the speakers task force on affordable natural gas*. United States House of Representatives. http://www.wilderness.org/NewsRoom/TipSheets/20030924.cfm.

Whitelaw, E., ed. 2003. *A letter from 100 economists to President Bush and the governors of eleven western states regarding the economic importance of the West's natural environment*. http://www.econw.com/pdf/120303letter.pdf.

Noon, B. R. 2002a. Expert comments on *Draft EIS for the Powder River Basin oil and gas project, Wyoming*, unpublished data.

———. 2002b. *Expert comment on Farmington resource management plan and draft environmental impact statement (DEIS),* unpublished data.

Perry, C., and R. Overly. 1976. Impact of roads on big game distribution in portions of the Blue Mountains of Washington. In *Proceedings of the*

Elk-Logging-Roads symposium, ed. S. R. Hieb, 62–8. Moscow, Idaho: Forest, Wildlife and Range Experiment Station, University of Idaho.

Peterson, G. L., Sorg, C. F. 1987. *Toward the measurement of total economic value. USDA Forest Service. GTR RM-148*. Fort Collins, Colorado: Rocky Mountain Forest and Range Experiment Station.

Pillard, D. A., D. L. DuFresne, J. E. Tietge, and J. M. Evans 1999. Response of mysid shrimp (*Mysidopsis bahia*), sheepshead minnow (*Cyprinodon variegatus*), and inland silverside minnow (*Menidia beryllina*) to changes in artificial seawater salinity. *Environmental Toxicology and Chemicals*. 18:430–35.

Power, T.M. 1996. *Lost landscapes and failed economies: The search for a value of place*. Covelo, California: Island Press.

Rasker, R. 1994. A new look at old vistas: The economic role of environmental quality in western public lands. *University of Colorado Law Review*. 52(2):369–99

Rig Data. 2004. *Rig counts*. Rig Data. http://www.rigdata.com

Root, D. H. E. Attanasi, R. F. Mast, and D. L. Gautier. 1997. *Estimates of inferred reserves for the 1995 USGS national oil and gas resource assessment, U. S. Geological Survey open-file report 95–75L*, Washington DC: U. S. Geological Survey.

Rost, G. R., and J. A. Bailey. 1979. Distribution of mule deer and elk in relation to roads. *Journal of Wildlife Management*. 43:634–41

Sawyer, H., F. Lindzey, and B. Jellison. 1997. Applying GIS technology to test an elk habitat effectiveness model in north-central Wyoming. In *Proceedings of the 1997 deer/elk workshop, Rio Rico, Arizona*, ed. J. de Vos, 176–83. Phoenix, Arizona: Arizona Game and Fish Department.

Sawyer, H., and F. Lindzey. 2000. *Jackson hole pronghorn study*. Laramie, Wyoming: Ultra Petroleum, Wyoming Game and Fish Department, U. S. Fish and Wildlife Service, U. S. Forest Service, U. S. Bureau of Land Management, and Teton Science School of Wyoming. Cooperative Fish and Wildlife Research Unit.

Schlesinger. W. H. 2002. *Expert comments on Montana statewide draft oil and gas environmental impact statement and amendment of the Powder River and Billings resource management plans*, unpublished data

Shanley, K. W., R. C. Cluff, L. T. Shannon, and J. W. Robinson. 2003. Controls on prolific gas production from low-permeability sandstone reservoirs in basin-centered regions: Implications from the Rocky Mountain Region for resource assessment, prospect appraisal, and risk analysis. In *2003 meeting of the American Association of Petroleum Geologists*. Tulsa, Oklahoma: American Association of Petroleum Geologists.

Trout Unlimited. 2004. *Gas and oil development on western public lands. Impacts on fish, wildlife, hunting and angling*. Trout Unlimited. http://publiclands.tu.org/topic.asp?index=3&topic=23&id=2.

Turner, M. G., R. H. Gardner, R. V. O'Neill, and S. M. Pearson. 1994. Multiscale organization of landscape heterogeneity. In *Volume II: Eastside forest ecosystem health assessment. ecosystem management: Principles and applications, USDA Forest Service, general technical report PNW-GTR-318*, eds. M. E. Jensen, and P.S. Bourgeron, 73–9. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

U. S. Bureau of Land Management. 1990. *Big Piney-LaBarge coordinated activity plan environmental assessment,* Rock Springs, Wyoming: Rock Springs District Office.

———. 1997. *BLM public land statistics*, *fiscal year 1997–2002*. Washington, DC: U. S. Bureau of Land Management.

———. 1999. *Draft EIS for the Pinedale anticline oil and gas exploration and development project*, *Sublette County, WY*. Pinedale, Wyoming: U. S. Department of the Interior, U. S. Bureau of Land Management, Pinedale Field Office.

———. 2000. *Draft EIS for the Jack Morrow Hills Coordinated Activity Plan*. Rock Springs, Wyoming: U. S. Department of the Interior.

———. 2001. *Wildlife monitoring protection plan for the Pinedale anticline project*. Laramie, Wyoming: TRC Mariah Associates Inc.

———. 2002. *Public land statistics for 2002*. U. S. Bureau of Land Management. http://www.blm.gov/natacq/pls02/.

———. 2003a. *BLM public rewards from public lands*. U. S. Bureau of Land Management. http://www.blm.gov/nhp/pubs/rewards/2003/index.htm.

———. 2003b. *Proposed RMPA/Final EIS for federal fluid minerals seasing and development in Sierra and Otero counties*, Las Cruces, New Mexico: Las Cruces Field Office.

———. 2003c. *Fish, wildlife, botany, and special status species program* *evaluation: Final report on evaluation findings and recommendations for action plan development*. Fish Wildlife and Botany Group, Renewable Resources and Planning Directorate, and U. S. Department of Interior. http://www.blm.gov/nhp/spotlight/wo230_eval/.

———. 2004a. *Stipulation exception data for raptor wintering and nesting, sage grouse lek, nesting, and winter range from the Pinedale, Wyoming Field Office*. U. S. Bureau of Land Management. http://www.wy.blm.gov/pfo/wildlife.htm

———. 2004b. *Annual report, fiscal year 2003*. Washington, DC: U. S. Bureau of Land Management.

———. 2004c. *Fiscal year 2004 budget justification*. Washington, DC: U. S. Bureau of Land Management.

U. S. Department of Interior and U.S. Department of Energy. 2003. *Scientific inventory of onshore federal lands' oil and gas resources and reserves and the extent and nature of restrictions or impediments to their development*. Washington, DC: U. S. Department of the Interior, U. S. Department of Energy.

U. S. Geological Survey. 1996a. 1995 *National assessment of United States oil and gas resources-results, methodology, and supporting data, U. S. Geological Survey digital data series DDS-30*. Reston, Virginia: U. S. Department of the Interior, U. S. Geological Survey.

U. S. Fish and Wildlife Service, U. S. Department of Commerce, and U. S. Census Bureau. 2002. *2001 National survey of fishing, hunting and wildlife-associated recreation*. Washington, DC: U. S. Fish and Wildlife Service, U. S. Department of Commerce, and U. S. Census Bureau.

U. S. Forest Service. 1996. *Interior Columbia Basin ecosystem management project, status of the interior Columbia Basin: Summary of scientific findings, general technical report PNW-GTR-385*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

———. 2004. *Fiscal year 2004 budget justification*. U. S. Forest Service. Washington, DC: U. S. Forest Service.

U. S. Geological Survey. 1996. *Impact of oil and gas activity on land-use management decisions, FS-016-97*. Energy Resource Surveys Program. http://energy.usgs.gov/factsheets/GIS/gis.html.

———. 2001. *Arctic National Wildlife Refuge, 1002 Area, petroleum assessment, 1998, including economic analysis. USGS fact sheet FS-028-01*. Reston, Virginia: U. S. Department of the Interior, U. S. Geological Survey.

Urban, D. L., R. V. O'Neill, and H. H. Shugart. 1987. Landscape ecology, a hierarchical perspective can help scientists understand spatial patterns. *Bioscience*. 37:119–27.

Weller, C. J, Thomson, P. Morton, and G. Aplet, 2002. *Fragmenting our lands: The ecological footprint from oil and gas development. A spatial analysis of a Wyoming gas field.* The Wilderness Society. http://www.wilderness.org/Library/Documents/FragmentingOurLands.cfm.

Western EcoSystems Technology, Inc. 2003. *An evaluation of the 1988 BLM Pinedale Resource Management Plan, 2000 BLM Pinedale Anticline final EIS and recommendations for the current revision of the Pinedale resource management plan.* Cheyenne, Wyoming: Wilderness Society, The Wyoming Outdoor Council and The Greater Yellowstone Coalition.

Wiens, J. A., and B. T. Milne. 1989. Scaling of 'landscapes' in landscape ecology, or, landscape ecology from a beetle's perspective. *Landscape Ecology*. 3: 87–96

Yuhnke, R. 2002. Air quality and CBM development. In *Coal-bed methane conference.* Boulder, Colorado: University of Colorado Natural Resources Law Center.

Session Four.
Fire Management: Burning Issues

Chair
Debra S. Pressman
U. S. Department of Agriculture, Forest Service
Washington, DC

Cochair
David Ledford
Rocky Mountain Elk Foundation
Missoula, Montana

The Challenging World of Fire Emissions, Wildlife Habitats, Hazardous Fuels and Ecosystem Health

Dennis Haddow
U. S. Fish and Wildlife Service
Denver, Colorado

James Lawrence
Council of Western State Foresters
and the Western Forestry Leadership Coalition
Lakewood, Colorado

Introduction

Wildfire and prescribed fire play a major role in the development and maintenance of many wildlife habitats and wildland ecosystems. In general, the ecological and societal needs for prescribed fire is increasing as wildlife habitats decline, fuel loadings increase, unnatural successional changes continue and research clarifies the role of fire in natural ecosystems. However, in many parts of the country, fire is a large, intermittent source of fine particulates that can have a significant short-term impact on public and firefighter health. Biomass smoke can also impair visibility in our nations' most scenic areas and on highways. New and proposed air pollution standards and regulations designed to protect human health and visibility are challenging the use of prescribed fire as a land and wildlife management tool. With informed participation in air regulatory processes, wildlife managers can have a significant role in minimizing this threat.

This paper briefly identifies the benefits of conducting prescribed burning and managing naturally ignited wildfires to meet resource management objectives, reducing the risk of catastrophic fires and minimizing air pollution impacts. It also:

- discusses existing and potential legal requirements for managing smoke
- explains the need for wildlife managers' participation in developing air quality policies, guidelines and regulations by informing the public and air regulatory agencies about the critical role of fire in managing wildlife habitat
- lists references where more information on current regulatory activities can be obtained.

Background

Fire's Role in Wildland Ecosystems

Fire regimes have evolved over centuries and have played a role in shaping the ecological components in most of North America's landscapes. According to fire ecologist Robert Mutch (2002:1), "Our ability to sustain/restore forest health (and meet resource objectives) rests in large part on understanding and applying knowledge of fire history, fire regimes, fire effects, fire information (and smoke management principles) to resolve complex resource management issues."

Beyond wildlife habitat issues, basic ecological and societal perspectives challenge us to conduct landscape-scale restoration and maintenance treatments, to address the costs and impacts of wildfire and to conduct project-size burns for biological purposes, while dealing with tremendous growth and development in the urban interface.

In many areas, particularly in lower elevations' dry forests and rangelands, fire regimes have been highly modified, resulting in abnormally destructive wildfire behavior (catastrophic wildfire), substantially increased fire suppression costs, undesirable fire effects, degradation of critical wildlife habitat and significant loss of life and property.

Further, although less dramatic than the recent wildfires of 2000, 2002 and 2003, tree mortality from bark beetles and other forest pests is also increasing beyond acceptable levels in many areas of the West due to continuing drought conditions and overcrowded forests. The primary causes for these declines in forest health are altered forest stand conditions due, in large part, to nearly a century of fire exclusion. Specifically, fire exclusion has resulted in increased tree and shrub densities, shifts in tree and understory species composition, an uncharacteristic age-composition distribution and an accumulation of dead fuels. These causes are subtle points for a North American public that generally associates more trees on the landscape with vigorous forest health. Unfortunately, the real picture, including loss of wildlife habitat, is more complex and troubling.

Decades of wildfire suppression, reduced active management on federal forestlands and the lack of cross-boundary, community-based restoration projects has hampered forestland managers' ability to proactively address forest health and catastrophic wildfire threats. The further we diverge from historic fire regimes and are forced to contend with unnatural fuel conditions, the more likely we are to fail in programs, such as wildlife management. The economic, ecological and social results of fire exclusion are unacceptable.

Fire effects on wildlife vary greatly by influencing cover, vegetative successional stages, animal and population responses at the time of fire, arrangement of habitat components and food. In general, according to Lyon et al. (2000), these effects include the following.

- Burning sets back plant development and succession, often increasing or improving forage for wildlife from a few years to more than 100 years, depending on vegetation type.
- Fires usually increase habitat patchiness, providing wildlife with a diversity of vegetation conditions from which animals may select food and cover.
- The biomass of forage plants usually increases after burning in all but dry ecosystems.
- The production of seeds by grasses and legumes is usually enhanced by annual or biennial fires. Most production is usually enhanced by a 5-year or longer burning cycle.
- Burning sometimes increases the nutritional content and digestibility of plants. This effect is short-lived, typically lasting only one or two growing seasons.

- Some wildlife species select a more nutritious diet from burned areas even though the average nutrient content of burned plants does not differ from that of unburned plants.

Because of these and other factors, prescribed fire is a critical tool in the management of wildlife habitat. The unique ecological benefits of fire are essential to conserving North America's wildlife heritage, whether by restoring native grasslands, wetlands and other ecosystems, by controlling invasive plants or by creating more natural and defensible conditions around refuges and critical habitat for threatened and endangered species. Without periodic fire, the health of fire-adapted ecosystems and the wildlife dependent on them can suffer greatly. Prescribed fire can also be the most cost-effective method of hazardous fuels reduction, necessary for the effective protection of communities in the wildland-urban interface and natural resources. For example, in fiscal year 2003, the cost of mechanical fuels treatment on lands managed by the U. S. Fish and Wildlife Service (FWS) averaged about 20 times the cost of treatment using prescribed fire.

Fire and Fuels Management Today

The western governors and the secretaries of the U. S. Department of Agriculture and the U. S. Department of the Interior have endorsed and transmitted to Congress (per congressional direction), *A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment: 10-Year Comprehensive Strategy, August 2001*, followed by the National Fire Plan (NFP), Implementation Plan, May 2002. The NFP is a long-term strategy based upon the core principles of collaboration, priority setting and accountability, and it has four primary goals:

- improve prevention and suppression of fires
- reduce hazardous fuels
- restore fire-adapted ecosystems
- promote community assistance.

A complement to the NFP is the Healthy Forests Restoration Act (HFRA), signed by President George W. Bush on December 3, 2003 (PL108-148). The bipartisan law is the first major piece of forestry legislation to pass

Congress in more than a decade. The six titles of the bill offer many opportunities and challenges for western state and federal land managers and for private landowners. Each of these goals focuses on restoring the United States' precious forests and rangelands.

The increased direction, funding and emphasis could dramatically increase the level of fuel treatment and restoration. According to the fiscal year 2002 NFP's performance report, the combination of prescribed fuel treatment and wildland fire use management of naturally ignited wildland fires to accomplish specific resource management objectives and ecosystem maintenance objectives and restoration resulted in 3.28 million acres (1.33 million ha) being treated (to mitigate hazardous conditions and to restore forest and rangeland health on federal lands). We can expect the treatment acres to increase with the streamlining of processes with potential funding shifts, increases authorized in the Healthy Forests Restoration Act (HFRA). However, even with the increased national attention on the need for fuels and fire management, there are a number of air quality programs which could challenge the use for fire because of public health and other potential smoke impacts.

Fire and Smoke Issues

Smoke from biomass burning is an air quality concern because of its extent and chemical make-up. Approximately 70 percent of the particulates in biomass smoke are less than 2.5 microns in diameter and, as a result, can be carried deep in to the respiratory system. As such, they are a concern to states and the Environmental Protection Agency (EPA), who have the responsibility to protect human health. These same particulates are of an optimum size to scatter light and can cause visibility impairment in scenic areas. That makes them a concern for federal land managers, states and the EPA, who have a legal responsibility to protect visibility in class I areas (those national parks and wildernesses over certain acreages that were in existence as of August 7, 1977). Very small particulates can also act as condensation nuclei and actually cause fog formation if the humidity is high. This can be a significant concern for highway safety.

There are hundreds of different chemicals that have been identified in biomass smoke; five of these (acetaldehyde, acrolein, (1–4) butadiene, formaldehyde and polycyclic organic matter) have been classified by EPA as air toxics because of public health concerns. The exact amounts of these pollutants released from biomass burning and their impacts on the public are presently unclear. Two of those pollutants, formaldehyde and acrolein, may be a concern for firefighter health, especially during long wildfire campaigns. In short, the extremely small size and chemistry of wood smoke particulates make them a concern relative to public health, safety and welfare.

Air Regulatory Issues

Fine Particulate Standards

As required by Congress, the EPA has recently developed new standards to limit ambient concentrations of those particulates less than 2.5 microns in diameter (PM-2.5) to protect human health. Most current epidemiological studies indicate that there is a much stronger relationship between increases in PM-2.5 concentrations and mortality and morbidity than there is with larger particulates. States are presently completing monitoring to determine which areas of the country may exceed these standards. For areas that do not meet the new health standards, states are required to develop regulations to reduce the amount and impact emissions from sources that generate PM-2.5. This will present a challenge for wildlife management and prescribed burning programs in certain areas of the country if those regulatory programs do not consider the social and ecological tradeoffs relative to fire.

Visibility Regulations

States, the EPA, federal agencies, industry and public interest groups are currently working to develop regulatory programs to protect visibility in 156 class I areas (Public Law 101–549. 1990). For those 156 areas, Congress has identified a national visibility goal: "the prevention of any future and the remedying of any existing, impairment of visibility in mandatory federal cass I areas from man-made air pollution" (42.U.S.C.S. 7469 a). In many cases, states and EPA consider smoke from prescribed fire to be human-made. In fact, states are required by federal regional haze regulations to consider fire and smoke management when developing their state visibility regulations. It should be noted that research conducted in some wildernesses and in national parks indicates that viewing the scenery through clean, fresh air is one of the most important attributes to recreationists. This presents an interesting challenge for federal land managers who want to use prescribed fire in those same areas.

The EPA Regional Haze Regulations (40 CFR Part 51) established five regional planning organizations (RPOs) (see map in Figure 1) to help develop more uniform regional haze programs among the states. Within individual federal agencies and bureaus, working with the five RPOs has been limited by availability of time and staff.

Figure 1. RPO Web Sites: **CENRAP**, Central Region Air Planning Organization, http:// www.cenrap.org; **MANE-VU**, Mid-Atlantic/Northeast Visibility Union http:// www.mane-vu.org; **Midwest Regional Planning Organization** http://www.ladco.org; **VISTAS**, Visibility Improvement—State and Tribal Association of the Southeast http://www.vistas-sesarm.org; **WRAP**, Western Regional Air Partnership, http:// www.wrapair.org.

The RPO that has taken the lead nationally in developing policy and technical programs is the Western Regional Air Partnership (WRAP). The WRAP was created to aid in the implementation of the recommendations of the Grand Canyon Visibility Transport Commission. Under the WRAP, wildland fire is being addressed by a specific forum (The Fire Emissions Joint Forum), whose activities can be reviewed on their Website, at http://www.wrapair.org. The WRAP has adopted a policy relative to the classification of fire as being either natural or anthropogenic. Basically, the policy states that any wildfire or any fire being managed to *maintain* the natural fire frequency will be classified as natural. Fire that is being ignited or managed to *restore* the natural fire frequency is anthropogenic. All fire, such as slash burning or agricultural burning, is also anthropogenic. Anthropogenic fire will probably be subject to more stringent regulations than natural fire. The policy and a number of technical and policy documents addressing visibility protection as it relates to fire and other sources can be found at the WRAP Website.

Wildlife Managers Air Quality Role

The general public's lack of knowledge about the role of fire in wildland ecosystems has the potential to result in more stringent air quality regulations, which, in the long term, could create severe changes to wildlife habitats and ecosystems in general. The public, including most air quality regulators, understands the need for good air quality much better than they understand the relationship of fire to wildlife habitat and populations, forest diseases, protection of rare and endangered species, management of national parks and wilderness, forest succession, and biological diversity. Few of the public are aware that many natural forest and rangeland ecosystems, including wildlife habitats, exist because of periodic fire rather than in spite of it. Unfortunately, the public is becoming more urban and, as a result, may have less of an understanding of biological processes and ecology than their rural counterparts.

The wildlife community can be an important and credible leader in communicating knowledge to the public about the role of and need for periodic fire to maintain wildlife habitats and populations. Any awareness and education efforts will also increase public sensitivity to the role of fire in overall ecosystem maintenance. Messages need to be developed that are clear and concise for different wildlife species, habitats and ecosystems. Based on their beliefs, knowledge and understanding of terms, the public is probably more supportive of programs to protect and manage wildlife than they are of programs to provide sustainable ecosystems. Wildlife and fire managers also need to let the public know that they are doing the best possible job of managing smoke from prescribed fire, and that they are aware of air quality concerns and that they take these concerns seriously.

Population and industrial growth in much of the country are increasing at the same time as the ecological need for the use of prescribed fire is increasing; however, given a finite atmosphere and a need to control air pollutants, the public must determine which sources will be allocated shares of the air resource. Public decisions on the allocation of the air resource will be based to some degree on the public's knowledge of the need and ecological tradeoffs for tolerating certain short-term pollutant levels. In short, the lack of understanding on the public's part has the potential to result in severe restriction on the use of prescribed fire, compared to restriction of other, more understood air pollution sources. In short, this lack of knowledge of the ecological need for fire could result in unnatural, unwanted and potentially catastrophic fuel loadings and ecosystem changes.

Wildlife managers can be an important partner in effectively informing the public and regulatory agencies of the potential nonair-quality impacts that may result from unnecessarily restricting fire in fire-dependent ecosystems. It is a major concern that control of one environmental problem, such as air pollution, might contribute to other environmental problems, such as loss of wildlife habitat, biological diversity or catastrophic wildfire.

Wildlife managers have the opportunity to partipate in the development of air quality policies, guidelines and regulations at the national, regional and state levels. The Websites for the regional planning orgainzations listed in this paper provide information on the status of policies, guidelines and regulations currently under development. As a result of public support for wildlife, wildlife managers can play a pivitol role in some negotiations. Wildlife and fire managers also have the opportunity to work together to develop messages for the public to help the public make informed decisions on where and how fire will be used for wildlife management.

Summary

Fire as a land management tool is critical for managing wildlife habitats and populations. Fire can also result in high, short-term levels of fine particulates that have potential to impact public health, safety and welfare. The public will ultimately decide which air pollution sources will be eliminated or restricted to meet air quality standards and public expectations. The public is generally supportive of effective wildlife management programs and good air quality, but it may not be aware of potential tradeoffs that need to be made in order to have both. Air quality regulations and programs are currently being developed that have the potential to unnecessarily impact the use of prescribed fire. Wildlife managers can play a critical role in providing information to the public on the role of fire to maintain wildlife habitats and populations so the public can make informed decisions. To carry out this role wildlife managers should become involved with the various regional planning organizations and work with fire managers to develop messages for the public on the role of fire in managing wildlife.

Reference List

Botkin, Daniel. 1992. *Discordant harmonies, a new ecology for the 21st century.* New York, New York: Oxford University Press.

Environmental Protection Agency. 1992. *EPA prescribed burning background document and technical information document for prescribed burning best available control measures, EPA-450/2–92–003.* http://www.epa.gov/ttncaaa1/t1bid.html.

Environmental Protection Agency. 1996. *The EPA natural events policy for particulate matter.* http://epa.gov/ttncaaat1/t1/memoranda/nepol.pdf.

———. 1998. The *EPA interim air quality policy on wildland and prescribed burning.* http://www.epa.gov/ttn/oarpg/t1/memoranda/firefnl.pdf.

Hardy, Colin C, Roger D. Ottmar, Janice L. Peterson, John E. Core, and Paula Seamon. 2002. *Smoke management guide for prescribed and wildland fire—2001 edition.* Boise, Idaho: National Wildfire Coordinating Group, National Interagency Fire Center.

U. S. House of Representatives, 108th Congress. 2003. *Healthy Forests Restoration Act.* http://thomas.loc.gov/cgi-bin/query/D?c108:9:./temp/~c108wyalSe::

Lyon, L. Jack, Mark H. Huff, Robert G. Hooper, Edmund S. Telfer, Schreiner, Scott David, and Jane Kapler Smith. 2000. *Wildland fire in ecosystems, effects of fire on fauna. General Technical Report RMRS-GTR-42-Vol.* 1. U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. http://www.fs.fed.us/rm/pubs.

Malm, William C. 1999. *Introduction to visibility, cooperative agreement number CA2350-97-001: T097-04, T098-06.* Fort Collins: Cooperative Institute for Research in the Atmosphere, NPS Visibility Program, Colorado Sate University.

Mutch, Robert W. 2002. *Lesson Plan.* Phoenix, Arizona: National Advanced Resource Technology Center.

National Fire Plan. 2001. *A Collaborative Approach for reducing wildland fire risks to communities and the environment, 10-year comprehensive strategy.* http://www.fireplan.gov.

———. 2003. *FY2002 National fire plan performance report.* http://www.fireplan.gov.

Sampson, R. Neil. 1997. Forest management and climate change policy issues in the 11 western states. Alexandria, Virginia: Forest Policy Center, American Forests.

Teie, William C., and Brian F. Weatherford. 2003. *Fire in the west—The national fire plan and fire season 2002, 5th edition, a report for the western state fire managers.* Rescue, California: Deer Valley Press.

Western Forestry Leadership Coalition. 2002. *Western bark beetle report, a plan to protect and restore western forests.* Fort Collins, Colorado: U. S. Department of Agriculture, Forest Service.

———. 2003. *Policy Statement: Forestlands at risk: Forest health and wildfire.* http://wwwwflccenter.org.

Western Regional Air Partnership. 2004. Western Regional Air Partnership. http://www.wrapair.org.

The Healthy Forests Restoration Act of 2003—Unforeseen Circumstances?

Jack Ward Thomas
Boone and Crockett School of Wildlife Conservation, College of Forestry and Conservation
University of Montana, Missoula

Nicholas Noble
College of Forestry and Conservation
University of Montana, Missoula

Forest Health as Related to Healthy Forests Restoration Act of 2003

The Healthy Forests Restoration Act of 2003 (HFRA) is the latest chapter in a continuing debate of how to deal with forest conditions; these forests encompass 192 million acres (77.7 million ha) on public land and have become "unhealthy" due to past management practices. Foremost among those practices has been long-term (more than 50 years) protection from fire. This period has encompassed a number of anticipated fire return intervals and has resulted in stocking densities far in excess of what would be considered normal.

Selective harvest of large trees or clear cutting coupled with fire exclusion has, in some cases, resulted in crowded stands of early successional, fire-sensitive species. In addition, these stand conditions have proven susceptible to outbreaks of diseases and insect infestations, leading to widespread and, sometimes, intense mortality.

Management approaches to dealing with these evolving circumstances have become the focus of technical, political and legal controversy. The debate has engendered a series of efforts to address the issue, including the cohesive Strategy (General Accounting Office 1999), the National Fire Plan (U. S. Department of Agriculture and U. S. Department of the Interior 2000), to the Healthy Forest Initiative ((U. S. Department of Agriculture, Forest Service 2001), which led to the Western Governors Action Plan, culminating in the HFRA.

The HFRA defines "forest health" as the creation and maintenance of forest stand conditions resistant to stand-replacement wildfires that threaten human life, property and water for human consumption in the wildland-urban interface (WUI) and in municipal watersheds—a relatively new and focused definition of forest health. Some $760 million per year was authorized in the HFRA for the U. S. Department of Agriculture (USDA), Forest Service (FS) and the U. S. Department of Interior (USDI) to carry out the described program, and President George W. Bush has included that amount in his proposed budget for 2005.

Filip (2002) noted that the definition of forest health varied, depending on the desired outputs from the forest system in question. Kolb et al. (1994) said the utilitarian definition sustained timber production as the key indicator. The FS (1993) has defined forest health as a condition wherein living and nonliving influences did not threaten achievement of management objectives (which could vary widely) in either the present or the future. Monning and Byler (1992) defined forests in good health as a fully functioning community of plants and animals and their physical environment.

Scientists have described three basic fire regimes (Agee 1993). High-severity fire regimes are associated with alpine, subalpine and coastal forest systems, with fire return intervals of 200 to 400 years with stand-replacement fires.

Mixed-severity fire regimes predominate in midelevation forests, with fire return intervals of 40 to 80 years that produce a mosaic of habitat types that have unique fire responses. Due to fire exclusion over the past century, some stands are more apt to experience stand-replacement fires.

Low-severity fire regimes predominate in low-elevation, dry forests that are shaped by frequent, low-intensity fires, occurring at 5- to 25-year intervals, which had less effect on larger trees with insulating bark. Past management and fire exclusion have produced stand stocking and fuel accumulations far outside the range of natural variability, making these forests prone to stand-replacement fires.

Political Rhetoric Prior to and upon Enactment

Both poles of opinion in the HFRA debate expressed support for treatments in the WUI. Beyond that support, political discord began (Shindler 2002). Some saw the program as not only protecting human lives, property and water supplies but also as providing jobs in rural communities; they believed the prescribed actions were too narrow in scope (Larazoff 2000). Others considered the program a charade to enhance yields of wood products and to weaken environmental laws (Alliance for the Wild Rockies 2000).

Top-level elected officials have weighed in. President George W. Bush, speaking at the site of a large-scale wildfire in Arizona, said, "Forest-thinning projects make a significant difference about whether or not wildfires will destroy a lot of property. We saw the devastation, we saw the effects of fire run wild, not only on hillsides, but also in communities, in burned buildings, lives turned upside down because of the destruction of fire" (Associated Press 2003).

Senator Joseph Lieberman (D-CT) countered, "Unlike our first president, George Bush just can't come clean about his plan to cut down trees . . . . He is using the need to clear brush and small trees from our forests as an excuse for a timber industry give-away, and Arizonans should make no mistake: This is logging industry greed masquerading as an environmental need" (Associated Press 2003).

Such rhetoric produced a political standoff until the cumulative impact of the fires of 2000 and 2002 and, especially, those of 2003, which stimulated the passage of the HFRA in late fall 2003. But, the likely results to emanate from application of the HFRA are more complex—ecologically, technically and politically—than it appears at first glance.

Application in the Wildland-Urban Interface

This discussion focuses on potential consequences related to carrying out provisions of the HFRA in WUIs that occur in mixed- and low-severity fire regimes. Management actions will be directed to the creation and maintenance of forest conditions now that will reduce the danger to human life and property from wildfire.

If the described forest management actually occurs at the time and space envisioned, it would entail the most significant land-management actions associated with national forests in the last several decades. If the $760 million authorized in President Bush's budget for 2005 is approved by Congress, it is anticipated that 4 million acres (1.6 million ha) will be treated in the first year. It is anticipated that, between 2001 and 2005, some 11 million acres (4.45 million ha) will have been treated. This paper focuses on predictions of effects on wildlife, wildlife habitats and associated social, political and economic consequences in the WUI.

The HFRA (2003:5) defines at-risk communities as, "a group of homes and other structures with basic infrastructure and services . . . within or adjacent to federal land; in which conditions are conducive to a large-scale wildland fire disturbance event; and for which a significant threat to human life or property exists." A WUI is defined as an area within or adjacent to an at-risk community as it is identified in a community wildfire protection plan. If there is no plan in effect, the definition includes, "an area extending ½ mile from the boundary of an at-risk community; or an area within 1½ miles of the boundary of an at-risk community" (5). WUIs usually occur where private land with homes and with associated infrastructure abut rugged, upland terrain that is largely covered with forests and is in federal ownership.

In addition—within areas at high risk of stand-replacement fire—the HFRA authorizes treatment of municipal watersheds, of habitats for threatened and endangered species, of areas of windthrow or blowdown and ice-storm damage, and of places of significant potential for an insect epidemic or a disease outbreak exists. These areas are not covered in this discussion, but these additions increase the potential treatment areas well beyond those associated with WUIs and municipal watersheds. Environmentalists see this as the "catch 22," or the escape clause, in otherwise tightly drawn constraints. There are provisions made to streamline the processes leading to more timely management actions by limiting the number of alternatives presented, constraining appellants to verified participants and accelerating court reviews. Now that the HFRA has been passed into law, it is well to examine potential consequences and ask two questions. The first question determines the extent of the likely impact, asking if the days of greatly intensified timber harvest from the national forests at hand? And, the second question asks, from the environmental view, is the sky really falling?

Silvicultural Treatments and Desired Future Conditions

There is ongoing debate over appropriate silvicultural treatments to achieve the desired results. Whatever the approach, it is certain that small diameter trees will compose the bulk of woody material removed from the WUI and from municipal watersheds. The HFRA provides assistance to develop economic uses for such material.

It will be essential for managers to decide the future condition for tree stands in the WUI and the basis for that decision. That decision will likely be made with reference to the "historic range of variability" (Hesburg et al. 1994, Swanson et al. 1999). The political sticking point, and the subsequent decision on management prescriptions, will revolve around how many trees of larger diameter will be produced, retained and removed in the process. The historic conditions that are to be mimicked are neither well described nor well understood (Tiedemann et al. 2000).

Unforeseen but Likely Consequences

Our intent is to surface some of the likely consequences of executing the HFRA that, so far, have not been recognized or discussed adequately. We hope these factors will be addressed in planning, assessment and mitigation as management is instituted. To be forewarned is to be forearmed.

Ungulate Response

Frequently, WUIs include areas where white-tailed deer (*Odocoileus virginanus*) are year-round residents associated with intermixed agricultural operations, woodlots and wooded riparian zones; these areas are, oftentimes, adjacent to upland forests and are in public ownership. In the mountain West, mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) commonly concentrate in lowland areas adjacent to such areas to spend the winter. Wildlife damage to human life and property in the United States was estimated at $3 billion per year in 1995 (Conover et al. 1995). Human incursions into wildland is steadily increasing. The WUIs in question here are part of such incursion.

Understory Response to Thinning

Forest management in the WUI will focus on thinning of densely stocked stands of trees to produce and maintain spacing between tree canopies to lessen the probability of wildfires taking human lives and destroying property. Reductions in understory vegetation and in residue from stand treatment activities will be routine—first through mechanical means and then with repeated controlled burns, through application of herbicides or with other means.

As a consequence, more sunlight and precipitation will reach the forest floor, favoring increased density and volume of grasses, forbs and shrubs. Periodic controlled burns, mechanical treatments, grazing or applications of herbicides—applied singly or in some combination—will maintain conditions resistant to stand-replacing fire. The resulting conditions may be highly attractive to ungulates.

In turn, predators, including black bears (*Ursus americanus*), mountain lions (*Felis concolor*) and coyotes (*Canis latrans*), will prey on wild ungulates or feed on their carrion. These predators will follow their food sources to where the sources are concentrated. In some areas in the northern Rocky Mountains, grizzly bears (*Ursus arctos*) and wolves (*Canis lupus*) will be included. Depending on locale, numbers of such predators have already markedly increased over the past several decades because of reductions in hunting, trapping and predator control activities.

Escalating Human/Wildlife and Political Conflicts

Exacerbation of human/wildlife conflicts can be expected as ungulates and their predators increase in and near WUI, where stand treatments occur. These conflicts will include some or all of the following: wild ungulates eating or damaging ornamental plants and gardens, collisions between wild ungulates and vehicles, chasing and sometimes killing of wildlife by domestic dogs, transfer of disease between wildlife and domestic animals, transfer of disease between animals and humans, predation on livestock and pets, perceived (and to a small degree real) danger to humans from predators, and increasing political consternation over how to deal with these problems.

Debates will emerge over which agencies should have authority or responsibility to deal with emergent problems, over which agencies should pay the bills, and over the means or techniques of management. Traditionally, states deal with nonmigratory wildlife. However, these new or exacerbated problems will be attributable to federal actions, which will create a demand for federal dollars for required wildlife management activities. The array of potential solutions, ranging from fencing to control of animal numbers to how those reductions should be carried out (such as lethal techniques versus live animal removal) will be recurring, expensive and controversial. For example, experience has shown that hunting in and near human-populated areas is fraught with problems, including safety concerns and political acceptability.

Some Wildlife Species Will Be Losers

There will be an intentional elimination or a significant reduction of standing dead trees (snags) in the WUI. Those snags, when ignited, can become a source of windborne embers that can spread fire. Snags can also be a hazard to humans. Dead and fallen woody material that adds to the spread and intensity of wildfire will likewise be targeted for removal.

A suite of vertebrate wildlife species associated with cavities in trees, primarily dead trees, will be adversely affected within treated areas. As an example, in the Blue Mountains of Oregon and of Washington, Thomas et al (1979) identified 30 birds and 23 mammal species that utilize tree cavities, and Maser et al. (1979) identified some 179 vertebrate species that made some use of woody vegetation, primarily logs, on the ground. Tiedemann et al. (2000) reported forest arthropods were much diminished in areas from which wood was harvested and then burned. While those effects are significant and should be recognized, the amount of the areas treated in the short-term will likely be small enough and scattered enough to minimize overall effect on the wildlife species in question, unless there are threatened or endangered species involved whose habitats are slated for treatment. However, these adverse effects on wildlife will be concentrated in forests in proximity to human habitation, making those areas relatively depauperate in wildlife diversity. However, the total acres anticipated to be treated (11 million acres [4.45 million ha] between 2001 and 2005) may be significant. The exact kind of treatment is not clear.

Stephenson (1999) and Rochelle (2002) suggested stand treatments to reduce danger of stand-replacement fire to maintain some habitat provided by snags and dead and fallen woody material. Such approaches are expensive and can be achieved only with some diminution in the effectiveness of treatments to preclude wildfire.

Exotic Weeds and Varied Responses

Trees removed in thinning operations will be predominantly small and, consequently, low in commercial value. Therefore, thinning operations will be commonly executed at the lowest feasible cost, using ground-based equipment with associated significant ground disturbance. Given the attributes of the WUI related to road density, human habitations, presence of domestic livestock and the feeding thereof, treated areas can be expected to be persistently invaded by noxious weeds with coincident threats to biological diversity and ecosystem function (Griffs et al. 2001). Any extensive or continuing use of herbicides in the WUI is likely to be controversial, expensive, politically difficult and, therefore, unlikely to be acceptable over the long-term. Mechanical and biocontrol methods for noxious weeds are more expensive and less effective. The chief of the FS has identified noxious weeds as a primary area of concern. That problem is likely to be at its worst in the WUI.

Flashy Fuels Added to the Mix—Now What?

Stands of trees thinned and periodically treated to maintain desired open stand conditions will produce an increased volume of grasses, forbs and shrubs. Unless removed, this biomass will accumulate and increase fuel loading. In the late summer, when the danger of wildfire is highest, these "flashy fuels" will exacerbate fire problems. When dried (particularly when several years of accumulation), such fuels are easily ignited and can support hot, fast-burning and fast-moving wildfire.

Traditionally, these fuels have been partially controlled through livestock grazing, which would likely continue on private land. However, livestock grazing on adjoining public land is under increasing attack with an uncertain future (Wuerthner and Matteson 2002). The contribution of livestock grazing to reduction of ground level fuels begs consideration. In any case, attention to periodic reductions in ground level fuels in treated areas will be essential to reduce fire danger from this source.

Efforts to maintain desired stand conditions, including depression of exotic weeds and reduction in ground level fuels following thinning, may depend on controlled burning at appropriate and frequent intervals. Given the presence of people, houses, buildings and other infrastructure in the WUI, such burning will be confined to periods and circumstances whereby prescribed fire can accomplish the objective of fuel reduction with acceptable levels of risk to human lives and property. In other words, a high degree of certainty must exist that such fires will not escape control or will not produce unacceptable levels of smoke for unacceptable periods of time.

Prescribed Fires—Neither Easy nor Certain

In most parts of the West, managers will conduct controlled burns in the spring or late in the fall, under conditions of relatively low wind speeds and low temperatures, coupled with relatively moist conditions. While safer to execute, such burns produce less heat and more smoke than hotter burns, which occur during periods when wildfires most commonly occur. As a result, controlled burns will most commonly take place under conditions outside the natural range of occurrence. The consequences of repeated off-season burning to the long-term vigor of native species of vegetation are largely unknown. Technical and social problems associated with such burning will increase over time because each year will see an increased number of acres brought into and then maintained in desired condition. The more acres that must be maintained in the desired condition the more treatments will be needed to maintain those conditions, including burning.

If or when smoke from controlled burns becomes socially or politically intolerable, it is likely that managers will fall back to an alternative (such as mechanical fuel treatments, grazing or herbicides) to maintain desired conditions. Those alternatives have their own inherent advantages and drawbacks (Sheley et al.1995).

Some prescribed burns, no matter how skillfully conducted, will, sooner or later, escape control and do damage—both to finances and to public confidence and tolerance. If the programs envisioned in the HFRA are to be long-lived and are to include fire in the array of management options, adequate resources must be in place and immediately available to promptly extinguish burns that do escape control.

Congressional Mandates—Will Future Actions Match the Mandates?

Many millions of acres of public forests are now outside their range of natural variability because of logging, fire suppression, weather cycles and failure to carry through scheduled forest management operations (Graves 1987). Examining the history of federal forest management raises caution. Congresses and Administrations routinely bless programs and fund activities that are politically attractive under the circumstances of that moment. Continued political and public support for any course of action cannot be taken for granted. Circumstances change, political players come and go, and power shifts. Constituencies with adequate political persuasion must be maintained if any land management activity of high cost is to continue long-term.

For example, from 1950 through the mid-1980s, federal timber sales and associated road building were consistently funded at or above levels requested by the FS. Road maintenance, road eradication, stand-tending (e. g., repeated thinning), fish and wildlife mitigation, and other associated items were funded below requested levels, i. e., full funding to assure a balanced program was put off. Too often, "another day" came late or, more commonly, never came at all.

Such reneging on clearly understood but nonbinding agreements to provide resources for active stand and road management (combined with changes in what the public wanted from the national forests) helped to produce the forest health crisis of today. The total effect of these sales, sans adequate attention to secondary effects, were below cost, both in the economic and environmental senses. Today's elected officials, who examine responsibility for the perceived crisis, will find that their predecessors helped to produce the circumstances that exist today. Now, it is these elected officials' pain to wonder if future congressional actions will assure that authorizations in the HFRA turn into consistent appropriations over decades to come to execute the mandated programs.

In for a Dime, in for a Dollar?

Once undertaken and pursued, stand treatments to reduce fire risks will create an entirely new set of stand conditions, very significantly, where the forest meets the people. Faithful maintenance of those conditions to retain their hoped-for resistance to stand-replacement fire, will be essential—year after year, decade after decade, century after century. Failure to maintain the open stand conditions that are created and maintained at such high costs will allow those same stands to evolve into a new set of conditions susceptible to stand-replacing fire danger equal to, or worse, than what existed before treatments ensued.

And, costs will quite likely increase well beyond current estimates if the stated objectives are to be met. Why? At first, the cumulative total of acres treated will increase rapidly as the vast majority of resources available will be devoted to initial treatments. But, after a time lag, an ever-increasing portion of the funds available will have to be used to maintain the inexorably growing number of acres brought into desired condition.

At some point, unless the budget routinely increases to deal with accelerating costs for maintenance, all available funds will be required for maintenance. How quickly that point is reached depends on the ratio of the costs of treatment to the cost of maintenance.

We should recognize that we are, in the poker player's parlance, in for a dime, in for a dollar. In other words, an initial investment carries with it the requirement to maintain the condition produced, even as costs rises. Otherwise, forest conditions produced at high cost to resist fires of a magnitude that threaten human lives and property will inevitably revert, upon cessation of scheduled treatments, to previous or worse conditions.

Do We Want More Wildland-Urban Interfaces?

Obviously, WUIs come into being—or increase in size—as more and more people build homes in the woods. In doing so, they place themselves and their property in harm's way. Clearly, if people ceased building homes in such locations, there would be fewer problems with forest health, in relation to protection of life and property. Because human populations in the West continue to increase faster than that of the nation as a whole, can a worsening problem related to expansion of the WUI be avoided? Prevention is far cheaper and more certain than fixing.

Will successful efforts to reduce the danger to lives and property from wildfires have the perverse effect of encouraging even more home building in such areas and creating more WUIs? If so, the stand treatments prescribed, unless coupled with governmental actions (federal, state and local) to discourage any additional building in the WUI, will magnify an already serious problem. Clearly, the fire fighting cadres of the federal land management agencies are, more and more, devoted to being the WUI's fire departments, either directly or through reimbursement of state and local fire departments for their efforts in this regard.

When it is considered that such actions as those described in the HFRA could encourage even more building, in spite of a clear and present danger of such developments being lost to wildfire, the plot thickens. Now that there is an acknowledged, governmental responsibility to reduce that danger to human life and property in the WUI, one might well consider the following question. Is there an assumed liability for those properties if such continued protection is not forthcoming on a timely and continual basis?

As a nation, we have faced similar problems with building in flood plains and in coastal areas. We should examine those precedents and act accordingly.

Placing Things in Perspective—No Need for Panic

It is well, at some point, to place the probable outcomes in perspective. The HFRA limits the federal land that can be treated to 20 million acres (8.1 million ha) and authorizes the appropriation of $760 million per year to cover authorized activities. In addition to treatments on federal land, this authorized appropriation can be used to cover treatments on land in nonfederal ownership that is covered by community wildfire protection plans, monitoring activities relative to executed treatments and grants to states and local governments, Native American tribes and other eligible recipients for related activities. Just how much will be available for treatments on federal land is unclear and problematical.

An authorization is not an appropriation. Being in President Bush's recommended budget does not assure enactment. And, a new appropriation is required each year. If appropriations are forthcoming to match the authorization, it is not clear how much of such appropriations will fund the activities authorized by the HFRA nor which acres will be treated. It would be unwise to jump to conclusions to the extent of the environmental effects logically anticipated from carrying out this legislation. Treatments in the WUI can be expected to be much less controversial than those in the back country and will, likely, receive priority treatment.

Given the deepening deficit of the federal budget, an unfavorable balance of trade, accumulating federal debt and the anticipated increases in government spending, such an appropriation or the continued support for such programs is no certainty. Fact sheets, emanating from the USDA and USDI, on what land management actions can be expected from HFRA activities make it impossible to anticipate achievements in reducing chances of stand replacement fires in the WUI. It is anticipated that some 3.7 million acres (1.5 million ha) of federal land will be treated in fiscal year 2004 (2.6 million acres [1.05 million ha] under the National Fire Plan and 1.1 million acres [.45 million ha] from other sources) to reduce fire danger on federal land. Obviously, there will be a significant difference in costs between types of treatments, e. g., relatively low cost per acre for controlled burns in rangeland, compared to the much higher per-acre costs of thinning crowded, second-growth forest and compared to dealing with ground fuels in forested WUIs. Further, many of the acres treated are the result of collateral benefits of other land management actions and are charged to other budget line items, such as (in the case of the FS) timber, vegetation and watershed, and wildlife and fish. Hazardous fuels reduction is slated for only $266 million (some 35%) of the $760 million dedicated in the HFRA to reduction of fire danger on federal land.

If the cost of thinning and ground fuel treatment approximates $1,000 per acre in the forested WUI (a conservative estimate) and one half ($133 million) of the allocation for treatment of hazardous fuels is spent in forested WUIs associated with national forests, the acres treated would be some 133,000 acres per year (53,846 ha/year). Unless the amount is increased each year to account for cost of maintenance, each year more money will go to maintenance and less will go to treatments.

Considering the relatively small portions of forested federal land to be subjected to thinning of forested stands in the short-term plus the constraints, and considering the limiting prescriptions detailed in the HFRA, it seems premature to paint awesome pictures of environmental "doom and gloom" associated with compliance. Assuming the application of the principles of adaptive management, adjustments reducing adverse effects that become obvious with increasing experience should become routine.

In Summary

The Sky is Falling! The Sky is Falling!—Not Really

The FS, seen by many as mired in increasing inaction in land management for nearly two decades, has, in the form of the HFRA, clear direction from both Congress and the President. That clear direction is accompanied by funding to proceed with the specified forest management tasks. Most certainly, FS leaders have been led to understand the level of public and political scrutiny that will be associated with execution of the mandates in the HFRA.

Our advice to both sides of the arguments surrounding the actions pending under the HFRA is to "get a grip." What is about to ensue is a set of "baby steps," constrained in time and space with an opportunity for continuous adjustment. Even then, clear and detailed thought and careful planning are required to execute the direction of the HFRA in a politically and legally acceptable fashion. The complexities and costs of acting on the legislation are yet to be thought out or revealed. Looking before leaping maybe old wisdom, but it remains most excellent advice.

Even with continuous efforts concentrated in high priority locations, large-scale stand replacement fires can be expected to occur, varying in amount from year to year, relative to climatic conditions. The present conditions took many decades to develop, and the circumstances will not be quickly altered.

When Draining the Swamp, Watch for Alligators!

Now that the HFRA is law, agency professionals should diligently prepare for unforeseen consequences. These consequences are most logically handled during the planning and assessment efforts that must be undertaken prior to on-the-ground activities.

The HFRA focused—as legislation designed to deal with perceived crisis is wont to do—on "draining the swamp" (the big picture), with only cursory consideration given to the "alligators" (the unforeseen consequences) lying submerged in that swamp with only their eyes and snouts showing above the surface. Those alligators are best dealt with before they are chewing at the backsides of those vigorously draining the swamp. Pointing out that necessity and a few of the alligators has been our intent.

Reference List

Agee, J. K. 1993. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades In *Eastside forest ecosystem health assessment, vol. III—Assessment*, compiler, P. F. Hessburg, 359–414. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Northwest Forest Experiment Station.

Alliance for the Wild Rockies. 2000. *Groups request congressional hearings on Forest Service $12 billion forest proposal.* News Release, August 29, 2000.

Associated Press. 2003. Bush presses for forest thinning during tour of Arizona Damage. *The News Review of Douglas County.* August 12.

Conover, M. R., W. C. Pitt, K. K. Kessler, T. J. DuBow, and W. A. Sanborn. 1995. Review of human injuries, illnesses, and economic losses caused by wildlife in the United States. *Wildlife Society Bulletin.* 23:407–14.

Filip, G. M. 2002. Wildfire and forest health In *Fire in Oregon's forests: Risks, effects, and treatment options: A synthesis of current issues and scientific literature.* ed. S. A. Fitzgerald. Corvallis, Oregon: The Oregon Forest Resources Institute.

General Accounting Office. 1999. *Western national forests: A cohesive strategy is needed to address catastrophic wildland fire threats.* Washington, DC: General Accounting Office.

Griffs, K. L., J. A. Crawford, M. R. Wagner, and W. H. Moir. 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. *Forest Ecology and Management.* 146:239–45.

Graves, G. R. 1987. Anti-conservation and federal forestry in the Progressive Era. *Dissertation Abstracts International; The Humanities and Social Sciences*. 48(6):1,527.

Hessburg, P. F., G. B. Smith, and R. B. Salter. 1999. Detecting change in forest spatial patterns from reference conditions. *Ecological Applications*. 9:1,232–52.

Healthy Forest Restoration Act. 2003. *Public law (HR 1904)*. 108–48. United States 108th Congress, first session.

Kolb, T. E., M. R. Wagner, and W. W. Covington. 1994. Concepts of forest health: Utilitarian and ecosystem perspectives. *Journal of Forestry*. 92(7):10–5.

Larazoff, C. 2000. Logging does not reduce fire risk, two studies conclude. *Environmental News Service*. September 5.

Maser, C., R. G. Anderson, K. Cromack Jr., J. T. Williams, and R. E. Martin. 1979. Dead and down woody material. In *Wildlife habitats in managed forests—The Blue Mountains of Oregon and Washington. Agriculture Handbook 553*, 78–95. Washington, DC: U. S. Department of Agriculture, Forest Service.

Monning, E., and J. Byler. 1992. *Forest health and ecological integrity in the Northern Rockies, report 92–7*. Missoula, Montana: U. S. Department of Agriculture, Forest Service.

Rochelle, J. A. 2002. Effects of fire on wildlife In *Fire in Oregon's forests: Risks, effects, and treatment options: A synthesis of current issues and scientific literature*, ed. S. A. Fitzgerald. Corvallis, Oregon: The Oregon Forest Resources Institute.

Sheley, R. L., B. H. Mullin, and P. K. Fay. 1995. Managing riparian weeds. *Rangelands*. 17(5): 154–7.

Shindler, B. 2002. Citizens in the fuel-reduction equation: Problems and prospects for public forest managers In *Fire in Oregon's forests: Risks, effects, and treatment options: A synthesis of current issues and scientific literature*, ed. S. A. Fitzgerald. Corvallis, Oregon: The Oregon Forest Resources Institute.

Stephenson, N. L. 1999. Reference conditions for Giant Sequoia forest restoration: Structure, process, precision. *Ecological Applications*. 9:1,253–65.

Swanson, F. J., J. A. Jones, D. O. Wallin, and J. H. Cissel. 1994. *Natural variability—Implications for ecosystem management in eastside forest ecosystem health assessment, vol. II: Ecosystem management: Principles and applications: General technical report GTR-PNW-318*, eds. M. E. Jensen, and P. S. Bourgeron,. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Thomas, J. W., R. G. Anderson, C. Maser, and E. L. Bull. 1979. Snags. In *Wildlife habitats in managed forests—The Blue Mountains of Oregon and Washington. Agriculture Handbook 553*, 60–77. Washington, DC: U. S. Department of Agriculture, Forest Service.

Tiedemann, A. R., J. O. Klemmendson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: Are forest productivity and wildlife at risk? *Ecology and Management*. 127:1–18.

U. S. Department of Agriculture, Forest Service. 1993. *Healthy forests for America's future: A strategic plan, report MP*-1513. Washington, DC: U. S. Department of Agriculture, Forest Service.

———. 2001. A collaborative approach for reducing wildland fire risks to communities and the environment, 10-year comprehensive strategy. Washington, DC: U. S. Department of Agriculture, Forest Service.

U. S. Department of Agriculture, and U. S. Department of the Interior. 2000. *Managing the impact of wildfires on communities and the environment: A report to the President in response to the wildfires of 2000*. Washington, DC: U. S. Department of Agriculture and U. S. Department of the Interior.

Wuerthner, G., and M. Matteson, eds. 2002. *Welfare ranching: The subsidized destruction of the American west*. Washington, DC: Island Press.

The Culture of Fire in the Southeast

William E. Palmer
Tall Timbers Research Station
Tallahassee, Florida

Kevin M. Robertson
Tall Timbers Research Station
Tallahassee, Florida

Ronald E. Masters
Tall Timbers Research Station
Tallahassee, Florida

Introduction

In most of the Southeast, natural plant and animal communities require relatively frequent fires to maintain their ecological integrity (Robbins and Myers 1992, Sparks and Masters 1996). Historically and until relatively recently, use of frequent fire was an integral part of rural southern life (Stoddard 1931, Pyne 1982). For thousands of years, Native and Euro-Americans purposely burned, or allowed to burn, vast acreages for agricultural and silvicultural objectives, wildlife management, pest control, wildfire protection and public attitudes toward the use of fire were favorable (Pyne 1982). Ecologically, the outcome of this fire culture was maintenance of pine and oak savannahs, prairies and other unique communities favoring fire-adapted plant and wildlife species (Stoddard 1931, Komarek 1964, Robbins and Myers 1992, Frost 1998). According to Pyne (1982), concerted legal and educational efforts to reduce burning began as commercial forestry on public and private land; this started to replace range- and row-crop based economies after the turn of the 20th century. At the same time, researchers and forest managers were demonstrating the importance of fire as an ecological process and as a management tool for wildlife and forestry. This began decades of debate within the land management community on the importance of fire for protection of forest resources and later for protection of biodiversity.

Today, the fire culture in the South persists in some pockets, but efforts to encourage our fire heritage may not be strong enough to increase fire use

(Brennan et al. 1998). Millions of acres of industrial pine plantation lands managed for pulp or other timber products exclude fire. Publicly-owned lands have suffered from decades of fire exclusion, or their ecological value has been greatly reduced because of low fire frequencies (Brennan et al. 1998). The loss of fire across most of the southern landscape has resulted in dramatic declines in some species, including the very bird Stoddard first used to demonstrate the importance of fire; the northern bobwhite (*Colinus virginianus*) (Church et al. 1993).

The purpose of this paper is to reiterate the ecological importance of frequent fire for management practices and maintenance of upland systems in the South. We present the Red Hills experience, where fire use has remained the dominant land management practice, an exception to regional and national trends. We summarize obstacles to conducting prescribed fire, based on phone interviews with wildlife and forestry professionals from 13 southern states. We also compare and contrast burning regulations across the Southeast to assess the regional consistency in the regulation and promotion of prescribed fire. Finally, based on the above information, we present some thoughts on the future of prescribed fire in the Southeast and what needs should be considered to expand the use of fire.

The Red Hills Example: Fire Remains Key

On the southern landscape, there exist patches where the use of fire in natural pine forests remains a land management tradition, in stark contrast to the rest of the Southeast (Brennan et al. 1998, Sheffield and Dickson 1998). In these areas the ecological benefits of a long tradition of fire use are obvious (Brennan et al. 1998). Fire is still a part of the southern experience in areas managed for wildlife, principally bobwhites, including Alabama, Florida, Georgia, Oklahoma and South Carolina, and in some agricultural communities, such as rangelands of central Florida. In the Red Hills Region of southern Georgia and northern Florida, Tall Timbers Research Station has been involved with land stewardship for nearly a half a century and thus has some documentation of outcomes of its continued use. Each year in the Red Hills, prescribed burning is applied to about 300,000 acres (120,000 ha) of private lands within a four-county area. The result of this frequent use of fire is low wildfire danger, high ecological integrity of fire-adapted plant communities, high populations of both hunted and nonhunted wildlife populations that are declining elsewhere, and high quality timber production (Masters et al. 2003). For instance, large portions of the Red Hills, where fire and timbering have remained the dominant land management practices, still retain the amount and quality of biodiversity that Stoddard recorded over 80 years earlier (Masters et al. 2003). Further, unlike massive declines throughout the rest of the South, high northern bobwhite populations have been maintained for over 100 years (Brennan et al. 2000).

In the Red Hills, because frequent fire keeps fuels to a minimum and, therefore, fire intensity low, it is not uncommon for an experienced burner to burn over 1,000 acres (400 ha) within a single day after properly establishing fire breaks. It is rare to witness a fire that scorches the pine canopy, unless that was the specific goal of the practitioner. Such burning literally borders public school properties, shopping centers and suburban communities and occurs within 10 miles of the urban centers of Tallahassee, Florida and Thomasville, Georgia. The take home message is where prescribed fire is recognized as important by the public, made a priority by public agencies, passed across generations and maintained at natural fire frequencies, it can still be used on private lands in modern landscapes to maintain fire-adapted ecosystems and meet multiple land management objectives.

Ecological Reality: Fire Frequency in Southern Pine Forests

Fire Frequency

Most ecosystems in the southern United States historically burned at relatively high frequencies, at least once every 12 years. In the southern Coastal Plain, upland pine ecosystems were adapted to a fire return interval of about every 1 to 4 years (Frost 1998). The same also is true for the interior highlands of Arkansas and Oklahoma (Masters et al. 1995). The few examples of long-term research (i. e., more than 10 years) on fire frequency in southern pine systems demonstrate that fire frequency is one of the most important variables determining the persistence of plant communities (Waldrop et al. 1992, Masters et al. 1993, Glitzenstein et al. 2003). In upland pine systems, research from Florida, Oklahoma and South Carolina clearly shows that fire return intervals of more than 3 years result in shrub and hardwood midstory increases (Waldrop et al. 1992, Masters et al. 1993, 1996, 1997; Hermann 1998; Sparks et al. 1998, 1999; Glitzenstein et al. 2003). Annual or biennial fires are needed to prevent hardwood shrub and hardwood midstory development (Masters et al. 1993, 1997, Glitzenstein et al. 2003). Changes in habitat in response to greater than 3-year fire return intervals can quickly result in a decline of important wildlife species, such as northern bobwhites, red-cockaded woodpeckers *(Picoides borealis)*, a host of neotropical migrant songbirds and small mammals (Engstrom et al. 1984; Wilson et al. 1995; Masters et al. 1989, 1995, 1998, 2002; Cram et al. 2002) as well as rare plant communities (Sparks et al. 1998, Gray et al. 2003).

Restoration

Following years of mismanagement with low fire frequency, recovery of pine systems may not be as simple as reintroducing fire (Masters et al. 1995, Provencher et al. 2001, Drewa et al. 2002). Hardwood and pine midstory development creates conditions suitable for shade-tolerant species at the expense of more pyrogenic shade-intolerant species. Even with mechanical reduction of midstory hardwoods, herbaceous plant communities may not recover quickly and may require management other than fire to reduce hardwood resprouting or shrub development (Sparks et al. 1999, Provencher et al. 2001, Brockway et al. 2003).

Season of Fire

The relevance of season of fire to maintenance of plant communities has become an important conservation issue (Komarek 1964; Platt et al. 1988; Glitzenstein et al. 1995; Sparks et al. 1998, 2002). Season of fire is likely more important in some plant communities than others (Platt et al. 1988, Heirs et al. 2000). However, current knowledge suggests that a conservative management approach is to apply fire at the time of year necessary to ensure fire frequency and acreage goals are met. For instance, mandating that a proportion of fires must occur within certain months to mimic natural processes (e. g., lightning season fire) can become a red herring to maintenance of biodiversity if the result is reduced fire frequency and increased hardwood encroachment. This has been the experience on Tall Timbers Research Station and certain public lands, where adjustments to fire season were required to achieve burn objectives. The ability to conduct late growing season burns can be constrained by the permitting process, by weather, by fuel type and conditions, by budgets, and by safety concerns (Sparks et al. 2002), which may result in extended fire intervals and at least temporary habitat degradation.

Obstacles to Frequent Use of Fire

To better understand the limits on use of prescribed fire on public and private land, we interviewed 15 state forestry professionals and wildlife biologists from 13 southeastern states. Interviewees were asked a series of 15 questions about the importance of fire, the actual use of fire on public and private lands, and their perceptions of obstacles to the use of fire. Also, they were provided an opportunity to comment on these topics in general. The survey was not random; rather, it was aimed at individuals who have significant experience in dealing with prescribed fire issues in their state. We also summarized regulations relating to prescribed burning by state to identify regional regulatory trends and to determine how different states are balancing fire suppression and fire promotion.

Results of Surveys

Importance of fire. Professionals we interviewed shared the view that fire is an important process for sustaining southern upland ecosystems. All respondents also thought more prescribed fire is needed to achieve that goal, as well as other goals, such as wildfire protection. All respondents listed species or ecosystems harmed by low fire frequency in their states. These common values were not surprising given the steady increase in fire ecology research during the past four decades, including 21 Tall Timbers Fire Ecology Conferences (Pyne 1982, Stevenson 1998).

The perception of how well fire is being used on federally-managed, state-managed and private lands to sustain natural communities varied predictably among respondents. Respondents ranked federal lands as most likely to meet the needs of these ecosystems, followed by state and private lands, in that order. Respondents' estimates of the amount of private lands being burned frequently enough to maintain fire-adapted plant communities ranged from less than one to five percent. On average, respondents thought 35 percent (range one to 90 percent) of state-owned lands received adequate burning for this purpose. However, when state forestry professionals were asked to rank the primary objective for burns conducted on state-owned lands, they ranked fuel reduction as number one, before timber management, game management or biodiversity management, in 10 of 13 cases. Burning for biodiversity was ranked either third or fourth (last) in 10 of 14 cases. This result coincides with previous literature that most prescribed burns on state-owned and private land in the Southeast are for fuel reduction, which suggests that most land is not being burned frequently enough to meet biodiversity goals (Brennan et al. 1998). Based on our conversations with wildlife and forestry professionals throughout the South, fire frequency goals set for state-owned lands are not consistently being met and most privately-owned forest lands are rarely being burned.

Attempts to promote prescribed burning—Legislation and liability. In 1990, Florida pioneered a legislative approach to promoting prescribed fire that would be imitated by most other southern states during the subsequent decade. The development of a set of laws that would be referred to as the Florida Prescribed Fire Act (Forest Service 590.125) was largely in response to pressure from regional prescribed fire councils. These councils are nongovernment task forces represented by private, nonprofit, and government-employed individuals who share a common interest in preserving the right to use prescribed fire, whether for hazard-reduction, wildlife management, timber management or biodiversity (Stevenson 1998). Their approach to protecting the future of prescribed burning was to: (1) increase public confidence in the practice of burning by specifying and by raising standards of safety and training required of those applying fire and (2) limiting the liability of prescribed fire practitioners. Thus, the legislature established a training program for certified burners who would be protected from liability related to smoke or fire escape unless negligence was proven (recently legal protection was increased to proving gross negligence [Brenner and Wade 2003]). Responsible use of fire was defined, in part, by possession of a written burn plan (prescription) to guide the decision of whether or not to burn and notification of consent from the state forestry agency.

Similar legislation subsequently has been established in Georgia (1992), Mississippi (1992), Louisiana (1993), South Carolina (1994), Alabama (1995), Virginia (1997), North Carolina (1999) and Texas (1999) and is pending votes from the legislature in Oklahoma. Florida and Georgia are unique in that they provide limited liability for noncertified as well as certified burners. Texas is unique in its requirement for certified burners to be insured. Other specific variations in legislation among the states are presented in greater detail elsewhere (Hauenstein and Siegel 1981, Haines and Cleaves 1999) and are partly summarized and updated in Table 1. Given the current trend, it is likely that other southern states will adopt similar legislation in the future.

It is currently not clear whether or not such prescribed fire acts have influenced the use of prescribed fire in the South. In general, the new laws have

Table 1. Prescribed burning permitting regulations and other services offered by state forestry commissions[a] from 13 southeastern states

Issue or Question	AL	AR	FL	GA	KY	LA	MS	NC	OK	SC	TN	TX	VA
Active prescribed fire council?	Y	N	Y	Y	N	N	N	N	N	Y	N	N	N
Prescription burn training?	Y	N	Y	Y	N	Y	Y	Y	Y	Y	N	Y	Y
Prescription burner certification?	Y	N	Y	Y	N	Y	Y	Y	Y	Y	N	Y	Y
Liability limited by law for certified burners?	Y	NA	Y	Y	NA	Y	Y	Y	P	Y	NA	Y	Y
Liability limited by law for noncertified burners?	N	N	Y	Y	N	N	N	N	N	N	NA	N	N
Burn permitting system (verbal or written)?	Y	N	Y	Y	N	N	Y	Y	Y	Y	Y	N	Y
Permits issued to noncertified burners?	Y	NA	Y	Y	NA	NA	Y	Y	Y	Y	Y	N	Y
Permits more flexible for certified burners?	N	NA	Y	N	NA	N	N	N	N	N	NA	Y	Y
Notification of agency required?	Y	N	Y	Y	N	Y	Y	Y	Y	Y	Y	Y	Y
Notification of neighbors required?	N	N	N	N	N	N	N	N	Y	N	N	N	Y
Fixed seasonal burn bans/restrictions?	N	N	N	N	Y	N	N	N	N	N	Y	N	Y
Fixed hour of day burn restrictions?	?	N	Y	N	Y	N	N	N	N	N	Y	N	Y
Agency retains right to declare burn bans?	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
Agency plowing service?	Y	N	N	N	N	Y	Y	Y	N	Y	N	N	N
Agency burning service?	Y	N	N	N	N	Y	N	N	N	Y	N	N	N
Demand for services met?	N	NA	NA	NA	NA	N	N	?	NA	N	NA	NA	NA
State agency incentive programs for prescription fire?	N	N	?	N	N	N	N	Y	?	N	N	N	N
Federal money used to promote/subsidize prescription fire?	?	?	Y	Y	N	?	Y	?	Y	?	N	Y	?

[a]Agencies: Alabama Forestry Commission, Arkansas Forestry Commission, Florida Division of Forestry, Georgia Forestry Commission, Kentucky Division of Forestry, Louisiana Department of Agriculture and Forestry, Mississippi Forestry Commission, North Carolina Division of Forest Resources, Oklahoma Department of Forest Services, South Carolina Forestry Commission, Tennessee Division of Forestry, Texas Forest Service, Virginia Department of Forestry.

not yet been tested in the courts (Haines and Cleaves 1999) and, thus, lack the power of precedent. According to our communications with state officials, it is the fear of liability (Table 2), not the lack of education, that remains the most significant obstacle to the application of prescribed fire by private land-owners (see also Cleaves and Haines 1995, Eshee 1995, Monroe 2002). Companies that offer insurance to prescribed fire contractors also seem unconvinced that burners are protected by these laws. Although the service of such private contractors is in the highest demand ever throughout the Southeast, contractors are few in number in the states where they have not disappeared entirely because of prohibitive insurance costs. Texas has taken the novel approach of limiting the sum for which a prescribed fire practitioner may be sued, provided they are certified. Although the effects of the Texas legislation are similarly unknown, such laws might eventually provide needed relief from insurance premiums if adopted by other states.

Table 2. Ranking of obstacles to burning on private lands by 15 state forestry and wildlife professionals from 13 southeastern states. Respondents were asked to rank potential obstacles to implementing prescribed fire on private lands as rarely important (1), sometimes important (2) and always important (3).

Potential obstacle	Mean score	Standard deviation	Overall ranking
Liability concerns	2.8	0.38	1
Limited availability of contractors	2.7	0.63	2
Development/human encroachment	2.5	0.66	3
Lack of interest in using fire	2.2	0.60	4
Knowledge of how to burn	1.9	0.49	5
Limited cost-share incentives	1.7	0.75	6
Knowledge of importance of fire	1.6	0.51	7
Permitting	1.5	0.66	8

According to our personal communications with state forestry agencies, liability is a less serious obstacle to burning for the agencies than it is for private landowners and contractors. In states where prescribed burning services are offered by the agency (at $8 to $15 per acre), it was reported that significantly more acres are burned by the agency than by private landowners. However, such services are offered only in Alabama, Arkansas, Louisiana, South Carolina and Tennessee. In some of these states, it was reported that the demand for burning is not being met, primarily because of increasing shortages in budgets and staffing. Also, tree-planting and fire suppression are mandated as the higher priorities of the agencies, and resource-consuming wildfires tend to occur when prescribed fires would most effectively be applied.

The support of government funding aimed at private landowners for the promotion and application of prescribed fire is generally indirect and not specifically required under the provisions of government programs. Federal cost-share programs, which potentially provide incentives for prescribed burning, require management plans to be developed and adhered to by the landowner under the supervision of a government agency representative. Specifically, management plans funded by the Wetlands Reserve Program (WRP), the Environmental Quality Incentives Program (EQIP) and the Wildlife Habitat Incentives Program (WHIP) are overseen by the U. S. National Resource Conservation Service (NRCS), plans for the Forest Land Enhancement Program (FLEP; no funding after 2004) are supervised by state forestry agencies, and the Conservation Reserve Program (CRP) is overseen by the U. S. Farm Service Administration. Thus, the degree to which these programs benefit prescribed fire depends heavily on the particular goals of landowners and the guidance provided by individuals representing the overseeing agencies. Specifications of the many individual management plans funded under these programs are difficult to access and, thus, can not be evaluated here for their relevance to prescribed fire. Generally, the programs offer a choice of management goals, which may include timber production, wildlife enhancement, soil conservation and other goals, which may or may not require prescribed fire. Thus, under current programs, attempts to direct federal funding toward prescribed burning should focus on persuading both landowners and program representatives of the critical role of fire in managing forests and rangelands.

Several southern states, including Alabama, Louisiana, Mississippi, North Carolina, South Carolina, Texas and Virginia, also provide economic incentive programs that can include prescribed burning as part of management plans authorized for assistance (Granskog et al. 2002). However, each of these programs focus on increasing forest productivity as the primary goal. In general, the goal of maximum short-term production of pine timber on uplands in the Southeast (usually in row-planted pine plantations) is not compatible with the goal of maintaining or restoring native plant communities (Hedman et al. 2000, Dagley et al. 2002) or of improving habitat for indigenous wildlife species (Lancia et al. 1989, Engstrom and Palmer 2003). Thus, the benefits of prescribed fire as authorized or required under these programs are likely limited to wildfire hazard reduction and site preparation.

An additional obstacle to burning is the establishment of county and municipal laws that add burn restrictions to those specified in state legislation. According to our communications, these restrictions can be prohibitive in relatively populated counties, whereas they are rarely an obstacle in rural areas. For example, 19 counties composing and surrounding the greater Atlanta area in Georgia have fixed seasonal burn bans, as do 18 counties in North Carolina in similarly populated districts. The number of counties falling under such restrictions is expected to grow as populations continue to rapidly increase around most urban centers throughout the South.

Obtaining permits. Interestingly, obtaining a permit to conduct a burn was not considered problematic. However, there were some differences in opinion between wildlife biologists and state forestry professionals on this topic. Biologists consistently ranked obtaining a burning permit as an important obstacle, whereas state forestry professionals consistently ranked permitting as not an obstacle. This difference in response seemed to be based on a biologist's interest in conducting burns based on site-specific burning conditions, whereas state foresters relied on information at a much broader scale in the permitting process for determining safe burning conditions. Respondents commented on cases where habitats on management areas were negatively affected by low fire frequency resulting from difficulties obtaining burning permits or different priorities among state agencies. Given the importance of frequent fire on maintenance of habitats and the high skill level of fire practitioners within state agencies, different agencies within states need to align policies to maximize the safe use of fire where possible on public lands.

Conclusions

Our results suggest that both wildlife and forestry agencies and professionals share common ideas about the importance of fire for maintaining southern pine ecosystems, and they agree that increasing the use of prescribed fire is needed. However, our results also indicate that fire frequency on most public lands is likely too low to sustain fire-adapted plant and wildlife communities; although, in some states, the trend is toward increasing fire frequency on public lands. States throughout the South have an opportunity to demonstrate the benefits of frequent prescribed fire to important fire-adapted plant and wildlife species. Given the significant declines in important fire-adapted wildlife species, native plant communities and needs for fuels management, we hope that changes in public perceptions will be sufficient to restore fire where practical on the landscape (Cole 1998).

On private lands, liability concerns of landowners and fire practitioners will likely continue to thwart efforts to increase prescribed fire, especially where the fire culture has faded. The need for cost-share, and more importantly incentive payments, to encourage adoption of prescribed fire is evident. In some states, the Safe Harbor Program, aimed a preserving habitats for red-cockaded woodpeckers, is a model of how funding can be used to promote burning in order to sustain natural communities and species of concern.

Without new initiatives that promote fire, reduce liability risk, and provide direct cost-share and incentive payments to landowners, we agree with Brennan et al. (1998) that most of the Southeast will continue to be essentially fire-excluded, with predictable effects on wildlife. The only notable exceptions are where a fire culture has persisted for a long period of time in small pockets of the Southeast, like the Red Hills Region. As states address long-term habitat restoration programs for fire-adapted species, such as northern bobwhites (e. g., Northern Bobwhite Conservation Initiative), efforts must be made to locate other areas where use of frequent fire is still obtainable.

Reference List

Brennan, J. M. Lee, and R. S. Fuller. 2000. Long-term trends of northern bobwhite populations and hunting success of private shooting plantations in northern Florida and southern Georgia. *National Quail Symposium.* 4:75–7.

Brennan, L. A., R. T. Engstrom, W. E. Palmer, S. M. Herman, G. A. Hurst, L. W. Burger, and C. L. Hardy. 1998. Whither wildlife without fire? *North American Wildlife and Natural Resources Conference.* 63:402–14.

Brenner, J., and D. Wade. 2003. Florida's revised prescribed fire law: Protection for responsible burners. *Tall Timbers Research Station Miscellaneous Publication.* 1:132–6.

Brockway, D., K. W. Outcalt, and B. L. Estes. 2003. Restoring southern pine forest ecosystems: Plant community response to mechanical midstory history reduction and prescribed fire on sandhills at Fort Benning, Georgia. *Ecological Society of America.* 88:46.

Church, K. E., J. R. Sauer, and S. Droege. 1993. Population trends of quails of North America. *National Quail Symposium*. 3:44–54.

Cleaves, D. A., and T. K. Haines. 1995. Regulation and liability risk: Influences on the practice and the price tag of prescribed burning. In *Proceedings of the environmental regulation and prescribed fire conference: Legal and social challenges*, ed. D. C. Bryan, 165–84. Tampa, Florida: Florida State University, Tallahassee.

Cole, F. T. 1998. Are political and philosophical issues limiting use of prescribed fire? *Tall Timbers Fire Ecology Conference*. 20:432–48.

Cram, D. S., R. E. Masters, F. S. Guthery, D. M. Engle, and W. G. Montague. 2002. Northern Bobwhite Population and Habitat Response to Pine-grassland Restoration. *Journal of Wildlife Management*. 66:1,031–9.

Dagley, C. M, T. B. Harrington, and M. B. Edwards. 2002. Understory restoration in longleaf pine plantations: Overstory effects of competition and needle fall. In *Proceedings of the eleventh biennial southern silvicultural research conference, GTR SRS-48*, ed. K. W. Outcalt, 487–9. Ashville, North Carolina: U. S. Department of Agriculture.

Drewa, P. B., W. J. Platt, and E. Barry Moser. 2002. Fire effects on resprouting of shrubs in headwaters of southeastern Longleaf pine savannas. *Ecology*. 82:757–67.

Engstrom, R. T., R. L. Crawford, and W. W. Baker. 1984. Breeding bird populations in relation to changing forest structure following fire exclusion: A 15 year study. *Wilson Bulletin*. 96:437–50.

Engstrom, R. T., and W. E. Palmer. 2003. *Two species in one ecosystem: management of northern bobwhite and red-cockaded woodpecker in the Red Hills, USDA, Forest Service PSW-GTR-191*. Albany, California: Pacific Southwest Research Station.

Eshee, W. D. 1995. Legal implications of using prescribed fire. In *Proceedings of the environmental regulation and prescribed fire conference: Legal and social challenges*, ed. D. C. Bryan, 126–30. Tampa, Florida: Florida State University, Tallahassee.

Frost, C. C. 1998. Presettlement fire frequency regimes of the United States: A first approximation. *Tall Timbers Fire Ecology Conference*. 20:70–81.

Glitzenstein, J. S., D. R. Steng, and D. D. Wade. 2003. Fire frequency effects on longleaf pine (*Pinus palustris)* vegetation in South Carolina and northeast Florida, USA. *Natural Areas Journal*. 23:22–37.

Glitzenstein, J. S., W. J. Platt, and D. R. Streng. 1995. Effects of fire regime and habitat on tree dynamics in north Florida longleaf pine savannahs. *Ecological Monographs*. 65:441–76.

Granskog, J. E., T. Haines, J. L. Greene, B. A. Doherty, S. Bick, H. L. Haney, Jr., S. O. Moffat, J. Speir, and J. J. Spink. 2002. Policies, regulations, and laws. In *Southern forest resource assessment, USDA GTR SRS–53*, eds. O. N. Wear, and J. G. Greis, 189–223.

Gray, J. B., T. R. Wentworth, and C. Brownie. 2003. Extinction, colonization, and persistence of rare vascular flora in the longleaf pine-wiregrass ecosystem: Responses of fire frequency and population size. *Natural Areas Journal*. 23:210–9.

Haines, T. K., and D. A. Cleaves. 1999. The legal environment for forestry and prescribed burning in the South: Regulatory programs and voluntary guidelines. *Southern Journal of Applied Forestry*. 23:170–4.

Hauenstein, E. B., and W. C. Siegel. 1981. Air quality laws in the southern states: Effects on prescribed burning. *Southern Journal of Applied Forestry*. 5:132–45.

Hedman, C. W., S. L. Grace, and S. E. King. 2000. Vegetation composition and structure of southern coastal plain pine forests: An ecological comparison. *Forest Ecology and Management*. 134:233–47.

Heirs, J. K., R. Wyatt, and R. J. Mitchell. 2000. The effects of fire regime on legume reproduction in longleaf pine savannas: Is a season selective? *Oecologia*. 125:521–30.

Hermann. S. M. 1998. Thirty-five year results from the Stoddard fire plots: A study of fire frequency in the Red Hills of North Florida and South Georgia. *Tall Timbers Fire Ecology Conference*. 20:31.

Komarek E. V. Sr. 1964. The natural ecology of lightning. *Tall Timbers Fire Ecology Conference*. 3:139–83.

Lancia, R. A., J. P. Roise, D. A. Adams, and M. R. Lennartz. 1989. Opportunity costs of red-cockaded woodpecker foraging habitat. *Southern Journal of Applied Forestry*. 13:81–5.

Masters, R. E., R. L. Lochmiller, and D. M. Engle. 1993. Effects of timber harvest and periodic fire on white-tailed deer forage production. *Wildlife Society Bulletin*. 21:401–11.

Masters, R. L., J. E. Skeen, and J. A. Garner. 1989. Red-cockaded woodpecker in Oklahoma: An update of Wood's 1974–77 Study. *Proceedings of the Oklahoma Academy of Sciences*. 69:27–31.

Masters, R. L., J. E. Skeen, and J. Whitehead. 1995. Preliminary fire history of McCurtain County Wilderness Area and implications for red-cockaded woodpecker management. In *Red-cockaded woodpecker: Species recovery, ecology and management*, eds. D. L. Kulhavy, R. G. Hooper, and R. Costa, 290–302. Nacogdoches, Texas: Center for Applied Studies, Stephen F. Austin University.

Masters, R. L., K. Robertson, B. Palmer, J. Cox, K. McGorty, L. Green, and C. Ambrose. 2003. *Red Hills forest stewardship guide, miscellaneous publication no. 12.* Tallahassee, Florida: Tall Timbers Research Station.

Masters, R., L. Lochmiller, S. T. McMurry, and G. A. Bukenhofer. 1998. Small mammal response to pine-grassland restoration for red-cockaded woodpeckers. *Wildlife Society Bulletin.* 28:148–58.

Masters, R. L., R. E., C. W. Wilson, D. S. Cram, G. A. Bukenhofer, and R. L. Lochmiller. 2002. Influence of ecosystem restoration for red-cockaded woodpeckers on breeding bird and small mammal communities. In *The role of fire in non-game wildlife management and community restoration: traditional uses and new directions: proceedings of a special workshop: General technical report NE–288*, eds. W. M. Ford, K. R. Russell, and C. E. Moorman, 73–90. Nashville, Tennessee: U. S. Department of Agruculture, Forest Service, Northeast Research Station.

Masters, R. L., R. E., C. W. Wilson, G. A. Bukenhofer, and M. E. Payton. 1996. Effects of pine-grassland restoration for red-cockaded woodpeckers on white-tailed deer forage production. *Wildlife Society Bulletin.* 24:77–84.

Masters, R. L., W. D. Warde, and R. L. Lochmiller. 1997. Herbivore response to alternative forest management practices. *Southeastern Association of Fish and Wildlife Agencies Proceedings.* 51:225–37.

Monroe, M. C. 2002. Fire. In *Human influences on forest ecosystems: The southern wildland-urban interface assessment. USDA, Forest Service GTR SRS-55*, eds. E. A. Macie, and L.A. Hermansen, 133–50. Asheville, North Carolina: Southern Research Station.

Platt, W. J., G. W. Evans, and M. M. Davis. 1988. Effects of fire season on flowering forbs and shrubs in longleaf pine forests. *Oecologica.* 76:353–63.

Provencher, L., B. J. Herring, D. R. Gordon, H. L. Rodgers, K. E. M. Galley, and L. A. Brennan. 2001. Effects of hardwood reduction techniques on longleaf pine sandhill vegetation in northwest Florida. *Restoration Ecology*. 9:13–27.

Pyne, S. J. 1982. *Fire in America: A cultural history of wildland and rural fire*. Princeton New Jersey: Princeton University Press.

Robbins, L. E., and R. L. Myers. 1992. *Seasonal effects of prescribed burning in Florida: A review, miscellaneous publication no. 8*. Tallahassee Florida: Tall Timbers Research Station.

Sheffield, R. M., and J. G. Dickson. 1998. The South's forestland—On the hot seat to provide more. *North American Wildlife and Natural Resource Conference*. 63:316–31.

Sparks, J. C., and R. E. Masters. 1996. Fire seasonality effects on vegetation in mid-, tall-, and southeastern pine-grassland communities: A review. *North American Wildlife and Natural Resources Conference*. 61:230–9.

Sparks, J. C., R. E. Masters, D. M. Engle, and G. A. Bukenhofer. 2002. Season of burn influences fire behavior and fuel consumption in restored shortleaf pine-grassland communities. *Restoration Ecology*. 10:714–22.

Sparks, J. C., R. E. Masters, D. M. Engle, M. E. Payton, and G. A. Bukenhofer. 1999. Influence of fire season and fire behavior on woody plants in red-cockaded woodpecker clusters. *Wildlife Society Bulletin*. 29:88–96.

Sparks, J. C., R. E. Masters, D. M. Engle, M. W. Palmer, and G. A. Bukenhofer. 1998. Effects of growing-season and dormant-season prescribed fire on herbaceous vegetation in restored pine-grassland communities. *Journal of Vegetation Science*. 9:133–42.

Stevenson, J. A. 1998. Evolution of fire management in Florida's state parks. *Tall Timbers Fire Ecology Conference*. 20:99–101.

Stoddard. H. L. 1931. T*he bobwhite quail: Its habits, preservation and increase*. New York, New York: Charles Scribner's Sons.

Waldrop, T. A., D. L. White, and S. M. Jones. 1992. Fire regimes for pine-grassland communities in the Southeastern United States. *Forest Ecology and Management*. 47:195–210.

Wilson, C. W., R. E. Masters, and G. A. Bukenhofer. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. *Journal of Wildlife Management*. 59:56–67.

Prescribed Burns and Large Carnivores in South Florida: Can Fire Be Too Much of a Good Thing?

David S. Maehr
University of Kentucky, Department of Forestry
Lexington

Jeffery L. Larkin
University of Kentucky, Department of Forestry
Lexington

Unlike the ancient influence of naturally started fire on many ecosystems and as a driver of evolutionary adaptations in individual species and entire biotic communities, the effect of modern, human-caused fire on the landscape in North America is relatively novel. Certainly, prehistoric humans might be considered a natural fire-wielding component of this history, but even our primitive ancestors had little time to affect the evolutionary trajectories of species and their communities; our species' presence in North America is miniscule compared to the ancient communities that existed before Clovis people arrived from Beringia. In southern Florida, where humans have been established since about 6000 BC (Carr and Beriault 1984), fire is one of the most important ecological forces on the biota and the landscape (Wade et al. 1980). As was observed by Hofstetter (1984:465), "Fire existed as a natural environmental factor in southern Florida before man arrived. Essentially all natural fires here are caused by lightning. Lightning is most common in association with convective storms in the rainy season, May to October, and is not common the rest of the year. Thus natural fires are a wet-season phenomenon. At that time, the soil is usually moist, the vegetation growing and turgid, and the air humid. Thus, the natural fires do not normally become roaring holocausts in the pinelands or peat-consuming ground fires in the wetlands." It was to these conditions, periodically adjusted by southern oscillation influences (Beckage et al. 2003), that Florida's large carnivores adapted. The predictability of summer fires combined with comparatively fire-free winter droughts allowed these species to safely give birth to helpless offspring in some of the most flammable plant communities on earth. Both the black bear (*Ursus americanus floridanus*) and the Florida panther (*Puma concolor coryi*) continue to prefer the dense thickets of saw palmetto

(*Serenoa repens*) for everything from winter hibernacula and feeding areas to daytime bed sites and natal dens (Maehr 1997), but modern land management that includes artificially ignited fires outside of the lightning season has the potential to change the relations between these species and the landscape.

The traditions of southern winter burning have their roots in pest control, range management (Hartman 1949, Burger 1973) and early game management (Stoddard 1935). Because fire returns nutrients to the soil and because it retards succession toward communities dominated by hardwoods, it became a popular tool throughout the region for range management (Stoddart et al. 1975, Hofstetter 1984) and for the perpetuation of edge-loving wildlife species (Peek 1986). South Florida was no different—a region that contained a number of fire-adapted biotic communities typical of the Everglades and Big Cypress Swamp regions.

Much has changed in southern Florida in the last 8,000 years. These changes have been most dramatic in the last few decades, and they include alterations to the fire regimes with which many organisms evolved. Egler (Hofstetter 1984:466) believed, "that the herbaceous everglade and surrounding pinelands were born in fires; that they can survive only with fires; that they are dying today because of fires." Robertson (1953:8) suggested that fire ecology had been so disrupted that, "ill-conceived land use practices of the past forty years ha[ve] brought the entire region to the point where its survival in any condition resembling the original is seriously in question." Today, most fires are set by humans outside of the growing season (Duever et al. 1979), and occur more frequently than before south Florida was settled by humans (Robertson 1962). The implications of human-caused fires (or their exclusion) in the loss of global biodiversity (Robertson 1953, Hansen et al. 1991, Bunting 1996, Harris et al. 1996, Quintana-Ascensio and Menges 1996, Myers 1997, Laurance and Williamson 2001, Newmark 2002) lead Terborgh (2002:34) to advise that one of the, "reforms . . . urgently needed to prevent the further degradation of public lands," was the, "restoration of semi-natural fire regimes."

Although Florida natural resource agencies are increasingly cognizant of the ecological role of lightning fires (e. g., Florida Department of Environmental Protection 2002), the tradition to burn frequently during the winter persists. During winter, fuels tend to burn more uniformly, weather conditions are more predictable and less damage is done to trees because air temperatures are cooler. The downside to winter fires is that drier conditions increase the possibility for a prescribed burn to spread into sensitive areas or to escape containment. Further, many plants have likely adapted to smaller, patchier and potentially hotter summer fires. For example, wire grass (*Aristida stricta*) and beard grasses (*Andropogon* spp.) often fail to flower unless they are burned when lightning is most apt to ignite a fire (Abrahamson 1984, Platt et al. 1988). Although Abrahamson and Hartnett (1990) suggested that prescribed fires do not dramatically alter the plant species composition of pine flatwoods, their effects on terrestrial vertebrates is less clear. There is little doubt that edge-loving game species, such as northern bobwhite (*Colinus virginianus*) and white-tailed deer (*Odocoileus virginianus*) can directly benefit from frequent winter fires in pine-dominated ecosystems, but management that increases food availability for the former might also be offset by declines in nesting habitat. Similarly, although a prescribed fire rotation of less than or equal to 4 years has been suggested for enhancing feeding opportunities for the Florida panther by improving nutritional conditions for deer, more open conditions might reduce the availability and quality of natal den cover for this endangered species. While nutritional benefits have often been touted as a benefit of fire, "when the effort is made to demonstrate it, the generality very often gives way to more intricate relationships" (Peek 1986:150). This paper examines the potential for conflict between prescribed fire practices in southern Florida and then two imperiled large carnivores. It begins by reviewing the role of Paleo-Americans in the genesis of terrestrial fires in southern Florida, then it examines the autecology of saw palmetto—an important cover plant that is used by both large carnivores. Then, we consider the consequences of modern fire management on the black bear, Florida panther and the landscape they inhabit.

The Pre-Columbian Fire Regime

Although the precise temporal and geographic distribution of fire in ancient southern Florida will never be known, records of regional lightning occurrence, the sizes of naturally occurring fires and suggestions by those familiar with fire ecology provide a basic framework to begin to understand fire in this region. Most pre-Columbian fires in southern Florida pine forests with saw palmetto-dominated understories were likely small. First, saw palmetto usually grows in distinct, restricted patches of a few acres or less. Second, because natural fires were caused primarily by summer lightning, most burns were restricted by standing surface waters, afternoon rains and high humidity (Duever et al. 1979, Hofstetter 1984). Third, although the Big Cypress Swamp experienced many fires annually, only five percent of recorded fires were caused by lightning; the remainder were caused by humans (Duever et al. 1979). Although humans used fires extensively elsewhere in the southeastern Coastal Plain (Hann 1991; Milanich 1994, 1995), the pattern was likely different in south Florida. Archaeological investigations demonstrate that the native Calusa and Tequesta cultures in southern Florida did not use fire for agriculture—indeed, farming was disdained (Hann 1991, Marquart 1992). Instead, abundant marine life that included shellfish, manatee (*Trichechus manatus*) and Atlantic right whale (*Eubalaena glacialis*), was supplemented with uncultivated terrestrial foods, including fruits from saw palmetto and cocoplum (*Chrysobalanus icaco*) (Larson 1980). Such bounty obviated the need for slash and burn agriculture. Pre-Columbian settlements in southern Florida tended to be coastal and relatively permanent (Larson 1980); their inhabitants were "fisher-hunter-gatherers" (Hann 1981:329). Thus, if interior forests and marshes burned primarily as the result of summer lightning, the evolutionary relations between nonhuman-induced fire and many plant and animal species was uninterrupted by pre-Columbian southern Florida humans and their relatively short history in North America.

Carnivore-Saw Palmetto Relations

The highly flammable saw palmetto is abundant (Hilmon 1968) and widespread (Little 1978) in the southeastern coastal plain. It is the most characteristic shrub in southern Florida pinelands (Tomlinson 1980), and it is a very important fuel for fires (Davison and Bratton 1988). Its fruit and apical meristems are important black bear foods throughout the year in Florida (Maehr 1997). Frequent fires reduce carbohydrate stores in its prostrate stems (Hough 1968), reduce the incidence of fruiting and flowering (Hilmon 1968), and may eliminate it from a given community (Langdon 1981). Although this species provides an extremely popular and effective herbal remedy for enlarged prostate and other human medical conditions (Maehr and Layne 1997), its ecological value—including a natural nursery shade for longleaf pine (*Pinus palustris*)(Allen 1956)—has escaped similar widespread recognition. Several studies of saw palmetto indicate that frequent fires change the species' fruiting phenology and structural characteristics that are important to black bear and panther. Although saw palmetto may flower profusely the year following fires

(Abrahamson 1999), fruit production becomes irregular with repeated burning (Gholz et al. 1999). Whereas palmetto responds with new growth immediately after a fire (Davison and Bratton 1988), substantial fruit crops may not be produced for up to 10 years following the last burn (Hilmon 1968, Carrington and Mullahey 1997). Further, while it may regain 80 percent of its crown coverage in the year following a fire, its relatively slow stem growth (Abrahamson 1995, Kennard et al. 2003) means that the dense horizontal cover (Figure 1) that is preferred by the Florida panther for natal dens (Maehr et al. 1990) and frequently used by the black bear may not return for many years (Sackett 1975).

Figure 1. Inside view of a typical Florida panther den site (with a 14-day-old kitten) in dense saw palmetto. (photo by D. Maehr)

If managers of public lands in southern Florida focus strictly on panther nutritional needs as their justification for short-rotation fire, other equally important carnivore habitat attributes may be neglected. Whereas frequently burned pine ecosystems may benefit the panther through locally concentrated deer, an entire upland landscape of recently burned pine and saw palmetto habitat would reduce a very important food resource for the black bear (Maehr et al. 2001), would eliminate the dense palmetto thickets that panthers prefer for natal dens and day beds (Maehr et al. 1990) (Figure 2), and would increase the probability of endangering neonates if fires are set during winter (Land 1994, Stratman 1998)(Figure 3). An extensive history of saw palmetto autecology

Figure 2. Outside view of a Florida panther den site located in a thicket of dense saw palmetto. (photo by D. Maehr)

Figure 3. A saw palmetto thicket one day following a winter prescribed fire in southwest Florida. This site was previously suitable as a large carnivore den location offering the conditions pictured in Figures 1 and 2. (photo by D. Maehr)

(Hilmon 1968; Davison and Bratton 1988; Abrahamson 1995, 1999; Gholz et al. 1999) and recent studies on large carnivore habitat relations (Maehr et al. 1990, Maehr 1997, Maehr et al. 2001) make a compelling argument for considering longer rotation burns during the growing season.

Although southern Florida is one of the most lightning-prone regions of the world, the probability that a detectable fire results from a strike is low. Further, the size of such fires under pre-Columbian conditions would be relatively small if they started after the onset of summer rains and concomitant increases in water level. Although the factors that determine the timing and extent of natural fires in southern Florida forests are complex, it is likely that many areas of fire-prone vegetation escape ignition for much longer than 4 years. Inasmuch as southern Florida's large carnivores depend heavily on uplands dominated by saw palmetto, current management plans for most lands inhabited by the bear and panther may not sufficiently consider all of their habitat needs. Although sufficient nutrition in the form of deer flesh may result from a 4-year fire rotation, it is unknown to what extent female bears and panthers must expend additional energy searching for suitable natal dens or how often litters might be lost due to direct mortality caused by winter burns. In addition, the stresses inflicted on saw palmetto plants by overly frequent fires may retard the development of viable reproductive tissues and reduce their capacity to support the black bear.

Discussion

We found no empirical evidence of a natural lightning ignition rate that approaches what is used on the Florida Panther National Wildlife Refuge and other public lands in southern Florida. In addition, published estimates (Duever et al. 1979) and anecdotal observations (J. Durrwachter, personal communication 2002) suggest that lightning-caused fires are much less common than those started by people. Clearly, managers of lands that are intended to provide refuge for imperiled species, such as the black bear and the Florida panther, would be justified in considering longer prescribed fire rotations in upland pine habitats. Ignition rates, if maintained on a short rotation, would likely reduce important food and cover characteristics for these species. The lower ignition rates may come closer to simulating the pre-Columbian fire regime and would leave unburned saw palmetto refugia that are preferred by panther and bear for cover and food.

The extensive fires in Yellowstone National Park during 1988 reduced grizzly bear (*Ursus arctos*) carrying capacity by eliminating important foods, such as whitebark pine (*Pinus albicaulis*), a species that may take more than 40 years to recover (Craighead et al. 1995). This largely natural conflagration is distinguished from the situation in southern Florida because Yellowstone National

Park's inhabited carnivore range is connected to other conservation lands in the region and because important bear foods may take several decades to recover. Southern Florida is further distinguished from Yellowstone National Park inasmuch as decades of management and the effects of the most recent fires can be relatively quickly reversed simply by changing burn prescriptions to promote fruit production and dense cover. Saw palmetto that serves as suitable carnivore natal den habitat and as food for the black bear can return in a matter of years rather than in decades. This is particularly important because the microhabitat features that seem to be important to these species have more to do with understory conditions than with species composition or height of the forest canopy. That is, a mature pine overstory is not required for a palmetto patch to have value to large carnivores in southern Florida.

Whereas a long-rotation, growing season approach to southern Florida fire can be defended because it makes ecological sense in a wilderness landscape, managers have more than just large carnivores to consider in their planning. Further, we recognize that even the best planned fires, conducted under optimal conditions, often burn much less (or much more) than is targeted in a prescription, and small patches of saw palmetto have a lower probability of burning in a given year than a large patch. However, given the tenuous status of large carnivores in the region, we believe that it is prudent to develop management plans that recognize the value of older age saw palmetto habitat and the services it provides bears and panthers, rather than gamble that a short-rotation prescription will not reduce important food and cover for these species. In other words, the maintenance of good large carnivore habitat in southern Florida should not be an accident. Although we might agree with Schortemeyer et al. (1991:524) who stated that frequent fire can, "provide maximum benefits for deer and other prey species," in southern Florida—a reduction in the extent of mature saw palmetto thickets and their associated structure may locally eliminate stalking cover and restrict kill success rates for the panther, even under increased prey conditions. Where boundary issues, such as smoke on highways and private property damage, are important concerns, fire prescriptions could be adjusted to allow winter burning and strategic fuel reduction in restricted peripheral areas. Elsewhere, efforts should be made to optimize older age saw palmetto habitats for large carnivores, and to avoid the use of fire during times when neonates are not fully ambulatory.

We believe that a mosaic of recently burned and relatively fire-free pine and palmetto habitat could maintain conditions that are conducive to reproduction and nutrition of Florida's large carnivores and could provide an opportunity to experimentally investigate the response of bears and panthers to these conditions. This approach targets the maintenance of some saw palmetto habitat in a fire-free condition for 20 years or longer. These areas should serve as focal points of management units that would include a network of patches that are burned in a rotating cycle (Figure 4). Patches should be maintained at post-fire intervals ranging from one year to the maximum rotation age. Until further research clarifies the best arrangement of various stages of saw palmetto and explicates the threshold below which the amount of old-stage saw palmetto is insufficient to provide stalking cover for panthers, food for bears and optimal den sites for both species, the core, old-stage patches should cover approximately 25 percent of each management unit—a rule of thumb developed for maximizing biodiversity benefits in old growth forest (Harris 1984). The ignition of the oldest patches (D in Figure 4) should occur only after an equal number of the next oldest patches (C in Figure 4) are available to replace them. Such a strategy would avoid a one-dimensional approach that ignores several important natural history requirements while retaining the landscape patterns in which southern Florida black bear and panther evolved.

We agree with Dees et al. (2001) that further investigation is needed to better understand the relation among fire frequency, fire season, southern Florida pinelands and native large carnivores. Radio-collared panthers and bears inhabiting the FPNWR could be closely monitored and their movements examined relative to the rotating fire mosaic. We predict that den use in both species will increase with increasing age of palmetto patches and that black bear use in fall will increase in palmetto patches that have had more than 4-years protection from fire. If allowed to become a long-term research project, such a study might reveal the age at which a saw palmetto patch senesces and offers diminishing returns to resident large carnivores. This could then be used to set the maximum age of a palmetto patch in a managed upland pine mosaic. Other research might investigate:

1. whether fires burn homogeneously through saw palmetto-dominated communities
2. how plant species composition and structure change relative to the frequency, season and extent of fire prescriptions
3. the interval and arrangement of fire in managed forests that promote the well-being and recovery of southern Florida's large carnivores.

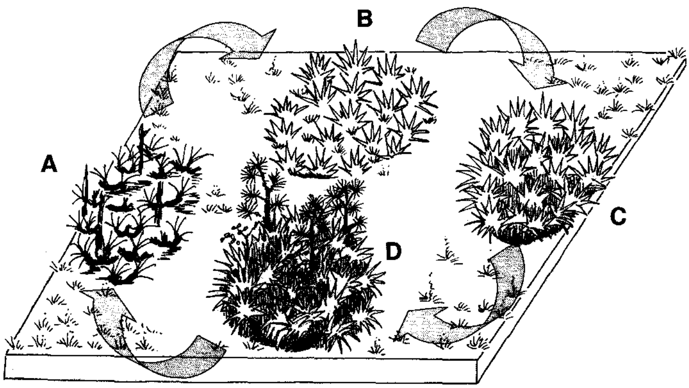


Figure 4. A long-rotation approach to managing saw palmetto habitats in southern Florida involves maintaining some saw palmetto habitat in a fire-free condition for up to 20 years. These areas should serve as focal points of management units that would include a network of patches that are burned in a rotating cycle (e. g., A is burned in year 1 then left unburned for 20 years; B is burned in year 6 then left unburned for 20 years; C is burned in year 11 and left unburned for 20 years; D is burned in year 16 and left unburned for 20 years). Prescribed fires could still be planned on an annual basis given a large enough management area, with prescriptions modified depending on the amount of lightning-caused fires. (line drawing by D. Maehr)

Clearly, basing landscape-scale management solely on a tendency for panthers or their prey to use recently and frequently burned areas ignores important attributes of habitats that have not recently burned and that might otherwise be expected to escape fire for relatively long periods as our modeling and other evidence suggest. Until the correlates of large carnivore reproductive success and nutrition are better understood (especially where bear and panther are the subject of management that targets their recovery), we encourage managers of southern Florida pinelands to be more conservative with their fire prescriptions. Southern Florida may be one of the few places in the southeastern United States where networks of public lands are large enough to adopt evolutionarily relevant fire management programs that are not obviously complicated by millennia of anthropogenic fire. As the autecology of important cover plants and the preferences of the large carnivores that use them attest, longer fire rotations should be considered as components of management plans that target the Florida panther and the black bear. To do otherwise would be to promote traditions based in convenience rather than to adopt enlightened approaches that stem from a greater appreciation for the ecology of fire in a frequently wet, patchy, forest landscape.

Reference List

Abrahamson, W. G. 1984. Species responses to fire on the Florida Lake Wales Ridge. *American Journal of Botany*. 71:35–42.

———. 1995. Habitat distribution and competitive neighborhoods of two Florida palmettos. *Bulletin of the Torrey Botanical Club*. 122:1–14.

———. 1999. Episodic reproduction in two fire-prone palms, *Serenoa repens* and *Sabal etonia* (Palmae). *Ecology*. 80:100–15.

Abrahamson, W. G., and D. C. Hartnett. 1990. Pine flatwoods and dry prairies. In *Ecosystems of Florida*, eds. R. L. Myers, and J. J. Ewel, 103–49. Orlando, Florida: University of Central Florida Press.

Allen, R. M. 1956. Relation of saw palmetto to longleaf pine reproduction on a dry site. *Ecology*. 37:195.

Beckage, B., W. J. Platt, M. G. Slocum, and B. Panko. 2003. Influence of the El Niño southern oscillations on fire regimes in the Florida Everglades. *Ecology*. 84:3,124–30.

Bunting, S. C. 1996. The use and role of fire in natural areas. *National parks and protected areas: Their role in environmental protection*, ed. R. G. Wright, 277–301. Cambridge, Massachusetts: Blackwell Science.

Burger, G. V. 1973. *Practical wildlife management*. New York, New York: Winchester Press.

Carr, R. S., and J. G. Beriault. 1984. Prehistoric man in southern Florida. In *Environments of south Florida: Present and past II*, ed. P. J. Gleason, 1–14. Coral Gables, Florida: Miami Geological Society.

Carrington, M. E., and J. J. Mullahey. 1997. *Effects of time since burning on saw palmetto* (Serenoa repens) *flowering and fruiting, Abstract*. Miami, Florida: Walt Dineen Society Annual Conference, Florida International University.

Craighead, J. J., J. S. Sumner, and J. A. Mitchell. 1995. *The grizzly bears of Yellowstone: Their ecology in the Yellowstone ecosystem, 1959–1992*. Washington, DC: Island Press.

Davison, K. L., and S. P. Bratton. 1988. Vegetation response and regrowth after fire on Cumberland Island National Seashore, Georgia. *Castanea*. 53:47–65.

Dees, S. D., J. D. Clark, and F. T. van Manen. 1999. Florida panther habitat use in response to prescribed fire. *Journal of Wildlife Management* 65: 141–7.

Duever, M. J., Carlson, J. E., Meeder, J. F., Duever, L. C., Gunderson, L. C., Riopelle, L. A., Alexander, T. R., Myers, R. L., Spangler, D. P. 1979. *Resource inventory and analysis of the Big Cypress National Preserve: Contract no. CZ500070899*, Atlanta, Georgia: U. S. Department of Interior, National Park Service.

Egler, F. E. 1952. Southeast saline Everglades vegetation, Florida, and its management. *Vegetatio Acta Geobotanica.* 3:213–65.

Florida Department of Environmental Protection. 2002. *The role of wildland fire in Florida state parks*. http://www.dep.state.fl.us/parks/ncr/wildfire.htm.

Gholz, H. L., D. N. Guerin, and W. P. Cropper. 1999. Phenology and productivity of saw palmetto (*Serenoa repens*) in a north Florida slash pine plantation. *Canadian Journal of Forest Research.* 29:1,24–53.

Hann, J. H., 1991. *Missions to the Calusa.* Gainesville, Florida: University of Florida Press. Harris, L. D. 1984. *The fragmented forest: Island biogeography theory and the preservation of biotic diversity.* Chicago, Illinois: University of Chicago Press.

Hansen, A. J., T. A. Spies, F. J. Swanson, and J. L. Ohmann. 1991. Conserving biodiversity in managed forests. *Bioscience.* 41:382–92.

Harris, L. D., T. S. Hoctor, and S. E. Gergel. 1996. Landscape processes and their significance to biodiversity conservation. In *Population dynamics in ecological space and time*, eds. O. E. Rhodes, R. K. Chesser, and M. H. Smith, 319–47. Chicago, Illinois: University of Chicago Press.

Hartman. A. W. 1949. Fire as a tool in southern pine. In *The yearbook of agriculture*, 517–27. Washington, DC: U.S. Department of Agriculture.

Hilmon, J. B. 1968. *Autecology of saw palmetto* (Serenoa repens) . Durham, North Carolina: Duke University.

Hofstetter, R. H. 1984. The effect of fire on the pineland and sawgrass communities of southern Florida. In *Environments of south Florida present and past II* , ed. P. J. Gleason, 465–76. Coral Gables, Florida: Miami Geological Society.

Hough, W. A. 1968. Carbohydrate reserves of saw palmetto: Seasonal variation and effects of burning. *Forest Science*. 14:399–405.

Kennard, D. K., K. Outcalt, and M. Carrington. 2003. *Serenoa repens* (Bartr.) Small Palmae. In *Wildland shrubs of the United States and its territories: Thamnic descriptions, general technical report, IITF-WB-1ed*, ed. J. K. Francis. U. S. Department of Agriculture, Forest Service. http://www.fs.fed.us/global/iitf/wildland_shrubs.htm.

Land, E. D. 1994. *Southwest Florida black bear habitat use, distribution, movements, and conservation strategy. Final report*. Tallahassee, Florida: Florida Game and Fresh Water Fish Commission.

Langdon, O. G. 1981. Some effects of prescribed fire on understory vegetation in loblolly pine stands. In *Prescribed fire and wildlife in southern forests*, ed. G. W. Wood, 143–53. Georgetown, South Carolina: Belle W. Baruch Forest Science Institute, Clemson University.

Larson, L. H. 1980. *Aboriginal subsistence technology on the southeastern Coastal Plain during the late prehistoric period.* Gainesville, Florida: University Press of Florida.

Laurence, W. F., and B. Williamson. 2001. Positive feedbacks among forest fragmentation, drought, and climate change in the Amazon. *Conservation Biology*. 15:1,529-35.

Little, E. L., Jr. 1978. *Atlas of United States trees. volume 5. Florida. USDA Forest Service, miscellaneous publication no. 1361*. Washington, DC: U. S. Department of Agriculture, Forest Service.

Maehr, D. S. 1997. The comparative ecology of bobcat, black bear and Florida panther in south Florida. *Bulletin of the Florida Museum of Natural History*. 40:1–176.

Maehr, D. S., and J. N. Layne. 1997. The saw palmetto, Florida's all-purpose plant. *The Palmetto*. 16(4):6–10.

Maehr, D. S., E. D. Land, J. C. Roof, and J. W. McCown. 1990. Day beds, natal dens, and activity of Florida panthers. *Proceedings of the Annual Conference of Southeastern Fish and Wildlife Agencies*. 44:310–8.

Maehr, D. S., T. S. Hoctor, L. J. Quinn, and J. S. Smith. 2001. *Black bear habitat management guidelines for Florida. technical report no. 17*. Tallahassee, Florida: Florida Fish and Wildlife Conservation Commission.

Marquart, W. H. 1992. Calusa culture and environment: What have we learned? In *Culture and environment in the domain of the Calusa,* ed. W. H. Marquart, 423–36. Gainesville, Florida: Institute of Archaeology and Paleoenvironmental Studies, University of Florida.

Milanich, J. T. 1994. *Archaeology of pre-Columbian Florida.* Gainesville, Florida: University Press of Florida.

Milanich, J. T. 1995. *Florida Indians and the invasion from Europe.* Gainesville, Florida: University Press of Florida.

Myers, N. 1997. The rich diversity of biodiversity issues. In *Biodiversity II*, eds. M. L. Reaka-Kudla, D. E. Wilson, and E. O. Wilson, 125–38. Washington, DC: Joseph Henry Press.

Newmark, W. D. 2002. Conserving biodiversity in East African forests: A study of the eastern Arc Mountains. *Ecological Studies.* 155:1–197.

Peek, J. M. 1986. *A review of wildlife management.* Englewood Cliffs, New Jersey: Prentice Hall.

Platt, W. J., G. W. Evans, and M. M. Davis. 1988. Effects of fire season on flowering of forbs and shrubs in longleaf pine forests. *Ecologies.* 76:353–63.

Quintana-Ascensio, P. F., and E. S. Menges. 1996. Inferring metapopulation dynamics from patch-level incidence of Florida scrub plants. *Conservation Biology.* 10:1,210–9.

Robertson, W. B., Jr. 1953. *A survey of the effects of fire in Everglades National Park.* Homestead, Florida: U. S. Department of Interior, National Park Service.

Robertson, W. B. 1962. Fire and vegetation in the Everglades. *Proceedings of the Tall Timbers Fire Ecology Conference.* 1:67–80.

Sackett, S. S. 1975. Scheduling prescribed burns for hazard reduction in the southeast. *Journal of Forestry.* 73:143–7.

Schortemeyer, J. L., D. S. Maehr, J. W. McCown, E. D. Land, and P. D. Manor. 1991. Prey management for the Florida panther: A unique role for managers. *Transactions of the North American Wildlife and Natural Resources Conference.* 56:512–26.

Stoddard, H. L., Sr. 1935. Use of controlled fire in southeastern upland game management. *Journal of Forestry.* 33:346–51.

Stoddart, L. A., A. D. Smith, and T. W. Box. 1975. *Range management. Third edition.* New York, New York: McGraw-Hill.

Stratman, M. R. 1998. *Habitat use and effects of prescribed fire on black bears in northwestern Florida.* M.S. thesis. Knoxville, Tennessee: University of Tennessee.

Terborgh, J. 2002. The "working" forest-Does it work for biodiversity? *Wild Earth.* 12(3):29–35.

Tomlinson, P. B. 1980. *The biology of trees native to tropical Florida.* Allston, Massachusetts: Harvard University Printing Office.

Wade, D., J. Ewel, and R. Hofstetter. 1980. *Fire in south Florida ecosystems. forest service general technical seport SE-17.* Asheville, North Carolina: U. S. Department of Agriculture, Southeastern Forest Experiment Station.

Real Fire Restoration Deserves Real Funding

Sean Cosgrove
Sierra Club
Washington, DC

Most of the recent political debate over fire management on federal public lands has centered on the use, location and effectiveness of commercial logging as a proper tool to protect homes and communities. Added claims by timber industry representatives and their allies were that commercial timber sales could also be used to prevent forest fires or to reduce their intensity across the landscape. Special emphasis was placed on claims that expediting commercial timber sales through environmental impact analyses would help stop or reduce catastrophic forest fires (American Forest Research Council 2004). In August of 2002, the Healthy Forests Initiative was proposed by the current Administration as a plan for, "wildfire prevention and stronger communities" (Bush 2004). The timber industry won last year's political debate when President George W. Bush signed the Healthy Forests Restoration Act of 2003 in December (Public Law 108-148). As a result, we can expect to see a dramatic increase in federal timber sales, with the stated purpose of fuel reduction, in areas within the wildland-urban interface (WUI) as well as remote roadless forests that are miles from the nearest house. While the Healthy Forests Restoration Act is limited to 20 million acres (8 million ha) of federal land, the institutional momentum of using commercial timber sales for fuel reduction will likely spur an increase in federal logging projects through the U. S. Department of Agriculture, Forest Service's (USFS's) forest products and vegetation management programs. Depending on the implementation of the new law, commercial timber sales could be more ubiquitous than ever for purposes other than the traditional claim of "providing fiber for the market." The logic given to agency managers seems to be that timber sales must be used to tame violent nature and possibly even replace the act of fire itself with thinning and logging. (Indeed, a USFS promotional video claims, "It will take active management all across the landscape to reduce the risk of big, dangerous fires and to restore our forests to a healthier condition where fire is a friend, rather than a foe" (2003). The federal agencies implementing the Healthy Forests Restoration Act will be looked at closely by politicians, Administration officials, conservationists, timber industry executives, scientists and citizens who love and use our public federal lands. The fundamental flaws of the Healthy Forests Initiative and the Healthy Forests Restoration Act are the importance and priority that these policies place on the use of commercial timber sales as a substitute for the natural role of fire and a method of fuel reduction. The commercial timber program gives agency managers incentives to increase timber harvest over other resource management goals through the ability to retain funds out of timber sales receipts. When a commercial timber sale is used for other objectives, such as fuel reduction, fire restoration and habitat improvement, the production of timber becomes the primary product. The best opportunities to restore the natural role of fire to federal public lands, to protect communities, to best use scarce federal funds and to build trust and participation with the public is not to use more commercial timber sales. Instead, redirecting federal funds and personnel into more direct fuel reduction and fire restoration will provide better economic and ecological benefits.

Protect Communities First

The Sierra Club believes that the number one priority of the federal fire management program should be to protect homes and communities from threat of wildfire. The research behind the Firewise Program (see http://www.firewise.org) has produced solid results that help homeowners and community leaders take action and derive a substantial degree of increased protection. Further, the research of the USFS Fire Research Laboratory (see http://www.firelab.org) proves that fuel reduction 100 to 200 feet around homes and other structures will provide a significant degree of increased safety from a surrounding wildland fire (Cohen 1999). For economic, ecological and political reasons we must achieve significant results towards providing funds and resources for responsible fuel reduction around homes and communities before any lasting progress can be made in addressing the concerns related to fire restoration. This does not mean that the planning of fire restoration projects should cease until each home, cabin and doghouse in North America is Firewise Program approved. But, until the engaged public and policymakers assure that serious steps are taken to protect homes and lives, the debate around fire management and public safety will continue to be contentious and divisive. Wildlife managers and ecologists interested in broader fire restoration goals should expect that a substantial amount of funding will be unavailable to their programs as long as the federal commercial timber sale program is used as an avenue for fuel reduction and fire restoration.

It's been said that the first step to overcoming a problem is to acknowledge that it exists. If we are to restore the role of natural fire to fire-adapted ecosystems and to derive the ecological benefits that come from natural fire, we should first try to recognize the scope of the problem. The amount of burned acreage is much higher than in recent decades. From 1919 to 1929, an average of 26 million acres (10.5 million ha) burned each year. In the following decade of 1930 to 1939 an average of 39 million acres (15.7 million ha) burned each year (National Interagency Fire Coordination Center 2003). When were natural fire cycles interrupted to such an extent they need to be restored? Medler (2004), a fire scientist, a former firefighter and a member of the Board of Directors of the Association for Fire Ecology, looked at USFS data of historic fire regimes and their historical fire return intervals for each regime. He conservatively estimated that an average 26.4 million acres (10.6 million ha) across the United States should burn each year in order to sustain fire-adapted ecosystems and to prevent them from degrading into a more severe condition class. Then, to ensure against underestimating the amount of acreage needed to be treated to protect communities from wildfire, Medler drew a buffer zone of a one-third mile (0.54 km) around the 4,135 communities identified by the U. S. Census Bureau in the western United States. This buffer zone was broad enough that in places it included nonflammable areas, such as the beaches of San Diego, California and highways around Boulder, Colorado. Subtracting the area within the total one-third mile (0.54 km), Medler calculated that approximately 12 million acres (4.8 ha) should be burning in the western United States each year in order to maintain fire-adapted ecosystems. Then, using the same calculations and his professional judgment of nonflammable areas inside the WUI, Medler determined that approximately 4.4 million acres (1.7 million ha) need to be treated to create community fire protection zones around every community in the western United States (M. Medler, personal communication 2003).

Commercial Timber Sales Cannot Replace the Ecological Benefits of Fire

For ecologists and wildlife managers, the use of commercial timber sales to attempt wildlife habitat prescriptions is not a rarity. However, any consequential benefits that a specific timber sale may produce for wildlife habitat have often come at a greater cost. There is no doubt that logging creates change in a forest ecosystem. The question often becomes whether the change is the change desired. Fire, as an ecosystem process, provides numerous ecological benefits that commercial logging cannot emulate. The beneficial effects of fire on forest and grassland ecosystems are prolific and are not accompanied by the negative impacts associated with logging, such as soil compaction and erosion. Although fire in the western United States may be associated with the spread of some invasive plant species, the dramatic ground disturbance caused by logging and logging roads create exceptional conditions for invasive species to take hold. Logging equipment and logging roads also become a notorious vector for most noxious weeds and some tree pathogens, such as the Port Orford cedar root disease. The spread of invasive species not only crowds out native plant species but can increase erosion, degrade wildlife habitat, reduce land values and cause economic losses (Westbrooks 1998).

When fires burn naturally or under carefully prescribed conditions, a vegetation mosaic of different forest types is created. This provides a greater diversity of vegetation and, consequently, a greater diversity of wildlife species. Prescribed fires can reduce the amount of combustible fuel buildup that may cause larger, more destructive fires. One major effect of fire that logging cannot replicate is a change in soil nutrients and soil temperature. Fire may be a chief factor maintaining productivity in colder soils where the lack of nutrients is a major factor limiting plant growth. Fires release nitrogen and other nutrients from woody vegetation back into the soil in the form of mineral-rich ash, which makes them readily available for new plant growth (Brown 2003). Serotinous cones are those that open only when exposed to extreme heat and trees that bear these cones, such as jack pine, require fire to propagate. Fire is efficient in seedbed preparation and in eliminating vegetative competition. Finally, many beneficial species rely on the ecosystem structures that are created by fire, such as down logs and snags. For example, many woodpecker species that control epidemic insect populations are dependent upon dense stands of fire-killed trees. Although logging and thinning can affect the structure and composition of forests, they fall short in producing the myriad benefits of natural fire. The Sierra Nevada Ecosystem Project Report states: "The many ecosystem functions of frequent low-to-moderate fire can be restored fully only through the use of fire. Silvicultural cuttings and other fire surrogates can substitute only partially for fire"

(Weatherspoon 1996). Logging and logging roads have other well-documented harmful impacts on fish and wildlife habitat that include fragmentation, forest type conversion, erosion and increased siltation. In an effort to produce better conditions for wildlife using commercial timber sales, the consideration for wildlife habitat more often becomes a consequence or a byproduct rather than the main product.

Excessive Costs of Mechanical Treatment Restrict Landscape Scale Application

The cost of mechanical fuel treatments for either home protection from fire or as preparation for the reintroduction of fire through prescribed burning can be expensive. Prescribed burning alone can have significant costs but is not usually as costly as mechanical thinning. Mechanical thinning can cost between $500 to $1500 per acre, or more, and prescribed burning can range generally between $50 and $500 per acre. If one is only considering treating the estimated 4.4 million acres (1.7 million ha) within the WUI of one-third mile around western communities and achieves a moderate cost per acre, the total cost could easily add up to billions of dollars. Considering both the nondesirable impacts of commercial logging and the cost of mechanical treatment, it is best to focus the use of mechanical thinning in the one-third mile WUI. In this area, mechanical thinning can be most effective in reducing fuels that can bring fire in connection with homes, and costs can be kept to a reasonable level.

One common perception of the benefit of using commercial timber sales is that selling merchantable trees or other fiber can offset fuel reduction costs. Small diameter, nonmerchantable fuels are the type of material most likely to increase fire risk, and they are most likely the material needing to be removed in community-safety fire protection projects. Timber companies, however, prefer to bid on timber sales that include large trees—not brush and trees less than 5 inches (12.7 cm) in diameter. Harvesting costs are often the primary issues in whether or not a stand treatment will pay for itself. This is not surprising considering the cost to operate a sale increases considerably when tree diameter sizes decrease (Hartsough et al. 2002). This dynamic often leads to fuel reduction thinning treatments being expanded in acreage, larger tree diameter or number of trees cut in order to increase economic gains. Costs could be offset in some projects, but even the basic cost to bring in equipment capable of hauling the trees away can greatly increase the costs of operating the sale. Hauling, loading, skidding and using a feller-buncher increases costs at each stage. Even the Healthy Forest Restoration Act, which includes expedited procedures for planning timber sales, will not produce any measurable revenue. The Congressional Budget Office (CBO) analysis of the bill had the following to say: "Enacting this legislation could affect offsetting receipts (a credit against direct spending), but CBO estimates that any such effects would total less than $500,000 a year" (Congressional Budget Office 2003). Simply put, materials from real fuel reduction projects are more apt to be used for mulch, rustic chairs and kiosks for the Winter Olympics (U. S. Department of Agriculture, Forest Service 2002). Those all may be credible uses for fuel reduction byproducts, but there is not enough of a market to economically support fuel reduction work on a nation-wide scale by selling byproducts. The incentive to increase economic gains can cause impacts or can increase fire risk and severity. We also cannot expect them to produce added funding for more fire restoration or wildlife habitat improvement projects. The USFS's dismal track record of returning receipts to the U. S. Department of the Treasury strongly suggests that if more commercial timber sales are used for fire restoration or fuel reduction, more taxpayers' funds are wasted.

Commercial Timber Program Creates Harmful Incentives and Wastes Funds

The waste and abuse of taxpayer funds spent in implementation of the USFS timber program is legendary. The USFS has been criticized by taxpayer advocates and by conservation groups, by Representatives and Senators, by its own employees, by other federal agencies and by independent economists as one of the most fiscally mismanaged agencies in the federal government. When investigated over the course of multiple years or even over a single-year the USFS timber program loses taxpayer funds. A September 2001 General Accounting Office (GAO) report (GAO-01–1101R) on the Timber Sales Program information reporting system (TSPIRS) illustrates this problem. GAO found that the Reporting System made it, "impracticable, if not impossible, for us or anyone to accurately determine the USFS's timber sales program costs" (Calbom 2001:1). The Wilderness Society found that, in fiscal year 1997, 83 of the 104 national forests with timber programs yielded a net loss to the taxpayers

(The Wilderness Society 1997). Randall O'Toole, of The Thoreau Institute, claims that, "where the Forest Service calculates an $88 million loss in 1997, the Institute calculates a $214 million loss before payments to counties in lieu of property taxes" (1997). The USFS responded to this challenge for fiscal responsibility by announcing it would end the preparation of TSPIRS reports. Two and a half years later the final TSPIRS report for fiscal year 1998 was released. From that report Taxpayers for Common Sense (2002) concluded that the USFS lost $407 million dollars in fiscal year 1998 and only 6 of 111 national forests generated enough revenue to cover the cost of their timber programs.

The USFS has argued that the money-losing timber program achieves broad forest stewardship goals and—in the long run—the public benefits by overall conditions created by logging. Another claim is that federal timber sales are needed to provide jobs and economic benefits to rural communities. Surely this is a laudable goal but—of all the jobs provided by the national forest system—a mere 2 percent are produced by the timber program (U. S. Department of Agriculture, Forest Service 1995).

Another major problem with the use of commercial timber sale contracts is the incentive for agency managers to design projects to appeal to timber purchasers instead of to seek the best resource management goals. Well-intentioned projects that may be designed to reduce fuel loads by removing small diameter fuels are often expanded to include larger diameter and more merchantable trees after they fail to attract bidders. One project was discovered in the Prescott National Forest this summer by Sierra Club members, who found large, old-growth ponderosa pine logs cut and stacked along a road that winds through a forest of thick unthinned, small, fire-prone trees. According to the local USFS managers, these sales, located on a forest road near Indian Creek Campground, were fuel reduction and beetle kill projects planned with a categorical exclusion, meaning there was little environmental review. The USFS also indicated that they sold the trees at the minimum rate, that they couldn't get anyone to bid on trees smaller than 12 inches (30.4 cm), and that there is no plan for future removal of the small flammable trees. "The old growth log decks on the Prescott are a clear reminder of why opening the forests to expedited logging will not promote forest health or reduce fire risks to communities," said Sandy Bahr (personal communication) of the Grand Canyon Sierra Club: "Given free reign and no public review, the industry will continue to take the oldest, largest trees, leaving the small fuels that are the problem behind" (Southwest Forest

Alliance 2003). There is an inherent problem when any portion of North America's rare, old growth or native forests are destroyed because of a faulty contract and bidding process.

Other costs and management problems occur when proper brush and slash disposal does not follow a commercial timber sale. The residual logging slash alone can often add more biomass and fuel load to the forest floor where it greatly increases fire risk. It would be easier to provide a direct payment to qualified contractors to perform fuel reduction and prescribed burning instead of using an inefficient and outdated commercial timber sale process. The relatively little fiber that is produced through the federal timber program is a paltry benefit compared to the environmental damage, financial waste and abuse of public trust that the timber sale program generates.

Budget Realignment Can Produce More Effective Fire Restoration

Responsible fire restoration will certainly be expensive, but, if the results are safer communities, are more productive for fish and wildlife habitat and save on future fire suppression, the expense could be worth it. Federal dollars are limited, but the demonstrated need for fuel reduction funding is well established. So, how can we make the best use of scarce federal resources to reduce fuels and restore fire? Three major areas for progress are:

1. increase funding for fuel reduction in the WUI
2. redirect funding from the USFS forest products and vegetative management line items to fuel reduction and fire restoration
3. create policies that curtail excessive spending on suppression and allow more use of wildland fir-use fires.

Obviously, increasing expenditures to fire restoration will help but only if the funds are spent in a direct and efficient manner. The greatest priority should be for fuel reduction in the WUI as defined by a quarter-mile around communities. The Sierra Club and other conservation organizations have proposed a figure of \$2 billion each year for five consecutive years to be spent on fuel reduction in the WUI. Any amount approaching this level of funding would help insure a program that provides the amount of security that communities need from fire threats. It would also avoid the hesitancy of implementing fuel reduction projects created by disinterested timber companies and the slow development of market uses for small diameter material. Whether or not funding on this level is politically feasible relies on Congress and the executive branch to conclude that safe homes are the most desirable outcome. The Healthy Forests Restoration Act authorized $760 million for implementation of the entire bill in each year, but it is not clear whether the Bush Administration or the Congress will provide new funds or shift funds from other line items. The early indication from the proposed fiscal year 2005 USFS budget is that much needed new funds will not be forthcoming. This has already caused some concern that the fuel reduction requirements will not be addressed and that other programs will suffer: "The need is for $760 million in new spending," said Jeff Olson (Associated Press 2004) of Rapid City, who is on the Black Hills National Forest Advisory Board. Olson represents the board and said the search for thinning money would hurt other programs: "My fear is they're going to raid wildlife and recreation," he said.

Bypassing the commercial and salvage logging programs and investing directly in fuel reduction in the WUI and in fire restoration will put the resources where they can produce the best result. Better preparation around communities will increase opportunities for wildland fire-use (WFU) and prescribed burning to be used to greater effect on non-WUI lands. Based on the Bush Administration's proposed budget for fiscal year 2005 a simple reprogramming of the USFS's forest products line item would free $274 million that could be used for fuel reduction and fire restoration. This is $8 million more than the Administration has proposed for hazardous fuels reduction and $38 million more than the agency received for fiscal year 2004 (Public Law 108-108). The hazardous fuels reduction funding level is to treat 1.8 million acres (0.7 ha). The fiscal year 2005 budget also proposed $194 million for vegetative management. This line item is also used for commodity producing salvage timber sales. In the last six years the average acreage burned on federal lands each year through prescribed burning was a little over 1.6 million acres (0.6 ha). Expanding the use of prescribed burning on non-WUI lands is a cost effective and ecologically beneficial manner to reduce fuels and to restore fire to the landscape. Applying increased funds and resources from the salvage and commercial logging programs will help to achieve fire restoration and to reduce the short- and long-term flammability factors, such as residual slash and increased roaded acres that logging contributes.

Use the Fire Nature Gave You

Federal land managers took an important step towards restoring the role of fire in fire-adapted ecosystems in 1994 when they signed the Federal Wildland Fire Management Policy. The Federal Wildland Fire Management Policy encouraged greater prescribed burning and ordered land management agencies to develop fire plans that help firefighters decide where and under what conditions they can let wildfires burn. Although fire planning has lagged, federal agencies have dramatically increased the number of acres they have treated through prescribed burning. Another important development is the agencies' attention to using WFU fires. These are naturally caused fires that are monitored and allowed to burn. The average acreage burned through the use of WFU fires in the last 6 years was roughly 120,000 acres (48,562.3 ha) (National Interagency Fire Coordination Center 2003). One example is the Bear Creek fire that burned near Vallecito, Colorado in 2003. The July 21 USFS update stated the fire was, "managed as a WUF to achieve resource benefits. Benefits include allowing fire to play its role in the ecosystem, removing dead and downed fuels, improving wildlife habitat and forage, and creating structural diversity in the forest. In the case of Bear Creek, the fire will also ultimately protect the community of Vallecito from a future catastrophic wildfire by removing the large quantities of dead fuel" (U. S. Department of Agriculture, Forest Service 2003). With the exorbitant cost of wildland fire suppression, the increased use of WFU fires could be a huge savings in annual funds, could provide increased fire restoration and even could help keep firefighters from some of the more dangerous backcountry blazes.

Building Trust Is Key to Making Progress

Fuel reduction and wildland fire restoration efforts must be done right in order to build the public's trust and participation. Sierra Club chapters and groups across the country are willing to work with federal and state agencies to promote fire safety education, to build public support for fire restoration and even to provide volunteers to help with the cutting and hauling of flammable material. In June of 2003, over 55 volunteers, most local Sierra Club members, helped haul slash and flammable brush away from nine homes in Silverthorne, Colorado. This was a fuel reduction project planned by the USFS with fire safety inspections by the local fire department and volunteer labor coordinated by a Sierra Club staff person that used tree cutters and wood chippers donated by local businesses. "This project shows what can happen when fire departments, local government, and community environmental groups get together to work on a project," says Gary Lindstrom, Summit County Commissioner (Valtin 2003).

With the Bush Administration's desire to increase commercial logging comes strong concern that terms and phrases, like "vegetation management," "forest health," "stewardship" and "salvage," could become just mere terms to justify large commercial timber sales. In many cases the USFS has become its own worst enemy and created tremendous distrust of forest restoration attempts by misnaming large-scale logging projects in sensitive and controversial areas as restoration projects. Unfortunately, the Healthy Forests Initiative started on the wrong foot. The Boswell Creek Watershed Project, on the Sam Houston National Forest in Texas, is 1 of 11 new demonstration projects nationwide designed to showcase the ideas behind the Healthy Forests Initiative. This demonstration project is ostensibly planned to reduce the threat of catastrophic wildfire to protect communities, firefighters and wildlife and to improve forest health. Although the project proposes prescribed burn, the USFS limited consideration to just two alternatives: commercial logging or no action. When the project was first announced, Texas Sierra Club members had great hope that a new focus towards fire restoration would blossom after years of contention over the commercial logging program in Texas' national forests. Our members wanted to see new resources and personnel put toward a science-based, nonextraction, fire restoration program. Sadly, their hope for a new opportunity of building partnerships for forest restoration has fled.

Even more effective at destroying trust with the conservation-minded public is the cynical packaging of a real fuel reduction or fire restoration project with the commercial logging of old growth or roadless area forests. The Kelsey Whiskey timber sale, planned by the Medford District of the U. S. Bureau of Land Management (BLM) would clearcut 530 acres (214.4 ha) of classic, ancient forest out of the BLM's largest forested roadless area in the nation. This is a supposed requirement in order to reduce fuels and risk. It is even more startling that the BLM (2003) proposed this continued logging of old growth forests after noting that "any project proposed in this area generates public controversy." What type of approach to building public involvement is it when an agency notes such a concern and then decides on the management alternative that will produce the largest amount of timber volume at the expense of dwindling old growth habitat? Another example is the East Rim Vegetation Management timber sale, on the Kaibab National Forest, which proposes to log 2,300 acres (930.8 ha) of the scarce fire resistant old growth ponderosa pine forests on the north rim of the Grand Canyon to decrease risk of catastrophic fire (Southwest Forest Alliance 2004). How can our leading federal conservation agencies attempt to build trust with an increasingly ecologically aware public by proposing such retrograde projects?

Building the public's trust and involvement is not only vital to the success of any individual project but must be the foundation for the forest restoration work of the coming decades. Forest fires can scare the general public, but there has been a great increase in the level of knowledge and support for restoring natural fire cycles. It would be tragic to see the trust of many in the conservation community and in the public destroyed by a spate of new commercial logging projects that use the support for fire restoration but only seek to take advantage of a political climate currently favored by commercial logging interests.

How Do We Restore Fire without a Federal Timber Program?

Despite years of rhetoric and misinformation, national and regional economies are not dependent on logging national forests. Our national forests and other federal public lands produce goods and services that are much more significant than the value of commercial logging. The need for community protection and forest fire restoration is abundantly evident and should be the number one priority of federal land management agencies. Achieving these goals will require serious investment of funds and personnel as well as an unprecedented amount of local government and public participation. The threats to communities and detrimental impacts caused by a further interruption of natural fire cycles are considerable. If we are to achieve these goals we must make a clean break from the failed and costly practices of the past. Serious steps to reform the mismanagement of the federal lands to protect our communities and to restore the role of fire should include the following.

- ***Focus fuel reduction activities within a quarter-mile of homes and communities.*** A serious investment of funds and personnel will help build public trust, will protect communities from fire, and will provide the margin of safety for increased use of wildland and prescribed fire.

- ***Eliminate expenditures of funds for commercial logging, post-fire salvage logging and logging road construction.*** Divert these funds to noncommercial fuel reduction, habitat management and prescribed burning programs.
- ***Fully implement the Roadless Area Conservation Rule, and protect all remaining old growth forests.*** Moving away from the expensive and controversial logging of rare habitats will allow funds to be used for other programs as well as build public trust.
- ***Create agency policies that allow increased use of wildland fire use fires.*** Slow the rate of initial fire suppression and allow natural fires in wilderness, roadless and other remote areas.

Restoring our national forests will help leave a legacy of clean air and water, wildlife habitat, recreation opportunities and protection from flooding and wildfire—a wild heritage that is worth more than can be measured by board feet and dollars. To improve fish and wildlife habitat on our national forests we need to create a fire restoration program that is scientifically credible, well funded, socially acceptable and economically viable. By investing in land health and restoration, we can ensure healthy and productive national forests for our families and for our future generations.

Reference List

American Forest Resource Council. 2004. *Comments needed on Bush's healthy forests initiative*. http://www.afrc.ws.

Associated Press. 2004. Tree thinning budget battle under way. Forest Conservation Archives. *Forest Conservation Portal.* http://forests.org/articles/reader.asp?linkid=29442.

Brown, James K., Elizabeth D. Reinhardt, and Kylie A. Kramer. 2003. *Coarse woody debris: Managing benefits and fire hazard in the recovering forest, general technical report RMRSGTR-105*. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Bush, George W., President. 2004. President Bush signs Healthy Forests Restoration Act into law. *The White House*. http://www.whitehouse.gov/infocus/healthyforests/.

Cohen, Jack D. 1999. *U. S. Department of Agriculture, Forest Service general technical report. PSW-GTR-173*. http://www.firelab.org/fbp/fbppubs/fbppdf/cohen/reducingwlfire.pdf

Congressional Budget Office. 2003. *Cost estimate of HR1904*. http://www.cbo.gov/showdoc.cfm?index=4234&sequence=0

Calbom, Linda M. 2001. Financial management: Annual costs of Forest Service's timber sales program are not determinable. U. S. General Accounting Office. http://www.gao.gov/new.items/d01101r.pdf

Hartsough, Bruce, Michael De Lasaux, John Shelly, Gary Nakamura, Ken Blonski, and Glenn Nader. 2002. *Small-parcel mechanical fuel treatment and utilization demonstration project*. http://www.forestprod.org/smallwood02_shelly.pdf

National Interagency Fire Coordination Center. 2003. *Statistics and summary*. www.nifc.gov/news/2003_statssumm/2003Stats&Summ.html

Sierra Club. 2004 *Forest protection & restoration: 7 point community protection plan*. http://www.sierraclub.org/forests/fires/7pt_communityplan.asp

Southwest Forest Alliance. 2003. *Prescott National Forest logs large old growth trees with forest health funds*. http://www.swfa.org/pr_2003/pr_07-29-03.html

Southwest Forest Alliance. 2004. *Conservation groups challenge timber sale*. www.swfa.org/pr_2004/pr_02–18–04.html.

Taxpayers for Common Sense. 2002. *Lost in the forest*. http://www.taxpayer.net/forest/lostintheforest/index.

Thoreau Institute, The . 1997. *National forest timber sale receipts and sales in 1997*. http://www.ti.org/97tspirs.html

Wilderness Society, The . 1998. *Commercial timber sales on national forests, fiscal year 1997*. October.

U. S. Bureau of Land Management. 2003. *Record of decision, Upper Kelsey timber sale: Kelsey Whisky landscape management plan, proposed Medford District resource management plan amendment and environmental impact statement*. http://www.or.blm.gov/Medford/planning/kelsey_wiskey_eis_home.htm#rod.

U. S. Department of Agriculture, Forest Service. 1995. *The Forest Service program for forest and rangeland resources: A long-term strategic* *plan. Draft 1995 RPA Program.* city, state: U. S. Department of Agriculture, Forest Service.

———. 2002. *State and private forestry technology marketing unit. Forest products laboratory. Madison, Wisconsin. 2001–2002 accomplishments.* http://www.fpl.fs.fed.us/tmu/2001_2002_Accomplishments.pdf.

———. 2003. *Bear Creek wildland use fire update. July 21. San Juan National Forest.* http://www.fs.fed.us/r2/sanjuan/bulletin_board/fire-archives/bearupdate21.htm.

Valtin, Tom. 2003. The fire next time. *Planet Newsletter.* http://www.sierraclub.org/planet/200305/fire.asp.

Weatherspoon, C. P. 1996. *Fire-Silvicultural relationships in Sierra forests. Sierra Nevada ecosystem project: Final report to Congress, vol. II, chap. 44.* Davis, California: University of California, Centers for Water and Wildland Resources.

Westbrooks, Randy G., 1998. *Invasive plants: Changing the landscape of America: Fact book.* Washington, DC: Federal Interagency Committee for the Management of Noxious and Exotic Weeds.

Session Five.
Wildlife on Wheels: The Marketing of Today's Outdoor Experience

Chair
Robert A. Clevenstine
U. S. Fish and Wildlife Service
Rock, Island, Illinois

Cochair
William Ohde
Iowa Department of Natural Resources
Wapello

Management Opportunities and Obligations for Mitigating Off-road Vehicle Impacts to Wildlife and Their Habitats

Robert H. Holsman
University of Wisconsin, Stevens Point.
Wisconsin

Introduction

Off-road vehicle (ORV) use on public land is poised to become the most contentious issue in the outdoor recreation arena, if it is not already. Although hunting certainly has a few vocal and passionate critics, the growing popularity of ORVs, especially all-terrain vehicles (ATVs), has sparked widespread concern over safety, noise, air and water pollution, scarred landscapes, disturbance to wildlife, and degradation to plant communities from the use of these vehicles. The outcry over these issues has recently been featured in numerous media outlets, highlighting the growing public discontent with ORV users (Williams 2003, Tanz 2004).

The controversy has also increased policy activity with respect to managing ORV use and its impact on both the environment and on non-ORV participants' enjoyment of the outdoors. At the time of this writing, one northern Wisconsin county was preparing a public referendum on the question of allowing ATV use on county forests. Several state agencies are exploring legislation to tighten rules regarding off-trail riding. Maine's governor recently appointed a state task force to study the growing problems regarding ATV trespass and destruction on private land (T. Peabody, personal communication 2003). Meanwhile, a group of Florida ORV riders have filed a lawsuit against the U. S. National Park Service for restricting their access to backcountry areas (Wilkerson 2002). These are just a few examples that illustrate the growing disruption and conflict associated with ORV use on public land.

The growth in the sale and use of ATVs over the past 20 years has been nothing short of phenomenal. ATV registrations now surpass snowmobile registrations in many northern states. According to the Motorcycle Industry Council, sales of ATVs between 1998 and 2002 more than doubled, to annual sales nearing 850,000. The growing popularity of ATV use has increased demands for additional trails and areas in which to operate them. In areas with few legal places to ride, ATV riders often carve out unplanned trail systems, taking advantage of closed areas that are not specifically posted as a trail system (Nelson 1996). Even where property has been properly posted against ORV use, illegal riding and sign vandalism are common problems (Wilkerson 2001). Both the authorized and unauthorized use of ORVs in undeveloped or wildland areas threaten certain wildlife management objectives, such as maintaining ecological integrity, biological diversity and wildlife recreation.

The two objectives for this paper are to review the research that documents the effects of motorized recreation on wildlife and their habitats and to discuss the current capacity of natural resource law enforcement to respond to harmful or unethical behaviors. Particular attention is given to ATVs due to their tremendous growth in sales and registrations. For clarity's sake, ATVs are defined as three- and four-wheeled vehicles, straddled by the driver and carried on low-pressure tires with engines that range from 50 to 500 centimeters of displacement (Rodgers 1999). The term off-road vehicle in this paper, refers generically to all types of motorized vehicles capable of off-road driving, including dirt-bikes, jeeps, dune buggies and four by four passenger vehicles.

Potential Impacts of Concern

Knight and Cole (1995) describe four categories of impacts that human behavior has on wildlife. These categories are disturbance, habitat modification, pollution and exploitation. Motorized recreation certainly has the potential to negatively impact wildlife in each of these four areas. The first type of impact, disturbance, includes both physiological and behavioral responses of wildlife to noise or surprise. In some cases, wildlife may become acclimated to disturbances, especially if it occurs in predictable locations (e. g., roads) and at regular time periods (Joslin and Youmans 1999). However, for some less tolerant species, regular disturbance may lead to elevated levels of stress, reduced access to food resources, disruption in breeding or parental care, or complete abandonment of all or a part of their home range. Ultimately, these types of disturbances have the potential to reduce fitness levels and contribute indirectly to mortality for wildlife.

Second, habitat impacts of motorized vehicles may take the form of outright loss of habitat or modification that reduces the value of available food, cover, or other key components. For example, repeated stream crossings by ATVs can increase water turbidity and can reduce the availability of spawning habitat for spawning trout or salmon. As a result, cascading impacts to the food chain may cause fewer spawning salmon to provide less food for grizzly bears *(Ursus arctos)*, mink (*Mustela vision*) and scavenging birds of prey. Besides outright habitat loss, the expansion of trail riding systems can increase fragmentation thereby increasing vulnerability of birds and small mammals to predation. Soil compaction or soil erosion from off-trail riding also may exacerbate the spread of exotic plant species.

The third impact category noted by Knight and Cole (1995) is pollution. All gas-powered machines and vehicles contribute greenhouse gases as well as particles and emissions that impact air quality. Since most ORV use takes place in rural and undeveloped areas with lower amounts of air pollution (compared with urban areas), their immediate impacts are difficult to measure. Yet, concerns have prompted the U. S. Environmental Protection Agency (2002) to regulate the nitrogen oxide and methane emissions from ORVs at manufacturer levels this year. Of larger concern is the use of ATVs in and around water bodies where oil and gas discharge may cause pollution. Havlik (2002) estimated that over 10 million gallons (37,850,000 l) of gasoline and motor oil enter the soils and waters of public land each year as a result of inefficient combustion of ATV engines. That figure does not include fuel transfer from undercarriages of ATVs driven through water bodies.

Finally, ORV use may alter exploitation rates and demographic structure for a variety of big game species, especially those that inhabit remote areas. ATVs, in particular, are designed and marketed to be able to go anywhere. The accessibility they provide likely enhances the chance of harvest for big game animals, such as elk (*Cervus elaphas*), bighorn sheep (*Ovis canadensis*) and mountain lions (*Felis concolor*), by moving hunters more easily into areas that previously required more stamina and physical fitness to access. In theory, improved access could mean more harvest efficiency and could alter the size and age structure of animals that are legally harvested. The extent to which the rise in ATV use among hunters has affected the population characteristics of game species has not been investigated.

A second way that motorized vehicles have the potential to increase exploitation of wildlife is through the illegal harvest of animals. National parks and wilderness areas have long been recognized as magnets for poaching trophy class big game animals. ORVs not only improve access to these remote areas, but they provide a much quicker means of removing illegally shot big game animals than would foot travel. In addition to increasing access, illegal use of ORVs as chase vehicles during hunts for pronghorn (*Antilocapra americana*) has been reported in the western United States (Canfield et al. 1999).

Although the affects of motorized aided hunting on harvest success, changes to population structure and increases in illegal harvest of wildlife have not been investigated, there is substantial literature to document the impacts of motorized recreation on wildlife disturbance and habitat degradation. The Montana chapter of The Wildlife Society recently compiled an extensive bibliography on the impacts of all types of wildlife recreations, including ORVs, on wildlife (Joslin and Youmans 1999). The following review draws heavily on relevant studies cited in that report. Additional studies were identified using keyword searches in the worldwide wildlife ecology, fisheries management, and environmental science and pollution management databases.

Research Findings

In the literature, the element of surprise seems to create especially high stress as a disturbance factor for many species. Consequently, some findings report that the flight responses of animals (e. g., cervids) is often of a higher intensity when induced by activities like cross-country skiing where the approach of humans is quieter and of longer duration than disturbance events created from motorized recreation (Freddy 1986, Cassirer et al. 1992). Some songbirds are apparently more disturbed from the approach of joggers than from passing or stopping vehicles (Bennett and Zuelke 1999). Taken out of context, it would be tempting to argue that the "silent sports" are more detrimental to wildlife than machine-based recreations. But, perspective can be regained from Boyle and Sampson's (1986) seminal review of nonconsumptive recreational impacts on wildlife where they concluded that most wildlife activities produce negative affects on wildlife. Therefore, rather than describing the impact of ORV driving relative to other activities, instances are reported where motorized disturbance has been found to alter the behaviors and fitness levels of a number of mammal and bird species. In other words, this investigation is not about which activities are most or least harmful; it is a review of the negative impacts of ORVs on wildlife. Aside from a few exceptions where ORV impacts were not found to be detrimental (e. g., Wolcott and Wolcott 1984), studies detailing the ways in which wildlife and natural habitats have benefitted from increased ORV use could not be located.

Disturbance Impacts

Both the behavioral and physiological responses to motorized recreation of many North American mammal and bird species have been investigated. Wildlife may either become acclimated to such disturbance, become nocturnal to avoid human pressure or abandon a preferred location altogether. All three options pose varying degrees of energy costs in terms of elevated heart rates or denied access to forage. Among the North American ungulates, bighorn sheep may be the most vulnerable to disturbance by human presence (Canfield et al. 1999). Studies have shown negative behavioral and physiological responses of bighorns to the presence of snowmobiles and overhead helicopters. Meanwhile, Bear and Jones (1973) found that ORV traffic negatively influences sheep distribution. Due to the sensitivity of these animals, one can assume the increased accessibility provided to their rocky, steep habitat would pose elevated stress levels. Consequently, Canfield et al. (1999) cited 25 studies, conducted since 1956, that have called for restrictions on vehicle access to sheep habitats.

Yarmoly et al. (1988) experimentally chased mule deer (*Odocoileus hemonus*) does with ATVs to test the affects of harassment on behavior and reproductive success. They found that mule deer chased on a daily basis through the forest (off-trail) for a three-week period became more nocturnal than control specimens and, on average, had fewer sets of twin fawns. Although this experiment may not reflect typical ORV use, other studies have found that habitat displacement along established ORV routes can occur for black-tailed deer (*Odocoileus hemonus columbianus*) (Ferris and Kutilek 1989) and elk (Marcum and Edge 1991).

There is also an extensive body of literature that shows displacement and avoidance of roads by many species, such as wolves (*Canis lupus*) and grizzly bears. Those studies are not reviewed here, yet it would not be an overextrapolation to reason that heavily used recreation motor trails may result in avoidance. Certainly, off-trail ORV use could be expected to disturb denning sites for large mammals that have been shown to abandon areas following increased motorized activity (Claar et al. 1999).

Several experimental studies have quantified the disturbance caused from ORVs on nesting bird species. Of particular concern are the risks posed to bird clutches by flushing females from the nests, exposing eggs or young to cold, and elevating predation risks. For example, Rodgers and Smith (1997) measured the flush distance of various shorebirds approached by four different types of human disturbance, including ATVs. Based on their data, they recommended establishment of buffer zones to minimize disturbance and to avoid the negative affects of repeated flushing of females from their nest sites. Establishment of ORV buffer zones around nest sites has also been recommended for some raptor species (Dubois and Hazelwood 1987) and Hamann et al. (1999) pointed out that some waterfowl species, such as black ducks (*Anus rubripes*) and harlequin ducks (*Histrionicus histrionicus*), require nesting areas free from any type of human disturbance.

ORV driving not only has the potential to disturb nesting birds, but also to run over the nest sites of ground nesting species. Palacios and Mellink (1996) located 29 potential breeding sites for the endangered least tern (*Sterna antillarum*) and found that ORV use was the main limiting factor for utilization of nesting sites by terns. Nesting plover species have exhibited a decline in reproductive success in areas where ORVs have been driven (Gaines and Ryan 1988, Strauus 1990, Wershler 1991) as defined by chick survival.

Hamann et al. (1999) stated that songbird response to being flushed from their nests by approaching ORVs ranges from habituation to nest abandonment. In general, interior forest species, like the brown creeper (*Certhia americana*), red-breasted nuthatch (*Sitta canadensis*) and hermit thrush (*Catharus guttatus*), were examples of species less tolerant to disturbance. Work done by Gutzweiller et al. (1997, 1998) noted that singing by forest songbirds was suppressed by regular, nonmotorized hiking and that the affects were especially apparent among species that nest on or near the ground. They have inferred that the suppression affects on reproductive behavior would hold true for ORV use as well; although, that hypothesis has not been specifically tested.

Habitat Degradation

Although the disturbance impacts of ORV use on wildlife are difficult to observe and measure, the visible and lasting effects of habitat modification and degradation are easier to document. Five broad and interrelated categories of ORV impacts on wildlife habitats and ecosystems will be described. ORVs have been shown to:(1) reduce diversity and resiliency of vegetation in sensitive plant communities, such as wetlands, bogs, deserts and beach dunes; (2) facilitate the spread of alien and noxious weed species; (3) increase soil erosion and compaction; (4) exacerbate siltation and disrupt streambeds; and (5) contribute to habitat fragmentation and edge effects through the development of trail systems. For the most part, these five categories of habitat effects can be considered deleterious to the maintenance of biodiversity and natural ecological processes.

Sensitive Communities

Certain ecosystem types appear less resilient to disturbance and, therefore, more vulnerable to impacts of vehicle traffic. For example, sandier soils, especially those in more arid environments, are less able to sustain damage and recover from ORV use (Belnap 2002). Reduced levels of floral diversity in desert and beach environments have been linked to ORV traffic. (Iverson et al. 1981, Adams et al. 1982, Anders and Leatherman 1987, Wester 1994). The mechanism for reduced plant diversity has not been identified in all cases, but reduced availability of nitrogen and associated destruction of lichens, fungi and algal crusts in compacted soils may explain reduced plant biomass and diversity (Wilshire 1983, Belnap 2002).

One such study compared plant and animal diversity in a dune community with plots open and closed to ORVs (Luckenbach and Bury 1983). They found reduced levels of herbaceous and perennial plants, arthropods, lizards and mammals in all ORV used plots. They tested both heavy and light ORV treatments and found that biota was reduced in light traffic areas and virtually nonexistent in the high traffic plots. Furthermore, the reduced plant species included several dune specialists that were considered rare or threatened.

Reduction in plant life has also been shown in wet communities as well where ATVs have been found to leave a lasting footprint in wetland. Hannaford and Resh (1999) compared the damage to pickelweed (*Salicornia virginica*) biomass from two types of ATVs (Argo Super and Lightfoot Super) driven lightly (2 passes) and heavily (20 passes). There was an immediate reduction in stem height and biomass for all treatments, but biomass did recover the following year. However, vehicle paths remained visible in the vegetation one year later for the heavy use trail left by the Lightfoot Super.

Racine and Ahlstrand (1991) experimentally compared the impacts of ATV riding and driving 2,640-pounds (1,200-kg) tracked Weasels on permafrost in Alaska and found effects differed seasonally. They concluded that ATV impacts could produce greater surface thaw than the heavier vehicles early in the season and that thawing depth increased with the number ATV rides taken.

Spreading Exotics

In addition to diminished biomass and reduced native plant diversity, ORV use has also been linked to the spread of exotic species. Clampitt (1993) blamed ORV traffic and fire suppression for the irreversible replacement of native prairie species with a less diverse assemblage dominated by weedy exotics. ATVs have been implicated in the spread of spotted knapweed (*Centaurcea maculosa*), an aggressive, exotic weed species. Lacey et al. (1997) found that thousands of seeds could hitchhike on ATV undercarriages and be transported for up to 10 miles (16 km). ATV riding along highway right-of-ways, especially on soft ground or newly sodded areas, disrupts native plant growth (CTC & Associates LLC 2003). Finally, ORV riding was, at least partially, to blame for the displacement of an endangered and endemic fern species in Hawaii by exotic weeds that thrived in disturbed soil (Wester 1994). The competitive dominance of exotics over native grasses and forbs generally reduces the quality and quantity of forage available to many North American herbivores.

Soil Erosion and Compaction

Griggs and Walsh (1981) used aerial photos, ground surveys and sediment load analysis to document an increase in sediment discharge, severe soil erosion and gullies caused over the course of a decade of ORV traffic in one arid California valley. Iverson et al. (1981) reported that surface water run-off was five times greater in arid landscapes allowing vehicle traffic than in controlled plots. They also found the sediment run-off was 10 to 20 times greater on vehicle-compacted soils. The authors estimated that recovery time for impaired arid environments was 100 years. Increased soil erosion due to ORVs has also been found to escalate storm erosion on beaches (Anders and Leatherman 1987). Adams et al. (1982) reported intense soil compaction in desert areas from ORVs on heavily used trails and campsites. The affects of ORV traffic on forested areas at different stages of succession appear to be an overlooked area of research.

Stream Damage

Brown (1994) measured sedimentation rates downstream from ORV crossings on two Australian river fords. In his work, Brown noted that ORV river crossings produce multiple factors that increase sedimentation rates downstream. These factors include the creation of wheel ruts that channel surface water run-off; compaction and the subsequent reduction in soil infiltration rates, leading to increased run-off; and the undercutting of banks by bow wave action.

Evans (2001) found significantly lower diversity and quantity of benthic macroinvertebrates when comparing sites downstream from ORV crossings to areas closed to ORVs on three streams in Texas. Benthic macroinvertebrates, such as mayflies (*Ephemeroptera* sp.) and stone flies (*Plecoptera* sp.), are commonly used to assess water quality, and their decline is generally associated with lower oxygen levels, higher sedimentation or increased inorganic pollutants.

In addition to the potential negative affects on aquatic invertebrates and fish species, Waller et al. (1999) pointed out that semiaquatic mammals, such as beaver (*Castor canadensis*), river otters (*Lutra canadensis*), muskrat (*Ondrata zibethicus*) and mink, are all vulnerable to disturbance and habitat degradation caused by ATV use along river banks and in riparian zones. For example, stream bank erosion and destabilization has been linked to elimination of beaver populations from localized areas (Bown 1988).

Edge Effects

The impacts noted so far theoretically apply to ORV traffic both on and off trail systems. The last category of habitat impacts discussed relates specifically to impacts created from trail systems. Whether for pedestrian hiking or ATV travel, trails create linear features that alter microhabitats, especially in forest and grassland communities (Askins 1994). For example, the impacts of snowmobile trails on wildlife have been widely studied. The results have been somewhat mixed. Wolves may use packed trails in the winter to conserve energy and access winter deer yards easier (Okarma 1995, Paquat et al. 1996). Lynx (*Lynx canadensis*) seem to tolerate moderate snowmobile activity but may ultimately lose out to other predators that can take advantage of packed snow trails to neutralize the competitive advantage of the cats' lighter foot loading (Mowat et al. 1999).

Seemingly, most ATV traffic on trails would not occur in winter, yet the trails affect communities with or without traffic present. Trails can break up forest patches, exposing bird species to the dual impacts of nest parasitism by brown-headed cowbirds (*Molothurus ater*) and increased predation rates by mesocarnivores (Miller et al. 1998). Rich et al. (1994) found these negative edge effects on utility corridor right-of-ways as narrow as 26.2 feet (8 m) wide. Meanwhile, Hickman (1990) reported hiking trails as narrow as 9.8 feet (3 m) negatively impacted forest nesting birds in Illinois. The implication is that all trails, regardless of their use, bring some negative consequences for local fauna.

Summary of Impacts

Research has identified a number of harmful impacts of ORV use on both aquatic and terrestrial habitats that include replacement of natural forage with weed species, stream bank erosion, stream siltation and soil compaction. Communities comprised of unique floral and fauna assemblages, such as wetlands, bogs and beach dunes have been shown to be particularly vulnerable to vehicle use. In addition, improved access through the creation of legal and illegal trails can alter wildlife behavior. Repeated disturbance during critical seasons or in critical areas may affect populations where stress leads to reduced fitness.

Riding Motivations and Their Consequences

Though my primary focus has been to review the ecological impacts resulting from ORVs, social aspects of ORV riding can undermine wildlife management objectives. For example, the growing intolerance for destructive ATV riding on private land in Maine has resulted in fewer landowners granting access to hunters, a trend that compromises the agency's ability to achieve population goals for white-tailed deer (*Virginianus odocoileus*) and moose (*Alces alces*) (T. Peabody, personal communication 2003). Although membership in ATV riding and hunting subcultures is not uniformly shared, some frustrated private landowners are denying access to everyone without distinguishing between segments.

Landowner or public perception that conflates ORV riders with hunters should not surprise anyone, given the intense marketing of ATVs in the hunting and outdoor media. Magazine ads and television commercials on The Outdoor Channel depicting mud-drenched warriors in camouflage with a buck draped on the back are what help sell these machines. Two consequences of these types of images emerge. One, slob behavior (e. g., littering) may not be differentiated by landowners who equate hunters with ATV riding. Two, the marketing also creates expectations among ATV riders (regardless of hunting participation) that are undesirable from a wildlife management perspective. How many fewer machines might be sold if ads portrayed images of families riding single-file down an established, gravel trail? Once you begin to consider motivations and appeals of riding ATVs, then you can grasp the real challenge in regulating the abuse of ecosystems. Although some enthusiasts contend that expanding trails alone will address problems raised in this paper, research suggests otherwise.

Tochner (1980) and Rodgers (1999) both identified riding on challenging terrain as one of the primary motivations of ATV riders. If challenging terrain means steep slopes, muddy banks and unblazed trails, then increasing educational efforts and building more riding trails will not likely ameliorate all of the problems caused by ATVs. It is naïve to assume that there will not remain a segment for which the desire to go ripping up the side of a highly erodable slope or "mudding" in the sphagnum moss is too strong to resist. After all, that is the very reason many people enjoy riding. Furthermore, a recent *New York Times* article interviewed some law enforcement officials who noted that some of the unethical ATV behavior was motivated by civil disobedience from a renegade segment of drivers who resent government restrictions on their rights to ride anywhere. Setting aside the question of riding motivations, the sheer number of ORV riders combined with the potential of their machines to cause lasting ecological damage suggests the need to revisit polices that govern ATV riding on state, federal and provincial land.

Policy, Education and Enforcement

Creating policies for managing wildlife disturbance and damage from ORV use will require a three-pronged attack: increased designation and development of legal ORV riding trails, more comprehensive educational programs, and increased funding and authority for law enforcement. First, a case can be made for the need to expand riding areas and trail networks for ATV enthusiasts. Everyone would also agree that planned trails are preferable to unplanned trails. Well-designed trails offer riders a place to go, while directing traffic away from more ecologically sensitive areas, such as riparian zones. The Governor's Task Force in Maine has recommended trail expansion as one of its strategies for curbing trespass by ATV users. Several states in the United States are currently pursing legislation to clarify laws regarding areas open and closed to ATV use. Such changes and an increased patrol presence may curtail some of the off-trail activity. In Michigan state forests, the number of illegal trails was reduced when the policy was changed to a "closed unless posted open" one (Nelson 1996).

Second, education should continue to promote responsible ATV use. The national Tread Lightly! campaign, as well as many local trail patrols and education programs sponsored by ORV organizations, are positive developments. Yet, few states require new riders to attend certification and education classes akin to the hunter safety programs. Only 16 states currently require certification programs for ATV use; no states require adult certification. Even where ATV education programs are being implemented, the emphasis is on user safety and not on environmental ethics. Although no one would argue that safety training is essential, more emphasis needs to be placed on the ecological and social repercussions of thrill riding through wetlands and other vulnerable areas.

Another challenge to education programs has broad policy implications as well. Fragmented jurisdictions and a patchwork of ORV rules at national, state and local levels frustrate law-abiding ORV riders. Some counties and townships may create local ATV ordinances that are more restrictive or more liberal than state laws. Such inconsistent policies between and among agencies and jurisdictions should be avoided. In the least, users need better education and information about where they can ride legally.

Third, the ability of state conservation officers to respond to ATV issues is limited by a lack of adequate penalties and funding for enforcement. As one example, current funding allows the average Wisconsin warden just 8 hours of ATV enforcement per year. Although county deputies also have enforcement authority for ATV operation, they seldom have the time or inclination to get involved in crimes against natural resources. A report by the U. S. General Accounting Office (1995) found that enforcement by federal agencies was also lacking, due, in part, to the lack of resources combined with insufficient prioritization by land managers. As a result, catching unethical ATV operators is a rare occurrence. Therefore, agencies should consider raising registration fees on ORV registrations and dedicating a larger portion of those fees toward active enforcement programs that are done in conjunction with local trail and riding clubs. This is a politically sensitive recommendation because ORV riders will inevitably pressure lawmakers to dedicate increased revenue to trail systems rather than enforcement.

A larger enforcement presence will only be effective if equipped with broader citation authority. Currently, most state conservation officers can issue citations for violations, such as failing to have proper registration, riding on public roads or riding in closed areas. Only the latter example begins to get at the goal of resource protection. Few laws are available for creating a deterrent for the reckless destruction of habitat. Violators of fish and wildlife crimes are sometimes charged a restitution fee in addition to the citation for the violation. The latter is the fine for the illegal act, but the former is the reimbursement to the citizens for the resource that was stolen. Perhaps, it is time that agencies enact tougher penalties for habitat destruction wrought by illegal and unethical ORV operation that include restitution for environmental damage.

In conclusion, the objective of this paper has been to call attention to documented impacts of ORVs on wildlife and their habitat and to suggest the need for more policy, education and enforcement action to curtail those problems. It was not my intention to indict all or even most ORV operators. As with any outdoor recreation, the vast majority might be law-abiding, well-meaning participants. However, the number of registered ORV riders exceeds hunters by well over a 2:1 ratio in the United States. The sheer volume of users alone dictates that we consider strategies that allow use within the constraints of maintaining a healthy environment. Their impacts—real and potential, deliberate or unintentional—can no longer be ignored. Although a poached deer represents a temporary and singular loss of wildlife benefits, ORV disturbance and damage can be lasting and pervasive. The capacity of ecosystems to absorb such a high impact deserves careful planning. In the final analysis, we must recognize that just because a machine is designed to traverse all-terrains does not mean it should always be allowed to do so.

Reference List

Adams, J. A., L. H. Stolzy, A. S. Endo, P. G. Rowlands, and H. B. Johnson. 1982. Desert soil compaction reduces plant cover. *California Agriculture*. 36(9–10):6–7.

Anders, F. J., and S. P. Leatherman. 1987. The effects of off-road vehicles on coastal foredunes at Fire Island, New York. *Environmental Management*. 11(1):45–52.

Askins, R. A. 1994. Open corridors in a heavily forested landscape: Impact on shrubland and forest interior birds. *Wildlife Society Bulletin*. 22:339–47.

Bear, G. D., and G. W. Jones. 1973. *History and distribution of bighorn sheep in Colorado*. Denver, Colorado: Colorado Division of Wildlife.

Belnap, J. 2002. Impacts of off-road vehicles on nitrogen cycles in biological soil crusts: Resistance in different U. S. deserts. *Journal of Arid Environments*. 52(2):15–165.

Bennett, K. A., and E. Zuelke. 1999. *The effects of recreation on birds: A literature review*. Smyrna, Delaware: Delaware Natural Heritage Program.

Bown, R. 1988. *Beaver habitat along rivers and reservoirs in central Montana*. M. S. Thesis. University of Montana, Missoula, Montana.

Boyle, S. A., and F. B. Samson. 1986. Effects of nonconsumptive recreation on wildlife: A review. *Wildlife Society Bulletin*. 13:110–16.

Brown, K. J. 1994. River-bed sedimentation caused by off-road vehicles at river fords in the Victorian Highlands, Australia. *Water Resources Bulletin*. 30(2):239–50.

Canfield, J. E., L .J. Lyon, J. M. Hillis, and M. J. Thompson. 1999. Ungulates. In *Effects of recreation on rocky mountain wildlife: A review for Montana*, eds. G. Joslin, and H. Youmans, 6.1–6.25. Helena, Montana: Committee on Effects of recreation on Wildlife, Montana Chapter of the Wildlife Society.

Cassirer, E. F. D. J. Freddy, and E. D. Ables. 1992. Elk responses to disturbance from cross country skiers in Yellowstone National Park. *Wildlife Society Bulletin*. 20(4):375–81.

Claar, J. J., N. Anderson, B. Boyd, B. Conrad, G. Hickman, R. Hompesch, G. Olson, H. Ihsle, T. Wittiger, and H. Youmans. 1999. Carnivores. In *Effects of recreation on rocky mountain wildlife: A review for Montana*, eds. G. Joslin, and H. Youmans, 7.1–7.62. Helena, Montana: Committee on Effects of recreation on Wildlife, Montana Chapter of the Wildlife Society.

Clampitt, C. A. 1993. Effects of human disturbance on prairies and regional endemic *Aster curtis* in western Washington. *Northwest Science*. 67(3):163–9.

CTC & Associates LLC. 2003. *Reducing ATV impact on highway right of ways. A synthesis report for the Bureau of Highway Operations Division of Transportation infrastructure*. Madison, Wisconsin: Wisconsin Department of Transportation.

Dubois, K., and R. Hazelwood. 1987. Raptors. In *Interagency Rocky Mountain front range wildlife monitoring and evaluation program*, 29–37. Lewiston, Montana: U. S. Bureau of Land Management.

Evans, D. A. M. 2001. *A benthic macro invertebrate and physiochemical analysis of the effects that off-road vehicles have on water quality at three stream crossing sites in the Angelina National Forest in east Texas*. M. S. thesis. Stephen F. Austin State University, Huntsville, Texas.

Ferris, R. M., and J. Kutilek. 1989. *Responses of black tailed deer to off-highway vehicles in Hollister Hills State Vehicular Recreation Area.*

San Jose, California: San Jose State University, Department of Biological Sciences.

Freddy, D. J. 1986. Responses of adult mule deer to human harassment in during winter In *Proceedings II: Issues and technology in the management of impacted western wildlife*, eds. R. D. Comer, T. G. Baumann, P. Davis, J. W. Monarch, J. Todd, S. VanGytenbeek, D. Willis, and J. Woodling, 286. Boulder, Colorado: Thorne Ecological Institute.

Gaines, E. P., and M. R. Ryan. 1988. Piping plover habitat use and reproductive success in North Dakota. *Journal of Wildlife Management*. 52(2):266–73.

Griggs, G. B., and B. L. Walsh. 1981. The impact, control and mitigation of off-road vehicle activity in Hungry Valley, California. *Environmental Geology*. 3(4): 229–43.

Gutzwiller, K. J., H. M. Marcum, H. B. Harvey, H. D. Roth, and S. H. Anderson. 1998. Bird tolerance to human intrusion in Wyoming montane forests. *Condor*. 100:519–27.

Gutzwiller, K. J., E. A. Forese, S. H. Anderson, and C. A. Wilkins. 1997. Does human intrusion alter the seasonal timing of avian song during breeding periods? *Auk*. 114:55–6

Hamann, B., H. Johnston, J. Gobielle, M. Hilles, S. Johnson, L. Kelly, and P. McClelland. 1999. Birds. In *Effects of recreation on Rocky Mountain wildlife*, Joslin, G., and H. Youmans, 3.1–3.34. Helena, Montana: Committee on effects of Recreation on Wildlife, Montana Chapter of The Wildlife Society.

Hannaford, M. J., and V. H. Resh. 1999. Impact of all-terrain vehicles on pickleweed (*Salicornia virginica L.*) in a San Francisco Bay wetland. *Wetlands Ecology and Management*. 7(4):225–33.

Havlick, D. G. 2002. *No place distance: Roads and motorized recreation on America's public lands*. Washington, DC: Island Press.

Hickman, S. 1990. Evidence of edge species attraction to nature trails within deciduous forest. *Natural Areas Journal*. 10:3–5.

Iverson, R. M., B. S. Hinckley, R. M. Webb, and B. Hallet. 1981. Physical effects of vehicular disturbance on arid landscapes. *Science*. 212 (4497):915–7.

Joslin, G., and H. Youmans. 1999. *Effects of recreation on rocky mountain wildlife: A review for Montana*. Helena, Montana: Committee on

Effects of recreation on Wildlife, Montana Chapter of the Wildlife Society.

Knight, R. L., and D. N. Cole. 1995. Wildlife responses to recreationists in Wildlife and recreationists: Co-existence through management and research, eds. Richard L. Knight, and Kevin J. Gutzwiller, 51–69. Washington, DC: Island Press.

Lacey, C. A., J. R. Lacey, P. K. Fay, J. M. Storey, and D. L. Zamora. 1997. *Controlling knapweed on Montana Rangeland.* Bozeman, Montana: Montana State University Extension.

Luckenbach, R. A., and R. B. Bury. 1983. Effects of off-road vehicles on the biota of the Algodones Dunes, Imperial County California. *Journal of Applied Ecology*. 20(1):265–86.

Marcum, C. L., and W. D. Edge. 1991. Sexual differences in the distribution of elk relative to roads and logged areas of western Montana In *Proceedings of the elk vulnerability symposium*, compilers A. G. Christiansen, L. J. Lyon, and T. N. Lonner, 142–8. Bozeman, Montana: Montana State University.

Miller, S. G., R. L. Knight, and C. K. Miller. 1998. Influence of recreational trails on breeding bird communities. *Ecological Applications*. 8:162–9.

Mowat, G. K. G. Poole, and M. O'Donoghue. 1999. Ecology of lynx in northern Canada and Alaska In *General technical report RMRS-GTR-30.* Boulder, Colorado: U. S. Department of Agriculture, Forest Service.

Nelson, C. M. 1996. Registered Michigan off-road use and users: Forest policy implications. *Northern Journal of Forestry*. 13(2):84–8.

Okarma, H., B. Jedrzejewski, W. Jedrzejewski, L. Milkowski, Z. Krasinski. 1995. The trophic ecology of wolves and their predatory role in ungulate communities in forest ecosystems in Europe. *Acta Theriologica*. 40:335–86.

Palacios, E., and E. Mellink. 1996. Status of the least tern in the Gulf of California. *Journal of Field Ornithology*. 67(1):48–58.

Paquet, P., C. J. Weiczhhowski, and C. Callaghan. 1996. *Summary report on the effects of human activity on gray wolves in Bow River Valley, Banff National Park, Alberta.* Ottawa, Ontario: Department of Canadian Heritage.

Racine, C. H., and G. M. Ahlstrand. 1991. Thaw response of tussock-shrub tundra to experimental all-terrain vehicle disturbances in south-central Alaska. *Arctic*. 44(1):31–7.

Rich, A. C., D. S. Dobkin, and L. J. Niles. 1994. Defining forest fragmentation by corridor width: The influence of narrow forest-dividing corridors on forest nesting birds in southern New Jersey. *Conservation Biology*. 8:1,109–21.

Rodgers, G. B. 1999. The characteristics and use patterns of all-terrain vehicle drivers in the United States. *Accident Prevention and Analysis*. 31:409–19.

Rodgers, J. A., Jr., and H. T. Smith. 1997. Buffer zone distances to protect foraging and loafing shorebirds from human disturbance in Florida. *Wildlife Society Bulletin*. 25(1):139–45.

Strauss, E. G. 1990. *Reproductive success, life history patterns, and behavior variations in a population of piping plovers subjected to human disturbance*. Ph. D. Dissertation, Tufts University.

Tanz, J. 2004. Making tracks, making enemies. *New York Times*. January 2.

Tocher, R. 1980. Four-wheel vehicles: Summary. In *Proceedings of off-road vehicle use: A Management Challenge*, 286–9. Ann Arbor, Michigan: University of Michigan, School of Natural Resources.

U. S. Environmental Protection Agency. 2002. *Summary and analysis of comments: Control of emissions from unregulated nonroad engines*. Washington, DC: U. S. Environmental Protection Agency.

U. S. General Accounting Office. 1995. *Federal lands: Information on the use and impact of off-highway vehicles*. Washington, DC: General Accounting Office.

Waller. A. J., C. A. Sime, G. N. Bissell, and B. G. Dixon. 1999. Semi-aquatic mammals In *Effects of recreation on rocky mountain wildlife: A review for Montana*, eds. Gayle Joslin, and Heidi Youmans, 5.1–5.25. Helena, Montana: The Wildlife Society, Committee on Effects of recreation on Wildlife, Montana Chapter.

Wester, L. 1994. Weed management and the habitat protection of a rare species: A case study of the endemic Hawaiian fern Marsilea villosa. *Biological Conservation*. 68(1):1–9.

Werschler, C. 1991. A management strategy for mountain plovers in Alberta. In *Proceedings of the 2nd endangered species and prairie conservation workshop, occasional paper No. 15*. Regina, Saskatchewan: Provincial Museum of Natural History.

Wilkerson, T. 2002. *On the beaten path*. National Parks. 75(3):34.

Williams, T. 2003. Off-road rage. *Audubon*. 105:78.

Wilshire , H. G. 1983. The impacts of vehicles on desert soil stabilizers In *Environmental effects of off-road vehicles: Impacts and management in arid regions*, eds. R. H. Webb, and H. G. Wilshire, 31–50. New York, New York: Springer-Verlag.

Wolcott, T. G., and D. L. Wolcott. 1984. Impact of off-road vehicles on macroinvertebrates of a mid-Atlantic beach. *Biological Conservation*. 29(3): 217–40.

Yarmoly, C., M. Bayer, and V. Geist. 1988. Behavior responses and reproduction of mule deer does (*Odocoileus hemionus*) following experimental harassment with an all-terrain vehicle. *Canadian Field Naturalist*. 102(3):425–9.

Youmans, H. 1999. Project overview. In *Effects of recreation on Rocky Mountain wildlife,* coordinators Joslin, G., and H. Youmans, 1.1–1.18. Helena, Montana: The Wildlife Society, Committee on Effects of recreation on Wildlife, Montana Chapter.

Safe and Responsible All-terrain Vehicle Use in the United States

Thomas S. Yager
Specialty Vehicle Institute of America
Irvine, California

Since 1983, the Specialty Vehicle Institute of America® (SVIA) has promoted the safe and responsible use of all-terrain vehicles (ATVs) through rider training programs, public awareness campaigns and state legislation. SVIA also serves as a resource for ATV research, statistics and vehicle standards. SVIA, based in Irvine, California, is a not-for-profit trade association sponsored by AlphaSports, Arctic Cat, Bombardier, Honda, John Deere, Kawasaki, Suzuki and Yamaha.

All-terrain Vehicles Defined

Just what is an ATV? ATVs are defined in American National Standard. This voluntary standard addresses design, configuration and performance aspects of ATVs. The standard defines an ATV as any motorized off-highway vehicle designed to travel on four low pressure tires, having a seat designed to be straddled by the operator and handlebars for steering control, and intended for use by a single operator and no passenger.

ATVs are subdivided into four categories as follows.

1. Category G (General Use Model) ATVs are intended for general recreational and utility use.
2. Category S (Sport Model) ATVs are intended for recreational use by experienced operators only.
3. Category U (Utility Model) ATVs are intended primarily for utility use.
4. Category Y (Youth Model) ATVs are intended for recreational off-road use under adult supervision by operators under age 16. Category Y ATVs can further be categorized as follows.
    a. Category Y-6 ATVs are category Y ATVs, which are intended for use by children age 6 and older.
    b. Category Y-12 ATVs are category Y ATVs, which are intended for use by children age 12 and older. (Specialty Vehicle Institute of America 2001).

Many states define ATVs differently from this published definition. However, for the purposes of this discussion, ATV refers to the ANSI SVIA definition.

Motorized Recreation Continues to Grow

Consumer demand for ATVs and off-highway motorcycles (OHMs) has grown significantly over the past 10 years. It's estimated that 5.6 million ATVs were in use nationwide in 2001, compared to 4 million in 1997. This represents a 40-percent increase from 1997 to 2001 (Levenson 2003).

New ATV sales experienced phenomenal growth in the early 1980s, then sharply declined from a peak of 550,000 units to under 150,000 in 1991. A resurgence in growth for new ATVs sales occurred from 1992 through 2003, with ATV sales in 2003 estimated at 800,000.

In 2003, sales of new OHMs and ATVs combined were 1,113,300 units, the highest level since 1982. In comparing sales by region from 1997 to 2003, the West saw the greatest increase, at 220 percent, followed by the East, at 176 percent, the Midwest, at 139 percent, and the South, at 87 percent. This yielded a 140 percent growth in sales nationally from 1997 to 2003.

Economic Value

The U. S. economic value of the OHM and ATV retail market for 1998 was $18 billion. New and used vehicle sales, parts, accessories and service contributed $6.1 billion and an additional $11.9 billion is estimated for state sales tax, dealer personnel salaries, income taxes, financing interest, insurance premiums and OHV trip expenses for 1998.

Who Owns ATVs

In 2003, the Motorcycle Industry Council (MIC) conducted the 2003 MIC Motorcycle/ATV Owner Survey (Motorcycle Industry Council 2004). This was a national probability telephone survey of all motorcycles and ATVs in use. The sample included 2,018 households and interviews of the primary vehicle rider were conducted over 12 months, from October 2002 through September 2003.

The average age of the ATV owners was 37 years, compared to 31 years for OHM owners. This age difference accounts to a large degree for many of the other demographic differences between ATV and OHM owners, such as marital status, occupation, education and income.

Almost 60 percent of ATV owners were married, compared to about 50 percent of the OHM owners. The majority of owners were male; 87 percent of the ATV owners were male, compared to 94 percent of the OHM owners. Female ownership was slightly higher among ATVs than OHMs.

Owners were asked about their principal occupation. Professional or managerial positions were most frequently held by both ATV and OHM owners, with little difference between ATV owners and OHM owners.

Laborers, semiskilled laborers, farm laborers, mechanics or craftsman were the next most common occupations among both ATV and OHM owners. The portion of retired or unemployed owners was much higher among ATVs. Students were more common among OHM owners. The age difference between these two owner groups accounts for some of the differences in occupation, education, marital status and household income.

ATV owners tend to have reached a higher level of education than OHM owners. About 40 percent of the ATV owners had at least some college education. Both ATV and OHM owning households tend to have a higher annual household income than the average U. S. household.

The median household income for ATV households was $53,800 and $55,900 for OHM households, compared to $39,000 for U. S. households (in 1998).

How ATVs Are Used

Part of what makes ATVs so popular is the wide variety of ways they are used. From replacing expensive tractors on the farm to riding fence lines to protecting our nation's security, ATVs are on the job.

In addition to the tremendous utility aspects of ATVs, they are used for recreation by individuals and families to enjoy the great outdoors. Whether traveling to that favorite hunting spot or fishing hole, an ATV is a valuable and fun way to go.

ATV owners were asked what other recreational interests or activities they used their ATV for. Pleasure riding (69 percent) followed by hunting (49 percent) and sightseeing (40 percent) were the most mentioned activities ATV owners engaged in. Camping and fishing were tied at 22 percent while 7 percent of ATV owners used them in conjunction with their hiking and backpacking activities.

Industry's Commitment to Responsible ATV Use

Rider Training

The ATV Safety Institute (ASI) was formed in 1988 as a division of SVIA to implement an expanded national program of ATV safety and awareness that SVIA initiated in 1983.

ASI's primary goal is to foster and promote the safe and responsible use of ATVs in the United States, thereby reducing accidents and injuries that may result from improper use. Our programs are designed to inspire rider awareness that promotes a commitment to safety and respect for the environment.

There may be a perception that, once an ATV is purchased, the new owner doesn't hear from the industry again. Nothing could be further from the truth.

In most cases, within 48 hours, new ATV owners are contacted by the ASI to encourage them to enroll in the free ATV RiderCourse.

At the time of sale, the dealer completes a rider training certificate and faxes it to ASI. ASI enters the purchaser information into a database where it is transferred to an automated predictive dialer phone system the following day. The dialer automatically dials ATV purchasers. When a connection is reached a trained enrollment representative explains the benefits of the ATV RiderCourse, answers questions, restates the safety messages and enrolls the purchaser and eligible family members in a class. Once enrolled, the buyer is sent a confirmation letter with the specific class information, instructor name and number, what to bring to class, and a map on the back of the letter to the training site.

Only licensed instructors are authorized to teach the ATV Rider Course. All ASI instructors must complete a comprehensive training program and meet specific ASI requirements to become licensed.

The ATV Rider Course is a hands-on, half-day program that is available free of charge to all individuals who have purchased a participating company's ATV, including the purchaser's eligible family members.

The ATV RiderCourse offers its students an opportunity to increase their safety knowledge and to practice basic riding skills in a controlled environment under the supervision of a licensed instructor.

The class is conducted outdoors and has a maximum class size of eight students for one instructor. The main themes in the ATV RiderCourse are safety and responsible use. Environmental ethics are taught as well as riding skills and state and local laws and regulations for operating ATVs.

ASI's instruction is targeted as much at the parent as it is to the child. As a first step in the process, we help parents make the decision as to whether ATVing is appropriate for their child through the use of a publication called *Parents, Youngsters and All-terrain Vehicles*. This booklet includes a readiness checklist that covers visual perception, motor skills development, physical development, social and emotional development, and reasoning and decision-making ability. If parents have determined that ATV riding is the right activity for their child, we will train the child with participation from the parent.

Our students practice basic safety techniques covering starting and stopping, turning—both gradual and quick—negotiating hills, emergency stopping and swerving, and riding over obstacles. Particular emphasis is placed on the safety implications relating to each lesson.

The course also covers protective gear, environmental responsibility, and state and local laws. Participants receive an ATV RiderCourse handbook, which reinforces the safety information and riding techniques covered during the ATV RiderCourse.

We have just introduced a new program that allows prospective purchasers to take a training course first, then get their training fee reimbursed when they buy a new ATV from a member company. We focus our efforts on the first time purchaser without prior riding experience. The Consumer Product Safety Commission has identified these riders as those most likely to benefit from rider training. It also is available to all ATV riders who don't qualify for free training, such as purchasers of used ATVs or other prospective riders, for a reasonable fee.

In 2003, nearly 50,000 students completed the ASI's half-day, hands-on ATV RiderCourse and over half a million students have completed since 1988.

Parents hold the safety of their children in their hands, and they can provide their children with a fun and safe ATV experience by ensuring that they enroll themselves and their children in an ATV rider training course. They should also purchase the right sized ATV for their child's age, provide their child with protective gear and always—always—supervise their children under 16 whenever they ride. The primary means to accomplish this mission is the ATV RiderCourse.

ASI also works with state, armed forces, independent agencies and corporations to present seminars, develop safety materials and coordinate training programs targeted to fill specific needs.

Ride Safe, Ride Smart Video

ASI has a VHS video, titled *Ride Safe, Ride Smart*, that provides a rider-friendly look at how to get a proper start in ATV riding.

ATV Rally

A CD-ROM computer game, titled *ATV Rally*, emphasizes proper use and responsibility. These were distributed to schools across the United States, and they were free.

Public Service Messages

In addition to the ATV RiderCourse, ASI is committed to increasing public awareness of ATV safety programs. It produces and distributes public service messages to ATV enthusiast magazines and other publications read by potential ATV riders.

ASI materials promote the golden rules. These rules are reinforced, beginning at the dealer, continuing throughout the training experience, and extending through educational materials. In summary the golden rules follow.

- Always wear a helmet and other protective gear.
- Never ride on public roads; another vehicle could hit you.
- Never ride under the influence of alcohol or other drugs.
- Never carry a passenger on a single-rider vehicle.
- Ride and ATV that's right for your age; the guidelines are: (1) age 6 and older, use 70 cc, (2) age 12 and older, use 70 to 90 cc, and (3) age 16 and older, use more than 90 cc.
- Supervise riders younger than 16; ATVs are not toys.
- Ride only on designated trails and at safe speeds.
- Take an ATV RiderCourse; to enroll call (800) 887-2887.

Tread Lightly!

The ATV and OHM industry is a long-time supporter of Tread Lightly! and the Tread Lightly! environmental ethic. In 2003 ASI included the new Tread Trainer program, which promotes respect of the environment and outdoors, into the ASI Professional Development Workshop.

Right Rider

The Right Rider campaign is targeted at both ATV and other off-highway recreationists. It was developed in cooperation with many organizations and agencies. The principle messages included in the campaign are:

- ride right
- know where you're permitted to ride and where you're not
- share the trails and make friends with others
- volunteer
- pack it in, pack it out
- always use a spark arrester.

Over one million copies of the Right Rider brochure have been distributed through user groups and public land management agencies.

When Education Isn't Enough

The ATV industry strongly advocates environmental responsibility. Although most ATV riders drive their vehicles responsibly and stay on trails, there are a few who do not follow rules on our nation's public trails. To address this, SVIA was instrumental in introducing enforcement legislation in the 107th Congress to address the issues caused by the few who do not follow rules for responsible use on U. S. public land. With 23 cosponsors, we plan to have the bill reintroduced during the current session.

The legislation would increase penalties for those who knowingly and willfully cause damage to federal land. The Americans for Responsible Recreational Access (ARRA) hailed this legislation as an important step in returning public land to the U. S. citizens for their enjoyment.

Many agencies frequently opt to close public land to the public because of inadequate enforcement penalties. If enacted, this legislation would provide federal agencies with strong enforcement tools to encourage appropriate behavior on public land.

The legislation also permits federal agencies to recoup the cost of restoring damage to public land by the imposition of fines against offenders. In addition, fines can also be used by the agencies for educational activities to encourage proper conduct on public land.

No other private industry has implemented such far-reaching, ongoing and creative approaches to encourage socially and environmentally responsible use of vehicles.

Reference List

Levenson, Mark S., 2003. *All-terrain vehicle 2001 injury and exposure studies*. Bethesda, Maryland: Consumer Product Safety Commission.

Motorcycle Industry Council, 2004. 2003 *MIC motorcycle/ATV owner survey*. Irvine, California: Motorcycle Industry Council.

Specialty Vehicle Institute of America, 2001. *American national standard for four wheel all-terrain vehicles—Equipment, configuration, and performance requirements*. Irvine, California: Specialty Vehicle Institute of America.

Wildlife on Wheels: Marketing Today's Wildlife Experience on The Outdoor Channel

Jake Hartwick
The Outdoor Channel
Temecula, CA

Chris Chaffin
The Outdoor Channel
Temecula, CA

Introduction

Part of our challenge in this session today is to initiate a dialog that represents various sectors of the outdoor industry affected or involved with the wildlife experience on wheels. Having been involved with the wildlife management and outdoor industry for almost 30 years, I've been exposed to many sides of this issue and know full-well that off-highway vehicles (OHVs), all-terrain vehicles (ATVs), dirt-bikes, swamp buggies, three-wheelers, four-wheelers, Argos, Gators and all the rest have many faces.

I'm guessing every last one of you has a reasonably strong opinion of ATVs. And, depending on your perspective, we can legitimately question if the vehicles collectively are valuable tools that generate more interest and involvement in hunting and fishing because they allow greater mobility. Are they simply a means to enable citizens to see and do more, so they are more actively involved and enjoy the outdoors to a greater extent? Or, are they the very personification of the devil's spawn, as I have heard expressed in some circles?

I won't attempt to answer in my allotted time today, but I hope I will provide some insight into why the OHV market has a place on an outdoor television network—a network that stands firmly behind conservation and the important work you undertake every day in looking after our wild resources and the associated recreational adventures we all feel so strongly about.

My Charge Today

Wildlife harassment, habitat damage and user conflicts on public land arise from the operation of personal watercraft, ATVs and sport utility vehicles (SUVs) by people in pursuit of outdoor recreational experiences. That's a fact of life, of course, and there's obviously no need to question the matter. On the other hand, there are tens of thousands of sportsmen and sportswomen—all across the country—finding hunting, fishing, camping or other outdoor recreational success and fulfillment, at least partially, through the use of motorized vehicles. The overall scenario clearly has two sides of the proverbial coin.

Before I proceed further, I need to clarify that I am only attempting to address the topic in relation to The Outdoor Channel (TOC) and our attitudes about and positions on the use of motorized vehicles in the pursuit of outdoor recreation and enjoyment.

And, for those of you who are not ATV fans, let me repeat that I am making these comments under the overarching philosophy that TOC stands fast in its support for the important conservation work state, federal and provincial wildlife management agencies embrace on behalf of all outdoorsmen and outdoorswomen. It is our belief that sound conservation ethics and practices are the common ground for insuring our outdoor heritage.

Having said that, we believe that drawing lines in the sand, legislative or otherwise—seeking to outlaw specific outdoor activities in favor of others you or I personally find more appealing—puts our future on the proverbial "slippery slope."

It is our position that all users of the outdoors should have adequate opportunity to recreate and enjoy the outdoors, as long as those uses are legal. We support the tenor of this panel, which is to develop a shared understanding of resource issues associated with off-road vehicle operation among all who are interested in and concerned about our shared outdoor resources and opportunities.

A Little Insight into the Outdoor Channel

I think it's important that you have a little background in how TOC operates before I get into more details on our perspective about the place motorized, outdoor vehicles have on outdoor television.

To do that, let me first answer the inevitable question about why we have gold prospecting shows on TOC. The short answer is that, until just recently, The Gold Prospector's Association of America (GPAA) was essentially our parent company. The Massie family was looking for a way to get more exposure for GPAA and, of course, knew television was the medium to use.

During one of those discussions in the early days in Temecula, California, current TOC Executive Vice President, Jake Hartwick, recognized that they would need a larger audience to succeed and said the famous words, "Well, how about making it part of something like an outdoor channel?" So, today, although TOC has become successful enough to take the lead in the corporation, and the hunting and fishing shows now dominate the schedule, it was the gold prospecting that got us off the ground.

From the business prospective, TOC is a publicly traded corporation, meaning we have all the corporate responsibilities of making a profit, answering to a board of directors who have a legal responsibility to see that we make a profit, and supporting nearly 100 employees and another 100 or so independent producers that make a living or at least part of a living by producing outdoor programming. To do that, those producers have to pay for their time slot on TOC, which is typically accomplished by partnering with sponsors and by selling advertising time on their programs.

In the simplest terms, if TOC doesn't attract viewers, the advertisers don't buy time, the producers can't sell their advertising slots and then, they won't have an outdoor television show for long. Ultimately, TOC wouldn't exist.

So, the bottom line is TOC, like all television networks, has to have programming that viewers want to see—the larger the number of viewers, the more advertisers are willing to pay for the opportunity to taut their products and services.

The business need for a large and growing audience has a very direct effect on the programming you'll see on TOC. Beyond the baseline philosophy of being an outdoor network that presents family-oriented, traditional, outdoor sports—read that hunting and fishing—we are developing and scheduling programming that embraces the general demographic of people who like and pursue activities in the outdoors.

We are a niche network that doesn't intend to conquer the entire world . . . just the outdoor television world. In addition to the traditional outdoor programming we run, you will also find shows on country music, NASCAR, rodeo and other topics that connect those outdoor people. It is our philosophy that, if we can attract those people to watch their favorite niche programming, they will be exposed to the traditional hunting and fishing shows—with every likelihood of generating interest and future participation in hunting and fishing activities.

Off-highway Vehicles s on The Outdoor Channel

And, that brings us to the presence of ATVs, SUVs and the like on TOC. We believe OHVs and advertising for OHVs have a rightful and legal place on television. OHV manufacturers have the right to advertise a legally manufactured product in an advertising medium that reaches their existing and potential market.

As you probably already know, the manufacturers of outdoor, motorized vehicles have a strong following and user-base. These are people who like the outdoors, and many of them, I feel completely justified in saying, also have strong feelings about wildlife and other outdoor values.

The vehicles and the activities associated with them have proven to be popular with a fairly wide variety of users: hunters, campers, those who race their machines and the thousands who simply go pleasure riding or even wildlife watching.

It is important to point out, from our perspective, that the off-road vehicle market generates a rather good advertising income for TOC and we like profit. In our world, profit is a good thing.

Another important ingredient of this mix is that the programming and the advertising of motorized outdoor vehicles clearly connects with our audience—both outdoors enthusiasts and potential outdoors enthusiasts. Connecting with this segment of our audience, in our opinion, increases the recruitment potential of our outreach efforts.

The Off-road Industry

What can I tell you about the off-road industry? Again, I won't pretend to represent them, but I can share this with you about them.

- Most industry contacts report some 875,000 ATVs were sold in 2003 in the United States; although, others claim over 1 million, not including dirt-bikes. Taking the conservative figure, if the average cost per dirt-bike is

$4,000, that represents $3.5 billion in purchases, without counting accessories, operation, maintenance, repair, travel, food, motels, etc. That's a significant economic factor.

- One of the major OHV organizations reported to me that, literally dozens of land managers across the nation observe that much of the off-trail ATV riding that occurs during the year is the result of scouting or game retrieval during hunting seasons.
- That same source, who happens, by their own description, to be avid hunters say that, "hunters in most cases are probably unaware of the regulations regarding cross-country travel, the impacts of their actions on the ground, and the negative impacts it has on OHV recreation." He went on to say, "However, I believe that the vast majority of hunters will make the right choices when they understand what the 'right' things to do are."

We believe the councils and coalitions representing a significant segment of this country's off-road riders are taking proactive steps to educate their members about the ethics and image of being good outdoor community citizens. I cite as examples.

1. In 2000, an OHV and Hunting Summit was organized to bring together hunting groups, the OHV community, agencies, OHV manufacturers and the environmental community. The goals of the summit were to identify inappropriate behaviors associated with ATV use during the hunt and to identify message points that could be used in all of the groups' educational materials to educate hunters on a consistent basis about ethical OHV and hunting practices. Only seven states were represented at the summit.
2. The results, according to one group, have been very positive. The main messages were boiled down to seven points, which are being printed in a number of state hunting and fishing regulations booklets and special permit applications. Many states have OHV and hunting brochures to hand out to hunters and at least one state has produced posters encouraging proper usage at trailheads and in campgrounds.
3. These same organizations are actively involved with programs like Tread Lightly!, which works to instill a sound, conservation-based outdoor ethic for everyone using the land, and they are working with The U. S. Forest

Service and the U. S. Bureau of Land Management to promote responsible land-use ethics and cooperatively establish and designate appropriate roads, trails and areas for motorized uses.

4. These same agencies are actively developing material and methodology to expose all users to the concepts and specific details of good outdoor ethics and OHV behavior. One such product, specifically aimed at hunters, explores ways to be more successful in using the machines while hunting and simultaneously demonstrating good manners and recognizing that others, who may not be on motorized vehicles are hunting in the same area.
5. All the material I reviewed was adamant about knowing and obeying all hunting and off-road vehicle regulations, including making it clear that hunters using ATVs should be prepared to backpack or horse pack game out of areas that do not have existing roads or trails or that do not allow travel off existing roads and trails for game retrieval. Clear advice was also given regarding crossing streams only at designated trail crossings, citing the damage by erosion and the potential harm to fisheries.

So, having stated that TOC and the organized off-road users feel there is a legitimate place and demand for off-road activities, vehicles and advertising, can we say there is any kind of conclusive argument for or against OHVs? Or, can we even conclude there is a clearly appropriate time and place for their use? I suspect such an attempt would be like trying to make all Republicans buy into the Democrat's point of view and vice versa. It's probably not going to happen any time soon.

Recognizing that condition and in an effort to be objective, can I say that all the advertising material the OHV manufacturers and industry associates put out is the best it can be, or that every scene represents best management practices for operating an off-road vehicle? No, I can't. I'd like to see fewer ATVs slamming their way across small creeks or grinding their way over fallen trees and less mud being splattered by what always looks like riders going all out in what could be taken to be a wetland or fragile mountain or desert habitats.

But, with a sense of the larger outdoor community and recognizing the serious challenges we face, I also have to acknowledge that—unlike their marketing people—I am not under the intense pressure to keep my ATV manufacturing company alive and profitable in a very competitive marketplace.

I should also point out that, as in most national advertising campaigns, the ad agencies more often than not, produce ads that strike very close to the images the buyers themselves want. That should give us a pretty clear target for future educational efforts relating to off-road vehicle users and manufacturers.

Conclusion

What I am hopeful of today, however, and ask all gathered here at this important wildlife and natural resources conference to-do, is to apply our impressive cognitive abilities to find ways to work together to solve existing and potential conflicts surrounding off-road vehicles, that we remember—even when the alligators are snapping at our butts—that we are stronger together than we are separately and that we almost always have more in common with others who enjoy being outdoors, than that which causes us to fret over differing preferences in how we recreate.

I feel it is also important to say that TOC and I are intensely attuned to our democratic way of life and to the system of capitalism that makes the United States the envy of much of the rest of the world.

TOC will continue to promote the best in outdoor programming. As part of that crusade, we will seek to encourage the highest outdoor ethics among all participants in the outdoors. We are firmly committed to the sound, science-based management of our wildlife and natural resources and to the continued existence and growth of our outdoor, angling and hunting heritage.

Effects of Off-road Vehicles on the Hunting Experience

Stan Rauch
Natural Trails and Waters Coalition
Victor, Montana

It really should be no surprise to anyone that if more and more people can easily get deeper into important wildlife areas, we're going to have to compensate with shorter seasons, reduced bag limits or controlled hunts. (Unsworth 2002)

The ever increasing number of dirt-bikes, all-terrain vehicles (ATVs) and other off-road vehicles and their many impacts on national forests and U. S. Bureau of Land Management (BLM) land have reached a critical level. Uncontrolled or unregulated off-road vehicle use damages the land, threatens wildlife and adversely affects millions of people who recreate on that land, including hunters.

Professional wildlife managers across the country are increasingly speaking out about the negative impacts of uncontrolled ATV use on big game and critical habitat. Many warn that, unless common sense limits are applied (including keeping some critical habitat vehicle-free and restricting vehicles to designated roads and routes), quality hunting opportunities are all but certain to decline. At the same time, traditional hunters are losing hunting opportunities in areas they have hunted for years because ATVs push into the most remote corners of the public land.

Hunters, General Public Increasingly Concerned

Hunters and the general public grow increasingly concerned about the impacts of uncontrolled, off-road vehicle use and believe that fish and game departments must focus on this problem.

In 2002, a survey of 1,100 randomly selected Idahoans found that state residents ranked enforcement of regulations for off-road vehicles as the third most important activity of the Idaho Department of Fish and Game. Only enforcement of hunting and fishing regulations and of managing big game ranked higher. Enforcement of off-road vehicle regulations was one of only four activities, out of a total of 49, to be identified as very important by more than 50 percent of all respondents.

In 1998, the Montana Department of Fish, Wildlife and Parks surveyed residents about hunter behavior. Forty-seven percent of those surveyed (195 of 413 responses) listed improper vehicle use or road hunting as the third most frequently observed problem during the previous hunting season. Only trespassing (204 of 413) and lack of respect for landowners (196 of 413) ranked higher. In addition, the survey offered respondents the opportunity to include any written comments. Based on summaries provided by the department, concern about widespread use of ATVs and their negative impact on the sport of hunting was the fifth most frequent comment.

In 2003, I conducted a baseline assessment survey to ascertain what various state-level hunting groups thought about the off-road vehicle issue as it pertains to hunting on public land. In obtaining this data, I spoke with the leaders of nearly 50 groups in 16 states. Based on my discussions, I reached several conclusions.

- Hunters and hunting groups are increasingly concerned about this issue, especially in the western United States.
- Hunters cite destruction of habitat, noise, disruption of wildlife, diminished hunting experiences and deterioration of hunting ethics as the most significant impacts.
- There is clear support for several fundamental policy changes, including:
    - prohibiting cross-country travel
    - limiting off-road vehicle use to designated roads and routes marked with signs or on maps indicating they are open for such use
    - increasing penalties for violating the rules.

Hunters Have Ample Access to Federal Public Land

Some contend that hunters are rapidly losing access to federal public land for hunting. Objective and in-depth analysis challenges this contention both in general as well as in the context of motorized use.

In 2003, the Congressional Sportsmen's Foundation released a comprehensive study of hunter access to public and private land in Colorado.

According to the foundation, it commissioned this study to take a detailed look at access to federal public land in Colorado by surveying licensed hunters to gauge their perceptions of access and to cross-reference that information with current and historical maps of U. S. Forest Service and BLM property. In this study, the phrase federal public land encompasses national forests, BLM land and national wildlife refuges. The major findings include the following.

Federal Public Land, Generally

- Overall, 71 percent of hunters who hunt on federal land in Colorado rated access as excellent or good.
- A majority (68%) of those who have hunted on federal public land in Colorado in the past 10 years have not had problems accessing federal public land in Colorado.
- Access to private land is more of a problem than access to federal public land in Colorado, and access to private land is becoming more of a problem.

National Forests and Grasslands

- National forests and grasslands had the highest percentage of respondents rating access as excellent or good (73%), compared to all other federal public land and private land.

Hunters Want More Traditional Access and Less Motorized Use

The study also assessed hunters' views about the appropriate balance between motorized and nonmotorized uses. Among hunters who had hunted on federal public land in the past 10 years, the study found that 49 percent wanted more access by foot while 32 percent wanted more access by horse. At the same time, 70 percent of these hunters stated that there should be less use of dirt-bikes while 56 percent support less ATV use for hunting.

The study concludes that the access issue is not about having all access to all places; it is more about having the right access to the right places.

Although the majority of hunters on federal public land reported they did not have conflicts with other users, the most frequently reported conflict on national forests and BLM land was with ATVs.

Public Land Is Highly Accessible Via Roads, and Hunters Rely on Roads

The study used maps and GIS technology to locate established roads and trails and to compare growth of roads and trails over time. In addition, the study found that hunters do not need to travel cross-country to reach hunting areas on federal public land.

- Overall between 91 and 97 percent of federal land in Colorado is within 1 mile (1.6 km) of a mapped road or trail. When looking at historical trends in Colorado, there are now more roads and trails available to hunters than in the past.
- Eight percent of U. S. Forest Service land is located farther than 1 mile (1.6 km) from any road.
- A majority of Colorado hunters (62%) who had hunted on federal public land in the past 10 years in Colorado always relied on established roads, and 44 percent of Colorado hunters always relied on established trails when accessing hunting areas on federal public land.

The foundation concludes that the issue is lack of information, not lack of access.

States Begin to Address Impacts

In some areas, particularly in the western United States, the impacts of uncontrolled, off-road vehicle use on critical wildlife habitat and quality hunting experiences have grown so great that fish and game agencies are taking action or giving the issue much greater attention.

Idaho Targets Cross-country Travel

In 2003, the Idaho Fish and Game Commission issued new regulations designed to address the many negative impacts associated with ATVs and other off-road vehicles traveling cross-country. The commission took this action following a chorus of complaints from hunters about opportunities and experiences ruined by ATVs noisily showing and disrupting elk and other big game. The commission acted based on science that demonstrates the negative impacts of roads and vehicle use on many game animals.

The commission's regulation reads in part: "motorized vehicle use as an aid to hunting is restricted to established roadways open to motorized vehicle traffic capable of travel by full-sized automobiles. A full-size automobile shall be defined as any motorized vehicle with a gross weight in excess of 1500 pounds [680.4 kg]" (Idaho Department of Fish and Game 2003:46). The regulation covered 16 management units in southwest Idaho. This regulation only applies to the use of off-road vehicles while hunting; it does not affect recreational use of these vehicles.

With serious impacts in other regions and strong hunter support for these common sense limits, the commission is considering expanding this regulation to as many as 16 more units statewide.

Montana, Wyoming and Idaho Join to Raise Awareness

Montana, Wyoming and Idaho have collaboratively produced a brochure, titled *Hunting and ATVs: Responsibility or Regulation?* The problems associated with ATV use while hunting presented in the brochure are: hunter conflict, noise, operating in closed areas, off-trail use and fair chase concerns. Other information imparted includes the effects of ATVs on habitat, wildlife behavior and the subsequent impacts on hunter success.

U. S. Forest Service Acknowledges the Problem

In a 2003 Earth Day speech, U. S. Forest Service Chief Dale Bosworth highlighted four great issues facing national forests, including unmanaged off-road vehicle use. He described a litany of adverse impacts caused by uncontrolled off-road vehicle use, including soil erosion, habitat destruction, damage to cultural and sacred sites and conflicts with millions of other visitors. Moreover, the threat posed by off-road vehicles is even more significant when one considers the role they play in spreading noxious and invasive weeds and fragmenting critical wildlife habitat—two of the other great issues he described.

Bosworth highlighted the explosion in renegade routes, many of which are illegal: "Each year, we get hundreds of miles of what we euphemistically refer to as 'unplanned roads and trails.' For example, the Lewis and Clark National Forest in Montana has more than a thousand unplanned roads and trails reaching for almost 650 miles [1,046 km]. That's pretty typical for a lot of national forests, and its only going to get worse" (Bosworth 2003:8).

The problem has become so serious that the U. S. Forest Service is developing new policies to manage off-road vehicles; in general, it is highlighting management policies that could better protect forest health and recreational experiences for all visitors by:

- prohibiting random, cross-country travel, except under limited circumstances
- keeping dirt-bikes, ATVs and other off-road vehicles on roads and routes specifically designated for their use
- issuing a uniform national policy of marking roads and routes available for off-road vehicle use with signs or on maps stating they are open for such use.

Proposed regulations could be published for public comment later this spring.

Opportunity to Support Common-sense Management

The U. S. Forest Service effort offers hunters, anglers and other sportsmen and sportswomen the opportunity to share their views about the most appropriate way to manage off-road vehicles on our national forests.

Reference List

Idaho Department of Fish and Game. 2003. *Big game seasons: Rules 2003*. Boise, Idaho: Idaho Department of Fish and Game.

Congressional Sportsmen's Foundation. 2003. *Hunter access to federal public lands in Colorado: A report from the Congressional Sportsmen's Foundation*. Washington, DC: Congressional Sportsmen's Foundation.

Montana Department of Fish, Wildlife and Parks. 1998. *Hunter behavior advisory council, final recommendations*. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Bosworth, Dale. 2003. *Managing the National Forest System: Great issues and great diversion*. San Francisco, California: U. S. Forest Service.

Unsworth, Jim. 2002. Tipping the Balance. *Spokesman Review*. December.

Wildlife for Persons with Disabilities: Making the Outdoors Accessible through the Use of Motorized Vehicles

Kirk Thomas
National Wild Turkey Federation, Wheelin' Sportsmen
Edgefield, South Carolina

Illana Burkhart
National Wild Turkey Federation, Wheelin' Sportsmen
Edgefield, South Carolina

The 2000 U. S. Census identified that there are over 57 million disabled people between the ages of 16 and 64 in the United States. In 2001 the U. S. Fish and Wildlife Service conducted a survey which reported that approximately 7 million people with disabilities participate in wildlife related recreation activities through the 2001 National Survey of Fishing, Hunting, and Wildlife Associated Recreation. This equals 12 percent of the disabled population who take part in wildlife-related recreation, compared to an estimated 39 percent of the nondisabled population. The National Survey of Recreation and the Environment, identified several reasons why the number of disabled individuals who engage in these activities is lower, but the most common barriers identified were physical limitations and access to mobility aids.

When talking about hunting, fishing and outdoor-related activities it is hard to differentiate between physical barriers and mobility aids in terms of a disabled person having access to the outdoors. Most individuals with disabilities who have some type of physical limitation require some type of mobility aid to enjoy the outdoors. Be it a walker, cane, wheelchair, or helping hand, something is necessary to move around the woods with a physical disability. And, sometimes those aids are simply not enough.

The use of all-terrain vehicles (ATVs) levels the playing field between disabled and nondisabled sportsmen and sportwomen. They allow the opportunity to experience the outdoors in a direct way. By providing more access and the freedom to explore the woods and countryside, ATVs and other motorized vehicles let the disabled see things first hand. They can actually drive up the side of a mountain or through the woods instead of stopping at the base of the hill or at the end of the road. This access and freedom has a tremendous impact on the quality of a disabled person's outdoor experience.

In many ways, ATVs enable a person with a disability to not only enjoy the outdoors but also to feel a sense of independence—independence from asking others to help and independence from having to stop at the limits of their physical abilities. The first time I explored the woods on an ATV after my accident, I felt a feeling I had not experienced since before I became paralyzed. I felt the freedom of going where I wanted to go and not being limited to how far into the woods I could push my chair. I had always enjoyed going where I wanted to before my accident, regardless of how dense or rough the terrain was. I could simply step over a branch or walk around a mud hole. I didn't have to end my journey with the end of the normal road. But, sitting in a chair, I found that these things were not so simple. I could simply push my way to wherever I felt like going. I was limited to access roads and paths. Using an ATV changed this and put an end to my frustration. It gave me back the sense of freedom and allowed me to, once again, explore the woods on my own terms.

There are many studies in the field of therapeutic recreation that support the claim that outdoor recreation benefits people with disabilities. To discuss them all would distract from the emphasis of this discussion, which is to state that there is a demonstrated need to provide more people with disabilities access to the outdoors. "Despite research in this area showing the benefits and importance of the natural environment to well-being, these settings seldom are readily accessible to people with disabilities" (Brown, Kaplan, & Quaderer, 1999).

Diane Groff and John Dattilo write about adventure therapy and relayed research in their recent textbook about therapeutic recreation techniques. Their overview of the theory suggests that action-centered approaches to treatment used within the field of therapeutic recreation can be very effective. Outdoor activities can create a climate in which individuals challenge their current perceptions and behaviors. Adventure therapy can be used to promote personal change, improve self-concept, improve perceived competence level and increase self-esteem. By overcoming an obstacle, patients gain a stronger sense of self-esteem and control over their lives. Adventure therapy uses outdoor recreation due to its inherent risk and uncertainty. By reflecting on the skills used in overcoming obstacles in adventure therapy, one can transfer those feelings of accomplishment to overcoming personal obstacles. Generalizing from Groff and Datillo's discussion of adventure therapy, facilitating greater involvement in outdoor activities helps achieve the same results. The feelings of accomplishment, freedom and self-control offered by using an ATV are similar to the benefits of adventure therapy. When participants are give, access to an ATV or to another motorized vehicle to extend their natural abilities in the outdoors, the same feelings of accomplishment and self-efficacy are present as in adventure therapy. By enabling participants to better enjoy and engage in their surroundings, ATVs become a necessary tool for disabled outdoor enthusiasts. An ATV can transport disabled individuals where wheelchairs are unable to go. Some chairs have limited mobility on nonfirm surfaces; some chairs are too heavy to cover rough trail conditions. Some disabled individuals lack the energy to go deep into the woods. Some disabled hunters have to be pushed through miles of wooded trails to access adequate locations. However, the use of an ATV helps disabled individuals carry equipment and retrieve game- activities that are impossible when pushing oneself in a wheelchair.

I will never forget the first deer I harvested after my accident. I shot a nice 8-point buck from 100 yards (91.4 m) on my property in Alabama. I was hunting from my truck, which is legal for disabled hunters in Alabama. The deer ran just a few feet before it came to rest. I was immediately overtaken with a powerful mix of joy and sadness. I felt joy because, after my rehabilitation, my Lord allowed me to go back to enjoying the things I loved. Also, I felt joy because this was a nice 8-point buck and a great harvest for any hunter. But, I was also sad because I was sitting in an access road, and I knew that there was no way that I could make it the 100 yards (91.4 m) through the woods in my chair to retrieve my trophy. Although I could clearly see where it lay, I had to wait until I could find someone who could to be able to retrieve my deer. I will never forget the feeling of helplessness.

That instance brought with it a determination to overcome the physical barriers before me. I now hunt with an ATV. The feelings of accomplishment and pride I feel by being able to scout, hunt and explore the woods on my own are very powerful. And, I am not alone. Many of our members tell me that, without an ATV or golf cart, they would be unable to enjoy outdoor activities. Imagine pushing a wheelchair up a mountain road. Or, imagine walking with a cane through swampy backcountry. It just is not possible.

There are 4.5 million ATVs being used in this country for planting food plots, riding enjoyment, scouting, accessing hunting locations, transporting game, transporting hunting equipment, retrieving game and many other uses. When riders adhere to a code of ethics, the use is perfectly acceptable. As long as they are used responsibly and legally on the more than 500,000 miles (804,672 km) of backcountry roads and trails, they should be made available for use by individuals with disabilities. The benefits far outweigh the concerns of those who may be opposed to their use.

Session Six. *Policy Implications from Long-term Studies of Mule Deer and Elk: A Synthesis of the Starkey Project*

Chair
Michael J. Wisdom
U. S. Department of Agriculture,
Pacific Northwest Research Station
La Grande, Oregon

Cochair
Martin Vavra
U. S. Department of Agriculture,
Pacific Northwest Research Station
La Grande, Oregon

The Starkey Project: Long-term Research for Long-term Management Solutions

Thomas M. Quigley
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
Portland, Oregon

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

The Starkey Project is a unique, long-term research program designed to study the effects of key resource uses of national forests on mule deer

(*Odocoileus hemionus*) and elk (*Cervus elaphus*) habitats and populations located at the Starkey Experimental Forest and Range (Starkey). The purpose of the project was to fill knowledge gaps that posed difficult impediments to effective management of ungulates and to facilitate transfer of this knowledge in mediums most useful to managers. The original studies were completed in the 1990s, but research to understand the effects of emerging resource uses on deer and elk continues today. In addition, new studies are underway to understand the role of these ungulates as disturbance agents that can dramatically alter the ecological patterns and processes in forest ecosystems.

The following papers at this special session of the 69th North American Wildlife and Natural Resources Conference address this evolution of research topics by the Starkey Project and associated research programs. This research now has a 22-year history. With this history is a compelling array of scientific accomplishments made possible through long-term commitments from a wealth of scientists and partners.

In our introductory paper, we summarize the long-term commitments of the Starkey Project, and we acknowledge the many partnerships that made this research possible. We also describe what we consider to be the key "ingredients" of a successful, long-term research program, using the Starkey Project as a case example. Finally, we discuss the accomplishments and credibility that result from long-term research involving diverse partnerships and public interests.

Ingredients for Successful, Long-term Research

Long-term research is rare and invaluable. Few research programs have the opportunity to operate, without interruption, for 5 years, let alone 10, 15 or 20 years. The history of the Starkey Project is an exception. Born of controversy over management of timber, grazing, roads and hunting, the planning stage alone took 4 years (1982–1985) (Wisdom et al. 2004a). Another 4 years (1986–1989) were required to establish the research facility (Rowland et al. 1997). Finally, the original studies took more than 5 years to complete (1990–mid-1990s).

Today, the research at Starkey (Wisdom et al. 2004a) continues to flourish as it diversifies (Thomas and Wisdom 2004). Known as the Starkey Project since inception (Rowland et al. 1997), the research was designed to address the most contentious problems regarding management of mule deer and elk on national forests in the western United States. While the original issues were addressed in research during the 1990s, a myriad of new issues has emerged. As a result, the Starkey Project has evolved to address these new issues, with continued focus on studies designed to gain a better understanding of the role of ungulates in managed ecosystems (Vavra et al. 2004).

The record of the Starkey Project provides a convincing example of what defines an effective, long-term research program. Our examination of its history (Rowland et al. 1997, Wisdom et al. 2004a) and achievements (Thomas and Wisdom 2004) led us to identify four key ingredients of the project's success: (1) diverse, stable partnerships, (2) long-term commitment, (3) high relevance to management and (4) effective technology transfer.

Diverse, Stable Partnerships

From inception, the Starkey Project sought support and involvement from all groups with strong interests and investments in management of national forests in the western United States. These interests included state wildlife agencies, federal land management agencies, timber companies, livestock associations, tribal nations and conservation groups. University partners became increasingly involved as the diversity of research topics increased and as opportunities for involvement of graduates studies were enhanced. Over 40 groups ultimately played an active role in planning and implementing the research (Figure 1, Table 1). Importantly, the distribution of different groups has been relatively even, with the highest number of partners having private, university and federal affiliations (Figure 1).

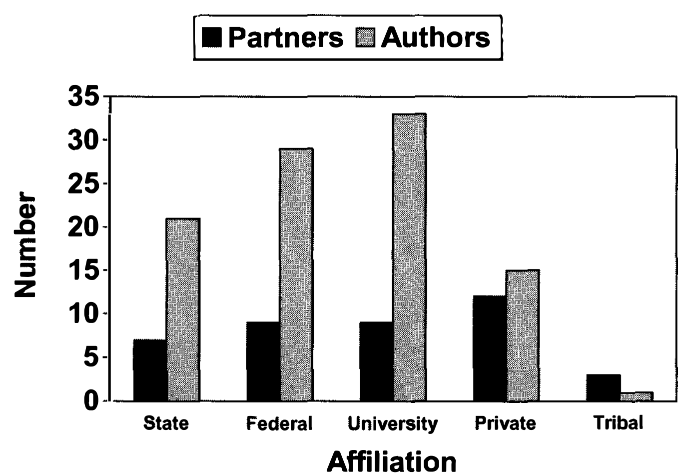


Figure 1. Number of partners in the Starkey Project and number of authors of Starkey Project publications, by affiliation (state agencies, federal agencies, universities, private groups and tribal nations).

Table 1. Research and management partners and their roles in the Starkey Project, 1982 to 2004.

Partner	Role
Oregon Department of Fish and Wildlife	Lead state agency in Starkey Project research from time of inception to the present.
U. S. Department Agriculture, Forest, Pacific Northwest Research Station	Lead federal research branch in Starkey Project research from time of inception to the the present.
U. S. Department of Agriculture, Forest Service, Region 6	Lead federal management branch supporting the establishment of Starkey Project telemetry system, other technologies and the technology transfer program.
Boise Cascade Corporation	Private company that provided the means to harvest timber as part of construction of the Starkey Project's enclosure and as an experimental treatment for the intensive timber management study. Ongoing partner in research through its staff of scientists. Led research on ungulate herbivore effects in Blue Mountains.
National Council of the Pulp and Paper Industry for Air and Stream Improvement	Private, nonprofit organization that has participated as a major research partner since inception of the Starkey Project. Led research on elk thermal cover and nutrition.
Oregon State University	Major academic partner in Starkey Project research. Oregon State University's Eastern Oregon Agricultural Research Center has led the cattle-related studies at the Starkey Project since the inception. Graduate students have completed, or are completing, a myriad of different studies used in eight Master's theses as part of the Starkey Project.
Rocky Mountain Elk Foundation	National nonprofit organization that has provided key logistical, funding, and technology-transfer support for Starkey Project research from time of inception to the present.
Wallowa-Whitman National Forest and La Grande Ranger District	Provided major funding and staffing in support of installation, maintenance and repairs of infrastructure required for the Starkey Project. Helped collect needed data on traffic counts and other field variables in support of original Starkey Project studies.
U. S. Department of Agriculture, Forest Service, Washington Office	Provided majority of funding for establishment of Starkey Project enclosure and other technologies to launch the start of the project.
Umatilla National Forest	Provided analytical and software support for the Starkey Project since inception. Have provided funding and helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.

Table 1 (continued). Research and management partners and their roles in the Starkey Project, 1982 to 2004.

Partner	Role
Oregon Department of Parks and Recreation	State agency that provided major research funding for study on effects of off-road recreation and helped plan the design of this study.
Ochoco and Malheur National Forests	Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
University of Alaska-Fairbanks	Major partner in recent research on the role of mule deer and elk in ecosystem processes for the Starkey Project. A Ph.D. student is nearing completion of this research, with multiple publications produced, in press or submitted.
University of Idaho	Major partner in a variety of Starkey Project studies. Graduate students have completed, or are completing, multiple studies that include two Master's theses and two Ph.D. dissertation.
University of Califomia-Berkeley	Provided major analysis and design for studies of diffusion modeling of animal movements with use of Starkey Project data. Graduate student is participating in study of effects of off-road recreation on mule deer and elk.
Purdue University	Partner in DNA-breeding studies of elk for the Starkey Project.
University of Minnesota	Partner in research on nutritional indices and condition for elk, using Starkey Project's tame elk.
University of Montana	Partner in research on effects of intensive timber management at Starkey. Graduate student completed a Master's thesis in support of this study.
Washington State University	Participated in the design of new herbivory research at Starkey, and have provided long-term veterinary care for tame deer and elk as part of the Starkey Project.
Washington Department of Fish and Wildlife	Major partner in elk diet selection grazing trials, using the Starkey Project's tame elk.
Eastern Oregon University	Provided field and office support for the Starkey Project since inception, in the form of biological intems under the Student Employment Training Program and in collaborative research efforts between faculty and Starkey Project staff.
Oregon State Police, Game Division	Provided law enforcement during the 58 hunting seasons conducted in support of research since the project's inception.
Idaho Department of Fish and Game	Major partner in research to validate and refine methods of aerial survey for elk as part of the Starkey Project.
Alaska Department of Fish and Game	Loaned and demonstrated the use of equipment for Starkey research.
La Grande Animal Health Center	Provided critical veterinary care for deer and elk used in Starkey Project research since its inception.

Table 1 (continued). Research and management partners and their roles in the Starkey Project, 1982 to 2004.

Partner	Role
Confederated Tribes of the Umatilla Indian Reservation	Helped design and participate in tribal hunts at the Starkey Project as part of the research. Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initative
Nez Perce Tribe	Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
Confederated Tribes of Warm Springs Indian Reservation	Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
Oregon Hunter's Association	Provided funding and helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
U. S. Department of the Interior, Bureau of Indian Affairs	Helped design tribal hunts at Starkey as part of the research. Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
U. S. Department of the Interior, Bureau of Land Management	Helped plan various studies for the Starkey Project as part of the Blue Mountains Elk Initiative.
Oregon Cattlemen's Association	Helped plan and provide support for the animal unit equivalency study at the start of the Starkey Project.
Dick Snow Ranch	Participated in cattle research as part of the Starkey Project since its inception.
Trimble, Incorporated	Provided major technologies needed for the Loran-C automated telemetry system.
Tracor, Incorporated	Provided major technologies needed for the Loran-C automated telemetry system.
Partney and Sons, Incorporated	Established the ungulate enclosure with use of innovative drilling technology.
West, Incorporated	Provided major statistical analyses in support of Starkey Project research.
Statistical Services, Incorporated	Provided major statistical analyses in support of Starkey Project research.
David Marx Consulting	Provided major statistical analyses in support of Starkey Project research.
U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station	Provided major analysis support for study of effects of off-road recreation on deer and elk at Starkey, and for studies of diffusion modeling of animal movements.

The diverse partnerships provided strong ownership in the research and subsequent results. Over 100 scientists, representing more than 40 partners, have participated in the publication process (Figure 1). The strong ownership and subsequent trust among so many partners has resulted in rapid and effective use of findings (Thomas and Wisdom 2004).

An additional benefit of the diverse partnerships has been the long-term stability that comes with a variety of funding sources. For example, federal and state budgets have fluctuated across the many budget cycles experienced by the Starkey Project, but the two lead agencies—the Oregon Department of Fish and Wildlife and the U. S. Department of Agriculture, Forest Service (Forest Service)—have invariably found ways to match resources with other partners in a cost-effective manner. Moreover, the diverse partnerships have provided a stable base, helping to minimize undesired changes in funding and research focus that can occur with dynamic budgets of any one partner or small group of partners.

The many partners also have contributed a variety of research skills, enhancing the scientific credibility of publications and subsequent management products. One example is the development of a model to allocate forage among ungulates for allotment management planning on public lands (Ager et al. 2004). Scientists involved with this work included field biologists, computer programmers, research analysts, ungulate specialists and landscape ecologists. Another example is the birth date-nutrition study of elk (Cook et al. 2004) that involved scientists with expertise in natural history, population genetics, ungulate ecology, manipulative experiments, animal nutrition and animal husbandry.

Long-term Commitment

The Starkey Project has a long and productive history. The research facility was formally established in 1987 with the completion of one of the largest ungulate-proof enclosures ever constructed (Rowland et al. 1997). The facility became operational in 1989 with the installation a novel, automated radio-telemetry system that could track the movements of more than 100 radio-collared ungulates accurately, frequently and regularly—24 hours a day (Rowland et al. 1997). Since then, more than 50 studies have been conducted, with more than 140 publications completed or in press (Thomas and Wisdom 2004).

A central and vital benefit from this long-term commitment has been the accelerating production of scientific publications (Figure 2). During the late 1980s

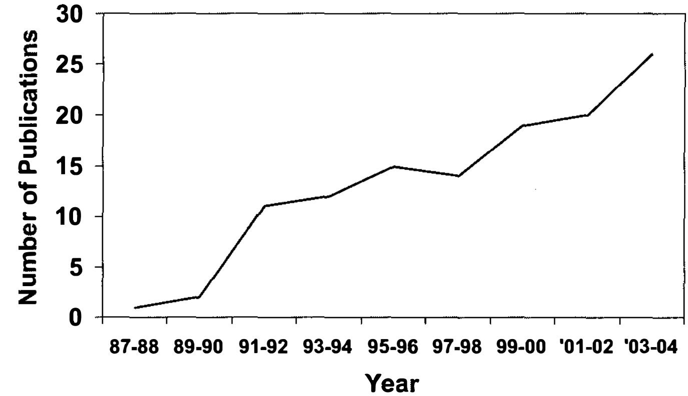


Figure 2. Number of Starkey Project publications since 1987, summed by two-year periods through early 2004. Only peer-reviewed publications appearing in journals, books, and graduate theses and dissertations are included (i. e., the summary excludes all published abstracts, "white papers," popular news and magazine articles and other "gray literature").

and early 1990s, the number of publications in peer-reviewed journals was low, owing to the need to focus on large capital investments in the facility and to develop, test and implement a series of innovative research designs (Rowland et al. 1997). As the original studies were completed during the 1990s, the number of publications increased substantially (Figure 2). This accelerated pace in publication rate became particularly apparent during the past 6 years, signaling a compelling return from the long-term investment. The publication rate for the Starkey Project now exceeds 12 peer-reviewed publications per year (not including abstracts, tours, workshops, symposia, videos, magazine articles and television features). This publication rate surpasses many other state or federal research projects of similar size and budgets.

Importantly, the large number of peer-reviewed publications produced during the past 6 years also reflects an increasing depth of study. Since 1998, most publications of the Starkey Project contained results from data collected over a period of 4 to 5 years. By contrast, our examination of publications appearing in the *Journal of Wildlife Management* during 2003 indicated that less than 5 percent of the articles were based on data collected over a period of 4 years or longer.

Research implemented over long periods is invaluable because results are robust to short-term anomalies that can confound results. For example, long-term results minimize the confounding effects of seasonal, annual and climatic variation that can overwhelm and bias results of short-term studies. Long-term studies provide valuable opportunities to validate the findings and hypotheses that emerge from the early years of research; that is, portions of larger data sets, collected over a longer period of time, can be "held out" to validate initial findings of a given study. Moreover, long-term data provide opportunities for a variety of follow-on analyses not planned under the original study objectives.

An example is the 2004 release of a voluminous data set by the Starkey Project (Kie et al. 2004). These data are now available for use by students, educators and scientists as a complement to past uses by the Starkey Project. Kie et al. (2004) and Wisdom et al. (2004a) describe these data and their original uses in Starkey Project studies.

High Relevance to Management

The Starkey Project was designed to address key knowledge gaps identified as impediments to effective management of ungulates on national forests in the western United States. The project operates under the concept of adaptive management (Holling 1984, Walters 1986). The scientific basis for adaptive management uses the following process: (1) managers identify knowledge gaps that prevent desired improvements in targeted resources; (2) managers and scientists jointly develop testable hypotheses to address the knowledge gaps; (3) scientists design and implement studies to test the hypotheses; (4) managers and scientists interpret and disseminate results from the studies for management use; (5) managers and scientists identify additional knowledge gaps and hypotheses for testing, based on study results and ensuing questions that arise from the results; (6) the cycle is repeated one or more times, if desired, using knowledge gained from earlier phases of study.

An example of the Starkey Project's use of adaptive management is the study of breeding efficiency of bull elk (Noyes et al. 2004). This study was jointly designed by managers and scientists to address the issue of whether hunter harvest of bull elk needed modification to improve survival of older bulls and to enhance their breeding performance. The study was conducted from 1989 through 1993 and was implemented over the entire elk population in the 19,180-acre (7,768-ha) Main Study Area. Hunting by the public was used to implement the research, with hunters working closely with scientists to achieve desired harvest and collect needed data. Managers then used results from the study to redesign harvest regulations for bull elk in many states and provinces in western North America. In turn, validation tests were conducted by repeating the study for another 5 years at Starkey. Results from the validation provided strong support for earlier changes in hunting regulations that were made in response to findings of the original study (Noyes et al. 2004).

Use of adaptive management continues in current research. For example, an emerging, national issue is the use of all-terrain vehicles (ATVs) on public lands. Land managers and public interests at national, regional and state levels worked closely with Starkey scientists in 2002 to design and implement new landscape research to evaluate effects of ATVs on deer and elk, as compared to hiking, mountain bike riding and horseback riding. Initial findings are now available (Wisdom et al. 2004b). Importantly, the findings are already being considered by managers, interest groups and Starkey Project's scientists to devise appropriate management applications. Given the high level of controversy surrounding off-road recreation on public lands, new study proposals are being drafted by managers and the Starkey Project scientists to validate aspects of the initial findings and to expand upon this study in future research.

Effective Technology Transfer

The Starkey Project was one of the first research programs in the Forest Service with full-time support from a technology transfer scientist (Rowland et al. 1997). This scientist functioned as part of the Starkey Project staff from 1987 to 1994 and served as the primary liaison between management and research. During this time, the Starkey Project shared technologies and results with more than 200,000 recipients encompassing local, regional, national and international organizations, groups and agencies. Transfer mediums included field tours, presentations, workshops, symposia, news releases, newspaper features, magazine articles, radio interviews and television coverage. This work helped garner widespread public acceptance and support of the project and its initial results (Rowland et al. 1997, Thomas and Wisdom 2004).

Today, the Starkey Project continues to share results and technologies with a wide range of resource managers. Scientists have averaged more than 15 field tours and more than 20 meeting presentations per year during the past decade. These communication mediums, beyond the formal scientific publications, have played an important role in the Starkey Project technology transfer program.

Challenges for Long-term Research

Many obstacles prevent successful implementation of long-term research. Short-term priorities, budget fluctuations, large capital investments, changing political agendas and impatience with the lack of immediate scientific production are example impediments. The Starkey Project has overcome these obstacles because of diverse partnerships, long-term investments, high relevance to management and effective delivery of results. Continued emphasis on these key ingredients will enable the Starkey Project to continue its success in serving land and population managers of deer and elk in the western United States.

Key to this continued success will be the project's reliance on adaptive management and effective synthesis of results in forms useful for management. Maintenance of healthy and diverse partnerships is imperative. Considering the wealth of completed and ongoing studies, the success of partnerships and the utility of the research facility, the future of the Starkey Project appears bright. The many papers presented at the 69th North American Wildlife and Natural Resources Conference attest to the project's accomplishments and relevance to resource management over the past 20 years.

Reference List

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:687–707.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Nutrition and parturition date effects on elk: Potential implications for research and management. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:604–24.

Holling, C. S., editor. 1984. *Adaptive environmental assessment and management*. New York, New York: John Wiley and Sons.

Kie, J. G., A. A. Ager, N. J. Cimon, M. J. Wisdom, M. M. Rowland, P. K. Coe, S. L. Findholt, B. K. Johnson, and M. Vavra. 2004. The Starkey database: Spatial-environmental relations of North American elk, mule deer and cattle at the Starkey Experimental Forest and Range in northeastern Oregon. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:475–90.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2004. Influences of age of males and nutritional condition on short- and long-term reproductive success of elk. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:572–85.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Thomas, J. W., and M. J. Wisdom. 2004. Has the Starkey project delivered on its commitments? *Transactions of the North American Wildlife and Natural Resource Conference.* 69:798–812.

Vavra, M., M. J. Wisdom, J. G. Kie, J. G. Cook, and R. A. Riggs. 2004. The role of ungulate herbivory and management on ecosystem patterns and processes: Future direction of the Starkey project. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:785–97.

Walters, C. 1986. *Adaptive management of renewable resources.* New York, New York: MacMillan.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004a. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:455–74.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004b. Effects of off-road recreation on mule deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:531–50.

Overview of the Starkey Project: Mule Deer and Elk Research for Management Benefits

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station
La Grande, Oregon

Mary M. Rowland
U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station
La Grande, Oregon

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Brian L. Dick
U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station
La Grande, Oregon

Introduction

Managers have long been concerned about the welfare of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) on public lands in the western United States. These two species generate millions of dollars annually to state wildlife agencies from sales of hunting licenses, and elk viewing generates millions of additional dollars to local and regional economies (Bolon 1994, Bunnell et al. 2002). By contrast, the potential for elk and mule deer to compete with livestock, to damage agricultural crops and to modify plant succession make the two species obvious sources of controversy among private and public land managers.

In the 1980s, wildlife managers began to focus on the potential effects of timber management, livestock grazing, road use and ungulate harvest strategies on mule deer and elk. At the same time, land managers were concerned about constraints associated with these management activities, based on the perceived needs of the two species. In particular, timber harvest, livestock production, road construction and motorized traffic were dominant and pervasive uses of public lands during the 1980s and earlier decades, but their effects on deer and elk were uncertain and widely debated. In addition, the dichotomy of managing for productive habitats (i. e., high biomass and quality of forage) versus secure habitats (i. e, dense cover and minimal human disturbance) was highly debated, with management trade-offs that were unclear (Lyon and Christensen 2002). As an outgrowth of these concerns and the lack of empirical data, the Starkey Project was initiated in the mid-1980s at the U. S. Department of Agriculture, Forest Service's Starkey Experimental Forest and Range (Starkey) in northeastern Oregon (Figure 1). The project was designed to evaluate mule deer and elk responses to the most common management activities occurring on national forests in the western United States.

In this paper, we provide an overview of the Starkey Project as an introduction for the papers that follow. Additional details about the Starkey Project and Starkey are found in Rowland et al. (1997, 1998), Skovlin (1991) and at the Starkey Project's Website (http://www.fs.fed.us/pnw/starkey/).

Controversies of Ungulate Management: Beginning of the Starkey Project

The Starkey Project was born from contentious debate about how best to manage habitats and populations of mule deer and elk in the western United States and about the degree to which traditional management practices on public lands needed modification to accommodate the two species' needs. The most prominent issues of debate related to (1) road and traffic management, (2) intensive timber production and thermal cover, (3) competition between wild ungulates and cattle, and (4) breeding efficiency of male elk in relation to population productivity. These four issues became the foundation of the Starkey Project's original studies that began in 1989 and ended during the 1990s (Table 1).

The issue of road management revolved around the uncertainty of whether mule deer and elk avoided roads open to motorized traffic and whether the rate of motorized traffic influenced ungulate distribution (Rowland et al.

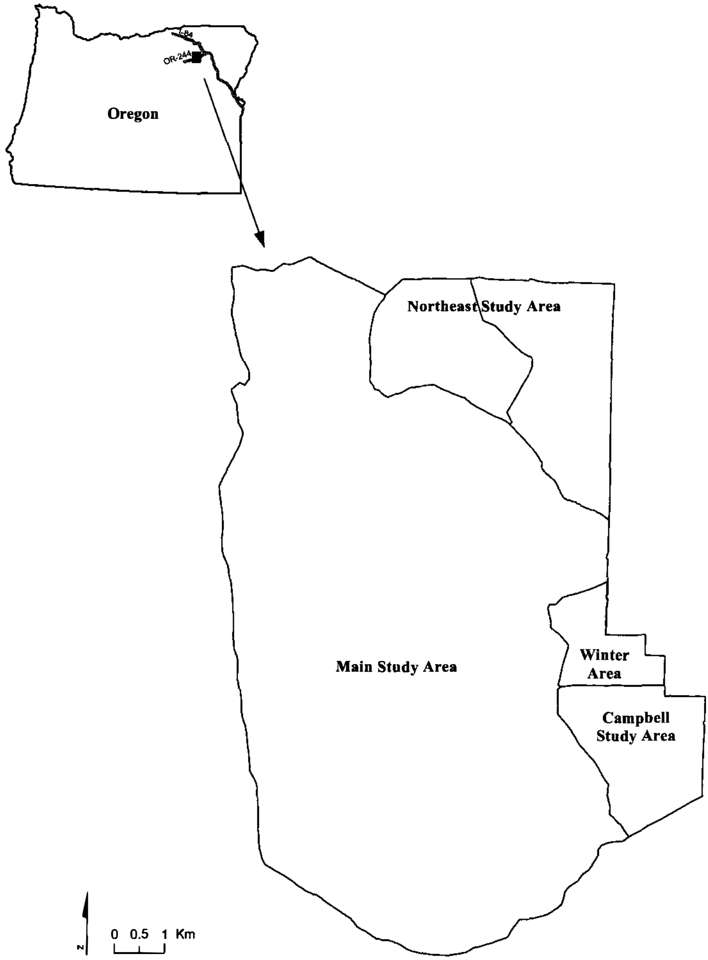


Figure 1. (Above) Location of Starkey Experimental Forest and Range in northeastern Oregon, 28 miles (45 km) southwest of La Grande. (At right) Boundaries of the ungulate-proof enclosure, internal ungulate fences and study areas. The 3,590-acre (1,453-ha) Northeast Study Area and the 19,180-acre (7,768-ha) Main Study Area are the two sites where research occurs from April through early December on free-ranging ungulates. The 655-acre (265-ha) Winter Area is the site of winter feeding and handling of deer and elk. The 1,537-acre (622-ha) Campbell Study Area is for the study of ungulate breeding and grazing.

1997). To address these issues, vehicle travel by the public was monitored with traffic counters during the 1990s in the Main Study Area (Figure 1) from May through December of each year (Table 1). Radio-telemetry locations of

Table 1. Chronology of major research activities of the Starkey Project, 1982 to 2004 (from Rowland et al. 1997 and updated to early 2004).

Year	Research Activity
1982-1985	Initial discussions regarding the Starkey Project take place among scientists from the Oregon Department of Fish and Wildlife and U. S. Department of Agriculture, Forest Service Pacific Northwest Research Station, resulting in proposals for the original studies.
1986	Environmental assessment is completed and approved for construction of Starkey Project enclosure.
1987	Twenty-seven miles (43 km) of New Zealand woven-wire fence is installed to enclose 24,962 acres (10,110 ha) of Starkey (Figure 1), encompassing summer range populations of mule deer and elk.
1988-1989	Over 65 traffic counters are installed at intersections of open roads in the Main Study Area for the roads and traffic study.
	The 655-acre (265-ha) Winter Area and handling facility is completed (Figure 1).
	Loran-C automated telemetry system (ATS) becomes operational in Northeast Study Area, allowing ungulates to be radio-collared and monitored remotely to start the intensive timber management study.
	Limited entry hunting of deer and elk begins in support of breeding bull study in the Main Study Area.
1990	Timber harvest contract is awarded for intensive timber management study in the Northeast Study Area.
	Loran-C ATS is completed for the Main Study Area, making the telemetry system operational in all areas of Starkey, and allowing the start of the animal unit equivalency and roads and traffic studies.
1991	Elk thermal cover study begins at Kamela, 30 miles (48 km) northeast of Starkey.
	Starkey's Institutional Animal Care and Use Committee is formed under new regulations promulgated from the federal Animal Welfare Act, formalizing the existing procedures for humane care and treatment of ungulates used for research (Wisdom et al. 1993).
	Road construction and logging begins late in the year in Northeast Study Area as part of intensive timber management study.
1992	Campbell Study Area (1,537 acres [622 ha]) is established (Figure 1) in support of new breeding bull study.
	Study of ungulate herbivory effects on plant succession and nutrient availability begins at exclosures established more than 25 years ago throughout northeastern Oregon.
	Loran-C ATS is upgraded, resulting in improved accuracy of animal locations.
1993	Logging is completed early in the year in Northeast Study Area; over 7 million board feet is harvested.
	Elk breeding bull efficiency study is completed in Main Study Area.
	Performance tests of Loran-C ATS are initiated.

Table 1 (continued). Chronology of major research activities of the Starkey Project, 1982 to 2004 (from Rowland et al. 1997 and updated to early 2004).

Year	Research Activity
1994	Animal unit equivalency study is completed in Main Study Area.
1995	Prototype forage allocation model is built, tested and published as part of animal unit equivalency study.
	Validation phase of elk breeding bull study begins in Main Study Area.
	Roads and traffic study is completed in Main Study Area.
	Elk thermal cover study is completed at Kamela study site.
1996	Results from the elk breeding bull study are published (Noyes et al. 1996).
	Intensive timber management study is completed and elk-ecosystem processes study is designed for northeastern study area.
1997	New fuels reduction study is designed for Main Study Area.
	Cross-fencing of Northeast Study Area into east (1,500 acres [608 ha]) and west (2,090 acres [846 ha]) pastures is completed to start new research about elk-ecosystem processes.
1998	Results from elk thermal cover study at Kamela are published (Cook et al. 1998).
	Performance tests and bias corrections for ATS are published (Johnson et al. 1998).
	Results from the traffic study are summarized in a Ph.D. dissertation (Wisdom 1998).
1999	Manipulative experiment to validate results from traffic study begins in Main Study Area.
	Fuels reduction study begins in Main Study Area.
2000	Results from elk-roads study are published (Rowland et al. 2000).
	Results from study on ungulate herbivory effects on plant succession are published (Riggs et al. 2000).
	Additional results from the animal unit equivalency study on mule deer and elk are published (Johnson et al. 2000).
	Manipulative experiment to validate results from traffic study is completed in Main Study Area.
2001	Elk-ecosystem processes research in Northeast Study Area is completed.
	Results from the intensive timber management study for elk are summarized in a Master's thesis (Rinehart 2001).
	Additional results from the animal unit equivalency study on cattle-deer-elk interactions are published (Coe et al. 2001).
	Symposium held at Eastern Oregon University to summarize findings of more than 10 years of research of the Starkey Project; attended by more than 150 scientists and managers from western North America.
2002	Results from the validation phase of the elk breeding bull study are published (Noyes et al. 2002).

Table 1 (continued). Chronology of major research activities of the Starkey Project, 1982 to 2004 (from Rowland et al. 1997 and updated to early 2004).

Year	Research Activity
2002 (cont.)	New study on effects of off-road recreational activities on deer and elk begins in the Northeast Study Area.
	Construction of two ungulate exclosures, each 18 acres (7.3 ha) in size, is completed in the Main Study Area to begin new research on ungulate herbivory effects on ecological processes.
	Replacement of the Loran-C ATS with a new GPS-based telemetry system is identified as a key need, with initial design and cost estimates completed.
2003	Results from elk birth date-nutrition study are published (Cook et al. 2004b).
	Three additional 18-acre (7.3-ha) ungulate exclosures are built in support of new herbivory research in Main Study Area.
	New GPS-based radio collars are used on 16 elk in the Northeast Study Area as part of study on effects of off-road recreation.
	Study to validate Starkey elk resource selection patterns begins at Sled Springs Demonstration Area on Boise Cascade Corporation lands in northeastern Oregon.
2004	Off-road recreation study is completed in Northeast Study Area and preliminary results are published (Wisdom et al. 2004a).
	Field sampling and grazing trials begin for new herbivory research in Main Study Area.
	Treatments for fuels reduction study are completed in Main Study Area.
	Over 10 years of research findings from the Starkey Project are summarized in a set of 20 papers presented at the 69th North American Wildlife and Natural Resources Conference, and subsequently published in the conference's *Transactions*, reprinted in a book by Allen Press.

ungulates were collected and analyzed in relation to road type (open, closed, restricted) and traffic rate (zero, low, moderate, high, very high). The result has been a set of compelling findings about deer and elk distributions, in relation to their distance to roads with varying traffic rates (Wisdom 1998, Wisdom et al. 2004b) and and about distance of elk to open roads (Rowland et al. 2000, 2004). Results are now used for management of roads and access throughout western North America (Thomas and Wisdom 2004).

Considerable controversy also existed about the effects of timber management on mule deer and elk (Hieb 1976, Rowland et al. 1997). Consequently, ungulate response to intensive timber production was assessed in the Northeast Study Area of Starkey (Figure 1). Radio-collared cattle, deer and elk were monitored from 1989 to 1996, spanning periods before, during and after timber harvest (Table 1). Over half of the forested area was subjected to intensive logging, combined with a doubling of road density. Animal responses were monitored in a variety of ways (Rinehart 2001, Wisdom et al. 2004c). Results provided little evidence of negative effects on ungulates from activities or changes brought about by intensive timber harvest on summer range during nonhunting periods (Wisdom et al. 2004c).

Throughout the 1980s, some studies documented high use of dense forests by elk (Leckenby 1984), and others postulated that elk were selecting these sites for thermal cover (Thomas et al. 1988). These ideas generated considerable debate about how forests should be managed and why elk were selecting dense cover. To address this issue, an experimental study was conducted at Kamela, Oregon, approximately 30 miles (48 km) northeast of Starkey (Cook et al. 1998). Here, the nutritional condition of tractable elk maintained in pens was monitored in relation to four treatments: (1) dense thermal cover, (2) moderately dense thermal cover, (3) no cover and (4) a combination of no cover and thermal cover. Results indicated no positive benefits to animal condition from thermal cover. Instead, a negative effect was associated with high levels of cover (Cook et al. 1998, 2004a). These results have changed the way that managers think about and plan for maintenance of elk thermal cover across western North America (Thomas and Wisdom 2004), in balance with the need to maintain dense cover for elk security, especially during hunting seasons (Lyon and Christensen 2002).

The issue of whether mule deer and elk compete with cattle for available forage on summer range was addressed by a long-term study of ungulate interactions in the Main Study Area (animal unit equivalency study, Table 1). By evaluating the spatial distributions, resource selection patterns and behavioral interactions of the three ungulates as cattle were rotated through livestock pastures on summer range (Coe et al. 2001, 2004; Stewart et al. 2002), scientists were able to devise a realistic allocation of forage among the ungulate species by month and season (Johnson et al. 1996, Ager et al. 2004). Based on these results and a subsequent study of diet overlap among the three species (Findholt et al. 2004), a new forage allocation model is now available for beta testing as part of allotment planning on national forests of the interior western United States (Ager et al. 2004).

The issue of whether elk productivity is affected by age of breeding bulls also was addressed in the Main Study Area (Table 1). From 1989 to 1993, breeding male elk were allowed to increase in age, beginning as 1.5 year-old (yearling) bulls in 1989. During each of these five years, this single cohort of male elk functioned as the only breeders in the study population. As bulls grew older, conception dates in female elk were earlier and more synchronous (Noyes et al. 1996). As a result, calves were born earlier and over a shorter time period each spring, which may provide a number of survival benefits (Noyes et al. 2004). Results have caused substantial changes in hunting regulations for bull elk across many states and provinces in western North America (Thomas and Wisdom 2004).

Continued Need for Information Drives Additional Studies

The original studies at Starkey were completed during the mid- and late 1990s (Table 1) and were used by management agencies soon after publication (Thomas and Wisdom 2004). Results also spurred follow-on research, such as validation tests of the original breeding bull study (Noyes et al. 2002) and original traffic study (Wisdom et al. 2004b). Results from the original studies also were the catalyst for subsequent studies on the interactions of nutrition and bull age in affecting elk condition (Cook et al. 2004b) and the assessment of ungulate diets (Findholt et al. 2004) to realistically allocate forage among cattle, mule deer and elk (Ager et al. 2004).

New research continues today to complement the original research. While past studies focused on mule deer and elk responses to management, new research also considers the effects of these ungulates on the ecosystem (Riggs et al. 2004, Vavra et al. 2004). In addition, new research has recently been completed, or is nearing completion, regarding deer and elk responses to off-road recreation (Wisdom et al. 2004a), fuels reduction (Vavra et al. 2004), hunting (Johnson et al. 2004), breeding-nutrition interactions on elk condition (Cook et al. 2004b) and fine-scale movement patterns of deer and elk (Ager et al. 2003).

New research also is underway on nonfederal ownerships near Starkey to complement past and current studies at Starkey. As an example, the resource selection models derived at Starkey for the animal unit equivalency study (Johnson et al. 2000) are being validated at the Sled Springs study area, on land owned by Boise Cascade Corporation (Coe 2003). This research at Sled Springs will strengthen and expand the inference space for the Starkey forage allocation model (Ager et al. 2004) to improve its use across larger areas of the western United States (Coe 2003).

New Technologies Support the Studies

The main goal of the Starkey Project has been to measure the habitat, behavioral and population responses of mule deer and elk to intensely managed forests and rangelands at the landscape scales at which management occurs. Meeting this goal for the plethora of studies that have taken place required innovative technologies and a unique research environment with the following characteristics:

1. a closed system to ungulates, with defined ungulate populations whose responses to management experiments on spring, summer and fall ranges could be measured accurately
2. an expansive area over which the experiments could be implemented, allowing inferences to be made at landscape scales to be used for management
3. an animal tracking system capable of accurate and continuous monitoring of animal movements
4. a network of traffic counters for close monitoring of human activities and movements
5. a winter facility, where mule deer and elk could be fed to minimize the confounding effects of winter weather
6. a database and mapping system capable of storing and displaying a comprehensive set of environmental variables in relation to animal movements and human activities
7. an effective strategy of hunter management and mule deer and elk harvest, using limited entry hunts to achieve desired management treatments on ungulate populations, to collect data from harvested animals, to meet research goals and to evaluate ungulate responses to varying levels and types of hunting pressure.

Establishing these technologies and the needed research facility at Starkey required innovation, persistence and persuasion on the part of the founders of the project (Rowland et al. 1997). Scientists, such as Jack Ward Thomas and Donavin Leckenby, with the U. S. Department of Agriculture, Forest Service's (Forest Service) Pacific Northwest Research Station and the Oregon Department of Fish and Wildlife, developed novel study plans not conceived in past wildlife research. The plans called for the most intensive radiotelemetry monitoring of ungulate populations ever attempted, and they called for novel, experimental treatments and controls for evaluating animal responses to management activities at landscape scales not considered possible.

Thomas then convinced a regional forester, John Butruille, of the worth of the proposed studies. Butruille responded by committing millions of dollars from the management branch of the Forest Service to establish the needed technologies and research facility. Larry Bryant, another scientist with the Pacific Northwest Research Station at the time, then directed the development, acquisition and implementation of every major technology and facility improvement during the late 1980s, each of which is still in use today.

Ungulate-proof Enclosure

Forty miles (64 km) of 8-foot-high (2.4 m), New Zealand woven-wire fence were constructed in 1987 to establish the Starkey ungulate enclosure (Bryant et al. 1993). An additional 27 miles (43 km) of internal fence were constructed to establish three separate areas (Figure 1): the Main Study Area (19,180 acres [7,768 ha]), the Northeast Study Area (3,590 acres [1,454 ha]) and the Winter Area (655 acres [265 ha]). In 1992, the 1,537-acre (622-ha) Campbell Study Area was built to accommodate additional research needs (Figure 1, Table 1).

The total area of 24,962 acres (10,110 ha) functions as one of the largest ungulate research enclosures in the world. Its completion established a closed but ecologically extensive system for ungulates. The result was a research environment that provided ungulates with unconstrained habitat choices over expansive areas, commensurate with the size of their spring, summer and fall ranges in the western United States, but also forced the ungulates to respond to the management experiments.

Importantly, the New Zealand woven-wire fence was designed as an effective but humane barrier to ungulates. When ungulates run into the fence, it acts as a trampoline, pushing animals away from the fence without injury (Bryant et al. 1993). The fence was designed to function more than 30 years before major repairs are needed.

Automated Telemetry System

Long range navigation (LORAN-C) technology developed by the U. S. Coast Guard for ship and airplane navigation was adapted by Starkey scientists to establish an automated telemetry system (ATS) in 1989 (Dana et al. 1989, Table 1). The Starkey ATS can generate up to one animal location every 20 seconds, 24 hours a day, from April through December each year. The frequency and order of obtaining animal locations is programmable, such that more than 100 radio-collared animals can be relocated at a frequency of approximately one location for each radio-collared animal/hour (Rowland et al. 1997, Kie et al. 2004). Or, a subset of animals can be relocated more frequently, such as one location for a given animal every minute, depending on the sampling schedule required to meet research needs.

Locations from the ATS have a mean accuracy of 164 feet (50 m) of the true location (Findholt et al. 1996). This location accuracy allows for reliable estimation of resource selection patterns and spatial distributions of ungulates in time and space to meet Starkey research goals (Findholt et al. 2002). Performance of the ATS has been well documented (Johnson et al. 1998, Garton et al. 2001). Over 2 million locations of elk, mule deer and cattle have been collected since 1989, with location data used in a variety of Starkey publications. Examples of uses include Johnson et al. (2000), Rowland et al. (2000), Garton et al. (2001) and Leban et al. 2001.

Winter Feeding and Handling

Mule deer and elk traditionally migrated to winter range at lower elevations far from Starkey. Construction of the Starkey enclosure in the summer and fall of 1987, however, surrounded the populations of mule deer and elk on their spring, summer and fall range, preventing migration to traditional winter range. Consequently, a 655-acre (265-ha) Winter Area (Figure 1) was established to feed animals and, in the process, minimize the confounding effects of winter weather and foraging conditions on deer and elk responses to the summer range experiments (Rowland et al. 1997).

Mule deer and elk are baited to the Winter Area or live-trapped and moved to the site each December. Deer and elk are fed alfalfa pellets and hay, respectively, from December through March, and then released back into the study areas for another year of research. This ration is fed ad libitum, thus reducing potential variation in physical condition of animals owing to variation in winter weather.

The Winter Area also has an elk handling facility that allows scientists to handle animals humanely and efficiently to check their condition and equip them with radio collars (Wisdom et al. 1993). Handling in December and January includes tests for pregnancy, disease, percent body fat and weight, for analysis of these data in relation to the summer range experiments.

Network of Traffic Counters

Over 65 traffic counters were installed at open road intersections throughout the Main Study Area in 1989 as part of the roads and traffic study (Table 1). The counters record the number of motorized vehicles that pass over an inductive loop buried underneath the road, as summarized by 15- or 30-minute intervals, 24 hours a day, from May through December of each year.

The counts have been used to estimate the rate of motorized traffic (number of vehicles per unit time) on each road at Starkey, as summarized by time of day for each season of each year. The traffic rates were used to evaluate mule deer and elk distributions in relation to distance from roads with varying rates of motorized traffic in the Main Study Area (Wisdom 1998, Wisdom et al. 2004b).

Use of Hunting for Research

From 1989 through 2003, 58 limited entry hunts of deer or elk have taken place in support of Starkey research goals (see Rowland et al. 1997). These hunts are used to maintain population characteristics (e. g., sex and age ratios) for mule deer and elk similar to those outside Starkey, so results are relevant to extant populations. Rowland et al. (1997:31) listed five additional reasons for use of hunting as part of Starkey Project research: (1) to collect data on animal condition, (2) to obtain baseline population data from harvest statistics, (3) to provide traditional recreational opportunity, (4) to provide data for a model of elk vulnerability to harvest and (5) to subject animals to hunting pressure and disturbance similar to that outside the enclosure. The incorporation of hunting as part of the Starkey Project research also has allowed scientists to evaluate deer and elk responses to being hunted, measured in terms of energy expenditures, changes in spatial distributions, changes in resource selection patterns and harvest rates in relation to hunter density (Johnson et al. 2004). Finally, the use of hunting as part of the research has facilitated detailed studies of hunters' willingness to pay for a variety of harvest opportunities and hunting experiences (Fried et al. 1995).

Data Collection and Mapping of Environmental Variables

Data for more than 100 environmental variables have been collected and mapped for Starkey Project research (Rowland et al. 1998). Example variables include slope, aspect, elevation, convexity, canopy closure, ecoclass, distance to water and distance to open roads (Kie et al. 2004). These variables have been used in a variety of Starkey publications, such as those as described here and in Table 1. Examples of uses include evaluation of resource selection patterns (Johnson et al. 2000, Coe et al. 2001); effects of roads (Rowland et al. 2000, 2004), traffic (Wisdom 1998, Wisdom et al. 2004b) and timber harvest (Rinehart 2001, Wisdom et al. 2004c); and forage allocation modeling (Johnson et al. 1996, Ager et al. 2004).

New Technologies Support Research Beneficial to Management

Use of the innovative technologies at Starkey allowed the original and subsequent studies to be completed during the 1990s (Table 1). Findings from the many studies are diverse and compelling. Highlights of key findings from the original studies are summarized here as an example of what has been learned. Details can be found in the papers cited.

- Elk avoid roads open to traffic, validating observational studies from the 1970s and 1980s that first identified this relation (Rowland et al. 2000, 2004).
- Elk avoidance of roads increases with increasing rate of traffic, providing further evidence that elk are not reacting to roads, per se, but to the activities associated with roads (Wisdom 1998, Wisdom et al. 2004b).
- Mule deer generally do not avoid roads and show increasingly strong selection toward areas near roads with increasing rate of traffic (Wisdom 1998, Wisdom et al. 2004b). This pattern is a result of mule deer avoiding elk, rather than deer selecting particular habitats near roads and traffic (Johnson et al. 2000, Coe et al. 2001).
- Elk avoid cattle and mule deer avoid elk, but the degree to which interference competition is operating is less clear (Johnson et al. 2000, Coe et al. 2001, Stewart et al. 2002). Elk may have ample opportunity to select a variety of habitats where cattle are not present, owing to rotation of cattle through a series of livestock pastures (Coe et al. 2001). Mule deer may not have these same habitat choices when avoiding elk, owing to the presence of elk throughout their summer range.
- Diets of cattle, mule deer and elk differ substantially during early summer, with cattle diets containing a higher composition of grasses,

deer diets containing more shrubs and forbs, and elk diets intermediate to those of cattle and deer (Findholt et al. 2004). Diets of the three ungulates become increasingly similar during late summer, when forage biomass and quality decline with the onset of summer drought, suggesting increased potential for exploitative competition.

- During the fall rut, the date of conception in female elk becomes progressively earlier and synchronous with increasing age of male breeders (Noyes et al. 1996, 2002, 2004). Breeding by yearling (1.5 year-old) bulls results in the latest dates of conception and highest variation in conception dates; breeding by mature (5.5 year-old) bulls results in earliest dates of conception. Earlier and closely synchronous dates of conception allow calves to be born earlier in spring and over a compressed time period, with potential benefits to survival (Noyes et al. 2004).
- In the absence of hunting, elk, mule deer and cattle do not appear to be negatively affected by intensive timber harvest (Rinehart 2001, Wisdom et al. 2004c), but timber harvest increases the vulnerability of elk to hunting (Wisdom et al. 2004c).
- Elk do not require or benefit from thermal cover to ameliorate their thermoregulatory requirements in response to extreme weather during winter and summer (Cook et al. 1998). Instead, a mix of open and closed-canopy habitats results in superior animal performance compared to homogeneous stands of thermal cover. Importantly, elk appear to select dense forest for security during both hunting and nonhunting seasons (Ager et al. 2003, Johnson et al. 2004).

Starkey's Future Appears Bright

The Starkey Project now has one of the most voluminous and intensive data sets ever collected on ungulates. Over 140 publications have been produced, with more than 50 studies underway or completed (Thomas and Wisdom 2004). More than 100 scientists have conducted the research, with support from the timber industry, livestock industry, hunting groups, conservation organizations and state and federal agencies (Quigley and Wisdom 2004). Results have been used widely to improve management of timber, grazing, roads and recreation in relation to the needs of mule deer and elk (Thomas and Wisdom 2004).

The future of ungulate research at Starkey and at associated, nonfederal sites appears bright. A unique set of technologies and research facilities remain in place, allowing scientists to investigate a wide range of management issues related to ungulates. Continued focus on studies to integrate the needs of mule deer and elk with economic, social and recreational interests in western North America represents an ongoing, compelling need that will continue to be addressed by the Starkey Project.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:687–707.

Bolon, N. A. 1994. *Estimates of the values of elk in the Blue Mountains of Oregon and Washington: Evidence from the existing literature, general technical report PNW-GTR-316.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Bryant, L. D., J. W. Thomas, and M. M. Rowland. 1993. *Techniques to construct New Zealand elk-proof fence, general technical report PNW-GTR-313*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Bunnell, S. D., M. L. Wolfe, M. W. Brunson, and D. E. Potter. 2002. Recreational use of elk. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 701–47. Washington, DC: Smithsonian Institution Press.

Coe, P. K. 2003. *Validating resource selection functions for elk in the Blue Mountains of northeast Oregon*. M.S. thesis proposal, Oregon State University, Corvallis.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:205, A51–A76.

Coe, P. K., B. K. Johnson, K. M. Stewart, and J. G. Kie. 2004. Spatial and temporal interactions of elk, mule deer and cattle. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:656–69.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004b. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs.* 155:1–61.

Cook, J. G., L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1998. Relations of forest cover and condition of elk: A test of the thermal cover hypothesis in summer and winter. *Wildlife Monographs.* 141:1–61.

———. 2004a. Thermal cover needs of large ungulates: A review of hypothesis tests. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:708–26.

Dana, P. H., W. Fowler and D. Hindman. 1989. Automated animal-tracking system: Tracking elk with retransmitted Loran-C. In *Proceedings 18th Annual technical symposium of the International Loran Association*, ed. J. M. Beukers, 53–61. Santa Barbara, California: International Loran Association.

Findholt, S. L., B. K. Johnson, D. Damiran, T. DelCurto, and J. G. Kie. 2004. Diet composition, dry matter intake and diet overlap of mule deer, elk and cattle. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:670–86.

Findholt, S. L., B. K. Johnson, J. W. Kern, A. A. Ager, L. L. McDonald, and R. J. Stussy. 2002. *Adjusting for radiotelemetry error to improve estimates of habitat use, general technical report PNW-GTR-555.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Findholt, S. L., B. K. Johnson, L. D., Bryant, and J. W. Thomas. 1996. Corrections for position bias of a LORAN-C radio-telemetry system using DGPS. *Northwest Science.* 70:273–80.

Fried, B. M., R. M. Adams, R. P. Berrens, and O. Bergland. 1995. Willingness to pay for a change in elk hunting quality. *Wildlife Society Bulletin.* 23:680–86.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. In *Radio tracking and animal populations*, eds. J. Millspaugh and J. Marzluff, 1–42. San Diego, California: Academic Press.

Hieb, S. R., ed. 1976. *Proceedings of the elk-logging-roads symposium.* Moscow, Idaho: Forest, Wildlife, and Range Experiment Station, University of Idaho.

Johnson, B. K., A. A. Ager, J. H. Noyes, and N. Cimon. 2004. Elk and mule deer responses to variation in hunting pressure. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:625–40.

Johnson, B. K., A. A. Ager, S. A. Crim, M. J. Wisdom, S. L. Findholt, and D. Sheehy. 1996. Allocating forage among wild and domestic ungulates—A new approach. In *Proceedings symposium on sustaining rangeland ecosystems*, *special report 953*, eds. W. D. Edge, and S. L. Olson-Edge, 166–9. Oregon State University Extension Service.

Johnson, B. K, A. A. Ager, S. L. Findholt, M. J. Wisdom, D. Marx, and L. D. Bryant. 1998. Mitigating spatial differences in observation rate of automated telemetry systems. *Journal of Wildlife Management.* 62:958–67.

Johnson, B. K, J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk in spring. *Journal of Wildlife Management.* 64:685–97.

Kie, J. G., A. A. Ager, N. J. Cimon, M. J. Wisdom, M. M. Rowland, P. K. Coe, S. L. Findholt, B. K. Johnson, and M. Vavra. 2004. The Starkey database: Spatial-environmental relations of North American elk, mule deer and cattle at the Starkey Experimental Forest and Range in northeastern Oregon. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:475–90.

Leban, F. A., M. J. Wisdom, E. O. Garton, B. K. Johnson, and J. G. Kie. 2001. Effect of sample size on the performance of resource selection analysis. In *Radio tracking and animal populations*, eds. J. Millspaugh and J. Marzluff, 291–307. San Diego, California: Academic Press.

Leckenby, D. A. 1984. *Elk use and availability of cover and forage habitat components in the Blue Mountains, Northeast Oregon 1976–1982, wildlife research report no. 14*. Portland, Oregon: Oregon Department of Fish and Wildlife.

Lyon, L. J., and A. G. Christensen. 2002. Elk and land management. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 557–81. Washington, DC: Smithsonian Institution Press.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management*. 66:1,301–07.

———. 2004. Influences of age of males and nutritional condition on short- and long-term reproductive success of elk. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:572–85.

Noyes, J. H., B. K. Johnson, L. D. Bryant, S. L. Findholt, and J. W. Thomas. 1996. Effects of bull age on conception dates and pregnancy rates of cow elk. *Journal of Wildlife Management*. 60:508–17.

Quigley, T. M., and M. J. Wisdom. 2004. The Starkey project: Long-term research for long-term management solutions. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:442–54.

Riggs, R. A., A. R. Tiedemann, J. G. Cook, T. M. Ballard, P. J. Edgerton, M. Vavra, W. C. Krueger, F. C. Hall, L. D., Bryant, L. L. Irwin, and T. DelCurto. 2000. *Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains Ecological Province, Forest Service research paper PNW-RP-527*. Portland, Oregon: U. S. Department of Agriculture.

Riggs, R. A., J. G. Cook, and L. L. Irwin. 2004. Management implications of ungulate herbivory in northwest forest ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:759–84.

Rinehart, J. M. 2001. Effects of intensive salvage logging on Rocky Mountain elk at the Starkey Experimental Forest and Range. M.S. thesis, University of Montana, Missoula.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: Implications for management in forested ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:491–508.

Rowland, M. M., P. K. Coe, R. J. Stussy, A. A. Ager, N. J. Cimon, B. K. Johnson, and M. J. Wisdom. 1998. *The Starkey habitat database for ungulate research: Construction, documentation, and use, general technical report PNW-GTR-430*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Skovlin, J. M. 1991. *Fifty years of research progress: A historical document on the Starkey Experimental Forest and Range, general technical report PNW-GTR-266*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83:229–44.

Thomas, J. W., D. A. Leckenby, M. Henjum, R. J. Pedersen, and L. D. Bryant. 1988. *Habitat-effectiveness index for elk on Blue Mountain winter ranges, general technical report PNW-GTR-218*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Thomas, J. W., and M. J. Wisdom. 2004. Has the Starkey project delivered on its commitments? *Transactions of the North American Wildlife and Natural Resources Conference*. 69:798–812.

Vavra, M., M. J. Wisdom, J. G. Kie, J. G. Cook, and R. A. Riggs. 2004. The role of ungulate herbivory and management on ecosystem patterns and processes: Future direction of the Starkey project. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:785–97.

Wisdom, M. J. 1998. Assessing life-stage importance and resource selection for conservation of selected vertebrates. Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004a. Effects of off-road recreation on mule deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:531–50.

Wisdom, M. J., B. K. Johnson, M. Vavra, J. M. Boyd, P. K. Coe, J. G. Kie, A. A. Ager, and N. J. Cimon. 2004c. Cattle and elk responses to intensive timber harvest. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:727–58.

Wisdom, M. J., J. G. Cook, M. M. Rowland, and J. H. Noyes. 1993. *Protocols for care and handling of deer and elk at the Starkey Experimental Forest and Range, general technical report PNW-GTR-311.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Wisdom, M. J., N. J. Cimon, B. K. Johnson, E. O. Garton, and J. W. Thomas. 2004b. Spatial partitioning by mule deer and elk in relation to traffic. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:509–30.

The Starkey Databases: Spatial-Environmental Relations of North American Elk, Mule Deer and Cattle at the Starkey Experimental Forest and Range in Northeastern Oregon

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Alan A. Ager
U. S. Department of Agriculture, Forest Service,
Umatilla National Forest
Pendleton, Oregon

Norman J. Cimon
U.S . Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Mary M. Rowland
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Priscilla K. Coe
Oregon Department of Fish and Wildlife
La Grande

Scott L. Findholt
Oregon Department of Fish and Wildlife
La Grande

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Martin Vavra
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

In the late 1980s, the Starkey Project was initiated to study interactions among North American elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*) and domestic cattle at Starkey Experimental Forest and Range (Starkey) in northeastern Oregon. As part of the Starkey Project, an automated radio telemetry system was developed to collect an unprecedented volume of location data on both wild and domestic ungulates (Rowland et al. 1997). That radio telemetry system was based on rebroadcast LOng RAnge aid to Navigation- (LORAN-) C signals. In this paper, we provide an overview of the databases resulting from this effort.

LORAN is a marine radio-navigation system that was well established before widespread use of the Navstar Global Positioning System (GPS). LORAN-A was the earliest version, developed during World War II, while today LORAN-C serves the civilian community and LORAN-D is used by the military (Logsdon 1992, West and Pittman 1993). LORAN systems operate by measuring the difference in arrival times of a radio signal broadcast at 100 kHz directly from a master station and after being relayed through several secondary stations. The time differences of arrival among the signals routed through the various secondary stations allow a LORAN receiver to determine its location on several hyperbolic lines of position. Where those lines cross is the estimate of the actual location of the receiver (Logsdon 1992, West and Pittman 1993).

The automated radio telemetry system at Starkey consists of a central computer base station, seven radio relay stations located throughout Starkey Forest and radio collars placed on individual elk, mule deer and cattle. Every 20 seconds, the base station pages one of the many radio collars deployed at any given time (Figure 1). As many as 150 animals have been included in the user-

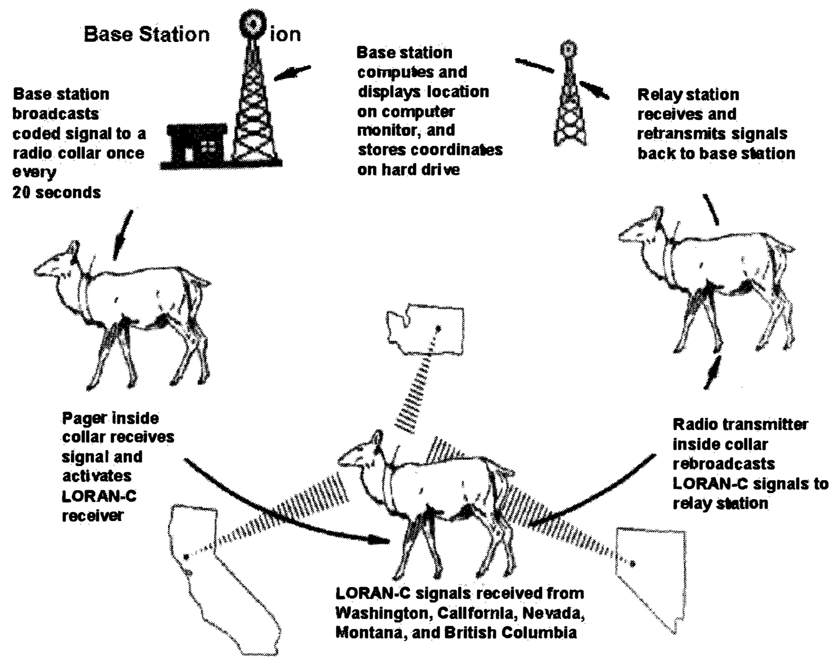


Figure 1. Automated, rebroadcast LORAN-C radio telemetry system used to collect location data on elk, mule deer and cattle at Starkey Experimental Forest and Range (from Rowland et al. 1997).

defined paging list. When a particular animal is paged, a LORAN-C receiver inside the collar responds by collecting raw data, which are then retransmitted via a very high frequency (VHF) radio link to one of the radio relay stations. The relay towers then transmit the data back to the central computer, where the raw data are decoded and information on the animal's location is stored electronically for future analysis (Figure 1). This LORAN-C system has a mean locational accuracy of about 55 yards (50 m); although, some errors can exceed 219 yards (200 m), particularly in the east-west direction (Findholt et al. 1996, 2002).

The Starkey Project has collected information on over 2 million animal locations since 1989. All animal data were collected following approval of protocols by an institutional animal use and care committee (Wisdom et al. 1993). This telemetry database represents one of the largest data sets of large-animal locations ever compiled. A variety of studies have been completed based on these data. Examples include research on ungulate interactions and resource selection (Johnson et al. 2000, Coe et al. 2001, Stewart et al. 2002), effects of roads

(Rowland et al. 2000) and traffic (Wisdom 1998) on elk, mule deer and cattle, spatial and temporal patterns of habitat use (Ager et al. 2003), effects of sample size on estimates of resource selection (Leban et al. 2001) and home range (Garton et al. 2001), and development of ungulate forage allocation models (Johnson et al. 1996).

To facilitate new analyses by other research wildlife biologists, we are providing access to a portion of the Starkey Project telemetry database over the World Wide Web at http://www.fs.fed.us/pnw/starkey/data. The telemetry database includes locations of animals in the main study area at Starkey Forest (Wisdom et al. 2004), recorded during spring and summer between 1993 to 1996, along with associated metadata. In addition, we are providing relevant geographic information systems (GIS) layers, such as habitat type, soil, a digital elevational model and the forest road network, in a second habitat database. The release of these data will allow other scientists to formulate and to test hypotheses that are best analyzed with comprehensive information about animal locations, such as that provided by the Starkey Project's data set.

Starkey Ungulate Telemetry Database

Records in the telemetry database consist of elk, mule deer and cattle locations (n > 287,000) collected in the 19,180-acre (7,762 ha) main study area at Starkey Forest (Wisdom et al. 2004) during spring and summer (April to mid-August) from 1993 to 1996 (Table 1). Animal locations are provided both as a point estimate in the Universal Transverse Mercator (UTM) coordinates (UTM Easting [E] and UTM Northing [N], all in UTM Zone 11) and also placed in the center point of each 33 by 33 yard (30 x 30 m) pixel that encompasses the point estimate (UTMGrid, UTMGridEast, UTMGridNorth). All UTM coordinates are given in North American Datum 1983 (NAD83). Animals are uniquely identified and were assigned a specific radio-collar number (RadNum) each year. The user should be aware that a given radio-collar number could be worn by different animals in different years. Time and date of each observation is provided in Greenwich Mean Time (GMTime) and in local time (LocTime, in Pacific Standard Time; no adjustments were made for daylight savings time). Time is also provided in a variable called Starkey Time, representing seconds from an arbitrary zero point at 0000 hours on 31 December 1987. Starkey Time is useful in determining elapsed time between any two observations. Sunrise and sunset

Table 1. Synthesized metadata description of variables included in the telemetry database of locations of elk, deer and cattle collected in Main Study Area, Starkey Experimental Forest and Range, spring and summer, 1993 through 1996.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
UTMGrid	Universal Transverse Mercator (UTM) Coordinates that identify the center point of each 33 by 33 yard (30 by 30 m) pixel within Starkey (Rowland et al. 1998); first six digits represent the UTM Zone 11 easting; second seven digits represent the UTM Zone 11 northing. Example: 373695 5014530 are the easting and northing coordinates, respectively, that identify the center point of a given pixel at Starkey.	Meters	Character			373695 5014470, 381885 5007540	Fixed (14)
UTMGrid East	Coordinates in UTM Zone 11 easting Example: 373695 are easting coordinates.	Meters	Numeric (long integer)			373695, 381855	Variable
UTMGrid North	Coordinates in UTM Zone 11 northing Example: 5014530 are the northing coordinates.	Meters	Numeric (long integer)			5005110, 5019120	Variable
ID	Unique alpha-numeric identification code assigned to each ungulate that was radio-collared. Example for deer: 890224EO4 Example for elk: 930318DO. Example for cattle: OSUX83041						Fixed (9)
Starkey time	Cumulative number of seconds that have elapsed since the Greenwich Mean Time of 00:00:00 on 31 December 1987, which represents a starting point of the Starkey ungulate research. Example: Starkey time equals 1 at Greenwich Mean Time of 00:00:01 on 31 December 1987.	Seconds	Numeric			168825628, 272245926	Variable

Table 1 (continued). Synthesized metadata description of variables included in the telemetry database of locations of elk, deer and cattle collected in Main Study Area, Starkey Experimental Forest and Range, spring and summer, 1993 through 1996.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
GMTime	Greenwich Mean Time expressed as hour: minutes:seconds; 8 hours ahead of Pacific Standard Time. Example: 18:31:47 is 31 minutes, 47 seconds after hour 18 in the 24-hour cycle, as set on GMTiine.	Hour:Minutes: Seconds		Character		00:00:00, 23:59:59	Fixed (8)
GMDate	Date associated with the Greenwich Mean Time, expressed as Year/Month/Day Example: 19930526 is 26 May 1993	Year/Month/ Day		Character		19930507, 19960815	Fixed (8)
LocDate	Date synchronized in relation to Pacific Standard Time, which is 8 hours behind Greenwich Mean Time. LocDate is expressed as year/mondi/day and is set for the Pacific Time Zone. Example: The Pacific Standard Time of 19:59:27 has an associated date for the Pacific Time Zone of 19930512, or 12 May 12 1993. However, the commensurate Greenwich Mean Time of 03:59:27 has an associated date of 13 May 1993.	Year/Month/ Day		Character		19930506, 19960815	Fixed (8)
LocTime	Pacific Standard Time, which is 8 hours later than Greewich Mean Time. No adjustment is made for Pacific Daylight Savings Time. Example: 19:59:27 is 59 minutes, 27 seconds after hour 19 of a 24-hour cycle, as set on Pacific Standard Time, which is commensurate with 03:59:27 of Greenwich Mean Time.	Hour:Minutes: Seconds		Character		00:00:00, 23:59:59	Fixed (8)

Table 1 (continued). Synthesized metadata description of variables included in the telemetry database of locations of elk, deer and cattle collected in Main Study Area, Starkey Experimental Forest and Range, spring and summer, 1993 through 1996.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
RadNum	Unique identification code assigned to a given radio-collar. Note that a given radio-collar could be wom by one or more animals within or across years and, thus, should not be confused with the unique identification code for each radio-collared animal, or ID.	Numeric				8,450	Variable
Species	Species of ungulate associated with the animal location, where C = cattle, D = deer, and E = elk.		Character	C, D, E	C = cattle, D = deer, E = elk.		Fixed(l)
UTME	Coordinates in the easting direction. Constitutes the x-coordinate of the point estimate of the animal location.	Meters	Numeric			373705, 381853	Fixed(6)
UTMN	Coordinates in the northing direction. Constitutes the y-coordinate of the point estimate of the animal location.	Meters	Numeric			5005110, 5019118	Fixed(7)
Year	Year in which the animal location was collected.	Year	Numeric			93,96	Fixed(2)
Grensunr	Time of sunrise at Starkey for a given date, expressed in Greenwhich Mean Time. See GMT variable.	Hour: Minutes: Seconds	Numeric			12:05:00, 13:35:00	Fixed(8)
Grensuns	Time of sunset at Starkey for a given date, expressed in Greenwhich Mean Time. See GMT variable.	Hour: Minutes: Seconds	Numeric			02:23:00, 03:49:00	Fixed(8)
Obswt	Correction to the bias in observation rate of an animal location, where observation rate is defined as the probability of obtaining an animal location from a radio-		Numeric			1.14,3.5	Variable

Table 1 (continued). Synthesized metadata description of variables included in the telemetry database of locations of elk, deer and cattle collected in Main sSudy Area, Starkey Experimental Forest and Range, spring and summer, 1993 through 1996.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
Obswt (cont.)	collared animal, when an animal location is requested by the Loran-C telemetry system. Application of Obswt (1 divided by the observation rate) increases the number of locations for a given area, according to the degree of bias in observation rate associated with that area (Johnson et al. 1998). Example: An Obswt of 2 would place twice the weight on a given animal location, whereas an Obswt of 1.5 would place 1.5 times the weight on a given animal location. These weights correct for animal locations in a given UTMGrid that were not observed, given the bias in observation rate among pixels (Johnson et al. 1998).						

[a] Refers to way in which data is entered, such as character (text), long integer, or floating point.

[b] Refers to whether data is of fixed or variable length; for fixed variables, maximum number of characters is displayed

times are given based on GMTime (Grensunr, Grensuns). Finally, a correction factor is provided to account for spatially dependent observation rates, as discussed by Johnson et al. (1998). The observation rate is defined as the probability of obtaining an animal location as a function of its location at Starkey. The correction factor (Obswt), defined as the inverse of the observation rate (1/observation rate), can be used to weight locations for assessing patterns of spatial use.

Starkey Habitat Database

Spatial layers for various features are provided in the habitat database (Table 2). Each record in the database is identified by UTM coordinates, and represents a unique 33 by 33 yard (30 x 30 m) pixel in main study area, Starkey. All UTM coordinates are given in North American datum 1983 (NAD83). Values for variables, such as ecoclass, soil depth and canopy closure, are provided. Details of the habitat database are available in Rowland et al. (1998).

In addition to providing habitat information in database form, habitat layers are included using ArcInfo (Environmental Systems Research Institute, Redlands, California) interchange format (Table 3). Raster files include ecoclass, soil, a digital elevational model, a canopy closure of overstory trees and black hole corrections for differences in telemetry observation rates as per Johnson et al. (1998). A polygon file is provided showing forest stand boundaries. Line files include roads, streams and fences (game-proof fences and barbed-wire cattle fences). Water sources are given as a point coverage. In addition, metadata is included for the Starkey map coverages. This metadata follows the general format of the Federal Geometric Data Committee's Content Standard for Digital Geospatial Metadata (Federal Geographic Committee 1998). It can be accessed, using a Web browser, as a hyperlinked list of "frequently asked questions" (FAQs), one list per map coverage.

Publications

Additional information about the Starkey Project is available on the World Wide Web at http://www.fs.fed.us/pnw/starkey. Two publications from the Starkey Project, to interpret the telemetry and habitat databases, are available there in PDF format; there is also a general history of the Starkey Project, which

Table 2. Synthesized metadata description of variables included in the habitat database for Main Study Area, Starkey Experimental Forest and Range.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
UTMGrid	Universal Transverse Mercator (UTM) coordinates that identify the center point of each 33 by 33 yard (30 by 30 m) pixel within Starkey (Rowland et al. 1998); first six digits represent UTM Zone 11 easting; second seven digits represent the UTM Zone 11 northing. All coordinates are given in NAD83. Example: 373695 5014530 are the easting and nothing coordinates, respectively, that identify the center point of a given pixel at Starkey.	Meters	Character			373695, 5014470, 381885 5007540	Fixed (14)
UTMGridEast	Easting coordinate in UTM Zone 11.	Meters	Numeric (long integer)			373695, 381855	Fixed (6)
UTMGridNorth	Northing coordinate in UTM Zone 11.	Meters	Numeric (long integer)			5005110, 5019120	Fixed (7)
SoilDepth	Soil depth to the restrictive layer (obtained from the Wallowa-Whitman National Forest soils resource inventory [SRI])	Centimeters	Numeric			9,60	Variable
PerSlope	Percent slope (Rowland et al. 1998)		Numeric			0,84	Variable
SINAspct	Sine of aspect (Rowland et al. 1998)		Numeric			-1,1	Variable

Table 2 (continued). Synthesized metadata description of variables included in the habitat database for Main Study Area, Starkey Experimental Forest and Range.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
COSAspct	Cosine of aspect (Rowland et al. 1998)		Numeric			-1, 1	Variable
Convex3	Convexity (Rowland et al. 1998)		Numeric			465.86, 524.64	Variable
DistCWat	Distance to the nearest water source from within a cattle pasture, including class I through III streams and water point sources such as stock ponds and springs	Meters	Numeric	-99	-99 denotes 33 by 33 (30 by 30 m) pixels in the pasture that are not available to cattle. These pixels should be excluded from any analyses of cattle distance to water.	-99,2714	Variable
Canopy	Total canopy closure (%) of all trees > 1 inch (2.5 cm) diameter at breast height)		Numeric			0.85	Variable
Elev	Elevation (Rowland et al. 1998)	Meters	Numeric			1121,1500	Fixed (4)
DistEWat	Distance to the nearest water source from within an ungulate-proof pasture (e. g., Main Study Area), including class	Meters	Numeric			0,1188	Variable

Table 2 (continued). Synthesized metadata description of variables included in the habitat database for Main Study Area, Starkey Experimental Forest and Range.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
DistEWat (cont.)	I through III streams and water point sources such as stock ponds and springs.						
EcoGener	General ecoclass description; modified from the original ecoclass variable (Rowland et al. 1998) to include only the dominant forest canopy species		Character	AB	Buildings, structures, roads		Fixed (10)
				CD	Douglas-fir,		
				CJ	Juniper,		
				CL	Lodgepole pine,		
				CP	Ponderosa pine		
				CW	White fir, grand fir,		
				GB	Bunchgrass vegetation,		
				MD	Dry meadow[c],		
				MM	Moist meadow[d],		
				MW	Wet meadow[e],		
				NR	Rocky land,		
				SD	Dry shrubland[f],		
				WR	Running water (stream, river, creek, ditch)		
DistOPEN	Distance to the nearest open road (Rowland et al. 1998)	Meters	Numeric			0,2419	Variable
DistRSTR	Distance to the nearest restricted access road (Rowland et al. 1998)	Meters	Numeric			0,2315	Variable

Table 2 (continued). Synthesized metadata description of variables included in the habitat database for Main Study Area, Starkey Experimental Forest and Range.

Variable name	Variable definition	Units	Storage type[a]	Codes	Code definitions	Range (min, max)	Data format[b]
DistCLSD	Distance to the nearest closed road (Rowland et al. 1998)	Meters	Numeric			0,2208	Variable
DistEFnc	Distance to the nearest ungulate-proof fence (Rowland et al. 1998)	Meters	Numeric			0,4243	Variable
CowPast	Cattle pasture (Rowland et al. 1998)		Character	BEAR HORSE SMITH-BALLY HALF-MOON MDW-CRK NE STRIP			Fixed
ForgProd	Forage production (biomass of understory species considered forage for ungulates; Hall 1973)	Kilograms/ Hectare	Numeric		0,2200	Variable	
DistEdge	Distance to nearest edge, based on the EcoGener polygons used for ecoclasses	Meters	Numeric		0,426	Variable	

[a] Refers to way in which data are entered, such as character (text), long integer, or floating point.

[b] Refers to whether data is of fixed or variable length; for fixed variables, maximum number of characters is displayed.

[c] Water table available part of the season.

[d] Water table available all growing season.

[e] Surface moist or wet all of growing season.

[f] Includes sagebrush and non-forest zone shrubland; not desert.

Table 3. Habitat layers in Arc Info export format for Main Study Area, Starkey Experimental Forest and Range.

Habitat layer name	Layertype	Description
vegetation-ecoclass	grid	Map of vegetation ecoclasses; 33 by 33 yard (30 by 30 m) cells
soils	grid	Map of soil types; classified by soil series and depth; 33 by 33 yard (30 by 30 m) cells
DEM	grid	Digital election model for Starkey; 33 by 33 yard (30 by 30 m) cells
Black-hole corrections	grid	Black-hole corrections for spatial differences in telemetry observation rates (Johnson et al. 1998); 33 by 33 yard (30 by 30 m) cells
canopy closure	grid	Canopy closure of overstory trees
vegetation forest-stands	polygon	Map of forest stands; classified by ecoclass
roads	line	Map of all roads on and immediately adjacent to Starkey; classified by road type
streams	line	Map of all stream drainages; classified by stream class, presence or absence of running water
game-proof fences	line	Map of game-proof fences only
all-fences	line	Map of all fences, including barbed-wire cattle fences and enclosures
water points	point	Location of water point sources such as stock troughs

contains information about telemetry data (Rowland et al. 1997) and details of the Starkey Habitat Database (Rowland et al. 1998). A complete list of Starkey Project publications, many available as full-text PDF files can be found at http://www.fs.fed.us/pnw/starkey/publications.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:205, A51–A76.

Federal Geographic Data Committee. 1998. Content Standard for Digital Geospatial Metadata. Reston, Virginia, U. S. Geographical Survey.

Findholt, S. L., B. K. Johnson, L. D. Bryant, and J. W. Thomas. 1996. Corrections for position bias of a LORAN-C radio-telemetry system using DGPS. *Northwest Science*. 70:273–80.

Findholt, S. L., B. K. Johnson, L. L. McDonald, J. W. Kern, A. Ager, R. J. Stussy, and L. D. Bryant. 2002. *Adjusting for radiotelemtry error to improve estimates of habitat use, general technical report PNW-GTR-555*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. In *Radio tracking and animal populations*, eds. J. J. Millspaugh, and J. M. Marzluff, 15–42. San Diego, California: Academic Press.

Hall, F. C. 1973. *Plant communities of the Blue Mountains in eastern Oregon and southeastern Washington, R6 area guide volume 3-1*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Johnson, B. K., A. A. Ager, S. A. Crim, M. J. Wisdom, S. L. Findholt, and D. Sheehy. 1996. Allocating forage among wild and domestic ungulates—A new approach. In *Proceedings of a symposium on sustaining rangeland ecosystems*, *Special Report 953*, eds. W. D. Edge, and S. L. Olson-Edge, 166–9. Corvallis, Oregon: Oregon State University Extension Service.

Johnson, B. K., A. A. Ager, S. L. Findholt, M. J. Wisdom, D. Marx, and L. D. Bryant. 1998. Mitigating spatial differences in observation rate of automated telemetry systems. *Journal of Wildlife Management*. 62:958–67.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk in spring. *Journal of Wildlife Management*. 64:685–97.

Leban, F. A., M. J. Wisdom, E. O. Garton, B. K. Johnson, and J. G. Kie. 2001. Effect of sample size on the performance of resource selection analysis. In *Radio tracking and animal populations*, eds. J. J. Millspaugh, and J. M. Marzluff, 291–307. San Diego, California: Academic Press.

Logsdon, T. 1992. *The NAVSTAR global positioning system*. New York, New York: Van Nostrand Reinhold.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and* *data collection methods for ungulate research, general technical teport PNW-GTR-396*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672-84.

Rowland, M. M., P. K. Coe, R. J. Stussy, A. A. Ager, N. J. Cimon, B. K. Johnson, and M. J. Wisdom. 1998. *The Starkey habitat database for ungulate research: Construction, documentation, and use, general technical report PNW-GTR-430*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Stewart, Kelley M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83:229-44.

West, G., and F. Pittman. 1993. *Boatowner's guide to marine electronics*, 3rd edition. Camden, Maine: International Marine.

Wisdom, M. J. 1998. Assessing life-stage importance and resource selection for conservation of selected vertebrates. Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., J. G. Cook, M. M. Rowland, and J. H. Noyes. 1993. *Protocols for care and handling of deer and elk at the Starkey Experimental Forest and Range, general technical report PNW-GTR-311*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:455–74.

Effects of Roads on Elk: Implications for Management in Forested Ecosystems

Mary M. Rowland
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon*

Michael J. Wisdom
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon*

Bruce K. Johnson
*Oregon Department of Fish and Wildlife
La Grande*

Mark A. Penninger
*U. S. Department of Agriculture, Forest Service,
Wallowa-Whitman National Forest
La Grande, Oregon*

The effects of roads on both habitat and population responses of elk (*Cervus elaphus*) have been of keen interest to foresters and ungulate biologists for the last half century. Increased timber harvest in national forests, beginning in the 1960s, led to a proliferation of road networks in forested ecosystems inhabited by elk (Hieb 1976, Lyon and Christensen 2002). Among disturbances to elk habitat, roads have been viewed as a major factor influencing distributions of elk across the landscape (Leege 1984, Lyon 1984, Lyon et al. 1985, Roloff 1998, Lyon and Christensen 2002, Wertz et al. 2004). Evidence from a variety of studies, such as those conducted at the Starkey Experimental Forest and Range (Starkey) in northeastern Oregon, has corroborated this view (Lyon 1983, 1984; Witmer and deCalesta 1985; Cole et al. 1997; Johnson et al. 2000; Rowland et al. 2000; Ager et al. 2003).

Early studies of elk were among the first to address effects of roads on wildlife, establishing a precedent for subsequent research on a wide range of terrestrial and aquatic species. These early elk-roads studies included those reported in a symposium on the topic in 1975 (Hieb 1976), the seminal studies of Jack Lyon in Montana and northern Idaho (Lyon 1979, 1983, 1984), the Montana Cooperative Elk-Logging Study (Lyon et al. 1985), and work by Perry and Overly (1977) in Washington and by Rost and Bailey (1979) in Colorado.

As research and analysis techniques have become more sophisticated, particularly with the advent of geographic information systems (GIS) and high-resolution remote imagery, the study of effects of roads on terrestrial and aquatic communities has evolved into a unique discipline of "road ecology" (Forman et al. 2003). Road effects are far more pervasive than originally believed and include such disparate consequences as population and habitat fragmentation, accelerated rates of soil erosion, and invasion of exotic plants along roadways. Indeed, "in public wildlands management, road systems are the largest human investment and the feature most damaging to the environment" (Gucinski et al. 2001:7). Summaries of the effects of roads on wildlife habitats and biological systems in general have been compiled by Forman and Alexander (1998), Trombulak and Frissell (2000), Gucinski et al. (2001), Forman et al. (2003) and Gaines et al. (2003).

Well-designed research that furthers our understanding of road effects and road management on key species, such as elk, and their habitats is critical for enhancing the long-term functioning of ecosystems impacted by the vast network of roads in North America. Moreover, addressing effects of roads on elk and elk habitat often is mandated on public land, e. g., through standards and guidelines developed for national forests.

Our goals in this paper are three-fold: (1) to describe current knowledge about effects of roads on elk, emphasizing results of research conducted at Starkey, (2) to describe an example in which a distance-band approach, rather than the traditional road density method, was used to evaluate habitat effectiveness (HE) for elk in relation to roads, and (3) to discuss the broader implications of road-related policies and land management with regard to elk.

Effects of Roads on Elk in Forested Ecosystems—What Do We Know?

Effects of roads on elk can be divided into two broad categories: indirect effects on habitats occupied by elk and direct effects on individual elk and their populations. Effects of roads in forested ecosystems in general have been well summarized (Gucinski et al. 2001, Gaines et al. 2003). With regard to elk habitat, the primary effect of roads may be habitat fragmentation; heavily roaded areas may contain few patches of forest cover large enough to function effectively as habitat for elk, especially where elk are hunted (Leege 1984, Rowland et al. 2000). The total loss of elk habitat from road construction is unknown; a rough estimate of 5 acres per linear mile (1.4 ha/km) of road is often applied (Forman et al. 2003). Across the United States, the area occupied by public roads and associated corridors is estimated to be 27 million acres (10.9 million ha); these numbers do not include private roads or unofficial roads on public land (Forman et al. 2003). Roads may also exert more subtle influences on habitat; for example, they may facilitate the spread of exotic vegetation (Gelbard and Belnap 2003), which may subsequently reduce quality and abundance of forage available to elk. Gaines et al. (2003) listed five road-associated factors in relation to elk: hunting, poaching, collisions, displacement or avoidance, and disturbance at a specific site.

The direct impacts of roads and associated traffic on elk, in addition to outright mortality from collisions with motorized vehicles, can be summarized as follows.

1. *Elk avoid areas near open roads.* A plethora of studies have demonstrated an increasing frequency of elk occurrence or indices of elk use, such as pellet groups, at greater distances from open roads (defined here as any road where motorized vehicles are allowed). This response varies in relation to traffic rates (Wisdom 1998, Johnson et al. 2000, Ager et al. 2003), the extent of forest canopy cover adjacent to roads (Perry and Overly 1977, Lyon 1979, Wisdom 1998, Wisdom et al. 2004b), topography (Perry and Overly 1977, Edge and Marcum 1991), and type of road (e. g., improved versus primitive; Perry and Overly 1977, Lyon 1979, Witmer and deCalesta 1985, Marcum and Edge 1991, Rowland et al. 2000, Lyon and Christensen 2002, Benkobi et al. 2004), which also correlates with traffic rates. Responses may also differ between sexes, with bull elk demonstrating a stronger avoidance of areas close to roads than do cow elk (Marcum and Edge 1991). Shifts in distribution of elk away from roads may occur across a range of temporal and spatial scales. For example, elk at Starkey were generally farther from open roads during daytime but moved closer to roads during nighttime (Wisdom 1998, Ager et al. 2003). This pattern was also observed in South Dakota (Millspaugh 1999). In addition, both daily movements and size of home ranges of elk may decrease when open road density

decreases. These reductions could lead to energetic benefits that translate into increased fat reserves or productivity (Cole et al. 1997). On a larger scale, entire ranges can be abandoned if disturbance from traffic on roads and the associated habitat loss and fragmentation exceed some threshold level. The ultimate effect of displacement of elk, by motorized traffic as well as other disturbances, is a temporary or permanent reduction in effective habitat for elk. Concomitant with loss of effective habitat are reduced local and regional populations (Forman et al. 2003).

2. *Elk vulnerability to mortality from hunter harvest, both legal and illegal, increases as open road density increases.* Many factors affect elk vulnerability to hunter harvest, but the evidence is compelling that survival rates of elk are reduced in areas with higher road density (Leege 1984, Leptich and Zager 1991, Unsworth et al. 1993, Gratson and Whitman 2000a, Weber et al. 2000, Hayes et al. 2002, McCorquodale et al. 2003). Closing roads offers more security to elk and may decrease hunter densities (fewer hunters may be willing to hunt without vehicle access). Also, poaching losses may decrease when roads are closed (Cole et al. 1997).

3. *In areas of higher road density, elk exhibit higher levels of stress and increased movement rates.* Higher levels of physiological indicators of stress, such as fecal glucocorticoids, have been observed in elk exposed to increased road density and traffic on roads (Millspaugh et al. 2001). In addition, the energetic costs of moving away from disturbance associated with roads may be substantial (Cole et al. 1997). Research to estimate such costs to elk in relation to recreational use on roads is underway at Starkey (Wisdom et al. 2004a). Conversely, elk may conserve energy by traveling on closed roads to avoid woody debris and downfall (Lyon and Christensen 2002).

Knowledge has been gained not only about elk response to roads, but also about modeling of this relationship. Results from research at Starkey suggested that a road-effects model based on distance bands provides a more spatially explicit and biologically meaningful tool than a traditional model based on road density (Rowland et al. 2000). This analysis, based on more than 100,000 radiolocations of cow elk during spring and summer, found no relation between numbers of elk locations and HE scores based on open road density in 15 elk analysis units. (We define habitat effectiveness as the "percentage of available habitat that is usable by elk outside the hunting season" [Lyon and Christensen 1992:4].) However, elk preference increased strongly (as measured by selection ratios) as distance to open roads increased. Such distance-to-roads analyses are readily accomplished using widely available spatial data layers in a GIS.

Despite the wealth of information about how roads and motorized traffic affect elk and their habitats, gaps in our knowledge remain. For example, although we know that elk response to roads generally varies depending on the level and type of motorized traffic, we have little knowledge about the precise levels of such disturbance that elicit a response and the duration of that response. Research at Starkey has demonstrated threshold rates of traffic above which a response by elk is elicited but below which open roads are functionally equivalent to closed roads (A. A. Ager, personal communication 2003; Wisdom et al. 2004b). Measurements of traffic rates and elk response to these rates are needed in other locations to better understand these thresholds. Though more costly to obtain than maps of roads, information about traffic rates can be used to improve management of roads in elk habitat in ways that are both cost-effective and beneficial to elk. Further research also is needed to better understand the interaction of roads, topography and forest cover in affecting elk distributions, primarily in relation to providing security for elk.

Also needed is a better understanding of the effectiveness of road closures; examples abound about the lack of effectiveness of closures on public land, especially when few resources are made available for enforcement (Havlick 2002, Wertz et al. 2004). More than half of 802 road closures inventoried on national forests in Idaho, Montana, Washington and Wyoming were found to be ineffective, even after accounting for administrative use (Havlick 2002). In Idaho, elk mortality was positively correlated to densities of both closed roads and open roads, suggesting that road closures were ineffective in reducing mortality from hunting (Hayes et al. 2002). Systematically collected data on use by all motorized vehicles, including off-highway vehicles, of closed roads would benefit management of elk and other resources (e. g., soils) affected by vehicle traffic on roads. And last, HE models for elk, including the roads variable, need further validation. Beyond the Starkey research (Rowland et al. 2000) and a few other studies (e. g., Roloff et al. 2001, Benkobi et al. 2004), such validation has not been conducted, especially of the most commonly applied models (Wisdom et al. 1986, Thomas et al. 1988). Given the continued widespread use of elk HE models in land-use planning on national forests and on other land occupied by elk, such validation is a critical research need.

A final cautionary note: much of what has been learned about elk and roads to date has resulted from field studies that had no experimental component and, thus, no sound basis from which to infer cause-effect relations. Experimental studies underway at Starkey—in which road densities and traffic rates are manipulated according to strict sampling protocols and distributions of elk are closely monitored—will greatly enhance our understanding of elk response to roads (Wisdom et al. 2004b).

Current Management Approaches to Elk-Roads Issues

In light of the deleterious effects of roads on elk as described above, both ungulate biologists and land managers have developed methods to address their respective concerns. During the 1970s and 1980s, biologists created a suite of models, based on empirical data, to predict effects of land management activities on habitat effectiveness for elk (e. g., Lyon 1979, 1983; Thomas et al. 1979, 1988; Leege 1984; Wisdom et al. 1986). All of these models incorporated a road-density component. In addition to the more general elk HE models, specific habitat guidelines related to roads were written. For example, guidelines developed in Montana specified that elk security areas be located more than 0.5 mile (0.8 km) from open roads (Hillis et al. 1991). Elk habitat models that include a roads component also have been used to evaluate the suitability of sites for restoration of elk populations (Didier and Porter 1999). Further, ungulate biologists have constructed resource selection models that include a roads variable to predict spatial distributions of elk (Cooper and Millspaugh 1999, Johnson et al. 2000).

Land managers, in turn, have incorporated concerns about elk and roads into formal planning processes through the application of standards and guidelines. How management agencies address elk-roads issues varies widely, however, both within and across agencies. For example, elk are designated as a management indicator species (MIS) within some national forests but not others. This designation, or lack thereof, subsequently affects how elk habitat is addressed in forest planning and environmental assessment.

Forest plans for many national forests contain specific standards and guidelines for elk HE, using one or more of the various elk HE models that have been developed. For example, the forest plan for the Wallowa-Whitman National

Forest in northeastern Oregon provides direction to maintain HE at greater than 0.5 during timber sale planning in management area 1 (MA1; timber production emphasis), but only, "where this can be done without reducing timber harvest volumes" (U. S. Department of Agriculture, Forest Service 1990b:4–57). (Habitat effectiveness scores range from 0 to 1.0 in most HE models.) Furthermore, the plan assumes that, in the long-term, elk HE will be maintained at 0.62 in MA1. Open road density in this management area is targeted not to exceed 2.5 miles per square mile (1.6 km/km^2) in general but no more than 1.5 miles per square mile (0.9 km/km^2) in selected elk summer and winter ranges. In the adjacent Umatilla National Forest, elk HE is projected to range between 0.67 and 0.70, and open road density from 2.0 to 2.2 miles per square mile (1.2–1.4 km/km^2), forest-wide during the five decades beyond 1990 (U. S. Department of Agriculture, Forest Service 1990a). In addition, the standard for elk HE on big game winter range is 0.70 (U. S. Department of Agriculture, Forest Service 1990a). Generally, if habitat for elk is identified as an issue for a proposed management activity, such as timber restoration, or if elk have been identified as a MIS, evaluation of elk habitat is mandated during the environmental assessment process. Such evaluation commonly entails the application of an elk HE model to the affected area under the various alternatives, with the results incorporated into an effects analysis for evaluation of alternatives.

A more recently developed approach incorporates evaluations of habitat effectiveness for elk into the initial stages of forest planning, rather than using HE models to evaluate effects of single management activities, such as timber harvests (Bettinger et al. 1999). This approach incorporates elk HE into the objective function of a mathematical forest-planning model. Various scenarios can be simulated, with maximization of elk HE scores, timber output, or both. Likewise, Roloff et al. (1999) developed a decision support system that allows evaluation of effects of various management strategies on habitat for elk and other wildlife within the context of forest planning models.

Applying a Distance-Band Model of Elk-Road Effects in Forest Planning: A Case Example

A method to evaluate effects of roads on elk using a distance-band approach has been suggested both by Roloff (1998) and by Rowland et al. (2000), as described above. Based on radiolocations of elk at Starkey, Rowland et al.

(2000) found no relation between number of elk locations and HE based on open road densities. By contrast, the authors found a strong, linear increase in selection ratios of elk as distance to roads increased. For this analysis, elk locations were assigned to 109-yard- (100-m-) wide bands away from open roads. Roloff (1998) also developed a road-effects module in which habitat adjacent to roads was buffered into distance bands in a GIS. Habitat effectiveness in the bands was adjusted according to level of security cover, as well as road use or road type. Regardless of the exact approach selected, ongoing planning efforts within national forests and other land that provide habitat for elk may benefit from consideration of a revised, spatially explicit road-effects variable.

The mechanics of calculating HE related to roads (HE_R) using distance bands are similar to those for another variable in elk HE models—the size and spacing of cover and forage (HE_S). Both variables involve buffering outwards from linear features—either roads, for HE_R, or the cover and forage edges, for HE_S—to create distance bands. Each band is assigned a weight, with lower weights corresponding to lower HE. A weighted average is then calculated, based on the proportion of the analysis area in each of the bands and the weight of the appropriate band (see Hitchcock and Ager 1992). The sum of these products yields the final HE value, which cannot exceed 1.0.

To examine how the method of calculation (i. e., the traditional road-density method versus distance bands) might affect HE_R for elk, we applied both methods in an evaluation of the effects of a timber sale in the Wallowa-Whitman National Forest in northeastern Oregon. The Dark Meadow Restoration Project was proposed to restore and enhance ecosystems within the project area, through thinning, prescribed fire and mechanical fuels-reduction treatments over the next 10 to 15 years (U. S. Department of Agriculture, Forest Service 2003). Project goals include reductions in fuel loading, promotion of old-growth habitat, improvement in big game habitat and initiation of tree regeneration. Under the two action alternatives of the project, open road density will be lower than that under the no action (existing condition) alternative (Table 1, Figure 1).

The Dark Meadow Restoration Project encompasses 17,700 acres (7,169 ha) of the Blue Mountains and is completely contained within the Starkey Game Management Unit. The elk population in this unit is estimated to be at the objective (5,300) set by Oregon Department of Fish and Wildlife. The area functions primarily as summer range for elk, with smaller portions used as

Table 1. Comparison of two methods for modeling effects of roads on elk habitat effectiveness (HE) under three alternatives in the Dark Meadow Restoration Project, Wallowa-Whitman National Forest, northeastern Oregon.

Variable	"No action" alternative[a]	Alternative 1	Alternative 2
Total miles (km) of open roads in analysis area[b]	138.1 (222.2)	114.2 (183.7)	106.5 (171.4)
Open road density in miles per square mile (km/km^2)	4.99 (3.09)	4.13 (2.56)	3.85 (2.39)
HE_R-ORD[c]	0.20	0.28	0.31
HE_R-DB[c]	0.17	0.19	0.20
HE_C[d]	0.60	0.59	0.61
HE_S[e]	0.84	0.79	0.80
Total HE (ORD method)[f]	0.47	0.51	0.52
Total HE (DB method)	0.45	0.46	0.47

[a]This alternative is the existing condition.
[b]Open roads include any road available to motorized traffic; these are roads officially designated as open as well as closed roads that have no promulgation.
[c]Habitat effectiveness for roads (HE_R) based on open road densities (ORD); HE_R-DB uses distance bands (DB) to calculate HE_R.
[d]Habitat effectiveness as related to cover quality.
[e]Habitat effectiveness as related to size and spacing of cover and forage areas.
[f]Total habitat effectiveness, which is the geometric mean of HE_R, HE_C, HE_S and HE_F. HE_F (habitat effectiveness as related to forage quality and quantity) was not derived empirically for this analysis; rather, a default value of 0.5 was input for this variable.

transitional or winter range. Lack of elk security habitat was identified as a key issue in planning for the Dark Meadow Restoration Project; thus, roads were a primary consideration in the crafting of alternatives (U. S. Department of Agriculture, Forest Service 2003).

To calculate HE_R for elk in Dark Meadow Restoration Project, all roads open to motorized vehicles were counted. No traffic rate data were available; thus, roads were not weighted according to level of use. We defined open roads as those officially designated as open as well as closed roads for which no promulgation was planned. Promulgated road closures are those for which the Code of Federal Regulations is applied; such closures are legal and enforceable. In the Wallowa-Whitman National Forest Plan, closed roads were assumed to be physically impassable to full-sized vehicles and also assumed to be seldom traveled by off-highway vehicles (U. S. Department of Agriculture, Forest Service 1990b). Roads designated as closed but not promulgated, however, are often traveled by off-highway vehicles (Havlick 2002).

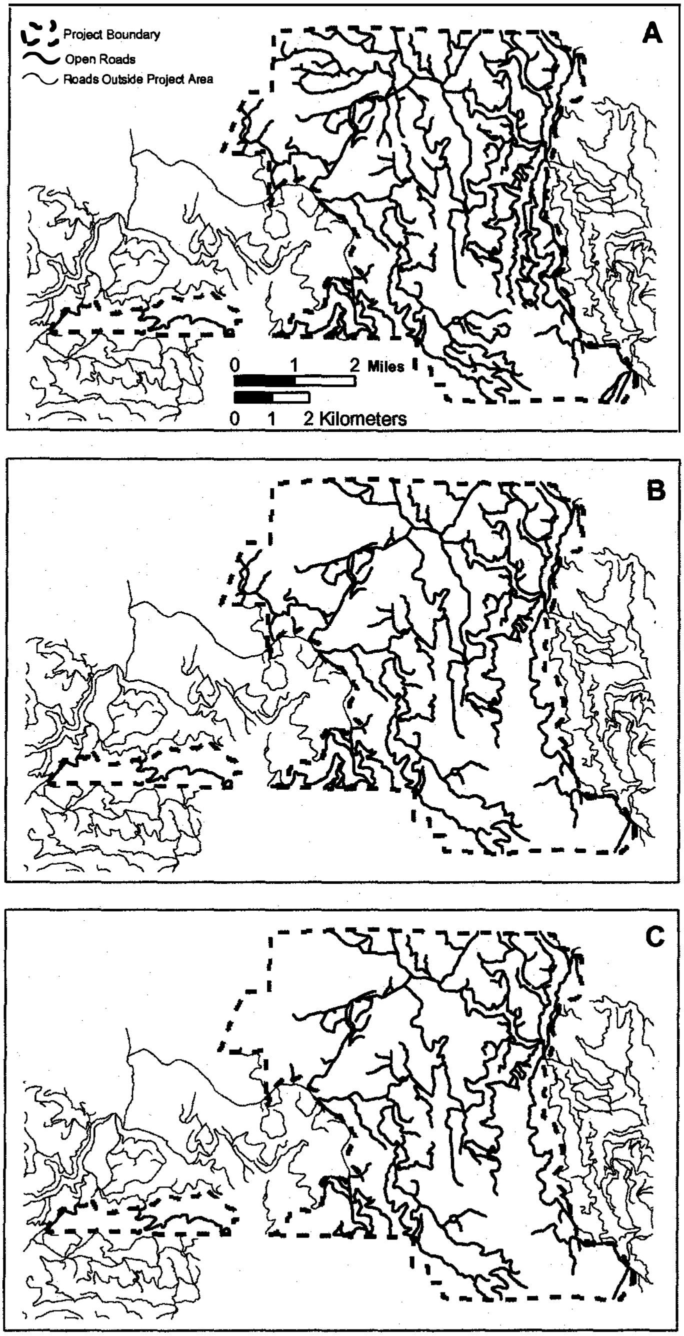


Figure 1. Open roads under three alternatives of the Dark Meadow Restoration Project, Wallowa-Whitman National Forest, northeastern Oregon: the "no action" alternative (A); Alternative 1 (B); and Alternative 2 (C). Open roads were defined as any road available to motorized traffic, including roads officially designated as open and closed roads that have no promulgation.

The HE_R variable based on open road densities (ORD)(hereafter referred to as HE_R-ORD) was then calculated with the equations of Hitchcock and Ager (1992) for the existing condition and the two action alternatives (Table 1). To calculate HE_R based on distance bands (HE_R-DB), all open roads were buffered in a GIS. The analysis area was partitioned into five bands, each 394 yards (360 m) wide, with the sixth band containing any area greater than 1,969 yards (1,800 m) from an open road. This distance (i. e., 1,969 yards) is equivalent to that at which elk response to open roads diminished markedly at Starkey (Rowland et al. 2000). Each band was assigned a weight, reflecting a linear increase in elk selection ratios as distance from open roads increased at Starkey: band 1 was 0.17, band 2 was 0.33, band 3 was 0.50, band 4 was 0.67, band 5 was 0.83 and band 6 was 1.0. HE_R-DB was then calculated as a weighted average, with the proportion of the analysis area in each band multiplied by the appropriate weight. Finally, we calculated total HE for the analysis area, based on the four variables of the elk HE model, with only HE_R differing between the two calculations (Table 1).

Open road density in the Dark Meadow Restoration Project area was relatively high under all three alternatives, and HE_R-DB was consistently lower than HE_R-ORD (Table 1). However, this difference was more pronounced with lower open road densities; under the no action alternative, HE_R-DB was only 15 percent less than HE_R-ORD, but, under the two action alternatives, this difference increased to at least 32 percent (Table 1). Compared to the no action alternative, the density of open roads declined 17 and 23 percent, respectively, under alternatives 1 and 2. Concomitant with this decline in road density were increases in HE_R-ORD of 40 and 55 percent for the two action alternatives, respectively; however, HE_R-DB increased only 12 and 18 percent (Table 1). These results suggest that the spatial arrangement of remaining open roads was such that the amount of effective habitat for elk improved only marginally (Figure 1). Thus, HE_R-ORD may overestimate habitat effectiveness for elk under certain conditions.

Because total HE is the geometric mean of all four input variables, differences in total HE between the two methods were not as substantial as were those for HE_R alone (Table 1). Among the four variables used to calculate HE, all of which are equally weighted in computing the mean, values for HE_R were substantially lower than those of the other three variables (Table 1). Thus, in the Dark Meadow Restoration Project, the relatively high open road densities were largely responsible for the low total HE scores. These scores exceeded only slightly the recommended standard of 0.5 for total HE in timber planning on the Wallowa-Whitman National Forest and only when HE_R-ORD was used for the roads variable (Table 1). By contrast, when HE_R-DB was used, total HE was below the standard for all alternatives (Table 1).

We did not alter band weights, or back buffer them, based on the level of security cover in each band (see Roloff 1998). This additional refinement may be warranted in situations where cover quality varies widely across the analysis area, or is predicted to vary under proposed management alternatives. In addition, band weights could be adjusted by accounting for topographic relief, such that areas providing topographic barriers to human disturbance would have weights adjusted upward, or by traffic rates, if such data were available.

Implications for Management and Policy Involving Elk-Roads Issues

Road management inevitably involves tradeoffs between the benefits of increased access that roads provide versus the ecological and economic costs associated with roads (Gucinski et al. 2001, Forman et al. 2003). Because the U. S. Department of Agriculture, Forest Service manages about 10 percent of the public road system in the United States (Forman et al. 2003), road-management decisions made by that agency strongly influence current road systems. U. S. Department of Agriculture, Forest Service policy regarding road closures and construction continues to engender controversy, exemplified by the multiyear debate over the national roadless rule. The rule, first published in the Federal Register in January 2001 (U. S. Office of the Federal Register 2001), has been challenged by at least nine lawsuits in federal district courts. Decisions about roads, including construction, reconstruction, closure, obliteration or decommissioning, are complex because they affect a multitude of resources, not just wildlife. All resource values in a watershed must be evaluated when making decisions about roads; these may include human safety (e. g., access to combat wildfires), soils, recreation, commercial timber harvest and restoration activities. In addition, decisions about roads are closely tied to available funding. Expenses are involved both in constructing, maintaining and decommissioning roads and in enforcing road closures (Forman et al. 2003). Complicating the issue of evaluating effects of roads is that roads in forested ecosystems currently are not well inventoried (Gucinski et al. 2001).

The potential implications of road-related policies for elk management are diverse and complex. Benefits of road closures may include:

- decreased energy expenditure by elk, a result of less frequent disturbance by motorized vehicles, with potential improvements in animal performance
- increases in total amount of effective habitat for elk in the area affected by the closures
- increased hunting opportunities on public land, when roads are closed on public land adjacent to comparatively less-roaded private land, thereby enticing elk to remain on public land rather than moving to private land where hunting may not be allowed or is prohibitively expensive (Wertz et al. 2004)
- decreased damage to crops and haystacks from elk on private land, due to lessened disturbance from traffic on public land, which in turn causes elk to remain on public land longer during the fall and winter seasons
- improvements in diet quality when elk are able to forage undisturbed in areas previously avoided due to excessive motorized traffic; these changes may translate into improvements in animal fitness and population performance
- increased hunter satisfaction, defined as either the ability to hunt in a roadless area or the access to roads and the use of all-terrain vehicles on closed roads or other off-highway sites (Gratson and Whitman 2000b)
- decreased vulnerability of elk during hunting seasons, due to fewer hunters willing to hunt without a vehicle or able to access the area.

Road closures alone may not be effective in eliminating effects of roads and traffic on elk because of inadequate enforcement. For this reason, the U. S. Department of Agriculture, Forest Service may promulgate road closures in addition to designating roads as closed, as in the Dark Meadow Restoration Project discussed above. Careful assessment of how roads are being used, rather than their official status, is important to credibly evaluate effects of roads on elk and other wildlife. Likewise, judicious closing of certain road segments, particularly road spurs (Forman et al. 2003), may retain or create blocks of habitat that serve as security areas for elk while allowing sufficient road access for other management needs. Spatially explicit models and tools are currently available to aid in evaluating among road closure alternatives.

Elk continue to exert tremendous impact on local economies, through their status as a premier game species, and on forested ecosystems, through their role as abundant, widespread large herbivores. Given the indisputable effect of roads on distribution of elk, roads and their management will undoubtedly remain, as stated by Lyon and Christensen (2002:566), "central to elk management on public and private lands."

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Benkobi, L., M. A. Rumble, G. C. Brundige, and J. J. Millspaugh. 2004. *Refinement of the Arc-Habcap model to predict habitat effectiveness for elk, research paper RMRS-RP-51*. Fort Collins, Colorado: U. S. Department of Agriculture, Forest Service.

Bettinger, P., K. Boston, and J. Sessions. 1999. Combinatorial optimization of elk habitat effectiveness and timber harvest volume. *Environmental Modeling and Assessment*. 4:143–53.

Cole, E. K., M. D. Pope, and R. G. Anthony. 1997. Effects of road management on movement and survival of Roosevelt elk. *Journal of Wildlife Management*. 61:1,115–26.

Cooper, A. B., and J. J. Millspaugh. 1999. The application of discrete choice models to wildlife resource selection studies. *Ecological Applications*. 80:566–75.

Didier, K. A., and W. F. Porter. 1999. Large-scale assessment of potential habitat to restore elk to New York State. *Wildlife Society Bulletin*. 27:409–18.

Edge, W. D., and C. L. Marcum. 1991. Topography ameliorates the effects of roads and human disturbance on elk. In *Proceedings elk vulnerability symposium*, eds. A. G. Christensen, L. J. Lyon, and T. N. Lonner, 132–7. Bozeman, Montana: Montana State University.

Forman, R. T. T., and L. E. Alexander. 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics*. 29:207–31.

Forman, R. T. T., D. Sperling, J. A. Bissonette et al. 2003. *Road ecology: Science and solutions*. Washington, DC: Island Press.

Gaines, W. L., P. H. Singleton, and R. C. Ross. 2003. *Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee National Forests, general technical report PNW-GTR-586.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Gelbard, J. L., and J. Belnap. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology.* 17:420–32.

Gratson, M. W., and C. Whitman. 2000a. Road closures and density and success of elk hunters in Idaho. *Wildlife Society Bulletin.* 28:302–10.

———. 2000b. Characteristics of Idaho elk hunters relative to road access on public lands. *Wildlife Society Bulletin.* 28:1,016–22.

Gucinski, H., M. H. Brooks, M. J. Furniss, and R. R. Ziemer. 2001. *Forest roads: A synthesis of scientific information, general technical report PNW-GTR-509.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Havlick, D. G. 2002. *No place distant: Roads and motorized recreation on America's public lands.* Washington, DC: Island Press.

Hayes, S. G., D. J. Leptich, and P. Zager. 2002. Proximate factors affecting male elk hunting mortality in northern Idaho. *Journal of Wildlife Management.* 66:491–99.

Hieb, S. R., ed. 1976. *Proceedings of the elk-logging-roads symposium.* Moscow, Idaho: Forest, Wildlife, and Range Experimental Station, University of Idaho.

Hillis, J. M., M. J. Thompson, J. E. Canfield, L. J. Lyon, and T. N. Lonner. 1991. Defining elk security: The Hillis paradigm. In *Proceedings elk vulnerability symposium,* eds. A. G. Christensen, L. J. Lyon, and T. N. Lonner, 38–43. Bozeman, Montana: Montana State University.

Hitchcock, M., and A. Ager. 1992. *Microcomputer software for calculating an elk habitat effectiveness index on Blue Mountain winter range, general technical report PNW-GTR-301.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk during spring. *Journal of Wildlife Management.* 64:685–97.

Leege, T. A. 1984. Guidelines for evaluating and managing summer elk habitat in northern Idaho. In *Wildlife Bulletin 11.* Boise, Idaho: Idaho Department of Fish and Game.

Leptich, D. J., and P. Zager. 1991. Road access management effects on elk mortality and population dynamics. In *Proceedings elk vulnerability symposium*, eds. A. G. Christensen, L. J. Lyon, and T. N. Lonner, 126–31. Bozeman, Montana: Montana State University.

Lyon, L. J. 1979. Habitat effectiveness for elk as influenced by roads and cover. *Journal of Forestry*. 79:658–60.

———. 1983. Road density models for describing habitat effectiveness for elk. *Journal of Forestry*. 81:592–95.

———. 1984. *Field tests of elk/timber coordination guidelines, research paper INT-RP-325*. Ogden, Utah: U. S. Department of Agriculture, Forest Service.

Lyon, L. J., and A. G. Christensen. 1992. *A partial glossary of elk management terms, general technical report INT-GTR-288*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

———. 2002. Elk and land management. In *North American elk: Ecology and management*, eds. D. E. Toweill and J. W. Thomas, 557–81. Washington, DC: Smithsonian Institution Press.

Lyon, L. Jack, T. N. Lonner, J. P. Weigand, C. L. Marcum, W. D. Edge, J. D. Jones, D. W. McCleerey, and L. L. Hicks. 1985. *Coordinating elk and timber management: Final report of the Montana Cooperative Elk-Logging Study*. Helena, Montana: Montana Department of Fish, Wildlife, and Parks.

Marcum, C. L., and W. D. Edge. 1991. Sexual differences in distribution of elk relative to roads and logged areas in Montana. In *Proceedings elk vulnerability symposium*, eds. A. G. Christensen, L. J. Lyon, and T. N. Lonner, 142–48. Bozeman, Montana: Montana State University.

McCorquodale, S. M., R. Wiseman, and C. L. Marcum. 2003. Survival and harvest vulnerability of elk in the Cascade Range of Washington. *Journal of Wildlife Management*. 67:248–57.

Millspaugh, J. J. 1999. *Behavioral and physiological response of elk to human activities in the southern Black Hills, South Dakota*. Ph.D. dissertation, University of Washington, Seattle.

Millspaugh, J. J., R. J. Woods, K. E. Hunt, K. J. Raedeke, G. C. Brundige, B. E. Washburn, and S. K. Wasser. 2001. Fecal glucocorticoid assays and the physiological stress response in elk. *Wildlife Society Bulletin*. 29:899–907.

Perry, C., and R. Overly. 1977. Impact of roads on big game distribution in portions of the Blue Mountains of Washington. In *Bulletin No. 11.* Olympia, Washington: Washington Game Department.

Roloff, G. J. 1998. Habitat potential model for Rocky Mountain elk. In *Proceedings 1997 deer/elk workshop, Rio Rico, Arizona*, ed. J. C. deVos, Jr., 158–75. Phoenix, Arizona: Arizona Game and Fish Department.

Roloff, G. J., B. Carroll, and S. Scharosch. 1999. A decision support system for incorporating wildlife habitat quality into forest planning. *Western Journal of Applied Forestry*. 14:91–99.

Roloff, G. J., J. J. Millspaugh, R. A. Gitzen, and G. C. Brundige. 2001. Validation tests of a spatially explicit habitat effectiveness model for elk. *Journal of Wildlife Management*. 65:899–914.

Rost, G. R., and J. A. Bailey. 1979. Distribution of mule deer and elk in relation to roads. *Journal of Wildlife Management*. 43:634–41.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Thomas, J. W., D. A. Leckenby, M. Henjum, R. J. Pedersen, and L. D. Bryant. 1988. *Habitat-effectiveness index for elk on Blue Mountain winter ranges, general technical report PNW-GTR-218*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Thomas, J. W., H. Black, Jr., R. J. Scherzinger, and R. J. Pedersen. 1979. Deer and elk. In *Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington, agricultural handbook no. 553*, ed. J. W. Thomas, 104–127. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*. 14:18–30.

Unsworth, J. W., L. Kuck, M. D. Scott, and E. O. Garton. 1993. Elk mortality in the Clearwater drainage of northcentral Idaho. *Journal of Wildlife Management*. 57:495–502.

U.S. Department of Agriculture, Forest Service. 1990a. *Umatilla National Forest land and resource management plan*. Pendleton, Oregon: U. S. Department of Agriculture, Forest Service.

———. 1990b. *Wallowa-Whitman National Forest land and resource management plan*. Baker City, Oregon: U. S. Department of Agriculture, Forest Service.

———. 2003. *Dark Meadow restoration environmental assessment*. La Grande Ranger District, Wallowa-Whitman National Forest, La Grande, Oregon: U. S. Department of Agriculture, Forest Service.

U. S. Office of the Federal Register. 2001. *Rules and regulations*. Federal Register 66:9 (Jan. 12, 2001): 3,244–3,273.

Weber, K. T., C. L. Marcum, M. G. Burcham, and L. J. Lyon. 2000. Landscape influences on elk vulnerability to hunting. *Intermountain Journal of Science*. 6:86–94.

Wertz, T. L., A. Blumton, and L. E. Erickson. 2004. Conflict resolution by adaptive management: moving elk where they want to go. In *Proceedings 2001 western states and provinces deer and elk workshop*, eds. J. Mortensen, D. G. Whittaker, E. C. Meslow, 59–66. Salem, Oregon: Oregon Department of Fish and Wildlife.

Wisdom, M. J. 1998. *Assessing life-stage importance and resource selection for conservation of selected vertebrates*. Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004a. Effects of off-road recreation on mule deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:531–50.

Wisdom, M. J., L. R. Bright, C. G. Carey, W. W. Hines, R. J. Pedersen, D. A. Smithey, J. W. Thomas, and G. W. Witmer. 1986. *A model to evaluate elk habitat in western Oregon, publication no. R6-F&WL-216-1986*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Wisdom, M. J., N. J. Cimon, B. K. Johnson, E. O. Garton, and J. W. Thomas. 2004b. Spatial partitioning by mule deer and elk in relation to traffic. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:509–30.

Witmer, G. W., and D. S. deCalesta. 1985. Effect of forest roads on habitat use by Roosevelt elk. *Northwest Science*. 59:122–25.

Spatial Partitioning by Mule Deer and Elk in Relation to Traffic

Michael J. Wisdom
U. S. Department of Agriculture Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Norman J. Cimon
U. S. Department of Agriculture Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Edward O. Garton
University of Idaho, Department of Fish and Wildlife Resources
Moscow

Jack Ward Thomas
Boone and Crockett Club and University of Montana
Missoula

Introduction

Elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*) have overlapping ranges on millions of acres of forests and rangelands in western North America. Accurate prediction of their spatial distributions within these ranges is essential to effective land-use planning, stocking allocation and population management (Wisdom and Thomas 1996). Many habitat variables influence the distributions of the two ungulates, making predictions a challenging and sometimes daunting task for resource managers (Johnson et al. 1996, Ager et al. 2004).

Distance to roads open to motorized vehicles has been identified as a significant predictor of deer and elk distributions (Thomas et al. 1979). Elk in particular have shown disproportionately less use of areas near roads open to motorized traffic (Lyon 1983; Rowland et al. 2000, 2004). As a result, extensive road closures have been implemented in the western United States in an attempt to mitigate the presumed reduction in elk use of habitat near open roads. Often, road closures are implemented under the presumption that any road open to traffic, regardless of its construction standard or traffic flow, will cause avoidance by elk and that closing such roads will mitigate the avoidance (Wisdom et al. 1986, Thomas et al. 1988). Moreover, mule deer have been assumed to avoid roads in the same manner as elk (Thomas et al. 1979), although empirical support for this assumption is limited and imprecise (see Perry and Overly 1977 versus Rost and Bailey 1979).

Despite the assumption that any road open to traffic elicits avoidance, researchers have suspected that the rate of traffic influences the magnitude of potential avoidance, especially by elk (Lyon and Christensen 2002). This was confirmed indirectly by Perry and Overly (1977), Rost and Bailey (1979), and Witmer and deCalesta (1985), among others, who found less elk use of areas near primary or main roads than near secondary or primitive roads, presumably due to a higher rate of traffic on primary roads and a higher level of human activity associated with the traffic. In addition, Rowland et al. (2000, 2004) found that elk showed increasingly strong selection toward areas with increasing distance from roads open to motorized traffic at the Starkey Experimental Forest and Range (Starkey). This research was especially compelling, owing to the large number of radio-collared elk and telemetry locations on which results were based during three years of study.

Until recently, however, no research has examined the explicit relationship of mule deer and elk distributions with traffic rates on jointly occupied range. Johnson et al. (2000) included roads with varying traffic rates as part of their analysis of resource selection by mule deer and elk at Starkey during spring. Such data are needed by resource managers charged with answering a myriad of questions about population and road management for these ungulates. For example, what traffic rate, if any, elicits avoidance? Is the response the same for the two species and the same during day versus night? Answers to these questions are needed to justify and guide efforts to mitigate any negative effects. Moreover, if traffic causes ungulate avoidance of areas near roads, the reduction in carrying capacity could be biologically significant (Johnson et al. 1996, Wisdom and Thomas 1996).

In this paper, we build on the earlier analyses by Johnson et al. (2000) at Starkey to further explore the spatial patterns of mule deer and elk in relation to roads of varying traffic rates and management. We specifically relate traffic rate with areas selected by mule deer and elk on spring and summer range. Our objectives were (1) to assess the degree to which mule deer and elk avoid areas near roads, based on variation in rates of motorized traffic, (2) to examine differences in response of mule deer versus elk to traffic, as an explicit test of the assumption made by earlier investigators that mule deer avoidance of open roads is similar to that of elk and (3) to describe the implications and potential uses of results for management.

Study Area and Technologies

Our study took place in the Main Study Area of Starkey in northeastern Oregon (Wisdom et al. 2004). Starkey features an automated telemetry system (ATS) that has been used to monitor the movements of a large percentage of female deer and elk (12 to 25 percent of females per species) in the Main Study Area since 1991 (Findholt et al. 1996, Rowland et al. 1997). Monitoring with the ATS has occurred during April to late December each year.

Starkey's Main Area was designed to facilitate large-scale studies of resource selection by deer and elk under population, habitat and human activity conditions that mimic conditions in national forests on spring, summer and fall ranges in the western United States. These design features are described in detail by Rowland et al. (1997) and Wisdom et al. (2004).

Methods

Deriving Traffic and Road Variables

Over 50 traffic counters were placed throughout the Main Area along roads not physically blocked to vehicle traffic. Each counter was associated with a unique road segment. Counters were located immediately beyond a segment's intersection with other roads, providing an explicit count of traffic for that segment. Each counter automatically tallied the number of vehicles passing over the associated road segment at 15-minute intervals, 24 hours a day, throughout the study. Rowland et al. (1997) described details about the counters and their use in collecting traffic data.

We used the counts of traffic to characterize the rate of traffic on each road segment for spring (mid-April to mid-June) and summer (mid-June to mid-August), 1993 to 1995, with the following steps. First, we summed the counts of traffic per counter for each hour of each day, and we summed these counts across like hours for each counter per season per year. Second, we used the summed counts to identify the 12-hour period of highest vehicle frequency versus the 12-hour period of lowest vehicle frequency for a generalized 24-hour day for each counter per season per year; we did this by calculating the ratio of traffic counted for all possible pairs of 12-hour periods for each counter per season per year, using a 12-hour moving window analysis, with each 12-hour window advancing in 30-minute intervals. Third, for a given season and year (season-year period), we defined day as being the 12-hour portion of the generalized 24-hour day that had the highest 12-hour ratio of traffic counts for the majority of counters, and we defined night as the opposite 12 hours. Starting times for the 12-hour portion defined as day ranged from 0530 to 0700, Pacific Standard Time, among season-year periods. Fourth, for each season-year period, we examined the frequency distributions of traffic counts for day and for night among counters and identified distinctive breaks in the distributions that resulted in five categories of traffic rate during day and three categories during night (Table 1). Traffic rates

Table 1. Definitions of 11 traffic and road variables and mean nearest distance of 86,000 pixels to each traffic and road variable, for each of six season-year periods encompassing spring (mid-April to mid-June) and summer (mid-June to mid-August), 1993 to 1995, Main Study Area, Starkey, northeastern Oregon. Values of mean nearest distance were used as estimates of the area available to deer and elk (VAVAIL, as defined in text) for MANOVA and ANOVA tests of selection in relation to traffic and road variables.

Traffic or road variable (No. vehicles/12 hrs	Nearest distance of pixels to traffic or road variable in meters					
	Spring 1993	Summer 1993	Spring 1994	Summer 1994	Spring 1995	Summer 1995
D5 (>10 day vehicles)	2,005	1,557	4,310	1,560	4,310	4,310
D4 (>4 day vehicles ≤ 10)	2,114	1,225	1,665	1,491	1,658	1,658
D3 (>1 day vehicles ≤ 4)	1,033	1,493	909	1,136	928	944
D2 (>0 day vehicles ≤ 1)	870	883	872	946	901	820
D1 (0 day vehicles)	280	280	269	280	279	280
N3 (>1 night vehicles)	1,785	761	4,471	1,388	4,471	4,470
N2 (>0 night vehicles ≤ 1)	701	713	585	531	597	591
N1 (0 night vehicles)	236	280	242	263	232	237
Open (All rates possible)	638	638	638	638	638	638
Restr. (All rates possible)	408	408	408	408	408	408
Closed (All rates possible)	398	398	398	398	398	398

were substantially higher during day, thus accounting for the larger number of traffic categories for day versus night (Table 1).

Each segment of road was then assigned to one of the five categories of traffic rate for day and one of the three categories for night, based on the category that was associated with that segment's traffic counter. The Main Area was then subdivided into 86,000 0. 22-acre (30- by 30-meter) pixels, and spatial analysis software (Ager and McGaughey 1997) used to calculate the distance of each pixel to the nearest road of each of the categories of traffic. Because segments of road often changed categories across season-year periods, the mean distance of the 86,000 pixels to the nearest road of each category of traffic often was unique for each season-year period (Table 1). Rowland et al. (1997, 1998) described additional details about the spatial database and the methods used to derive distance estimates for the traffic variables.

In addition to the traffic variables, we calculated the distance of each pixel to the nearest road open to motorized vehicles, the nearest road closed to motorized vehicles and the nearest road restricted to administrative traffic (Table 1, Rowland et al. 1997, 1998). Estimates of these road variables were an important complement to the traffic variables because such road variables are used as imprecise but presumably unbiased indices of traffic rate on national forests in northeastern Oregon, as well as in large areas of the western United States, as part of road management for deer and elk. Thus, we wanted to determine how well the patterns of ungulate selection accounted for by the traffic variables might also be indexed by the road variables. Unlike the dynamic nature of the traffic variables, whose distance estimates changed across season-year periods with shifts in traffic rate across road segments, the mean distance of the 86,000 pixels to nearest road of each type remained static across all season-year periods (Table 1).

Monitoring Animal Movements

We used the ATS during spring and summer, 1993 to 1995, to collect more than 160,000 locations from 12 to 31 radio-collared females per species per season-year period. Animals were systematically located approximately once every 3 to 4 hours ($\bar{x}$ = 3. 7 hours among season-year periods, *SE* = 0.6), which generated an average of 447 locations per female per season-year period (*SE* = 69).

The ATS computed each animal location in Universal Transverse Mercator (UTM) coordinates. Point estimates of these locations were placed within the center of the nearest 0.22-acre (30- by 30-meter) pixel. Location accuracy of the ATS was ± 58 ($\bar{x}$ = 53 meters, *SE* = 5. 9) (Findholt et al. 1996). Findholt et al. (1996), Johnson et al. (1998) and Rowland et al. (1997, 1998) described additional details about use of the ATS to collect animal locations.

Assessing Deer and Elk Selection in Relation to Traffic and Road Variables

For each season-year period, use for a given radio-collared animal for a given traffic or road variable was calculated as the mean of all distance values for the variable taken across all pixels in which the animal was located. Each location was weighted by a spatially explicit algorithm that corrected for spatial differences in the rate at which telemetry locations were successfully obtained (Johnson et al. 1998). Estimates of use (i. e., mean distance values) for all traffic and road variables for each animal in each season-year period were then placed in a use vector, defined as V_{USE}.

Similarly, availability of each traffic and road variable for each season-year period was calculated as the mean of all distance values for the variable taken across all 86,000 pixels (Table 1). Estimates of availability for all traffic and road variables for each season-year period were then placed in an availability vector, defined as V_{AVAIL}. Each vector of availability was unique to each season-year period (Table 1) due to the dynamic nature of traffic rates across seasons and years.

Within-species Selection. For each season-year period, we calculated a selection vector (V_{SELECT}) of use minus availability of all traffic and road variables for each radio-collared animal. Specifically, V_{SELECT} was calculated as a difference vector of selection values (use minus availability) of all traffic and road variables resulting from V_{USE} minus V_{AVAIL} for each animal and period. Each selection vector for a radio-collared animal in a season-year period was used as the unit of observation, or statistical replicate, to evaluate patterns of within-species selection by the population of deer or elk in relation to the traffic and road variables.

We used a two-step process to evaluate within-species selection. First, we used a fixed-effects, factorial multivariate analysis of variance (MANOVA)(Type III sum of squares [SS], generalized linear procedure [PROC GLM], SAS Institute, Inc. 1990), with season and year as factors, to initially test whether within-species selection differed by season, year or both. This MANOVA was described in detail by Wisdom (1998).

Second, with the finding of a significant factor effect ($P < 0.05$) of season, year or their interaction, we examined each individual analysis of variance (ANOVAs) to identify which traffic and road variables contributed to these effects in a biologically significant manner. We defined a biologically significant effect as any traffic or road variable that was statistically significant ($P < 0.05$) for a given factor and whose mean value for that factor was inconsistent in sign. For example, if use minus availability by season for the open roads variable was statistically significant and had a positive sign for spring (suggesting selection of areas farther from open roads than available) but a negative sign for summer (suggesting selection of areas closer to open roads than available), results for the two seasons were presented separately. Simple main effects (individual season-year periods analyzed separately) and partial main effects (some season-year periods pooled) were reported for any variable having biologically significant effects due to season, year or both.

All traffic and road variables deemed not to contribute to season, year or interaction effects in a biologically significant manner were then brought forward to test the main effects of within-species selection, all seasons and years pooled. This test was conducted using a one-sample, fixed-effects, completely randomized MANOVA (Hotelling's T2, no intercept option, Type III SS, PROC GLM, SAS Institute, Inc. 1990), as described by Wisdom (1998).

Because we had an unequal number of radio-collared replicates among seasons and years, we used the least square means option (Lsmeans, PROC GLM, or equivalent programming in PROC MEANS, SAS Institute, Inc. 1990) for all ANOVAs so that variation from radio-collared replicates for each season-year period contributed equally to univariate tests. We also weighted radio-collared replicates within each season-year period such that within-species variation by trap location (radio-collared animals trapped in main study area versus winter feed ground) was represented equally. Because of large differences in categories of traffic by day versus by night, we conducted separate MANOVAs for day and for night.

Between-species Selection. We also evaluated patterns of between-species selection by deer versus elk in relation to traffic and road variables. We used a fixed-effects, factorial MANOVA (Type III SS, Proc GLM, SAS Institute, Inc. 1990), with season, year, between-species selection, and we used their interactions as factors as outlined by Wisdom (1998).

With the finding of a significant interaction ($P < 0.05$) between any factors in the MANOVA, we examined the individual ANOVAs to identify which traffic and road variables contributed to the interaction in a biologically significant manner. Biologically significant interactions were defined as any statistically significant interaction that also contained "crossing" (Kirk 1982:356--359) of the treatment level means for between-species selection and whose treatment level means for between-species selection were inconsistent in sign and direction. For example, if the interaction of between-species selection and season for the open roads variable was statistically significant and indicated that elk were closer than deer to open roads during spring but farther from open roads during summer, results for the two seasons were presented separately.

As done for tests of within-species selection, we used the Lsmeans, procedure (PROC GLM, SAS Institute, Inc. 1990), so variation from radio-collared replicates from all season-year periods contributed equally to ANOVA tests of main effects and all interactions for between-species selection.

Results

The vector of traffic and road VSELECT by mule deer and by elk was significantly different ($P < 0.05$) from zero, both day and night, for the main effects of within-species selection (Wisdom 1998). Moreover, MANOVA tests of difference in selection between the two species also were significant ($P < 0.05$), as described in detail by Wisdom (1998). Univariate contributions of each traffic and road variable to the significant main effects and to the biologically significant interactions are described below for all within-species and between-species MANOVAs.

Within-species Selection: Deer

Both day and night, mule deer selected areas closer to roads that had day rates of more than 4 vehicles per 12 hours (D4 and D5; Figure 1a, b), closer to roads that had night rates of more than 1 vehicle per 12 hours (N3; Figure 1c, d), and closer to roads open to motorized traffic (Open; Figure 1e, f). Mule deer also selected areas closer to roads that had day rates of more than 1 but less than or equal to 4 vehicles per 12 hours except during summer, when selection was not significant (D3; Figure 1a, b).

The magnitude of selection toward D5, D4, N3 and D3 roads appeared to increase with increasing rate of traffic; roads with the highest rate—D5—had

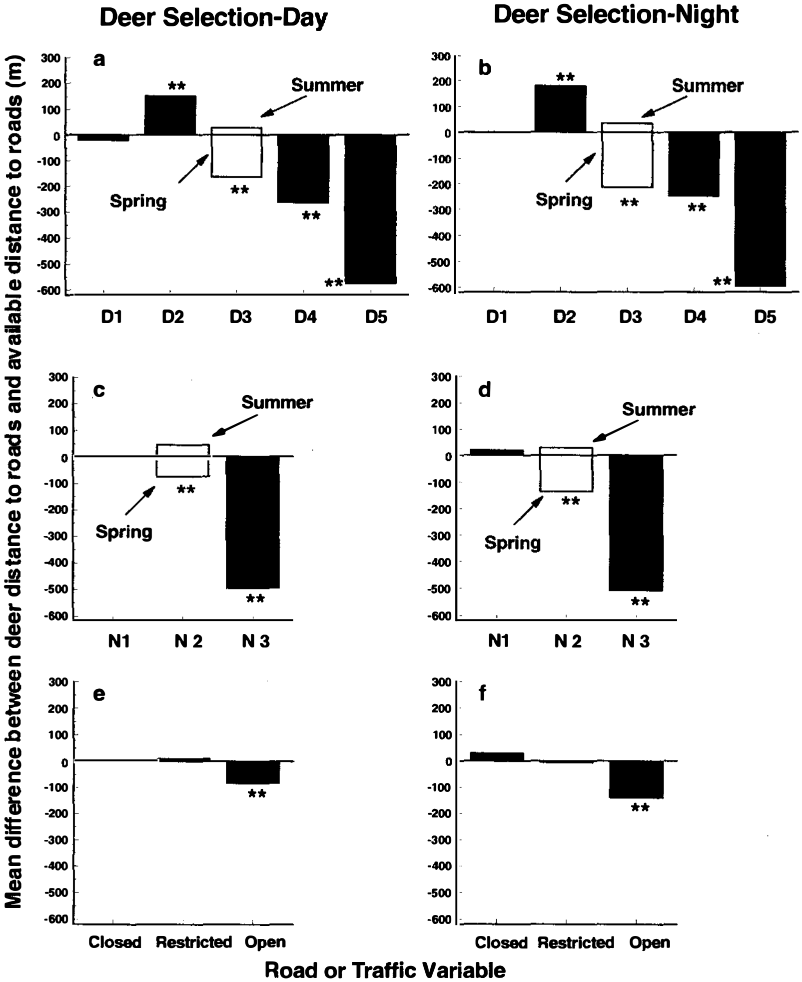


Figure 1. Areas selected by mule deer during day (a, c, e) and night (b, d, f) in relation to roads of different rate of traffic (D1–D5, N1–N3) and type (closed, restricted, open to vehicles). D1, D2, D3, D4 and D5 were roads with 0, more than 0 but fewer than or equal to 1, more than 1 but fewer than or equal to 4, more than 4 but fewer than or equal to 10, and more than 10 vehicles per 12 hours during day. N1, N2 and N3 were roads with 0, more than 0 but less than or equal to 1, and more than 1 vehicle per 12 hours during night. Main effects (all seasons and years pooled) are shown in black; partial main effects (some seasons and years pooled) are shown in white. Partial main effects are shown for variables whose parameter estimates could not be pooled across certain seasons or years due to interactions by season, year or both. Significant differences in selection are shown with asterisks (* = $P < 0.05$, ** = $P < 0.01$). Positive differences indicate selection away from roads. Negative differences indicate selection toward roads. Descriptive and test statistics are provided by Wisdom (1998).

the greatest mean difference of use versus availability, followed by N3, D4 and D3. The consistency of selection toward these roads also appeared to increase with increasing rate of traffic; roads with the highest rates—D4 and D5—had no significant season or year interactions.

In contrast to selecting areas closer to open roads and to roads that had higher rates of traffic, mule deer selected areas farther from roads that had day rates of greater than 0 but less than or equal to 1 vehicles per 12 hours (D2; Figure 1a, b) but selected areas closer to roads that had night rates of greater than 0 but less than or equal to 1 vehicle per 12 hours during spring (N2, Figure 1c, d). Finally, mule deer showed no selection in relation to roads that received no traffic (D1; Figure 1a, b; N1; Figure 1c, d) and no selection in relation to roads that were closed to vehicles or restricted to administrative traffic (Closed and Restricted; Figure 1e, f).

Within-species Selection: Elk

Both day and night, elk selected areas farther from roads that had day rates of more than 1 vehicle per 12 hours (D3, D4 and D5; Figure 2a, b). Elk also selected areas farther from roads that had night rates of more than 1 vehicle per 12 hours (N3; Figure 2c, d) and farther from roads open to motorized traffic (Open; Figure 2e, f). During night, summer 1995, however, elk showed no selection in relation to roads having night rates of more than 1 vehicle per 12 hours (N3; Figure 2d). Elk also showed no selection during night in relation to open roads for spring, all years (Figure 2f).

The consistency of elk selection away from roads appeared to increase with increasing rate of traffic; roads with the highest rates—D3, D4 and D5—had no significant season or year interactions. Roads with the highest rate of traffic—D5—also had greatest mean difference between use and availability.

In contrast to typically selecting areas farther from open roads and farther from roads that had higher rates of traffic, elk generally selected areas closer to roads that had little or no traffic. However, the pattern was not consistent among all season-year periods. Specifically, elk were closer to roads that had day rates of greater than 0 but less than or equal to 1 vehicle per 12 hours (D2; Figure 2a, c, except for spring 1994 during day), were closer to roads with no traffic (D1; Figure 2a, b, except for spring 1993; and N1; Figure 2c, d, except for spring 1993) and were closer to roads that were closed (Closed, Figure 2e, f). However, elk showed either an opposite or an inconsistent pattern of selection, day versus night,

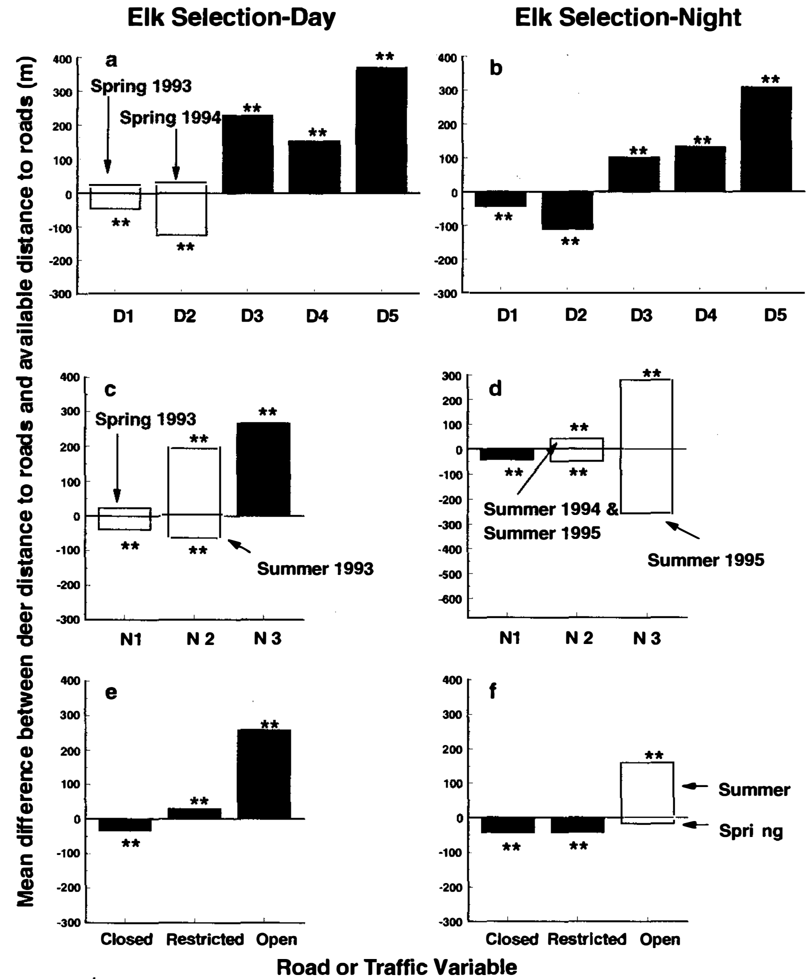


Figure 2. Areas selected by elk during day (a, c, e) and night (b, d, f) in relation to roads of different rate of traffic (D1–D5, N1–N3) and type (closed, restricted, open to vehicles). D1, D2, D3, D4 and D5 were roads with 0, more than 0 but fewer than or equal to 1, more than 1 but fewer than or equal to 4, more than 4 but fewer than or equal to 10, and more than 10 vehicles per 12 hours during day. N1, N2 and N3 were roads with 0, more than 0 but fewer than or equal to 1, and more than 1 vehicle per 12 hours during night. Main effects (all seasons and years pooled) are shown in black; partial main effects (some seasons and years pooled) are shown in white. Partial main effects are shown for variables whose parameter estimates could not be pooled across certain seasons or years due to interactions by season, year or both. Significant differences in selection are shown with asterisks (* = $P < 0.05$, ** = $P < 0.01$). Positive differences indicate selection away from roads. Negative differences indicate selection toward roads. Descriptive and test statistics are provided by Wisdom (1998).

in relation to roads that had night rates of greater than 0 but less than or equal to 1 vehicle per 12 hours (N2; Figure 2c, d). During day, elk also selected areas farther from roads restricted to administrative traffic (Restricted; Figure 2e) but selected areas closer to these same roads during night (Restricted; Figure 2f).

Between-species Selection: Deer versus Elk

Both day and night, elk were farther than deer from roads that had traffic rates of more than 1 vehicle per 12 hours (D3, N3, D4, and D5; Figure 3a, b, c, d) and farther than deer from open roads (Open; Figure 3e, f). By contrast, deer generally were farther than elk from roads that had lower traffic rates or that were restricted or closed, especially during night. Specifically, deer were farther than elk from roads having day rates of greater than 0 but less than or equal to 1 vehicle per 12 hours (D2; Figure 3a, b), farther than elk during night from roads having night rates of more than 0 but less than or equal to 1 vehicle per 12 hours (N2; Figure 3d, except for summer 1993), and farther than elk during night from roads that were restricted or closed (Restricted, Closed; Figure 3f). During day, however, deer were closer than elk to N2 roads (Figure 3c, except for summer 1993) and closer than elk to restricted roads during spring, all years (Restricted; Figure 3e). Finally, we found no difference in deer versus elk selection in relation to roads with zero traffic (D1, N1; Figure 3a-d), except during night, when deer were farther than elk (N1; Figure 3d). We also found no difference in deer versus elk selection during day in relation to closed roads (Closed; Figure 3e).

Discussion

Selection Patterns

A number of strong and surprising patterns emerged from our results. First, deer and elk selected areas in opposite ways in relation to rate of traffic, with the magnitude of difference increasing with increasing rate of traffic. Second, thresholds existed for both species in terms of direction in selection: elk were generally farther from roads with traffic rates more than 1 vehicle per 12 hours, both day and night, while deer were closer. By contrast, selection by both species was inconsistent in relation to roads with little or no traffic (less than or equal to 1 vehicle per 12 hours). Third, the type of road often correctly indexed the direction in selection shown by both species in relation to rates of traffic; that is, elk were farther from and deer were closer to roads that were open to all

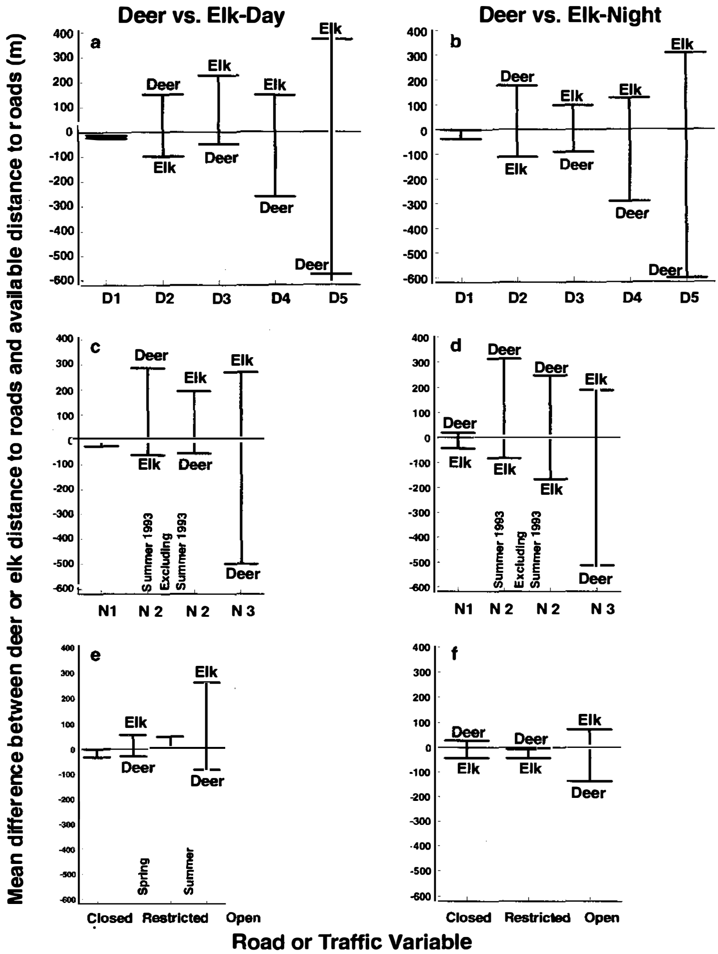


Figure 3. Differences in areas selected by mule deer versus elk during day (a, c, e) and night (b, d, f) in relation to roads of different type (closed, restricted, open to vehicles) and rate of traffic (D1–D5, N1–N3). D1, D2, D3, D4 and D5 were roads with 0, more than 0 but fewer than or equal to 1, more than 1 but fewer than or equal to 4, more than 4 but fewer than or equal to 10, and more than 10 vehicles per 12 hours during day. N1, N2 and N3 were roads with 0, more than 0 but fewer than or equal to 1, and more than 1 vehicle per 12 hours during night. Main effects (all seasons and years pooled) are shown, except when parameter estimates could not be pooled due to interactions by season, year or both. In such cases, partial main effects (some seasons and years pooled) or simple main effects (specific season-year periods analyzed separately) are shown and explicitly identified. Significant differences ($P < 0.05$) in selection are denoted by the names of each species above and below the bar. The species whose name is above the bar indicates it was significantly farther from a road than the species named below the bar. Descriptive and test statistics are provided by Wisdom (1998).

vehicular travel, which agreed with the overall direction in selection shown by both species in relation to most roads that had nonzero rates of traffic (i. e., all roads having more than 1 vehicle per 12 hours). Moreover, both species showed a weak or inconsistent pattern of selection in relation to closed or restricted roads, which agreed with the inconsistent pattern of selection shown by both species in relation to roads with little or no traffic (≤1 vehicle per 12 hours). And fourth, the type of road sometimes failed to index the magnitude of selection in relation to the traffic variables. For example, mule deer were approximately 100 to 150 yards (91–137 m) closer to open roads than available, yet deer were more than 250 yards (229 m) farther from roads of second-highest traffic rate (D4) and more than 550 yards (505 m) farther from roads of highest traffic rate (D5).

Implications for Roads and Habitat Models

Our results have direct bearing on a number of key assumptions and relationships contained in habitat models for elk. First, our results support the inclusion of an open-roads variable in spring-summer habitat models, such as done in models by Thomas et al. (1979), Leege (1984), Lyon et al. (1985), Wisdom et al. (1986) and Johnson et al. (1996). Second, our results corroborate the assumption by some elk habitat modelers (e. g. , Thomas et al. 1979, Wisdom et al. 1986) that increasing rate of traffic exerts an increasingly strong effect on elk selection; this implies that accuracy of elk habitat models could be improved by substituting a variable on traffic rate for the currently used variable on open roads. Third, our findings do not support the assumption (Thomas et al. 1979) that selection by mule deer is similar to that by elk in relation to traffic and road variables; instead, our findings suggest that separate modeling terms are needed for each species. Fourth, our results question the use of carrying capacity models that are strictly nutrition- or forage-based, such as those by Nelson (1984), Cooperrider and Bailey (1984), Van Dyne et al. (1984), and Schwartz and Hobbs (1985), due to the strong effect that traffic and traffic-indexed human activities may have on modeling results. In contrast to these models, our findings justify the inclusion of traffic variables as a potential decrement to any projections of carrying capacity for both species.

Our results would be particularly useful when considered as part of distance band assessment of roads, as recently proposed by Rowland et al. (2004). Under such an assessment, all roads open to traffic are mapped, and the associated landscape is subdivided into distance intervals or bands, according to the distance of each band to the nearest open road. The probability that elk will use each distance band, based on prior research on elk distributions in relation to distance from open roads (Rowland et al. 2000), is then assigned to each band. These probabilities are then weighted by the area of each distance band occurring on the landscape being evaluated, and an overall probability is calculated. If open roads could be further characterized by traffic rates, probabilities of elk use by distance to nearest road of each rate could be considered. This refinement in the road-distance band assessment deserves further consideration in future modeling of elk habitat use at landscape scales, such as watersheds.

Are Elk Displacing Deer?

Perhaps our most intriguing finding relates to the hypothesis of interference or disturbance competition where, "the mere presence of an animal intimidates or annoys another animal into leaving the area" (Nelson 1982:416). In the past, this hypothesis has been tested in relation to potential displacement of wild ungulates by domestic livestock. For example, many studies have shown that elk avoid or decrease their use of areas with the onset of cattle grazing (Knowles and Campbell 1982, Lonner and Mackie 1983, Wallace and Krausman 1987, Frisina 1992, Yeo et al. 1993, Coe et al. 2001, 2004), suggesting that interference competition may be operating.

The interference competition hypothesis, however, has not been tested rigorously between sympatric species of wild ungulates in North America. In potential support of this hypothesis in relation to mule deer and elk, a number of researchers (Cliff 1939, Mackie 1981, Nelson 1982) inferred that elk may out-compete and potentially displace mule deer on winter ranges that are limited in size and available forage. Nelson (1982) also believed that mule deer may leave or avoid areas of heavy use by elk, even if forage is abundant and dietary overlap with elk is low. Wisdom and Thomas (1996) also suggested that elk may displace mule deer on jointly occupied ranges when elk exist at moderate to high densities, due to a number of behavioral and physiological advantages that elk presumably have over deer. Finally, Cowan (1950:582), working in western Canada, inferred that, "mule deer and moose have decreased since elk became abundant, and the causal nature of the two events, though not established, seems probable."

Lack of direct, cause-effect evidence, however, limits firm conclusions. On the other hand, an additional analysis at Starkey (M. J. Wisdom, unpublished data) supports the assumption that mule deer change their distributions within our study area in relation to opposite changes in distribution by elk. In this additional analysis, elk were significantly farther ($P < 0.05$) from roads that had highest rate of traffic (D5) during summer 1994 (use minus availability = 338 yards), while mule deer were closer (use minus availability = - 412 yards). During a one-month, either-sex bow season on elk that immediately followed summer 1994, however, elk shifted their distributions significantly closer to these same roads (use minus availability = - 227 yards) while deer moved significantly farther away (use minus availability = 388 yards). Simultaneous with the shift by elk toward the D5 roads during the bow hunt was the inclusion of the D5 roads within a no-hunting area that extended 400 yards outward from these roads. Thus, the most plausible, logical explanation for these distributional shifts is that elk are sensitive to traffic but are more sensitive to hunting pressure, and mule deer are sensitive to the presence of elk. The results, we believe, are distributions of mule deer and elk that exist and shift in dynamic, opposite ways, in both direction and magnitude and, in agreement with the interference competition hypothesis.

Similar findings and inferences were made from earlier analyses of mule deer and elk interactions at Starkey by Johnson et al. (2000), Coe et al. (2001, 2004) and Stewart et al. (2002). In each of these analyses, investigators found strong evidence, although observational, that mule deer were avoiding elk. For example, Johnson et al. (2000) found that the strongest coefficient in explaining resource selection by mule deer was resource selection by elk. While mule deer occupied areas largely avoided by elk, the opposite was not true. That is, resource selection by elk could not be explained by selection patterns of mule deer.

Although such discussion is compelling, manipulative experiments are needed to formally test and validate the interference competition hypothesis under varying levels of deer and elk densities and rates of traffic, under the presumption that certain ungulate densities and certain traffic rates work together to cause elk to avoid areas near roads and to cause mule deer to select areas near roads as a means of avoiding elk. Ideally, such manipulative experiments would be designed to measure effects on population performance of both species. To date, analyses of the effect of vehicle access on survival (such as Cole et al. 1998 for elk) and reproduction of both species on jointly occupied range has not been conducted.

It also is important to note that some researchers have speculated that mule deer are attracted to areas near roads when roadsides have been seeded with nutritious grasses and forbs (Wallmo et al. 1976). However, attraction to forage along roadsides would not plausibly account for the large differences in mule deer selection that we observed in relation to varying rates of traffic. Moreover, the distributional shifts described above for mule deer versus elk before versus during the 1994 bow hunt provide compelling support for the interference competition hypothesis, which does not accommodate the notion that mule deer selected areas near roads due to superior roadside forage.

Limits of Inference

Our findings particularly are relevant to spring and summer ranges that are jointly occupied by deer and elk under conditions of moderate to high densities of elk. These are the conditions under which our study was conducted. However, these conditions are common across large areas of western North America. By contrast, inferring results from our study to other areas where mule deer are common and elk are absent or sparse, would be inappropriate and likely unreliable. Inferring results of our study to spring and summer ranges occupied solely by mule deer, in particular, could be especially unreliable, given the high potential for mule deer distributions in our study area to be affected strongly by selection patterns of elk. In the absence of moderate or high densities of elk, mule deer may exhibit different distribution and selection patterns in relation to roads and traffic than we observed at Starkey.

Management Implications

Differences in selection by mule deer and elk in relation to traffic could be considered in the management of motorized vehicles and traffic-related human activities on spring-summer ranges where both species occur, and where elk exist at moderate to high densities. Our results suggest that spring-summer habitat models for elk may not account for patterns of resource selection by mule deer on jointly occupied range and that resource needs of each species must be addressed separately.

Our results also suggest that inclusion of road or traffic variables in habitat models is essential to accurate portrayal of selection patterns for both species. Forage- or nutrition-based habitat models that exclude road or traffic variables have the potential to be highly inaccurate, given the large magnitude of difference in selection shown by sympatric populations of deer and elk in relation to rates of traffic and types of roads. Manipulative experiments are needed to validate the presumption that rate of traffic acts as a mechanistic cause for differences in selection patterns between mule deer and elk and to validate these selection patterns across a diversity of environments in which both species occur.

Reference List

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:687–707.

Ager, A. A. , and R. J. McGaughey. 1997. *UTOOLS: Microcomputer software spatial analysis and landscape visualization, general technical report, PNW-GTR-397.* Portland, Oregon: U. S. Department of Agriculture Forest Service, Pacific Northwest Research Station.

Cliff, E. P. 1939. Relationships between elk and mule deer in the Blue Mountains of Oregon. *Transactions of the North American Wildlife and Natural Resources Conference.* 4:560–9.

Coe, P. K. , B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:205, A51–A76.

Coe, P. K., B. K. Johnson, K. M. Stewart, and J. G. Kie. 2004. Spatial and temporal interactions of elk, mule deer and cattle. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:656–69.

Cole, E. K. , M. D. Pope, and R. G. Anthony. 1998. Effects of road management on movement and survival of Roosevelt elk. *Journal of Wildlife Management.* 61:1,115–26.

Cooperrider, A. Y., and J. A. Bailey. 1984. A simulation approach to forage allocation. In *Developing strategies for rangeland management*, 525–60. Boulder, Colorado: Westview Press.

Cowan, I. M. 1950. Some vital statistics of big game on over-stocked mountain range. *Transactions of the North American Wildlife and Natural Resources Conference.* 15:581–9.

Findholt, S. L. , B. K. Johnson, L. D. Bryant, and J. W. Thomas. 1996. Corrections for position bias of a LORAN-C radio-telemetry system using DGPS. *Northwest Science*. 70:273–80.

Frisina, M. R. 1992. Elk habitat use within a rest-rotation grazing system. *Rangelands.* 14:93–6.

Johnson, B. K. , A. Ager, S. A. Crim, M. Wisdom, S. L. Findholt, and D. Sheehy. 1996. Allocating forage among wild and domestic ungulates—A new approach. In *Proceedings of sustaining rangeland ecosystems symposium, SR 953,* eds. W. D. Edge, and S. L. Olson-Edge, 166–9. Corvallis, Oregon: Oregon State University.

Johnson, B. K. , A. Ager, S. L. Findholt, M. J. Wisdom, D. B. Marx, J. W. Kern, and L. D. Bryant. 1998. Mitigating spatial differences in observation rate of automated telemetry systems. *Journal of Wildlife Management.* 62:958–67.

Johnson, B. K, J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk in spring. *Journal of Wildlife Management*. 64:685–97.

Kirk, R. E. 1982. *Experimental design: Procedures for the behavioral sciences*, s*econd edition.* Pacific Grove, California: Brooks/Cole Publishing Company.

Knowles, C. J. , and R. B. Campbell. 1982. Distribution of elk and cattle in a rest-rotation grazing system. In *Wildlife-livestock relationships symposium: Proceedings 10*, eds. J. M. Peek, and P. D. Dalke, 46–60. Moscow, Idaho: Wildlife and Range Experiment Station, University of Idaho.

Leege, T. A. 1984. *Guidelines for evaluating and managing summer elk habitat in northern Idaho, wildlife bulletin number 11*. Boise, Idaho: Idaho Department of Fish and Game.

Lonner, T. N. , and R. J. Mackie. 1983. On the nature of competition between big game and livestock. In *Proceedings of forestland grazing symposium*, eds. B. F. Roche, Jr. , and D. M. Baumgartner, 53–8. Pullman, Washington: Washington State University.

Lyon, L. J. 1983. Road density models describing habitat effectiveness for elk. *Journal of Forestry*. 81:592–5.

Lyon, L. J. , and A. G. Christensen. 2002. Elk and land management. In *North American elk: Ecology and management*, eds. D. E. Toweill and J. W. Thomas, 557–81. Washington, DC: Smithsonian Institution Press.

Lyon, L. J. , T. N. Lonner, J. P. Weigand, C. L. Marcum, W. D. Edge, J. D. Jones, D. W. McCleerey, and L. L. Hicks. 1985. *Coordinating elk and timber management, final report of the Montana cooperative elk-logging study, 1970–1985*. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Mackie, R. J. 1981. Interspecific relationships. In *Mule and black-tailed deer of North America*, ed. O. C. Wallmo, 487–507. Lincoln, Nebraska: University of Nebraska Press.

Nelson, J. R. 1982. Relationships of elk and other large herbivores. In *Elk of North America: Ecology and management*, eds. J. W. Thomas and D. E. Toweill, 415–41. Harrisburg, Pennsylvania: Stackpole Books.

———. 1984. A modeling approach to large herbivore competition. In *Developing strategies for rangeland management,* 491–524. Boulder, Colorado: Westview Press.

Perry, C. , and R. Overly. 1977. *Impact of roads on big game distributions in portions of the Blue Mountains of Washington, 1972–1973, applied research bulletin number 11*, Olympia, Washington: Washington Department of Game.

Rost, G. R. , and J. A. Bailey. 1979. Distribution of mule deer and elk in relation to roads. *Journal of Wildlife Management.* 43:634–41.

Rowland, M. M. , L. D. Bryant, Bruce K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical Report, PNW-GTR-396.* Portland, Oregon: U. S. Department of Agriculture Forest Service, Pacific Northwest Research Station.

Rowland, M. M. , M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management.* 64:672–84.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: Implications for management in forested ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:491–508.

Rowland, M. M. , P. Coe, R. Stussy, B. K. Johnson, M. J. Wisdom, and N. J. Cimon. 1998. *The Starkey habitat database for ungulate research: construction, documentation, and use, General Technical Report, PNW-GTR-430.* Portland, Oregon: U. S. Department of Agriculture Forest Service, Pacific Northwest Research Station.

SAS Institute, Inc. 1990. *SAS/STAT user's guide, volumes 1 and 2*. Cary, North Carolina: SAS Institute, Incorporated.

Schwartz, C. C., and N. T. Hobbs. 1985. Forage and range evaluation. In *Bioenergetics of wild herbivores*, eds. R. J. Hudson, and R. G. White, 26–51. Boca Raton, Florida: CRC Press.

Stewart, K. M. , R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83:229–44.

Thomas, J. W. , D. A. Leckenby, M. Henjum, R. J. Pedersen, and L. D. Bryant. 1988. *Habitat effectiveness index for elk on Blue Mountain winter ranges, general technical report, PNW-GTR-218*. Portland, Oregon: U. S. Department of Agriculture Forest Service, Pacific Northwest Research Station.

Thomas, J. W., H. Black, R. J. Scherzinger, and R. J. Pedersen. 1979. Deer and elk. In *Wildlife habitats in managed forests—The Blue Mountains of Oregon and Washington*, *agricultural handbook number 553,* ed. J. W. Thomas, 104–27. Washington, DC: U. S. Department of Agriculture Forest Service.

Van Dyne, G. M. , P. T. Kortopates, and F. M. Smith. 1984. Quantitative frameworks for forage allocation. In *Developing strategies for rangeland management*, 289–412. Boulder, Colorado: Westview Press.

Wallace, M. C. , and P. R. Krausman. 1987. Elk, mule deer, and cattle habitats in central Arizona. *Journal of Range Management*. 40:80–3.

Wallmo, O. C. , D. F. Reed, and L. H. Carpenter. 1976. Alteration of mule deer habitat by wildfire, logging, highways, agriculture, and housing developments. In *Proceedings, mule deer decline in the West, a symposium*, eds. G. W. Workman, and J. B. Low, 37–47. Logan: Utah State University.

Wisdom, M. J. 1998. *Assessing life-stage importance and resource selection for conservation of selected vertebrates*. Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J. , and J. W. Thomas. 1996. Elk. In *Rangeland wildlife*, ed. P. R. Krausman, 157–81. Denver, Colorado: Society For Range Management.

Wisdom, M. J. , L. R. Bright, C. G. Carey, W. W. Hines, R. J. Pedersen, D. A. Smithey, J. W. Thomas, and G. W. Witmer. 1986. *A model to evaluate elk habitat in western Oregon, publication no. R6-F&WL-216–1986.* Portland, Oregon: U. S. Department of Agriculture Forest Service.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:455–74.

Witmer, G. W. , and D. S. deCalesta. 1985. Effect of forest roads on habitat use by Roosevelt elk. *Northwest Science.* 59:122–5.

Yeo, J. F. , J. M. Peek, W. T. Wittinger, and C. T. Kvale. 1993. Influence of rest-rotation cattle grazing on mule deer and elk habitat use in east-central Idaho. *Journal of Range Management.* 46:245–50.

Effects of Off-road Recreation on Mule Deer and Elk

Michael J. Wisdom
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon*

Alan A. Ager
*U. S. Department of Agriculture, Forest Service,
Umatilla National Forest
Pendleton, Oregon*

Haiganoush K. Preisler
*U. S. Department of Agriculture, Forest Service,
Pacific Southwest Research Station
Berkeley, California*

Norman J. Cimon
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon*

Bruce K. Johnson
*Oregon Department of Fish and Wildlife
La Grande*

Introduction

Off-road recreation is increasing rapidly in the United States, especially on public land (Havlick 2002, U.S. Department of Agriculture Forest Service 2004). An expansive network of roads provides easy access to much public land, which facilitates off-road uses in the form of all-terrain vehicles (ATVs), horses, mountain bikes and foot traffic. No research, however, has evaluated effects of these off-road activities on vertebrate species in a comparative and experimental manner (see review by Gaines et al. 2003). One recent study (Taylor and Knight 2003a) evaluated bison (*Bison bison*), pronghorn (*Antilocapra americana*), and mule deer (*Odocoileus hemionus*) responses to mountain biking and hiking. This study, however, did not include ATV or horseback riding, nor did it include experimental controls needed to assess cause-effect relations.

To address these knowledge gaps, we initiated a manipulative, landscape experiment in 2002 to measure effects of off-road recreation on mule deer and elk (*Cervus elaphus*), two charismatic species of keen recreational, social and economic interest across western North America. Our objectives were to (1) document cause-effect relations of ATV, horseback, mountain bike and hiking activities on deer and elk, using these off-road activities as experimental treatments and periods of no human activity as experimental controls; (2) measure effects with response variables that index changes in animal or population performance, such as movement rates, flight responses, resource selection, spatial distributions and use of foraging versus security areas; (3) use these response variables to estimate the energetic and nutritional costs associated with each activity and the resultant effects on deer and elk survival; and (4) interpret results for recreation management.

Our research began in 2002 and ended in 2004. In this paper, we present findings from 2002 to address parts of objectives 1, 2 and 4. We specifically focus on changes in movement rates and flight responses of mule deer and elk in relation to the off-road activities, compared to periods of no human activity. We then describe potential uses of the results for recreation management.

We present findings from our first year of study because of the urgent need for timely management information to address the rapid growth in off-road recreation (U. S. Department of Agriculture, Forest Service 2004). For example, ATV use on public land has increased seven-fold during the past 20 years, and many conservation groups are calling for widespread restrictions on ATV travel (U. S. Department of Agriculture, Forest Service 2004). Yet, no studies have evaluated the role of ATVs compared to other off-road activities, such as mountain biking and horseback riding, which also are increasing rapidly. Without comprehensive studies of ATV effects in relation to other recreation, the debate over ATV use is likely to intensify. Our study was designed to measure a variety of ungulate responses to address this debate, so results can be used to identify compatible mixes of different off-road recreational opportunities in relation to deer and elk management.

Throughout our paper, we refer to off-road recreation, both motorized and nonmotorized, that occurs on trails, primitive (unpaved) roads, or areas without trails or roads. This definition complements the phrase off-highway vehicle (OHV) use, which refers to motorized vehicle use on any surface beyond highways (U. S. Department of Agriculture, Forest Service 2004), but which does not include other forms of nonwinter recreation that typically occur on primitive roads and trails, such as hiking, horseback riding, and mountain biking.

Study Area and Technologies

We conducted our research in northeastern Oregon at the Starkey Experimental Forest and Range (Starkey, Figure 1), a facility equipped to evaluate real-time and landscape-level responses of deer and elk to human activities under controlled experimentation (Rowland et al. 1997, Wisdom et al. 2004a). The facility encompasses spring, summer and fall ranges typical of those used by mule deer and elk in the western United States. Timber harvest, livestock grazing, motorized traffic, hunting, camping and other public uses of Starkey also are managed like those on national forests in the western United States, providing a large inference space for research findings (Rowland et al. 1997, Wisdom et al. 2004a).

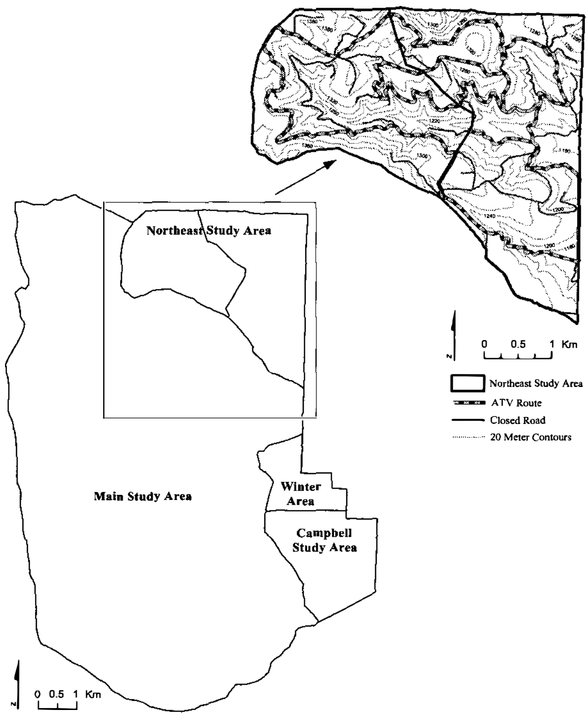


Figure 1. Boundaries of ungulate-proof enclosures at the Starkey Experimental Forest and Range in northeastern Oregon (bottom left) and location of transects used for ATV activities in the 3,590-acre (1,453-ha) Northeast Study Area (upper right), the site of the off-road recreation study. Transects were similar in length and location for mountain biking, hiking and horseback riding as those shown here for ATV activities.

An essential research component at Starkey is the ungulate-proof enclosure, one of the largest in the world, which allows scientists to evaluate ungulate responses to human activities over large areas and under controlled conditions (Bryant et al. 1993, Rowland et al. 1997). Another key technology is the automated tracking system (ATS), which can generate up to one animal location every 20 seconds, 24 hours a day, from April through December each year (Rowland et al. 1997, Kie et al. 2004). Additional technologies include maps and databases of more than 100 environmental variables to relate animal movements to the landscape experiments, as well as supporting methods and software to analyze these data (Rowland et al. 1997, 1998).

Implementing the Recreation Treatments

To meet our objectives, a network of off-road transects was established and run in 2002, using ATV, horseback, mountain bike and hiking activities as experimental treatments in the 3,590-acre (1,453-ha) Northeast Study Area (Figure 1). Approximately 20 miles (32 km) of transects were established (Figure 1), over which ATV, horseback, mountain bike and foot traffic was experimentally applied from mid-April through October. Locations of each transect were established with global positioning system (GPS) units (Figure 1). Transects were located on flat or moderate terrain typically used by off-road activities. Primitive roadbeds, like those often established by off-road vehicles (U. S. Department of Agriculture, Forest Service 2004), were included in the transects. Use of roadbeds and trails to implement human activities is referred to as a tangential experimental approach because animals are not targeted directly by the activities (Taylor and Knight 2003b). This is in contrast to a direct experimental approach, such as testing the reaction of nesting birds to designed encounters with humans at nest sites.

A sufficient number and length of transects were established to encompass all portions of the Northeast Study Area (Figure 1). Each off-road activity was run on a given transect twice daily, once in the morning and once in the afternoon, during a 5-day period; this daily frequency of activity corresponds to traffic frequency on Starkey roads that produced an avoidance response by elk in earlier research (Wisdom 1998, Wisdom et al. 2004b).

A particular activity for a given morning or afternoon was completed by one to three people who rode ATVs (four-wheelers or quads), mountain bikes, or horses, or who hiked as a group. On most days, group size consisted of two people moving as a pair; that is, by two people hiking or each riding ATVs, mountain bikes or horses. A group size of two, with a range of one to three people, often is typical for these recreation activities in nonwilderness portions of national forests (D. Barrett, personal communication 2002). Group size can vary substantially, however, with larger groups of 5 to 10 ATV riders or horseback riders, for instance. We had neither the resources nor the experimental options to include these larger groups as treatments in our study. Moreover, group size of mountain bikers and hikers often does not approach 5 to 10 people, and we wanted to maintain approximately the same group size across all four activities. A group size of two people, with a range of one to three people, provided this consistency.

For ATV travel, a pair of riders could easily cover the 20 miles (32 km) of transects during a given morning or afternoon. A pair of mountain bike riders, however, could cover about 50 percent of the 20 miles (32 km) in a morning or afternoon. Horseback riders and hikers could cover about 30 percent. Because we wanted to standardize the experiment by the same number of transect runs or passes (twice daily) among all four off-road activities, two different groups of mountain bikers and three groups of horseback riders or hikers were used to obtain complete coverage of transects for a given morning or afternoon. For mountain biking, the transects were divided in half, with each of the two groups assigned to ride a different half of the 20 miles (32 km) in a morning or afternoon. Similarly, three groups of horseback riders or hikers, each assigned to travel a different third of the transect length, were used for each morning and afternoon to obtain complete coverage of transects.

Each of the four off-road activities was implemented under an interrupted movement design, where humans were allowed to momentarily stop to view animals for less than 1 minute when animals were observed. This is in contrast to a continuous movement design, where human activities are not delayed or stopped when animals are observed (Taylor and Knight 2003b).

Each 5-day period of off-road activity was followed by a 9-day control period, during which no human activities occurred in the study area. This pattern was followed from mid-April through October, resulting in three replicates of each of the four off-road activities. Each 5-day replicate of an off-road activity thus was paired with a 9-day control period that immediately followed the replicate. Only one type of off-road activity (ATV, horseback, mountain bike or hiking) occurred on transects during a given 5-day replicate. The chronological order of each off-road activity, in terms of which activity occurred during the first 5-day replicate in late April, versus the next 5-day replicate in early May, and so on, was randomly chosen.

Throughout the experiment, all human entry beyond the four off-road activities, including administrative use of roads, was prohibited to eliminate the confounding effects of other human activities with animal response to the off-road activities. Consequently, human activities such as timber harvest, road traffic, camping and hunting did not occur during the study because of their confounding effects.

Measuring Animal Responses

To monitor animal responses, 12 female mule deer and 12 female elk were radio-collared among a larger population of approximately 25 female deer and 100 female elk present in the Northeast Study Area in early April. Movements of these radio-collared animals were monitored with the ATS (Rowland et al. 1997). During periods of off-road activity, locations of each radio-collard deer or elk were generated at approximately 10-minute intervals. Locations of humans engaged in each off-road activity were generated at approximately 1-minute intervals, using GPS units carried by one of the persons in each group of hikers or riders of ATVs, horses or mountain bikes. Use of the automated telemetry system to track animal movements, combined with the use of GPS units to track human movements, provided real-time, unbiased estimates of the distances between each ungulate and group of humans.

Our method of estimating distances between ungulates and humans contrasts strongly with the use of direct observation, using rangefinders or other devices, to measure distances. Direct observation as a means of estimating distances between ungulates and humans is likely to be biased by the proportion of deer or elk whose reactions to human activities cannot be observed because such reactions are different than those of animals that can be observed. For example, some animals may run from human activity at distances beyond the view of observers, while other animals may react at close distances to, and in view of, observers. This bias in observed distances would result in underestimation of the true distance at which animals react to the human activity. In other cases, animals may flee from humans at close distances but not be viewed because such animals seek dense cover during flight; this bias would result in overestimation of distances. We avoided such biases with the use of our automated telemetry system and GPS units to continuously monitor the movements of ungulates and humans throughout our study.

We also located radio-collared animals during the 9-day periods of no human activity, or control period. Approximately two locations of each radio-collared animal were obtained every hour during control periods, to establish baseline information about areas of deer and elk use, habitat selection, movement rates, and flight responses in the absence of human activities. For this paper, we analyzed two types of animal reactions: (1) movement rate and (2) probability of flight response. We evaluated movement rate and probability of flight response because both can ultimately be used to estimate the energetic costs of animal reactions to off-road activities (see Conclusions and Interpretations).

Estimating Movement Rates

We defined movement rate as the speed of animal movement (yards moved per minute), estimated hourly, 24 hours per day, for a given species, treatment and control period. We calculated the speed of animal movement for each radio-collared deer or elk for each pair of successive locations; that is, the horizontal distance between two successive locations divided by the elapsed time between locations (Ager et al. 2003). Each measurement of animal speed for a given radio-collared animal was assigned to the time recorded for the first location of each pair of animal locations used in the calculation.

Only successive locations with consistent elapsed times were included in the calculation of movement rates to eliminate the bias of excessively short and long elapsed times. Short elapsed times (e. g., fewer than 5 minutes) between locations falsely inflate the movement rate because of random location errors in the ATS over such short time periods (Findholt et al. 1996, 2002). Long elapsed times (e .g., more than 35 minutes) between locations allow animals to move back and forth between the documented locations, thus biasing the estimate of movement rate downward (Ager et al. 2003).

To estimate overall patterns of movement rates for each species, rates calculated for each individual radio-collared animal were averaged among all animals, for mule deer and for elk, by hourly interval, for each off-road treatment and the paired control period that immediately followed that treatment. For this analysis, we minimized random variation by summarizing results across each 5-

day treatment and across each subsequent 9-day control. We did this after exploratory plots of data provided no evidence of change in movement rates of animals from day 1 through day 5 of each treatment period, or for day one through nine of each control period, as examined on an hourly basis. We then pooled hourly results for each species across the three replicates of each off-road activity, and across control periods, after finding no evidence of differences in like replicates across time, or in control periods across time.

Estimating Probabilities of a Flight Response

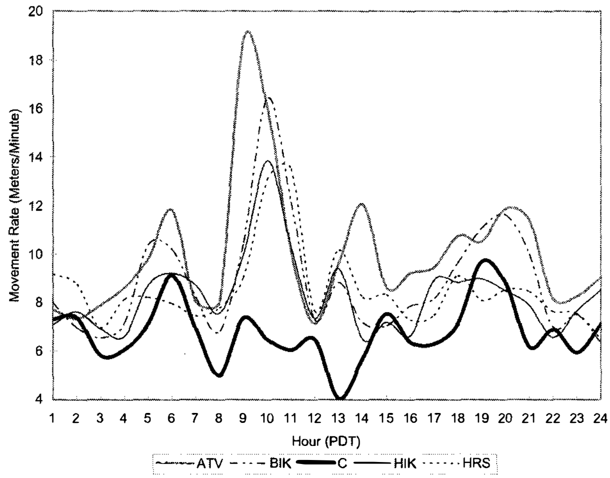
We used a stimulus response model to estimate the probability of a flight response by a deer or elk with changing distance between each animal and off-road activity. We defined a flight response as the speed of animal movement, or movement rate, that exceeded the 95th percentile of all deer or elk speeds calculated for each hour from data collected during the control periods. Specifically, a flight response was any animal movement for a given hour of day that exceeded the 95th percentile of all deer or elk speeds calculated for that same hour of day during the paired 9-day control period that immediately followed a given 5-day period of off-road activity. Thus by definition, when no stimulus was present (no human activity), a deer or elk would register a response (i. e., travel at speeds greater than the 95th percentile of all deer or elk speeds for that hour during the control period) 5 percent of the time. Probabilities of response were estimated using logistic regression within the generalized additive model framework (Hastie and Tibshirani 1990).

Each estimated probability of a flight response for a given radio-collared animal was linked to the estimated distance between that animal and each group of humans conducting an off-road activity, allowing an examination of how probabilities changed with distance between animals and humans. As with our analyses of movement rates, we pooled the probability data for each species across the three replicates of each off-road activity and across control periods. We pooled data after initial analyses showed that results for deer and elk were similar across the three replicates of each off-road activity and across all control periods.

Movement Rates of Elk

Movement rates of elk were substantially higher during periods of all four off-road activities, compared to periods of no human activity (Figure 2). Responses of elk to the morning and afternoon runs were clearly evident, with

Figure 2. Mean movement rate (speed, meters per minute) of elk, estimated hourly on a 24-hour basis, Pacific Daylight Time (PDT), during periods of no human activity (C) versus periods of ATV activity (ATV), hiking (HIK), mountain bike riding (BIK) and horseback riding (HRS), from April through October, 2002, in Northeast Study Area of Starkey.



the most pronounced increase in movement rates observed during the hours when each off-road activity occurred (Figure 2). For example, our morning pass on transects began between 0830 and 0930 Pacific Daylight Time (PDT), and highest movement rates for elk occurred in the hours immediately after, from 0900 to 1100, during all four activities (Figure 2). Moreover, lunch break for participants in the experiment occurred at or near noon, and movement rates for elk dipped to their lowest level at noon during all activities. Finally, we resumed each activity at 1230 to 1300 PDT, and movement rates for elk substantially increased immediately after (Figure 2).

Movement rates were substantially higher for elk during the morning pass, compared to the afternoon pass, for all four activities (Figure 2). Movement rates of elk during the afternoon pass, however, stayed well above the rates observed during the periods of no human activity (control period, Figure 2). Movement rates during the afternoon pass declined after 1500 PDT, when afternoon activities ended.

For the morning pass, movement rates of elk were highest during ATV riding, second-highest during mountain-bike riding and lowest during hiking and horseback riding (Figure 2). Movement rates of elk also stayed higher, over a longer period, during the afternoon ATV run, compared to rates during afternoon horseback riding, mountain-bike riding and hiking. Peak movement rates of elk during the morning pass were highest for ATV riding (21 yards per minute [19 m/min]), followed by mountain bike riding (17 yards per minute [16 m/min]) and horseback riding and hiking (both about 15 yards per minute [14 m/min]). For the afternoon run, movement rates of elk again were highest during ATV riding (13 yards per minute [12 m/min]), followed by horseback riding (about 11 yards per minute [10 m/min]) and hiking and mountain bike riding (about 10 yards per minute [9 m/min]).

By contrast, peak movement rates of elk during the control periods did not exceed 9 yards per minute (8 m/min). Moreover, peak movement rates during the control periods stayed below 8 yards per minute (7 m/min) during daylight hours of 0800 to 1500, the comparable period of each day when off-road treatments were implemented.

Interestingly, movement rates of elk also were higher than control periods at times encompassing sunrise and sunset for the days in which an off-road activity occurred, even though humans were not present at these times of day (Figure 2). These higher movement rates near sunrise and sunset suggest that elk were displaced from preferred security and foraging areas as a result of flight behavior during the daytime off-road activities. In particular, movement rates of elk at or near sunrise and sunset were higher during the 5-day treatments of mountain bike and ATV activity (Figure 2). This finding will be studied in detail in future analyses.

Flight Responses of Elk

The estimated probability of elk flight from a human disturbance was highly dependent on distance. When elk and humans were close to one another, the maximum probability of a flight response was approximately 0.65 during ATV, mountain bike and hiking activity, and 0.55 during horseback riding (Figure 3). Higher probabilities of flight response occurred during ATV and mountain bike activity, in contrast to lower probabilities observed during hiking and horseback riding (Table 1). Probability of a flight response declined most rapidly during hiking, with little effect when hikers were beyond 550 yards (500 m) from an elk. By contrast, higher probabilities of elk flight continued beyond 820 yards (750 m) from horseback riders and 1,640 yards (1,500 m) from mountain bike and ATV riders (Figure 3).

Movement Rates of Deer

In contrast to elk, mule deer showed less change in movement rates during the four off-road activities compared to the control periods (Figure 4).

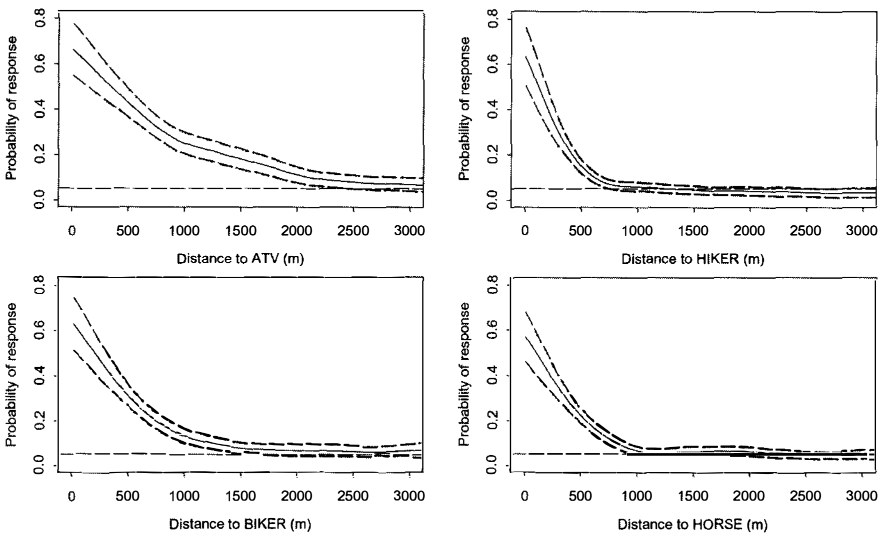


Figure 3. Estimated probability (solid line encompassed by dashed lines of the approximate 95 percent pointwise confidence interval) of a flight response by elk during 2002 in relation to distance (meters) from humans riding ATVs, mountain bikes, horses or hiking. A flight response is defined as an animal movement with a speed exceeding the 95th percentile of speeds observed during periods of no human activity (control period). The horizontal dashed line at the bottom of each graph is the probability of a flight response by elk during periods of no human activity, and this line represents the background, or the null condition, above which significant elk response to the off-road activities exists.

During the period of day from 0800 to1500 when off-road activities occurred, movement rates of deer during ATV riding were similar to rates during control periods. By contrast, daytime movement rates of deer were higher, compared to control periods, during mountain bike riding, horseback riding and hiking, especially in the morning (Figure 4).

Interestingly, the increased movement rates observed for elk near sunrise and sunset also were evident for mule deer. Movement rates at these times were particularly high during all four activities as well as during the control periods, suggesting that these times were peak foraging periods (Ager et al. 2003).

Flight Responses of Deer

Estimated probabilities of flight response for mule deer were similar among all four activities versus control periods (Table 1, Figure 5). These

Table 1. Estimated probabilities (and approximate 95 percent confidence limits) of a flight response by elk and mule deer as a function of distance between animals and humans riding all-terrain vehicles (ATV), mountain bikes (BIKE), horses (HORSE) or hiking (HIKE). On average there were 128 deer or elk locations obtained during a given day of each off-road activity (treatment periods). During periods of no human activity (control periods), the null probability of a flight response is 0.05. Thus, any values greater than 0.05 reflect an increased probability of a flight response in relation an off-road activity.

Distance[1]	ATV	Bike	Horse	Hike
109 yards (100 m)	0.62	0.58	0.50	0.52
from elk	(0.52–0.73)	(0.46–0.68)	(0.40–0.59)	(0.42–0.64)
545 yards (500 m)	0.43	0.31	0.22	0.15
from elk	(0.36–0.49)	(0.26–0.35)	(0.19–0.26)	(0.12–0.18)
1,090 yards (1,000 m)	0.25	0.13	0.07	0.06
from elk	(0.20–0.30)	(0.10–0.16)	(0.05–0.08)	(0.04–0.08)
All distances	0.19	0.14	0.11	0.08
from elk	(0.17–0.21)	(0.12–0.16)	(0.09–0.12)	(0.07–0.10)
109 yards (100 m)	0.06	0.08	0.11	0.10
from deer	(0.01–0.11)	(0.02–0.14)	(0.03–0.19)	(0.04-0.17)
545 yards (500 m)	0.05	0.07	0.05	0.04
from deer	(0.02–0.07)	(0.04–0.10)	(0.03–0.07)	(0.02–0.05)
1,090 yards (1,000 m)	0.03	0.06	0.04	0.04
from deer	(0.01–0.06)	(0.03–0.08)	(0.02–0.06)	(0.02–0.06)
All distances	0.03	0.05	0.04	0.04
from deer	(0.02–0.05)	(0.04–0.07)	(0.03–0.05)	(0.03–0.06)

[1] Distance between an animal and human during each off-road activity.

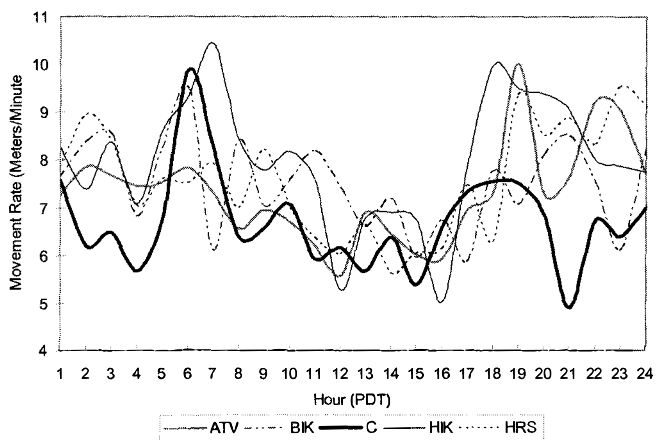


Figure 4. Mean movement rate (speed, meters/minute) of mule deer, estimated hourly on a 24-hour basis, Pacific Daylight Time (PDT), during periods of no human activity (C) versus periods of ATV activity (ATV), hiking (HIK), mountain bike riding (BIK) and horseback riding (HRS) during 2002 in the Northeast Study Area of Starkey.

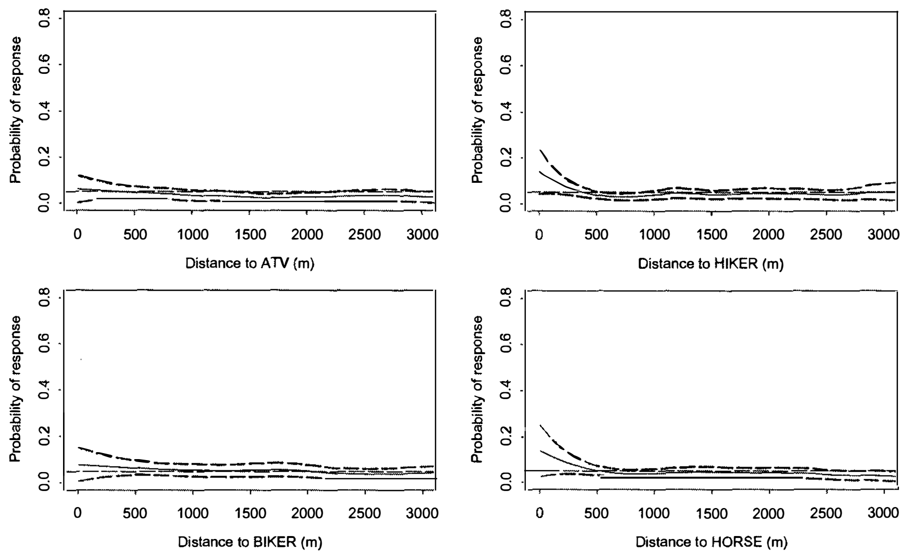


Figure 5. Estimated probability (solid line encompassed by dashed lines of the approximate 95 percent pointwise confidence interval) of a flight response by mule deer during 2002 in relation to distance (meters) from humans riding ATVs, mountain bikes, horses or hiking. A flight response is defined as an animal movement with a speed exceeding the 95th percentile of speeds observed during periods of no human activity (control period). The horizontal dashed line at the bottom of each graph is the probability of a flight response by deer during periods of no human activity, and this line represents the background, or null, condition, above which significant deer response to the off-road activities exists.

probabilities were nearly identical among all four activities and not significantly different than the null probability of 0.05 set for control periods, suggesting that deer were not exhibiting the same tendency for flight as shown by elk in relation to off-road activities (Table 1).

Conclusions and Interpretations

Elk

Movement rates and probabilities of flight response for elk were substantially higher during all four off-road activities, compared to control periods of no human activity. Consequently, off-road recreational activities like those evaluated in our study appear to have a substantial effect on elk behavior. The energetic costs associated with these treatments deserve further analysis to assess potential effects on elk survival. For example, if the additional energy required to flee from an off-road activity reduces the percent body fat of elk below 9 percent as animals enter the winter period, the probability of surviving the winter is reduced (Cook et al. 2004). Animal energy budgets also may be adversely affected by the loss of foraging opportunities while animals respond to off-road activities, both from increased movements and from displacement from foraging habitat. These potential effects will be evaluated as part of future analyses.

Our results from 2002 also show clear differences in elk responses to the four off-road activities. Elk reactions were more pronounced during ATV and mountain bike riding, and they were less so during horseback riding and hiking. Both movement rates and probabilities of flight responses were higher for ATV and mountain bike riding than for horseback riding and hiking.

Interestingly, the maximum probability of flight was approximately 0.65 for the treatments, meaning that, about 35 percent of the time, elk did not exhibit a flight response when close to an off-road activity. Most likely the response depends on local topography, cover and other factors that we have not yet analyzed as part of our flight response model. Future work will include terrain and vegetation measures as covariates in the probability models to examine whether these effects can be detected and quantified (see Taylor and Knight 2003b).

It is important to note that designing our study to maintain the same number of daily passes on transects among all four activities required the most effort for hiking and horseback riding, and the least effort for ATV riding. Specifically, to accomplish two runs per day required three groups of hikers or horseback riders (with each group hiking approximately 33 percent of transect length) but only one group of ATV riders. By contrast, accomplishing two runs per day required two groups of mountain bikers (with each group covering approximately 50 percent of transect length).

Our results for elk might have been different had we designed the study to test animal response to an equal number of groups, or equal density, of people engaged in the four off-road activities (i. e., the same number of groups of people engaged in each activity, regardless of the number of passes that could be accomplished), rather than testing for effects of equal saturation of the study area (i. e., two daily passes on transects for all four activities). In future analyses, we plan to explore the use of the amount of time spent by each off-road activity as a covariate and possibly weight the movement rates and probabilities of flight response by the inverse of time spent by each of the four off-road activities. This weighting would help account for differences in effort required among the four activities to achieve equal saturation of the study area.

Our results may also change if elk eventually become habituated to some or all of the off-road activities. We will evaluate this possibility in future analyses by formally testing for replicate and year effects under a random effects model, with repeated measures taken on radio-collared animals over time (Kirk 1982). Analyses to test for animal habituation to the off-road activities will be possible when all three years of data are collected.

Mule Deer

In contrast to elk, mule deer showed little measurable response to the off-road treatments. Movement rates increased slightly, however, during periods of all four-off road activities except ATV riding. Deer may well be responding to the treatments with fine-scale changes in habitat use, rather than substantial increases in movement rates and flight responses.

For example, it is possible that deer may respond to an off-road activity by seeking dense cover, rather than running from the activity. If mule deer are spending more time in dense cover, in reaction to any of the off-road activities, this could result in reduced foraging opportunities and a subsequent reduction in opportunities to put on fat reserves during summer that are needed for winter survival. Such potential responses will be evaluated as part of future analyses.

Utility of Response Variables

Taylor and Knight (2003b) defined a variety of terms for measuring animal responses to human activity. Neither movement rate nor probability of a flight response was defined, however, because these types of animal responses apparently have not been measured in past research. We measured these two responses to human activity because both variables can ultimately be used to estimate the energetic costs of animal reactions to human activities. For example, movement rate can be used as a background index of the rate of animal speed without human activities, versus periods of human activities, to estimate the additional energetic costs of increased movement, if any, in relation to human activities (Ager et al. 2003).

Similarly, the probability of a flight response indicates how likely an animal is to move at high speed in relation to its distance from a human. This probability indicates how likely an animal is to run from a human activity, and thereby disrupt the animal's activities related to energy acquisition (foraging) or energy conservation (resting). Any movement away from an area in relation to human activity has the potential to disrupt these foraging and resting patterns and, thereby, to cost energy (Johnson et al. 2004).

Future analyses will focus on the energetic costs, if any, to mule deer and elk from exposure to each off-road activity. Additional analyses also will include estimates of (1) the distance moved by an animal, given a flight response; (2) the time required for an animal that exhibits a flight response to return within a specified distance of the animal's location before the flight; (3) the change in space use by an animal, during or following periods of human activity, which may suggest or reflect an animal seeking greater refuge from the human activity, as compared to background, or null, use of space during periods of no human activity; and (4) the degree to which animals spend time in forage areas, gaining energy, versus time spent in nonforaging areas, during each off-road activity versus control periods.

Implications for Recreation Management

Laws and policies of public land management emphasize multiple resource uses. Management of timber, grazing, roads, minerals, and wilderness are examples of traditional uses on lands administered by the U. S. Department of Agriculture, Forest Service (Forest Service) and U. S. Department of Interior, Bureau of Land Management (BLM), the two largest federal landowners in the United States. Public land managers now face the additional challenge of serving a variety of off-road recreational uses that are increasing rapidly, and that can be difficult to accommodate on the same land area at the same time (Taylor and Knight 2003a).

New planning approaches are underway in the Forest Service to accommodate increasing off-road recreational demands while mitigating the negative effects on species like elk (U.S. Department of Agriculture Forest Service 2004). These approaches could consider two related concepts: (1) off-road use rates and (2) off-road recreational equivalents. We define off-road use rates as the number of passes per unit of time on a given linear route (primitive road or trail that we referred to as transects) traveled by an off-road activity. Our results show that one pass per day by any of the four off-road activities causes increased movement rates and flight responses by elk.

We define off-road recreational equivalents as the ratio of ATV riders, mountain bikers, horseback riders and hikers that results in approximately the same effect on a given resource, given the same off-road use rate. In the case of elk, movement rates and probabilities of flight were highest during ATV riding and lowest during horseback riding and hiking. These effects were a result of one group of ATV riders, two groups of mountain bikers and three groups of horseback riders or hikers required to complete one pass on the transects each morning or afternoon. Consequently, the stronger effects posed by ATV riding, combined with differences in the number of groups required of each activity to achieve one pass on the transects, suggest that recreational equivalents would exceed three groups of horseback riders or hikers to every one group of ATV riders, and exceed two groups of mountain bike riders to every group of ATV riders.

Although the formal methods of calculating the specific recreational equivalents could be a subject of lengthy debate, the idea that different levels of each off-road activity are required to approximate the same effect on a given resource is logical and defensible. Accordingly, off-road use rates and recreational equivalents could be tested as potential concepts in helping allocate recreational activities within and across watersheds on a given national forest or BLM field office. These concepts may be particularly relevant when derived from a combination of response variables or resource uses. For example, effects of each off-road activity on water quality, soil productivity, invasion of exotic plants and species sensitive to human activities could be considered in deriving use rates and recreational equivalents.

Such an approach would demand a substantial increase in research on effects of off-road activities. For management of elk, results from our study will be most useful when estimates of the energetic costs, if any, are derived for each of the four off-road activities in terms of use rates and recreational equivalents. Energetic costs to elk from one pass per day on a given linear route traveled by a given off-road activity could be estimated, and the equivalent energetic costs, given the same use rates, could be estimated among all off-road activities.

Although these details are not yet available, managers could begin to consider holistic management strategies for all off-road activities based on our current findings. Some watersheds might feature opportunities for ATV or mountain bike riding, for example, while other watersheds might focus on opportunities for horseback riding or hiking. Importantly, the watersheds identified for horseback riding or hiking could accommodate a substantially higher number of groups engaged in these off-road activities before realizing the same effects on elk as would be expected in watersheds where ATV or mountain bike riding are featured. This type of holistic management of different mixes of all off-road activities contrasts with management approaches that focus on a single off-road activity, without consideration of all off-road uses and their cumulative effects.

Other strategies for watershed planning might simply focus on restricting each recreational activity to specified trails or roads. In this case, our results suggest that the effectiveness of such a strategy would depend on how much area is affected by the network of trails or roads allowed for use. If the linear distance of trails or roads open to recreation is small, relative to the total area of the watershed, the effect on elk is likely to be minor or negligible. If the linear distance is large, relative to the size of the watershed, the negative effect on elk could increase substantially. The specific effects could be analyzed in the same manner as outlined for estimating effects of motorized road traffic on elk, as done with distance band models (Rowland et al. 2004).

Effective and defensible strategies to meet off-road recreation demands, while also mitigating negative resource effects, are likely to require a substantial increase in budgets of public land agencies for research, management and monitoring of these activities. Managers currently have little knowledge with which to develop effective strategies in partnership with the many public recreation users. Without such knowledge, the debate about off-road recreation is likely to intensify, with few scientifically based options for resolution in relation to mitigating potential negative effects on species like elk that are sensitive to human activities.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy.* 84:1076–88.

Bryant, L. D., J. W. Thomas, and M. M. Rowland. 1993. *Techniques to construct New Zealand elk-proof fence, general technical report PNW-GTR-313.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs* 155:1–61.

Findholt, S. L., B. K. Johnson, L. D. Bryant, and J. W. Thomas. 1996. Corrections for position bias of a LORAN-C radio-telemetry system using DGPS. *Northwest Science.* 70:273–80.

Findholt, S. L., B. K. Johnson, L. L. McDonald, J. W. Kern, A. A. Ager, R. J. Stussy, and L. D. Bryant. 2002. *Estimating habitat use from error associated with radiotelemetry positions, forest service, research paper PNW-RP-555.* Portland, Oregon: U. S. Department of Agriculture.

Hastie, T., and R. Tibshirani. 1990. *Generalized additive models.* New York New York: Chapman and Hall.

Havlick, D. G. 2002. *No place distant: Roads and motorized recreation on America's public lands.* Washington, DC: Island Press.

Gaines, W. L., P. H. Singleton, and R. C. Ross. 2003. *Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee national forests, general technical report PNW-GTR-586.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Johnson, B. K., A. A. Ager, J. H. Noyes, and N. Cimon. 2004. Elk and mule deer responses to variation in hunting pressure. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:625–40.

Kie, J. G., A. A. Ager, N. J. Cimon, M. J. Wisdom, M. M. Rowland, P. K. Coe, S. L. Findholt, B. K. Johnson, and M. Vavra. 2004. The Starkey database: Spatial-environmental relations of North American elk, mule deer and cattle at the Starkey Experimental Forest and Range in northeastern Oregon. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:475–90.

Kirk, R. E. 1982. *Experimental design, second edition.* Pacific Grove, California: Brooks/Cole Publishing.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collections methods for ungulate research, general technical report PNW-GTR-396.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: Implications for management in forested ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:491–508.

Rowland, M. M., P. C. Coe, R. J. Stussy, A. A. Ager, N. J. Cimon, B. K. Johnson, and M. J. Wisdom. 1998. *The Starkey habitat data base for ungulate research: Construction, documentation, and use, general technical report PNW-GTR-430.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Taylor, A.R., and R. L. Knight. 2003b. Behavioral responses of wildlife to human activity: Terminology and methods. *Wildlife Society Bulletin.* 31:1,263–71.

Taylor, A. R., and R. L. Knight. 2003a. Wildlife responses to recreation and associated visitor perceptions. *Ecological Applications.* 13:951–63.

U. S. Department of Agriculture, Forest Service. 2004. *Managing the National Forest System: Great issues and great divisions.* La Grande, Oregon: U. S. Department of Agriculture, Forest Service, Northwest Research Station.

Wisdom, M. J. 1998. *Assessing life-stage importance and resource selection for conservation of selected vertebrates.* Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004a. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:455–74.

Wisdom, M. J., N. J. Cimon, B. K. Johnson, E. O. Garton, and J. W. Thomas. 2004b. Spatial partitioning by mule deer and elk in relation to traffic. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:509–30.

Issues of Elk Productivity for Research and Management

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Introduction

Elk (*Cervus elaphus*) populations in western North America have been intensively managed for the past century. The species' popular appeal as an animal for hunting and viewing, and its potential to damage agricultural crops and to compete with livestock make it a species that is closely scrutinized by managers and many public interest groups. Today, elk continue to have significant ecological, cultural and economic values. For example, the species provides substantial revenue for rural communities from hunting and viewing. Since the 1960s, elk in some areas of the western United States cause economic damage to farming and ranching operations. Elk managers spent considerable effort to maintain populations at levels compatible with these private land uses, while also striving to provide sufficient hunter opportunities on public land to meet recreational demands.

More recently, concern about elk has changed from how to manage increasing populations to how to maintain populations. Elk recruitment, defined here as number of calves per 100 females that survive to one year of age, has declined in many areas of the western United States. Elk recruitment gradually declined since the 1960s in areas of northeastern Oregon (Oregon Department of Fish and Wildlife 2003) from more than 50 calves per 100 females to fewer than 20 calves per 100 females in some areas during the 1990s, resulting in lower elk populations and fewer hunting tags issued. In Wallowa County of northeastern Oregon, the number of antlered elk tags was reduced from 7,030 in 1995 to 4,620 in 2000 and from 4,140 to 350 for hunting antlerless elk (Oregon Department of Fish and Wildlife 2001). Elk populations declined 25 percent in the Clearwater Basin in Idaho, 50 to 70 percent in the North Rainier, 30 to 50 percent in the South Rainier and 30 percent in the Blue Mountain elk herds in Washington. Most of these declines were concurrent with low recruitment (Cook et al. 1999).

The focus of our paper is to describe biological factors that affect elk population dynamics and focus on some potential causes of recent declines in elk recruitment. We identify interacting factors that may explain recent declines in elk recruitment, and we describe the most likely hypothesis as a combination of several factors. We provide this discussion as context for the many subsequent papers that focus on elk productivity or related issues as part of research conducted by the Starkey Project and its partners.

Factors Affecting Elk Recruitment and Populations

The question of what regulates elk productivity is of practical interest because effective management depends on understanding how the multitude of factors may limit or interact to affect populations. Questions have centered on whether factors that most affect ungulate productivity are density dependent or density independent and if predation can limit elk populations. Following Skogland (1991), we define regulation as any positive, density-dependent (effects increase as density increases) process that tends to stabilize population numbers over time. Any process that changes population size is phrased limitation process. Limitation processes normally operate independently of density and, thus, do not stabilize populations. Both regulatory and limitation factors have been proposed to explain the long-term decline in elk recruitment. Factors that are usually listed as regulatory include food, disease and other key habitat resources that may be in short supply as populations increase. Changes in elk population characteristics, such as reproduction and survival, have been correlated with changes in population size or density (Houston 1982). Factors that often limit ungulate populations include human harvest, stochastic events (usually weather related) and predation. Predation effects can be density dependent, density independent or inversely density dependent (Messier and Crête 1985, Skogland 1991, Messier 1994). Because territoriality of predators will restrict their density at some point, there should be a point at which predation will no longer be positively density dependent as prey populations increase (Skogland 1991). Messier (1994) identified three scenarios where predation can have varying effects on ungulate populations. Predation can be (1) density dependent at low prey densities when predation rates increase as prey density increases, (2) density independent when predation rates remain constant and (3) inversely density dependent when predation rates decrease as populations increase often at higher prey densities (Messier and Crête 1985).

Density independent and density dependent factors affect population parameters (e. g., rates of birth, growth, fertility, mortality), and vital rates quantify these parameters (Caswell 2001). Temporal variation in vital rates has a major effect on population growth of elk. Eberhardt et al. (1996) estimated a maximum rate of increase (ë) of 1.33, or about 27 percent per year. Survival of adult female elk is the most important vital rate for long-term stability of populations (Wisdom and Cook 2000) followed by fecundity of prime-aged adults, fecundity of primaparous females and, finally, juvenile survival (Gaillard et al. 2000). In practice, however, survival of juveniles is an important component of ungulate population dynamics even though this vital rate has disproportionately less effect on population growth than other vital rates (Gaillard et al. 1998, 2000; Eberhardt 2002).

How might density independent and density dependent factors act separately or interactively to control vital rates and growth of ungulate populations? The literature contains theoretical and empirical works that address ungulate productivity. They are often contradictory when read in the absence of a broad examination of the issue. Moreover, the literature states that density dependent and density independent factors may act simultaneously, such that effects of density independent factors (drought or severe winter) increase with higher population density (Gaillard et al. 2000).

A related concept to density dependence is that of compensatory versus additive sources of mortality. Compensatory sources of mortality are those that have little influence on population growth because other factors interact or negate their effect (e. g., Bartmann et al. 1992). By contrast, additive sources of mortality are direct, minimally influenced by other factors, cause a predictable increase in mortality and reduce population growth. Predation can be an additive source of mortality when density dependent effects are minimal, but it can be a compensatory source at high population densities if food is limiting (Messier

1994). Hunting also can have an additive effect in heavily exploited populations (Dusek et al. 1992).

Density Dependent Effects

Effects of density dependence on population growth are well-documented in theoretical and experimental work (Fowler 1981, 1987). Density dependence in ungulates has been most evident in isolated populations, often on islands and in predator-free or nearly predator-free environments (Grubb 1974, McCullough 1979, Coulson et al. 1997, Gaillard et al. 2000, but see Houston 1982, Lubow et al. 2002, Taper and Gogan 2002 for exceptions with elk). Density dependent regulation evidently was caused by intraspecific competition for food that decreased reproductive performance, increased juvenile mortality rates and, in some cases, increased adult mortality rates. Density dependent changes in vital rates were most obvious in ungulates when populations are close to carrying capacity (Fowler 1981). Eberhardt (2002) summarized how density dependent regulation of ungulates manifests itself via four sequential steps as populations approach carrying capacity: (1) juvenile mortality increases, (2) age at first reproduction increases, (3) reproductive rate of adult females decreases and (4) mortality of adults increases.

Nutrition fundamentally controls reproduction in ungulates, with effects that can carry over from one season or year to the next. Inadequate nutrition of cow elk in winter and spring reduced calf weight and survival at birth (Thorne et al. 1976). Reduced spring ambient temperature also reduced calf birth weight (Smith and Anderson 1996). Poor condition of moose (*Alces alces*) cows resulted in low birth mass of their calves (Keech et al. 2000). Nutrition in summer has marked effects on growth and development of deer fawns (Holter and Hays 1977, Verme and Ozoga 1980) and elk calves (Cook et al. 1996, 2004). Steinheim et al. (2002) found that lifetime reproductive performance of free-ranging domestic sheep (*Ovis aries*) was related to body mass at birth; ewes with an initial low body mass produced fewer offspring at first and at second parturition. Summer nutrition influenced black-tailed deer (*Odocoileus hemionus*) body fat and weight dynamics (Parker et al. 1999) and probability of survival in winter (Parker et al. 1999, Cook et al. 2004). This effect might be most pronounced in ecosystems occasionally or frequently experiencing relatively harsh winters.

Nutritional demands of lactation during late spring through early autumn can affect pregnancy rates in autumn because nutritional condition (e. g., body fat) of cow elk is the prime determinant of successful breeding (Trainer 1971, Kohlmann 1999, Cook et al 2001c, Cook et al. 2004). The energetic demands of lactation are high, and, if the digestible energy in summer forage is low or marginal relative to requirement, cow elk may not rebuild sufficient body reserves by autumn to enter estrous. However, Cook et al. (2004) found that if the cow loses its calf early after parturition, she can accumulate sufficient fat reserves to successfully breed in autumn even when forage quality in summer is markedly inadequate for lactating cows. Trainer (1971) suggested that at a kidney to fat ratio (KFI) less than 60 (9 percent body fat [Cook et al. 2001a]), probability of pregnancy was reduced. Cook et al. (2004) also found that female elk in poor condition would not conceive and suggested that 8 to 9 percent body fat was a threshold for cow elk in autumn, below which pregnancy rates decline.

It is not clear about the degree to which density-dependent nutritional effects in winter versus those in summer most influence populations. Density dependent mortality has been well-established in Yellowstone National Park as a function of elk density on winter range (Houston 1982, Taper and Gogan 2002). Lubow et al. (2002) also concluded that density dependence on winter range regulated the elk herd in Rocky Mountain National Park. Beuchner and Swanson (1955) suspected density dependent influences on primaparity of young cow elk in southeast Washington. Merrill and Boyce (1991) reported significant correlations between elk calf survival and abundance of green phytomass during summer in Yellowstone National Park. Crête and Huot (1993) reported marked effects of nutrition during summer on reproductive processes in caribou at relatively high herd density.

Whatever the case may be regarding seasonal influences, it follows that, as elk populations approach carrying capacity, recruitment should decrease because increased intraspecific competition for high quality forage lowers nutrition. Moreover, selective grazing at high herbivore density may effectively erode the nutritional value of plant communities available to herbivores, enhancing the food-based density effect over time (Irwin et al. 1994, Riggs et al. 2000). Forest succession patterns also may exacerbate density dependent effects despite little or no change in animal numbers because amount of forage may decline markedly from early-successional stages to advanced stages in forest ecosystems (Hett et al. 1978, Peek et al. 2001). Such influences might be pronounced where wildfire and logging have been reduced and where precipitation is adequate to support high rates of forest succession. These effects may require many years to develop and, thus, may be hard to document with typical short-term studies (but see Peek et al. 2002). Finally, nutrition may be inadequate at low animal densities because available forage, no matter how abundant, may not adequately satisfy requirements. This density independent aspect of nutritional influences is rarely recognized, but it may be important in some ecosystems (Cook et al. 2004).

Density Independent Effects

Climate is typically identified as having a density independent effect on ungulates (Gaillard et al. 2000) that can limit populations. Effects of climate, however, can be more severe when populations are close to carrying capacity (Garrott et al. 2003). Variation in climate-caused juvenile survival varied widely even though adult survival was high (Gaillard et al. 1998). Sæther (1997) argued that climate is the most important density independent variable in the absence of predation. In temperate and montane habitats, climate is characterized by distinct warm and cold seasons, with corresponding variation in forage quality and quantity. During late spring and early summer, forage typically is abundant and high in quality, but, during winter, quality is low and abundance may be low.

Among seasons, temperature and precipitation may be highly variable, resulting in summer drought or severe winter conditions. Summer drought can decrease forage quantity and quality (Vavra and Phillips 1980), thus reducing fat accumulations in yearling and lactating cow elk that can delay or cancel estrus (Cook et al. 2004). The result can be lower pregnancy rates of yearling cow elk (Bruce Johnson, unpublished data 2003) and of prime-aged elk, and it can be increased susceptibility to harsh winter conditions. Variation in winter severity can cause wide variation in survival of elk calves but have little effect on survival of prime-aged females (Garrott et al. 2003). Severe winter climate followed by cold, wet springs that delayed green up of vegetation lowered survival of neonate caribou (*Rangifer tarandus*) (Adams et al. 1995) and caused higher predation on neonates in semidomestic reindeer (Tveraa et al. 2003).

Diseases and parasites can infect elk (Thorne et al. 2002); however, most of the common livestock diseases do not affect elk recruitment. The one exception is brucellosis that can reduce pregnancy rates up to 12.5 percent annually (Thorne et al. 2002). Elk populations in Oregon have been monitored for brucellosis, and the disease has not been detected in any sample. In general, elk populations are little affected by parasites, and problems are often restricted to local situations (Thorne et al. 2002).

Predation

Predation can have highly variable effects on ungulate population growth, and, in many areas of western United States, predator populations are increasing (Mech et al. 2001, Keister and Van Dyke 2002). Predator to prey ratios are not sufficient to describe the effects of predation on ungulate populations because the functional response of predation may vary with prey density (Messier 1994). Manipulative experiments with ungulates and predators can be difficult to accomplish because of the long time periods required. Several manipulative or correlative studies, however, have been conducted. Sinclair et al. (2003) described a 40-year experiment in Africa where small- to medium-sized ungulate populations increased when predators were poached or poisoned in the Serengeti Park in Tanzania from 1980 to 1987. When predator populations were allowed to return to previous levels, the ungulate populations returned to their former levels. Jedrzejewski et al. (2002) found that wolf (*Canis lupis*) predation had an inverse density-dependent effect, in that predation limited red deer (*C. e. elaphus*) numbers but did not regulate the population; by eliminating a large portion of the juvenile age class, wolves dampened the rate of deer population growth. Jedrzejewski et al. (2002) also summarized the effects of wolf predation on red deer in temperate forests of Poland from 1850 to the present. Wolves were extirpated twice. During both periods of wolf extirpation, populations of red deer increased dramatically to the point where density dependent effects were obvious. When wolf populations were reestablished, deer populations decreased in proportion to the increase in wolf populations.

Hayes et al. (2003) manipulated wolf densities across 10 sites in Canada to evaluate the population responses of woodland caribou (*R. t. caribou*), moose and Dall sheep (*Ovis dalli*) and found that predation lowered caribou and moose recruitment and adult moose survival. Populations of Dall sheep, however, did not respond to changes in wolf density. Kunkel and Pletscher (1999) found that when wolves colonized a study area in northern Montana, survival of white-tailed deer (*Odocoileus virginianus*) and elk decreased and concluded that wolf predation was an additive mortality.

Predation does not necessarily affect sympatric prey species in the same manner (Hayes et al. 2003). Robinson et al. (2002) compared recruitment parameters and cause-specific mortality factors for sympatric white-tailed deer and mule deer (*Odocoileus hemionus*). Although there were no differences in fetuses per female or fawn survival between the species, survival of adult white-tailed deer was higher. Predation by cougar (*Puma concolor*) was the major source of adult mortality for each species, but predation rates on adult mule deer were two times higher than on white-tailed deer. As a result, the mule deer population was about 30 percent lower than that of white-tailed deer.

Further complications with interpreting effects of predation on ungulates involve sources of alternative prey and multiple predators on one prey species. Messier and Crête (1985) and Messier (1994) suggested that wolf predation of moose was less pronounced when multiple species of prey were present. If a second predator was present, however, effects of predation may increase. By contrast, Wakkinen and Johnson (2001) suggested that woodland caribou numbers were declining in the Selkirk ecosystem because of predation by cougar. They hypothesized that the dependable, alternate prey source provided by the expanding white-tailed deer population allowed the cougar population to remain high, which in turn resulted in increased predation on caribou. Kunkel and Pletscher (1999) found that the survival of moose increased when white-tailed deer and elk were present when cougars, grizzly bears (*Ursus arctos*), wolves and black bears (*Ursus americanus*) were sympatric with the three prey species. Finally the degree of seasonal migration may affect the extent to which predation acts on prey populations (Crête and Huot 1993).

The composite picture from these studies suggests that effects of predation can operate in multiple ways, depending on the suite of prey species present and the number and abundance of predator species. Prey population responses to predation can vary, and few, if any, conclusions can be drawn without site-specific information on the predator and prey species that occur. Skogland (1991) stressed that, although predators may limit prey populations, there is little evidence that predators regulate prey populations. Territoriality of predators may place upper limits on predator densities, potentially precluding predation as a regulating factor.

Hunting

Hunting can have an important influence on ungulate populations, diminishing the evidence of density dependence by reducing density below carrying capacity. For example, Swihart et al. (1998) evaluated the nutritional condition and pregnancy rates of white-tailed deer from areas with and without hunting. In areas without hunting, pregnancy rates, body size and body fat were lower than in areas where hunting controlled deer numbers. In addition, Swihart et al. (1998) found that pregnancy rates varied inversely with density among the five sites evaluated. Messier and Crête (1985) warned that ungulate populations appearing to be healthy (high recruitment) may change if human exploitation increases in the presence of predator populations. If predator to prey ratios change, predation can shift from inverse density dependence to density dependence, resulting in a further decrease in ungulate density and resulting in predation being sufficient to keep ungulate populations suppressed (Messier and Crête 1985). In recognition of such effects, Hayes et al. (2003) suggested hunter harvest of moose and caribou populations be set at low levels (2–5 percent) in areas where wolf and bear populations were high.

Pervasive Human Disturbances

Recreational activities on public land are increasing as human populations increase, and the activities may decrease animal fitness or expose animals to higher rates of mortality (Knight and Gutzwiller 1995). Since the 1950s, road construction on public land of the western United States has provided access to public land, resulting in increased use of areas that were previously undisturbed (Trombulak and Frissell 2000). Examples of increased recreational activities include mushroom and berry picking, firewood gathering, mountain biking, all terrain vehicle (ATV) use, cross-country skiing, backpacking, camping and snowmobiling. Elk have responded by moving away from roads open to the public (Rowland et al. 2000, 2004), especially roads with higher rates of traffic (Wisdom 1998). Elk have also moved away from off-road recreation activities, especially ATV and mountain bike riding (Wisdom et al. 2004). Conner et al. (2001), Vieira et al. (2003) and Wertz et al. (2004) found that displaced animals moved to areas where disturbance was minimal.

During hunting seasons, energetic consequences of the increased disturbance include increased energetic costs associated with movements (Johnson et al. 2004) and perhaps shifts to habitats where foraging conditions are diminished (J. G. Cook, unpublished data 2003). Disturbance during parturition and calf rearing resulted in higher calf mortality (Phillips and Alldredge 2000) or in decreased reproductive performance of mule deer in the following year (Yarmoloy et al. 1988). These added energy costs could lead to higher winter mortality rates as animals deplete stored fat reserves to avoid human activities.

Competing Hypotheses to Explain Declines in Elk Productivity

Several factors or hypotheses may explain long-term decline in elk recruitment across large areas of the northwestern United States. Each factor is plausible but not likely to operate independently of other factors. Exact mechanisms causing declines likely vary from location to location, depending on habitat, climate and predator populations. We hypothesize that recent declines in elk productivity are a result of a combination of factors that interact in both density dependent and independent ways. Our hypothesis is built on the following assumptions.

1. Intraspecific competition for forage—elk populations have gradually increased concomitant with reductions in forage biomass and quality over the last six decades in much of western United States. In turn, this may have increased intraspecific competition for nutritious forage, resulting in lower pregnancy rates and higher mortality rates of elk. Effective fire suppression on forestland over the last 60 years has had a substantially greater, negative effect on forage production than the counteracting, positive benefits of wildfire, timber harvest and insect-caused mortality of trees that has occurred in the last two decades. The total effect may be a slow erosion of carrying capacity on many elk ranges. For example, the elk populations increased dramatically in the 1980s and subsequently declined in the late 1990s following the volcanic irruption at Mount Saint Helens in Washington.
2. Decrease in highly palatable forage—elk may exert strong and highly selective grazing pressure on the forage species that are relatively palatable and available, especially if animals are displaced because of human activities. Coupled with increasing density, nutritional limitation occurs, reducing population parameters.
3. Increase in predator populations—cougar, black bear and wolf populations have increased substantially during the past four decades, resulting in lower recruitment of elk.
4. Summer drought and winter severity—abiotic factors of summer drought and winter severity may not operate in the traditional, density independent manner. Instead, these factors may increase the vulnerability of elk to predation or may increase winter mortality due to nutritional constraints.
5. Legal and illegal hunting—in situations where recruitment is lowered due to predation, cow elk hunting has an additive and substantial effect on growth rate of elk populations.

6. Human activities—human disturbance activities have increased across all elk ranges and may be exerting a negative effect on elk populations not observed in the past. The energetic costs of elk avoiding pervasive human disturbances can be substantial and have been overlooked in regards to elk nutritional condition.

Considering the above assumptions together, we hypothesize that the underlying basis for elk productivity is nutrition and that predation, hunting, weather variation and human disturbance currently are additive factors in their effects on the decline in productivity. Also, the relative effects of each may vary among herds. Where limiting factors are prevalent, density dependent effects may not be apparent. Ratios of predators to prey are critical in understanding how the several factors outlined above result in increasing or decreasing rates of population growth. At high levels of nutrition and appropriate (but unknown) predator to prey ratios, we hypothesize that effects of predation, human disturbance and hunting would reduce but not limit population growth. That is, elk recruitment would respond positively even with these sources of mortality under high levels of nutrition, thus providing an inherent resilience to population change. If, however, predator to prey ratios were shifted dramatically due to reduction of elk populations due to hunting, severe winter mortality of elk or increases in predator populations, then predation could keep elk at a lower equilibrium such that elk populations and recruitment remain low. The point at which mortality factors tip the balance between growth and decline of the prey population should vary along a gradient of nutritional adequacy.

Challenges to Research and Management

We believe our hypothesis deserves careful examination within a research framework. No single factor can explain either the variability of or the long-term decline in elk recruitment across the northwestern United States. Several factors interact, in different ways and in different areas, making it difficult to impose any one management strategy that can ensure recovery of elk populations.

One of the principal goals of the Starkey Project was to examine management activities that affected elk productivity. The original studies of the Starkey Project were directed at understanding how single factors affected elk fitness, distribution or reproduction (e. g., Cook et al. 1998, Rowland et al. 2000, Noyes et al. 2002). Subsequent research at Starkey and at associated study sites was then expanded to examine how two or more factors might interact to affect elk reproduction, survival and recruitment (e. g. Cook et al. 1999, 2004; Johnson and Jackson 2001). The next step is to further expand research to find the full complexity of factors and their interactive effects on elk productivity and population growth.

We suggest three complementary approaches to gain better knowledge about elk productivity: (1) adaptive management, (2) retrospective and meta analysis and (3) simulation modeling. Under adaptive management, research would be designed and tested as manipulative experiments in real-world conditions (Walters 1986). Researchers and managers would jointly develop hypotheses for testing and implementing treatments and would work together to measure and interpret the results. For example, a set of watersheds could be selected for intensive habitat improvements and subsequent reduction in human disturbances, and another set of similar watersheds could be used as controls. Researchers could measure and compare a variety of population responses of elk to the habitat improvements and the reduction in human disturbances. On the other hand, predator populations could be reduced, and responses of prey populations monitored, probably providing a more rapid assessment that would not require waiting on habitat changes.

Retrospective and meta analysis provide alternative analyses of potential value. Meta analysis could be used to analyze the multitude of studies that have evaluated the effects of nutrition, predation and weather on population dynamics of ungulates as case studies (e. g., Messier 1994, Linnell et al. 1995, Sæther 1997, Unsworth et al. 1999, Gaillard et al. 2000, Hayes 2003). The synthesis of many studies of elk recruitment (Schlegel 1976, 1983; Smith and Anderson 1996; Singer et al. 1997; Myers et al. 1998; Gratson and Zager 2000; Jedrzejewski et al. 2002; Zager and White 2003) could yield general, predictive patterns about the interactive effects of various factors on ungulate productivity and population dynamics. However, most of these studies had no measure of nutritional condition of the prey populations or had imprecise estimations of predator populations, making analysis and interpretation more difficult.

Simulation modeling also can provide insights about the potential interactions of factors that affect elk productivity and growth rates of populations. For example, a series of models could be constructed under different competing hypotheses, and each model could be run under a set of standard simulations to gain a more formal understanding of potential effects. Model parameters could reflect both empirical data and hypothesized data to further understand the range of potential effects and to identify the most plausible effects and outcomes. The challenge will be to link habitat conditions, prey populations and predator populations—an undertaking that has rarely been attempted. Hobbs (1989) and Weisberg et al. (2002) provided examples of models that link weather and habitat conditions with animal population characteristics, but neither incorporated predation effects. Typical predator-prey models (Skogland 1991) rarely explicitly addressed contributions of habitat and nutrition. Further, if simulation models are to consider habitat and nutritional conditions on summer ranges in forest ecosystems, then plant succession patterns following disturbance and the interaction of herbivore numbers and plant succession patterns will be key elements of models.

Without these types of new research, managers face the difficult challenge of dealing with the uncertain proximate and ultimate causes of declines in elk recruitment. Even then, long-term monitoring of elk recruitment, survival and nutritional condition are critical to understanding when predation, weather variation, and hunting may be limiting or when nutrition is regulating. Nutritional condition of elk populations can be measured inexpensively by using cow elk hunters to collect reproductive tracts by using suitable indices of condition and by using mammary tissue for animals harvested in autumn (Kohlmann 1999; Cook et al. 2001a, b). In situations where hunter collections are inadequate to satisfy research and monitoring objectives, nutritional condition of live elk can be assessed reliably (Cook et al. 2001a, b).

To deal with periods when elk recruitment is low or declining, managers need to know how density dependent factors, long-term vegetative successional trends, variation in summer precipitation and winter severity, hunting and predation may interact. If nutritional characteristics and climatic variability are implicated, then management alternatives to address habitat conditions are necessary. And, where hunting or predation appear to be limiting recruitment, alternative actions may increase recruitment. Public acceptance of predator management evidently is waning, challenging managers to meet the social and economic needs of the various public groups interested in wildlife. Consequently, a strong partnership between managers and researchers, with the use of modeling and adaptive management approaches, appears to be the most suitable strategy for understanding and for effectively dealing with emerging challenges of declining elk productivity.

Reference List

Adams, L. G., B. W. Dale, and L. D. Mech. 1995. Wolf predation on caribou calves in Denali National Park, Alaska. In *Ecology and conservation of wolves in a changing world, number 35*, eds. L. N. Carbyn, S. H. Fritts, and D. R. Seip, 245–60. Edmonton, Alberta: Canadian Circumpolar Institute, Occasional Publications.

Bartmann, R. M., G. C. White, and L. H. Carpenter. 1992. Compensatory mortality in a Colorado mule deer population. *Wildlife Monographs*. 121:1–39.

Buechner, H. K., and C. V. Swanson. 1955. Increased natality resulting from lowered population density among elk in southeastern Washington. *North American Wildlife Conference*. 20:560–7.

Caswell, H. 2001. *Matrix population models: Construction, analysis, and interpretation. 2nd edition*. Sunderland, Massachusetts: Sinauer Associates, Inc.

Conner, M. M., G. C. White, and D. J. Freddy. 2001. Elk movement in response to early-season hunting in northwest Colorado. *Journal of Wildlife Management*. 65:926–40.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs*. 155:1–61.

Cook, J. G., L. J. Quinlan, L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1996. Nutrition-growth relations of elk calves during late summer and fall. *Journal of Wildlife Management*. 60:528–41.

Cook, J. G., L. L. Irwin, L. Bender, B. Johnson, and D. Boyce, Jr. 1999. *Assessing landscape-scale nutritional influences on elk herd productivity in the Northwest—A proposal for east- and west-side studies*. La Grande, Oregon: National Council for Air and Steam Improvement.

Cook, J. G., L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1998. Relations of forest cover and condition of elk: A test of the thermal cover hypothesis in summer and winter. *Wildlife Monographs*. 141:1–41.

Cook, R. C., D. L. Murray, J. G. Cook, P. Zager, and S. L. Monfort. 2001c. Nutritional influences on breeding dynamics of elk. *Canadian Journal of Zoology*. 79:845–53.

Cook, R. C., J. G. Cook, D. L. Murray, P. Zager, and B. K. Johnson. 2001b. Nutritional condition models for elk: Which are the most sensitive, accurate, and precise? *Journal of Wildlife Management*. 65:988–97.

Cook, R. C., J. G. Cook, D. L. Murray, P. Zager, B. K. Johnson, and M. W. Gratson. 2001a. Development of predictive models of nutritional condition for Rocky Mountain elk. *Journal of Wildlife Managemen*t. 65:973–87.

Coulson, T., S. Albon, F. Guiness, J. Pemberton, and T. Clutton-Brock. 1997. Population substructure, local density, and calf winter survival in red deer (*Cervus elaphus*). *Ecology*. 78:852–63.

Crête, M., and J. Huot. 1993. Regulation of a large herd of migratory caribou: summer nutrition affects calf growth and body reserves of dams. *Canadian Journal of Zoology*. 71:2,291–6.

Dusek, G. L., A. K. Wood, and S. T. Stewart. 1992. Spatial and temporal patterns of mortality among female white-tailed deer. *Journal of Wildlife Management*. 56:645–50.

Eberhardt, L. L. 2002. A paradigm for population analysis of long-lived vertebrates. *Ecology*. 83:2,841–54.

Eberhardt, L. E., L. L. Eberhardt, B. L. Tiller, and L. L. Cadwell. 1996. Growth of an isolated elk population. *Journal of Wildlife Management*. 60:369–73.

Fowler, C. W. 1981. Density dependence as related to life history strategy. *Ecology*. 62:602–610.

———. 1987. A review of density dependence in populations of large mammals. In *Current mammalogy, volume 1*, ed. Genoways, H. H., 401–41. New York, New York: Plenum Press..

Garrott, R. A., L. L. Eberhardt, P. J. White, and J. Rotella. 2003. Climate-induced variation in vital rates of an unharvested large-herbivore population. *Canadian Journal of Zoology*. 81:33–45.

Gaillard, J. M., M. Festa-Bianchet, and N. G. Yoccoz. 1998. Population dynamics of large herbivores: Variable recruitment with constant adult survival. *Trends in Ecological Evolution*. 13:58–63.

Gaillard, J. M., M. Festa-Bianchet, N. G. Yoccoz, A. Loison, and C. Toigo. 2000. Temporal variation in fitness components and population dynamics of large herbivores. *Annual Review of Ecological Systematics*. 31:367–93.

Gratson, M. W., and P. Zager. 2000. *Elk ecology. Study IV. Factors influencing elk calf recruitment, job progress report, project W-160-R-26*. Boise, Idaho: Idaho Department of Fish and Game, Federal Aid in Wildlife Restoration.

Grubb, P. 1974. Population dynamics of the Soay sheep. In *Island survivors: The ecology of Soay sheep on St. Kilda*, eds. P. A. Jewell, C. Milner, and J. M. Boyd, 242–72. London, England: Athlone Press.

Hayes, R. D., R. Farnell, R. M. P. Ward, J. Carey, M. Dehn, G. W. Kuzyk, A. M. Baer, C. L. Gardner, and M. O'Donoghue. 2003. Experimental reduction of wolves in the Yukon: Ungulate responses and management implications. *Wildlife Monographs*. 152:1–35.

Hett, J., R. Taber, J. Long, and J. Schoen. 1978. Forest management policies and elk summer carrying capacity in the *Abies amabilis* forest, western Washington. *Environmental Management*. 2:561–6.

Hobbs, N. T. 1989. Linking energy balance to survival in mule deer: development and test of a simulation model. *Wildlife Monographs*. 101:1–39.

Holter, J. G., and H. H. Hayes. 1977. Growth of white-tailed deer fawns fed varying energy and constant protein. *Journal of Wildlife Management*. 41:506–10.

Houston, D. B. 1982. *The northern Yellowstone elk: Ecology and management*. New York, New York: Macmillan Publishing Company.

Irwin, L. L., J. G. Cook, R. A. Riggs, and J. M. Skovlin. 1994. *Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystem, general technical report PNW-GTR-325*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Jedrzejewski, W., K. Schmidt, J. Theuerkauf, B. Jedrzejewska, N. Selva, K. Zub, and L. Szymura. 2002. Kill rates and predation by wolves on ungulate populations in Bialowieza Primeval Forest (Poland). *Ecology*. 83:1,341–56.

Johnson, B. K., A. A. Ager, J. H. Noyes, and N. Cimon. 2004. Elk and mule deer responses to variation in hunting pressure. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:625–40.

Johnson, B. K., and D. H. Jackson. 2001. *Evaluating the effects of predation and nutrition on recruitment of elk in Oregon—A study plan, federal aid project 98-R-1*.La Grande, Oregon: Oregon Department of Fish and Wildlife.

Keech, M, A., R. D. Bowyer, J. M. Ver Hoef, R. D. Boertje, B. W. Dale, and T. R. Stephenson. 2000. Life-history consequences of maternal condition in Alaskan moose. *Journal of Wildlife Management*. 64:450–62.

Keister, G. P, Jr., and W. A. Van Dyke. 2002. A predictive population model for cougars in Oregon. *Northwest Science*. 76:15–25.

Knight, R. L., and K. J. Gutzwiller, eds. 1995. *Wildlife and recreationists, coexistence through management and research*. Washington, DC: Island Press.

Kohlmann, S. G. 1999. Adaptive fetal sex allocation in elk: Evidence and implications. *Journal of Wildlife Management*. 63:1,109–17.

Kunkel, K., and D. H. Pletscher. 1999. Species-specific population dynamics of cervids in a multipredator ecosystem. *Journal of Wildlife Management*. 63:1,082–93.

Linnell, J. D. C., R. Aanes, and R. Andersen. 1995. Who killed Bambi? The role of predation in the neonate mortality of temperate ungulates. *Wildlife Biology*. 1:209–23.

Lubow, B. C., F. J. Singer, T. L. Johnson, and D. C. Bowden. 2002. Dynamics of interacting elk populations within and adjacent to Rocky Mountain National Park. *Journal of Wildlife Management*. 66:757–75.

McCullough, D. R. 1979. *The George Reserve deer herd*. Ann Arbor, Michigan: University of Michigan Press.

Mech, L. D, D. W. Smith, K. M. Murphy, and D. R. MacNulty. 2001. Winter severity and wolf predation on a formerly wolf-free elk herd. *Journal of Wildlife Management*. 65:998–1,003.

Merrill, E. H., and M. S. Boyce. 1991. Summer range and elk population dynamics in Yellowstone National Park. In *The greater Yellowstone ecosystem: redefining America's wilderness heritage*, eds. R. B. Keiter, and M. S. Boyce, 263–74. New Haven, Connecticut: Yale University Press.

Messier, F. 1994. Ungulate population modes with predation: A case study with North American moose. *Ecology*. 75:478–88.

Messier, F., and M. Crête. 1985. Moose-wolf dynamics and the natural regulation of moose populations. *Oecologia*. 65:503–12.

Myers, W. L., B. Lyndaker, P. E. Fowler, and W. Moore. 1998. *Investigations of calf elk mortalities in southeast Washington: Study completion report, project W-96-R*. Olympia, Washington: Washington Department of Fish and Wildlife.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management*. 66:1,301–7.

Oregon Department of Fish and Wildlife. 2001. *2001 Big Game Statistics*. Portland, Oregon: Oregon Department of Fish and Wildlife.

———. 2003. *Oregon's elk management plan*. Portland, Oregon: Oregon Department of Fish and Wildlife.

Parker, K. L., M. P. Gillingham, T. A. Hanley, and C. T. Robbins. 1999. Energy and protein balance of free-ranging black-tailed deer in a natural forest environment. *Wildlife Monographs*. 143:1–48.

Peek, J. M., B. Dennis, and T. Hershey. 2002. Predicting population trends of mule deer. *Journal of Wildlife Management*. 66:729–36.

Peek, J. M., J. J. Korol, D. Gay, and T. Hershey. 2001. Overstory-understory biomass changes over a 35-year period in southcentral Oregon. *Forest Ecology and Management*. 150:267–77.

Phillips, G. E., and A. W. Alldredge. 2000. Reproductive success of elk following disturbance by humans during calving season. *Journal of Wildlife Management*. 64:521–30.

Riggs, R. A., A. R. Tiedemann, J. G. Cook, T. M. Ballard, P. J. Edgerton, M. Vavra, W. C. Krueger, F. C. Hall, L. D. Bryant, L. L. Irwin, and T. Delcurto. 2000. *Modification of mixed conifer forests by ruminant herbivores in the Blue Mountains ecological province, general technical report PNW-RP-527*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research, Station.

Robinson, H. S., R. B. Wielgus, and J. C. Gwilliam. 2002. Cougar predation and population growth of sympatric mule deer and white-tailed deer. *Canadian Journal of Zoology*. 80:556–68.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: Implications for management in forested ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:491–508.

Sæther, B. E. 1997. Environmental stochasticity and population dynamics of large herbivores: A search for mechanisms. *Trends in Ecology and Evolution*. 12:143–9.

Schlegel, M. W. 1976. Factors affecting calf elk survival in northcentral Idaho: a progress report. *Proceedings of the Annual Conference of the Western Association of State Game and Fish Commissions*. 56:342–55.

———. 1983. *Factors affecting calf survival in the Lochsa elk herd, project W-160-R*. Boise, Idaho: Idaho Department of Fish and Game.

Sinclair, A. R. E., S. Mduma, and J. S. Brashares. 2003. Patterns of predation in a diverse predator-prey system. *Nature*. 425:288–90.

Singer, F. J., A. Harting, K. K. Symonds, and M. B. Coughenour. 1997. Density dependence, compensation, and environmental effects on elk calf mortality in Yellowstone National Park. *Journal of Wildlife Management*. 61:12–25.

Skogland, T. 1991. What are the effects of predators on large ungulate populations? *O ikos*. 61:401–11.

Smith, B. L., and S. H. Anderson. 1996. Patterns of neonatal mortality of elk in northwest Wyoming. *Canadian Journal of Zoology*. 74:1,129–37.

Steinheim, G., A. Mystrerud, O. Holand, M. Bakken, and T. Adnoy. 2002. The effect of initial weight of the ewe on later reproductive effort in domestic sheep (*Ovis aries*). *Journal of Zoology* (London). 358:515–520.

Swihart, R. K., H. P. Weeks, Jr., A. L. Easter-Pilcher, and A. J. DeNicola. 1998. Nutritional condition and fertility of white-tailed deer (*Odocoileus virginianus*) from areas with contrasting histories of hunting. *Canadian Journal of Zoology*. 76:1932–41.

Taper, M. L., and P. J. P. Gogan. 2002. The Northern Yellowstone elk: Density dependence and climatic conditions. *Journal of Wildlife Management*. 66:106–22.

Thorne, T., R. E. Dean, and W. J. Hepworth. 1976. Nutrition during gestation in relation to successful reproduction in elk. *Journal of Wildlife Management*. 40:330–5.

Thorne, T., E. S. Williams, W. M. Samuel, and T. P. Kistner. 2002. Diseases and parasites. In *North American elk: Ecology and management*, eds. Toweill, D. E., and J. W. Thomas, 351–88. Washington, DC: Smithsonian Institution Press.

Trainer, C. E. 1971. *The relationship of physical condition and fertility of Roosevelt elk* (Cervus canadensis roosevelti) *in Oregon*. M. S. thesis, Oregon State University, Corvallis, Oregon.

Tveraa, T., P. Fauchald, C. Henaug, and N. G. Yoccoz. 2003. An examination of a compensatory relationship between food limitation and predation in semi-domestic reindeer. *Oecologia*. 137: 370–6.

Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*. 14:18–30.

Unsworth, J. W., D. F. Pac, G. C. White, and R. M. Bartmann. 1999. Mule deer survival in Colorado, Idaho, and Montana. *Journal of Wildlife Management*. 63:315–26.

Vavra, M., and R. L. Phillips. 1980. Drought effects on cattle performance, diet quality, and intake. *Proceedings, western section, American society of animal science*. 31:157–60.

Verme, L. J., and J. J. Ozoga. 1980. Effects of diet on growth and lipogenesis in deer fawns. *Journal of Wildlife Management*. 44:315–24.

Vieira, M. E. P., M. M. Conner, G. C. White, and D. J. Freddy. 2003. Effects of archery hunter numbers and opening dates on elk movement. *Journal of Wildlife Management*. 67:717–28.

Wakkinen, W. L., and B. K. Johnson. 2001. *Selkirk ecosystem project, threatened and endangered species project E-14–6*. Boise, Idaho: Idaho Department of Fish and Game.

Walters, C. 1986. *Adaptive management of renewable resources*. New York, New York: MacMillan Press.

Weisberg, P. J., N. T. Hobbs, J. E. Ellis, and W. B. Coughenour. 2002. An ecosystem approach to population management of ungulates. *Journal of Environmental Management*. 65:181–91.

Wertz, T., A. Blumton, and L. J. Erickson. 2004. Conflict resolution by adaptive management: Moving elk where they want to go. In *Proceedings 2001 western states and provinces, deer and elk workshop*, ed. J. Mortensen, 59–66. Salem, Oregon: Oregon Department of Fish and Wildlife.

Wisdom, M. J. 1998. *Assessing life-stage importance and resource selection for conservation of selected vertebrates*. Ph. D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004. Effects of off-road recreation on mule deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:531–50.

Wisdom, M. J., and J. G. Cook. 2000. North American Elk. In *Ecology and management of large mammals in North America*, eds. Demarais, S., and P. R. Krausman, 694–735. Upper Saddle River, New Jersey: Prentice Hall.

Yarmoloy, C., M. Bayer, and V. Geist. 1988. Behavior responses and reproduction of mule deer, *Odocoileus hemionus*, does following experimental harassment with an all-terrain vehicle. *The Canadian Field-Naturalist*. 102:425–9.

Zager, P., and C. White. 2003. *Elk ecology. Study IV: Factors influencing elk calf recruitment, progress report, project W-160-R-30*. Boise, Idaho: Idaho Department of Fish and Game.

Influence of Age of Males and Nutritional Condition on Short- and Long-term Reproductive Success of Elk

James H. Noyes
Oregon Department of Fish and Wildlife
La Grande

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Brian L. Dick
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

Rocky Mountain elk (*Cervus elaphus*) populations in some areas of northeastern Oregon have experienced declines in spring calf to cow ratios of nearly 80 percent over the last 40 years. Among the potential causes of these declines, the effects of age of male sires and the nutritional condition of females on conception dates and pregnancy rates have received the most attention from biologists and wildlife managers. Reliance on younger males as primary breeders can result in later conceptions and a prolonged rut period (Follis 1972; Hines and Lemos 1979; Noyes et al. 1996, 2002). Mechanisms involved with delayed conception due to male age have included late maturity of young males (Hines et al. 1985), female preference for older males (Gibson and Guinness 1980, Squibb 1985) and delayed timing of estrus in the absence of older males (Komers et al. 1999). The rate of conception (pregnancy), rather than the timing, does not appear to be dependent on the presence of older male sires (Follis 1972; Noyes et al. 1996, 2002).

Female nutritional condition during breeding influences date of conception (Trainer 1971, Mitchell and Lincoln 1973) and pregnancy rate (Trainer 1971, Albon et al. 1986). The importance of nutritional condition applies to males as well as females. It has been shown that age at puberty (Hines et al. 1985) and annual or long-term reproductive success of males depends on body size and dominance, both of which are influenced by birth date and nutritional condition (Green and Rothstein 1993, Komers et al. 1999). Although age of male sires, nutritional condition of females (and males to a lesser extent), predation and other determinants of ungulate productivity have been examined in many studies, interactions in Rocky Mountain elk have only recently been addressed (see Cook et al. 2004).

We conducted a study in two trials, from 1989 to 1993 and from 1995 to 1999, to assess the effects of male age and female nutritional condition on conception dates and pregnancy rates of female elk in northeastern Oregon (Noyes et al. 1996, 2002). Results of both trials showed a significant influence of male age on conception dates but not on pregnancy rates. Results from the interaction of male age and female nutritional condition pooled across trials likewise verified the importance of male age in affecting conception dates across a range of nutritional conditions.

The most commonly referenced manifestations of skewed sex ratios and nutritional limitations on elk reproduction are short-term (annual) differences in conception dates, pregnancy rates and calf survival. Beyond their short-term effects on calf survival, early conceptions and birth dates may be of greater significance in affecting long-term reproductive success and herd demographics. Benefits of early births may include higher lifetime reproductive success in female red deer (*C. e. elaphus* [Clutton-Brock et al. 1987]), bison (*Bison bison* [Green and Rothstein 1993]), and moose (*Alces alces* [Saether et al. 2003]). Understanding the importance of maintaining older males in elk populations, in conjunction with knowledge of nutritional condition and their interactions should assist wildlife managers in making informed, effective decisions about harvest management.

Study Area

We conducted our study within a 30 square mile (78 km^2) study area at the U. S. Forest Service's Starkey Experimental Forest and Range (Starkey) in northeastern Oregon, about 21 miles (35 km) southwest of La Grande. Elevation ranged from 3,680 feet (1,116 m) to 4,960 feet (1,502 m). Vegetation was a mixture of grassland, regenerating forests and older forest stands. Grand fir (*Abies grandis*), Douglas fir (*Pseudotsuga menziesii*) and lodgepole pine (*Pinus contorta*) dominated the north aspects and higher elevations; ponderosa pine (*P. ponderosa*) was the dominant forest vegetation at lower elevations. Bluebunch wheatgrass (*Agropyron spicatum*) and Idaho fescue (*Festuca idahoensis*) typically dominated grassland vegetation. Mean annual precipitation was 20 inches (50.8 cm), and average mean temperatures were 24.8° Fahrenheit (-4° C) in January and 64° Fahrenheit (18° C) in July. Starkey was enclosed by an 8-feet (2.6-m) tall, game-proof fence that allowed us to adjust the population size and structure of a free-ranging elk herd. Further descriptions of the study area can be found in Noyes et al. (1996) and in Rowland et al. (1997).

Methods

Herd Management

We managed the elk population during both trials to allow a single cohort of males to function as principal herd sires as they matured from 1 to 5 years of age. We estimated our elk population size with a model described in greater detail in Noyes et al. (1996).

We maintained a bull:cow ratio of the study cohort between 16:100 and 21:100 during both trials to minimize the effects of numbers of males. We conducted hunts for yearling male elk in early August (except 1989 and 1995) to reduce the number of yearling males prior to the breeding season. Males that were younger than the study cohort were trapped annually and released outside of Starkey.

We fed a maintenance ration of alfalfa hay to those elk that moved to the winter feed ground. Elk were returned to the study area in similar nutritional condition each year to minimize the influence of variable winter severity on elk reproduction. Noyes et al. (1996, 2002) provide further descriptions of herd management. Our research was conducted under approved animal welfare protocols (Wisdom et al. 1993).

Reproductive Data

We collected reproductive tracts (uteri and ovaries), udders, lower incisors and kidneys with associated fat from female elk killed by hunters in early

December. We determined conception dates, pregnancy status, lactation status and female age and nutritional condition (KFI), as described by Trainer (1971). We conducted blood tests for leptospirosis (*Leptospira* spp.) and for brucellosis (*Brucella abortus*) to identify presence of diseases that may have affected elk reproduction.

Statistical Analysis

Conception Date

We compared conception dates of females among male ages (years) in each trial with analysis of variance and accounted for female nutritional condition with analysis of covariance. We compared conception dates of lactating and nonlactating females older than or equal to 3 years old with *t*-tests. We used stepwise multiple regressions to predict conception date.

Pregnancy Rate

We excluded females younger than age 3 or older than age 13 when summarizing pregnancy rates pooled by male age across trials because of age and lactation status effects.

Female Nutritional Condition

We used analysis of variance to compare KFI among years for females greater than or equal to 2 years old and compared KFI of lactating and nonlactating females greater than or equal to 3 years old with *t*-tests. We also used *t*-tests to evaluate nutritional condition of lactating and nonlactating females between the two trials. We tested for correlation between KFI for all females and May to August precipitation.

Results

Conception Date

Conception dates varied with age of male sires in both trials and also when conception dates were pooled among trials and adjusted for female nutritional condition. The largest differences in mean breeding dates were between 5-year-old sires and yearling or 2-year-old sires. Females bred by males older than or equal to 3 years of age conceived earlier than females bred by younger males in both trials, and they had similar conception dates. Median dates of conception for females bred by yearling males were approximately 2 weeks later than when 5-year-old males were sires. Lactating females conceived about 9 days later than nonlactating females.

Conception dates became more synchronous as male age increased (See Figure 1 in Noyes et al. 1996 for the chronology of conceptions, which applies to both trials). The duration of the rut differed by an average of 31 days between years with yearling and 5-year-old male sires (Noyes et al. 1996, 2002). We discarded the latest 10 percent of conceptions in each year to reflect that portion of the annual reproduction with the most management relevance. The date by which 90 percent of pregnant females were bred by yearling males was approximately 3 weeks later than when 5-year-old males were the sires. The cumulative percent of conceptions moved toward earlier dates as male age increased (Figure 1).

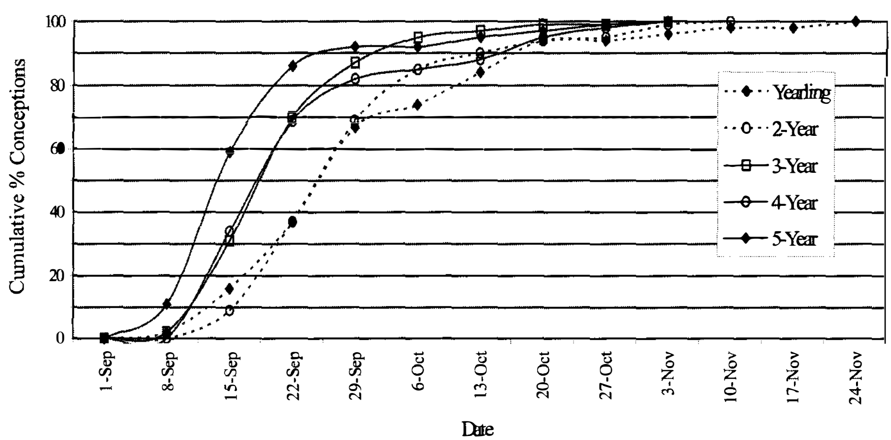


Figure 1. Cumulative percent conceptions for adult female elk (2 years or older) bred by males of five different ages at Starkey Experimental Range, Oregon, 1989 to 1999. Data were pooled from two identical 5-year trials. Age of sires increased each year from yearlings to 5-year-olds.

Pregnancy Rate

Pregnancy rates of females greater than or equal to 2 years old did not differ by male age in either trial. Pregnancy rates of females between 3 and 13 years of age pooled across trials, ranged between 89 percent and 94 percent (from Noyes et al. 1996, 2002). Pregnancy was related to KFI but not male age in pooled trials.

Female Nutritional Condition

The KFI of adult females differed among years and was significantly greater in 1989 and 1995, when yearling males were the primary herd sires. The KFI for all adult females during the second trial was less than KFI during the first trial. Pregnant, lactating adult females were in especially low nutritional condition in both trials when breeding was by 4-year-old males. KFI was lower in lactating females than in nonlactating females. KFI was correlated with May to August precipitation, and precipitation during those months was lower in the second trial than recorded in the first trial.

Discussion

We acknowledge the complex interactions among variables affecting reproduction in elk, but, for the purposes of this paper, we will emphasize the contribution of the age of male sires to differences in calf survival and long-term reproductive success.

We demonstrated the importance of mature males as sires in order to achieve early and synchronous conception in elk, but we found no relationship between age of males and pregnancy rates. In our study, female nutritional condition was significantly higher during the 2 years that immature males were sires than during the other years. Because of this, we were not able to determine if pregnancy rates of females bred by immature males might be lower if nutritional conditions were comparable to the other years. Holand et al. (2003) assessed the effects of skewed sex ratios and male age structure on calving rates of female reindeer (*Rangifer tarandus*) in excellent condition, and he questioned how the results might vary if females were in poor condition.

Early breeding by older male sires has been documented for elk (Follis 1972, Hines and Lemos 1979), moose (Saether et al. 2003), fallow deer (Komers et al. 1999) and reindeer (Holand et al. 2003). Conception dates for females in both of our trials were strongly influenced by the age of male sires across a range of nutritional conditions. After adjusting conception dates for differences in female nutritional condition, conception dates became increasingly early as male sires matured from yearlings to 5-year-olds.

Another influence of male age results from conception synchrony and its effect on female reproductive success and neonate survival. The significance of synchronous births on survival of neonates likely depends on the strategy for optimizing neonate survival employed by different species (Geist 1982, Clutton-Brock et al. 1987, Kiltie 1988).

For a discussion on the effects of male age on calf survival and long-term reproductive success, we will assume a constant gestation length and, consequently, will consider the phrases birth date and conception date to be interchangeable. Results of previous research on the length of the gestation period in ungulates and the ability of females to adjust the length vary, depending on the species, the nutritional condition of females, supplemental feeding, the effects on calf weight and survival, and perhaps a host of other unknown factors. No consistent patterns between nutritional condition and gestation length appear to be available. Guinness et al. (1978) stated that differences in calving time are likely due to factors affecting the time of conception, and we will assume that early and late conceptions are reflected in early and late births.

Birth date has been closely linked to the probability of offspring survival. Late calving and reduced survival has been associated with lower body weight in autumn in reindeer (Holand et al. 2003) and in winter in moose (Saether et al. 2003). Delays in breeding as a result of highly skewed sex ratios and the subsequent delays in birthing may reduce survival of offspring. Winter survival of red deer was related to birth date and population density (Guinness et al. 1978, Loison and Langvatn 1998). Cook et al. (2004) suggested that winter survival of captive-reared calves under varied nutritional condition was not related to birth date but indicated that other potential causes of mortality (e. g. predation) were not present. Clutton-Brock et al. (1987) reported mortality of red deer calves increased by 1 percent for each day the calf was born after the median birth date. Captive red deer hinds that bred much later than normal did not lactate normally following the late births, and Guinness et al. (1978) suggested that their calves in the wild would quickly die.

The ability of young ungulates to exhibit compensatory growth is an important variable that should be considered in assessing the relationship between birth date and survival. Research has shown conflicting patterns of compensatory growth in ungulates. Late-born captive calf elk on low or medium quality diets reached weights in winter similar to early-born calves, while late-born calves on a high quality diet had significantly lower weights than early-born calves during the first year of a study (Cook et al. 2004). Compensation in the low and medium nutrition groups may have been influenced by growth rate differences between sexes; late-born calves were composed of 62 percent males, compared to 31 percent in early-born calves. This pattern of compensatory growth was reversed during the second year, however, and the advantage of early birth was present in the low and medium nutrition groups but not in calves in the high nutrition group. Guinness et al. (1978) observed a relationship between winter weights and birth dates of red deer under high nutritional conditions. Late-born calves had lower weights, further illustrating the difficulty in assessing the presence of compensatory growth under sometimes widely different conditions. Holand et al (2003) reported that late calving in reindeer was associated with lower body weight in autumn but suggested caution when applying results from enclosure experiments to wild, free-ranging populations.

The relative importance of birth date differences resulting from age of male sires and their interaction with nutritional condition varies among studies and whether they involve wild or penned ungulates. Ginsberg and Milner-Gulland (1994) reported that breeding delays of one estrous cycle (18 days) in red deer can result in a 36-percent decline in reproductive success of females. Cook et al. (2004) suggested that birth date differences of 3 weeks (the effect of male age differences in our study) were insufficent to influence winter survival of calves under captive conditions. Mean differences of 1 week or less may be biologically significant for reindeer in environments with short seasons (Holand et al. 2003).

Perhaps more important than the effects of birth date and nutrition on survival of young are the long-term cumulative effects on herd productivity. Most studies that have failed to identify cumulative effects are short-term studies and, thus, are not able to determine these effects beyond 1 or 2 years. Green and Rothstein (1993) conducted a 9-year study to evaluate the relationship between birth date, long-term growth and reproductive success in bison. They documented beneficial effects of early birth that endured for the length of the study. Early-born females had significant fitness advantages and were more fecund during their first 9 years (Green and Rothstein 1993). Body weight differences related to birth date persisted throughout the study in females, suggesting that adult body size is enhanced by early birth. Links between birth date and body mass of moose calves in winter have been reported by Saether et al. (2003), who suggested that there may be long-term consequences of skewed sex ratios. Early births enhanced the probability that females would give birth in the succeeding year; late births increased the likelihood of subsequent reproductive failure.

To illustrate the range of interpretation of research results, Cook et al. (2001) stated that marginally deficient nutrition was responsible for delayed breeding of prime-aged lactating females despite high pregnancy rates. In our study of 17 prime-aged lactating females in deficient nutritional condition (less than 9 percent body fat), pregnancy rate was 94 percent, and conception dates were not delayed. This is another indication that age of male sires is an important influence on pregnancy rates and conception dates and is implicated in herd demographics due to reduced survival of late-born calves.

A longer, slower trend toward lower productivity in ungulate populations may exist with late conceptions because hinds that conceived early or produced heavy calves had higher lifetime success (Clutton-Brock et al. 1983). Several mechanisms are plausible to explain reduced long-term reproductive success. Delayed births may not allow females to recover from the demands of lactation in order to ovulate early in the rut or at all (Laflamme and Connor 1992). Female red deer that have delayed conception by one estrous cycle (18 days) may experience a 36 percent decline in reproductive success. In bison (Green and Rothstein 1993) it has been shown that the effects of late birth and the resulting decreased body mass may have lasting consequences that are likely not evident from the results of annual reproductive performance.

Reproductive success of males is related to body mass. Early-born bison calves are socially dominant to late-born calves and may have increased reproductive success (Lott 1979, Green and Rothstein 1993). Many life history characteristics are closely related to body mass of young females (Saether et al. 2003), which in moose is related to their body mass as calves, which is related to birth date. Because of this same relationship in male calves, the effects of late births result in reduced body mass of yearling males, which again can result in lower reproductive success. We could speculate that this pattern might function as a mechanism by which a subtle cycle of later births, lower body mass and decreased reproductive success is repeated to the long-term detriment of elk productivity.

The age of male sires can also influence reproductive dynamics and survival by means other than dates of conception. Young males may be either socially (behavior) or physiologically immature during the rut period. Female fallow deer avoided subadult males more than mature males (Komers et al. 1999).

Females with subadult males lost more weight (4.2 percent) during the rut than females with mature males (1.9 percent), and reduced body reserves could compromise female survival. Male moose in Norway exhibited a long-term decline in mean body mass as the proportion of adult males decreased (Solberg and Saether 1994) . They also found that the oldest male age groups experienced the largest declines. The relationship between male body mass and reproductive success has been reported for red deer and bison. Early birth affects the social dominance status and breeding success in male bison.

Young females may be especially sensitive to skewed sex ratios (Solberg et al. 2002, in Saether et al. 2003). In our study the behavioral immaturity of yearling males might explain the reproductive performance of 2-year-old females. There were approximately equal numbers of 2-year-old females for each of the five age classes of male sires (60 females total). Pregnancy rates were similar, ranging from 82 percent to 100 percent, and nutrition was significantly higher when yearling males were sires; however, conception dates of 2-year-old females were again about 3 weeks different between immature and mature male sires.

Management Implications

Effective management of elk populations might involve providing mature males to obtain early and synchronous conceptions, and interact with nutritional condition to enhance the survival of calves and long-term productivity. The importance of dates of conception might also be to set the stage for the influences of female nutritional condition. Without early conception, high nutritional condition of females cannot solely determine the probability of calf survival. We have restricted our discussion to interactions among male age and the nutritional condition of females and males. The nutritional condition of elk herds can be reduced for reasons other than habitat quality. Preliminary results from ongoing studies evaluating the energetic effects of recreational use (Wisdom et al. 2004) and increased movements related to hunting seasons (Ager et al. 2004) indicate energetic costs that also may affect herd demographics. Preliminary results from 4 years of a 6-year study evaluating archery disturbance during the rut indicate lower pregnancy rates and asynchronous conceptions (J. H. Noyes, unpublished data 2003).

We have identified several variables and their interactions that affect elk reproduction in slightly different ways. The challenge to wildlife managers is to provide the conditions that will allow the most opportunities for increased production over the greatest range of conditions. Managing for mature bulls through harvest regulations is much more straightforward than predicting precipitation and annual forage production. The presence of mature males in ungulate populations may be warranted for reasons other than their effects on short-term productivity. The significance of mature males in elk populations, regardless of the interpretations of research results, should not be ambiguous because of the evolutionary doctrine that states natural selection operates to provide conditions that enhance the survival of species. Mature bulls ordinarily function as principal sires of polygamous harems (Bubenik 1982). We seek to understand some of the complex interactions that are present today under conditions that have been greatly altered. Wildlife managers might consider all variables affecting the productivity of elk herds (male age, nutritional condition, predation, human disturbance and others). Challenges lay in adapting management options to the variety of social and environmental conditions that currently exist.

Reference List

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:687–707.

Albon, S. D., B. Mitchell, B. J. Huby, and D. Brown. 1986. Fertility in female red deer (*Cervus elaphus*): The effects of body composition, age, and reproductive status. *Journal of Zoology (London).* 209:447–60.

Bubenik, A. D. 1982. Physiology. In *Elk of North America: Ecology and management*, eds. J. W. Thomas, and D. E. Toweill, 125–79. Harrisburg, Pennsylvania: Stackpole Books.

Clutton-Brock, T. H., F. E. Guinness, and S. D. Albon. 1983. The costs of reproduction to red deer hinds. *Journal of Animal Ecology.* 52:367–83.

Clutton-Brock, T. H., S. D. Albon, and F. E. Guinness. 1987. Early development and population dynamics in red deer:Density-dependent effects on juvenile survival. *Journal of Animal Ecology.* 56:53–67.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monograph.* 155:1–61.

Cook, R. C., D. L. Murray, J. G. Cook, P. Zager, and S. L. Monfort. 2001. Nutritional influences on breeding dynamics in elk. *Canadian Journal of Zoology.* 79:845–53.

Follis, T. B. 1972. *Reproduction and hematology of the Cache elk herd.*, 72–8. Salt Lake City, Utah: Utah Division of Wildlife Resources.

Geist, V. 1982. Adaptive behavioral strategies. In *Elk of North America: Ecology and management,* eds. J. W. Thomas, and D. E. Toweill, 219–78. Harrisburg, Pennsylvania: Stackpole Books.

Gibson, R. M., and F. E. Guinness. 1980. Differential reproduction among red deer (*Cervus elaphus*) stags on Rhum. *Journal of Animal Ecology.* 49:199–208.

Ginsberg, J. R., and E. J. Milner-Gulland. 1994. Sex-biased harvesting and population dynamics in ungulates: Implications for conservation and sustainable use. *Conservation Biology.* 8:157–66.

Green, W. C. H., and A. Rothstein. 1993. Persistent influences of birth date on dominance, growth and reproductive success in bison. *Journal of Zoology (London).* 230:177–86.

Guinness, F. E., T. H. Clutton-Brock, and S. D. Albon. 1978. Factors affecting calf mortality in red deer (*Cervus elaphus*). *Journal of Animal Ecology.* 47:817–32.

Hines, W. W., and J. C. Lemos. 1979. *Reproductive performance by two age-classes of male Roosevelt elk in southwestern Oregon, wildlife research report no. 8.* Portland, Oregon: Oregon Department of Fish and Wildlife.

Hines, W. W., J. C. Lemos, and N. A. Hartman. 1985. *Male breeding efficiency in Roosevelt elk of southwestern Oregon, wildlife research report no. 15.* Portland, Oregon: Oregon Department of Fish and Wildlife.

Holand, O., K. H. Roed, A. Mysterud, J. Kumpula, M. Nieminen, and M. E. Smith. 2003. The effect of sex ratio and male age structure on reindeer calving. *Journal of Wildlife Management.* 67:25–33.

Kiltie, R. A. 1988. Gestation as a constraint on the evolution of seasonal breeding in mammals. In *Evolution of life histories of mammals*, ed. M. S. Boyce, 257–89. New Haven, Connecticut: Yale University Press.

Komers, P. E., B. Birgersson, and K. Ekvall. 1999. Timing of estrus in fallow deer is adjusted to the age of available mates. *American Naturalist.* 153(4):431–6.

Laflamme, L. F., and M. L. Connor. 1992. Effect of postpartum nutrition and cow body condition at parturition on subsequent performance of beef cattle. *Canadian Journal of Animal Science.* 72:843–51.

Loison, A., and R. Langvatn. 1998. Short- and long-term effects of winter and spring weather on growth and survival of red deer in Norway. *Oecologia.* 116:489–500.

Lott, D. F. 1979. Dominance relations and breeding rate in mature male American bison. *Zeitschrift-Fuer-Tierpsychologie*. 49(4):418–32.

Mitchell, B., and G. A. Lincoln. 1973. Conception dates in relation to age and condition in two populations of red deer in Scotland. *Journal of Zoology (London*). 171:141–152.

Noyes, J. H., B. K. Johnson, L. D. Bryant, S. L. Findholt, and J. W. Thomas. 1996. Effects of bull age on conception dates and pregnancy rates of cow elk. *Journal of Wildlife Management* 60:508–517.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management.* 66 (4):1,301–7.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Saether, B.-E., and M. Heim. 1993. Ecological correlates of individual variation in age at maturity in female moose (*Alces alces*): The effects of environmental variability. *Journal of Animal Ecology*. 62:482–9.

Saether, B.-E., E. J. Solberg, and M. Heim. 2003. Effects of altering sex ratio structure on the demography of an isolated moose population. *Journal of Wildlife Management*. 67:455–66.

Solberg, E. G., and B.-E. Saether. 1994. Male traits as life-history variables: Annual variation in body mass and antler size in moose (*Alces alces*). *Journal of Mammalogy*. 75:1,069–79.

Solberg, E. J., T. H. Ringsby, B.-E. Saether, and M. Heim. 2002. Biased adult sex ratio can affect fecundity in primipareous moose. *Wildlife Biology.* 8:117–28.

Squibb, R. C. 1985. Mating success of yearling and older bull elk. *Journal of Wildlife Management.* 49:744–50.

Trainer, C. E. 1971. *The relationship of physical condition and fertility of female Roosevelt elk (*Cervus canadensis roosevelti*) in Oregon.* M.S. thesis, Oregon State University, Corvallis.

Wisdom, M. J., J. G. Cook, M. M. Rowland, and J. H. Noyes. 1993. *Protocols for care and handling of deer and elk at the Starkey Experimental Forest and Range, general technical report PNW-GTR-311.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004. Effects of off-road recreation on mule deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:531–50.

Nutritional Condition Indices for Elk: The Good (and Less Good), the Bad and the Ugly

Rachel C. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Dennis L. Murray
Trent University, Departments of Biology
and Environmental Resource Studies
Peterborough, Ontario

Pete Zager
Idaho Department of Fish and Game
Lewiston

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Michael W. Gratson (deceased)
Idaho Department of Fish and Game
Lewiston

Introduction

Research on captive ruminants has clearly established the role of nutrition on virtually all aspects of individual and herd productivity, but assessment of nutritional effects on population dynamics of free-ranging ungulates is rare. Understanding influences of nutrition on wild ungulate herd demographics has been limited by a lack of practical, reliable and cost-effective techniques for monitoring elk condition and nutrition (Cook 2002). Assessing nutritional quality of forage is difficult, unreliable and expensive; whereas, assessing nutritional condition of animals has been impractical in the field, inaccurate or inadequately tested (Robbins 1983, Harder and Kirkpatrick 1994, Saltz et al. 1995).

The most rigorous approach to test the value of condition indices involves comparing various indices to actual fat and protein levels of the homogenized carcass. Statistical analysis of indices generally involved correlation between the indices and a body component, usually ingesta-free body fat. Nonlinear relationships often were transformed to facilitate analysis using general linear models (e. g. , Finger et al. 1981, Watkins et al. 1991). The final value of indices usually was determined via comparison of correlation coefficients or coefficients of determination. However, these methods of analysis often are incomplete, leaving unanswered questions related to reliability, sensitivity and applicability of such indices across space and time.

Moreover, past studies often failed to distinguish between indices appropriate for nutritional assessment versus those appropriate for evaluating nutritional condition. Nutrition is defined as the rate of ingestion of assimilable energy and nutrients, and nutritional condition is the state of body components (e. g., fat, protein), which in turn influence an animal's future fitness (Harder and Kirkpatrick 1994).

Cook et al. (2001a) used captive-raised cow elk (*Cervus elaphus nelsoni*) fed a variety of diets to induce a wide range of body conditions to develop predictive models of body fat. For live animals, they assessed serum and urine chemistry, a body condition score, thickness of subcutaneous rump fat and bioelectrical impedance analysis. For dead animals, they assessed femur and mandible fat, two carcass scoring techniques and three different kidney fat indices. They assessed relations between indices and percent fat and developed models to predict nutritional condition. Cook et al. (2001b) also evaluated range of usefulness, bias, precision and sensitivity to small changes in body condition for each model deemed to be most useful by standard methods. Herein we summarize these findings and report major results.

Methods

Seventeen 1.5-year-old, 19 2.5-year-old and 7 adult (5- or 7-year-old), nongravid cow elk were housed in pens near Kamela, Oregon. These elk originated from wild stocks in northeast Oregon and were either bottle- or dam-raised in captivity (Cook, J. G. et al. 2004). Pens were devoid of vegetation and contained a barn designed for individual feeding and collection of blood, urine and fecal samples.

Within each age group, we randomly assigned elk to one of three processing dates, mid-September, late-December or mid-March, which corresponded with times that managers most often handle ungulates in the wild. Beginning 2.5 months before each processing date, we subdivided animals within each age group into three nutrition treatments to create divergent condition levels. Dietary manipulation involved varying quantity and quality of food rations using alfalfa, mixed-grass hays and pellets (Cook et al. 2001a). A high nutrition diet was designed to maintain high nutritional condition during the 2.5-month feeding trial. We formulated medium and low diets to induce average body-mass losses of 8 to 10 percent and greater than 15 percent. We placed all animals on identical diets 7 days prior to data collection to alleviate potential confounding from effects of short-term nutritional influences on relations between condition indices and percent fat. We fed all elk a 35:65 ratio of high-quality pellets and high-quality hay, respectively, at a maintenance level.

At the end of each 7-day period, we collected urine samples via a galvanized metal pan placed under each stall's floor. The next day, animals were brought into the barn and anesthetized with xylazine hydrochloride. Blood samples were obtained within 10 minutes of the drug injection by jugular venipuncture. Seven urine and 23 serum variables were included in the analysis (Cook et al. 2001a).

We collected all live-animal measurements while elk were anesthetized. We used a body condition scoring (BCS) system, which averaged three separate scores derived from palpation of the ribs, withers and rump areas (scoring criteria described by Cook 2000). We calculated a body reserve index (BCS x body mass, Gerhardt et al. 1996) and followed methods of Farley and Robbins (1994) for bioelectrical impedance analysis. Subcutaneous rump fat thickness (MAXFAT) was measured using ultrasonography (Stephenson et al. 1998).

Elk were then euthanized via jugular injection of sodium pentobarbital. Carcass fat, musculature and visceral fat were visually scored via the Kistner score (Kistner et al. 1980) and the Wyoming Index (Lanka and Emmerich 1996), both of which were developed for deer (*Odocoileus* spp.) and modified for elk by Cook et al. (2001a).

Cows were eviscerated and weighed. We halved each carcass from nose to tail along the vertebrae (Stephenson et al. 1998). One half of the carcass, along with the hide and hair, was sectioned, stored at minus 2 degrees Fahrenheit (-20° C) and later homogenized to determine body composition. The other half was used as a source for collection of the femur and the mandible for bone marrow analysis.

We trimmed perineal fat, according to Riney (1955), and we weighed the kidneys, remaining fat and trimmed fat. We evaluated total kidney fat mass and calculated kidney fat indices (KFI) based on total fat mass (KFI_{full}) and trimmed fat mass (KFI_{trim}). For all analyses, we estimated KFIs separately for each kidney and calculated the average. We combined the removed organs and blood with the trachea, larynx, diaphragm, esophagus and all contents of the pleural and peritoneal cavities, exclusive of the ingesta; we weighed and stored them frozen.

Half carcasses and viscera were homogenized separately in a whole-body grinder (Autio 801 B with a Falk 50 hp grinder) at the University of California. We collected two samples of each ground tissue and stored them frozen until chemical analysis of fat, protein, ash and water content (see Cook et al. 2001a for assay descriptions). Body components were converted to a whole body, ingesta-free basis for subsequent analyses.

Statistical Analysis

Data were linearly transformed when necessary, and all indices with a coefficient of determination less than 0.25 with body fat were excluded from further analyses. We then assessed influences of age and season on relationships between each index and ingesta-free body fat using analysis of covariance (ANCOVA) (general linear model procedure [PROC GLM], SAS Institute 1988) with season and age as covariates. This identified the need for separate equations for each level of the two factors. If not significant, one equation was generated across all ages or seasons.

Second, we created two single-variable indices from arithmetic combinations of two original indices, each with different ranges of predictive ability. As described for deer (Connolly 1981), we combined femur marrow fat and KFI to form the CONINDEX. We combined the variables BCS and MAXFAT in a similar manner to produce a new variable: LIVINDEX (Cook et al. 2001a). We also combined MAXFAT and only the rump portion of the BCS in the same manner to produce rLIVINDEX. We then developed single variable predictive models for ingesta-free body fat with an accompanying coefficient of determination.

Evaluation of range of usefulness and sensitivity. To address some of the criticisms of past validation studies (see Robbins 1983, Hobbs 1987, Cederlund et al. 1989, Harder and Kirkpatrick 1994), we evaluated usefulness of 18 models having a coefficient of determination greater than or equal to 0.70 (Cook et al. 2001b). Our evaluation consisted of two types of analyses reflecting criteria of Robbins (1983) and Hobbs (1987): (1) a range of usefulness evaluation to identify specific types of relations between indices and body fat (i. e., identify biological relations that provide insights for what levels of condition the models apply); (2) an analysis of model sensitivity to test variation in the index relative to variation in the dependent variable.

To compare the range of usefulness among indices (i. e., identify biological relations that provide insights for what levels of condition the models apply), we graphed levels of fat with a depletion ratio (DR) of each index. We inserted percent fat (*y*-values) into each single-variable predictive equation for each index (see Figure 1 for general equations) and solved for *x*, providing index values we refer to as IN. DR was then calculated as:

- DR = (IN - LV)(HI - LV), where
- IN = value of index for any given level of fat
- LV = value of index at lowest value of fat in our data set (1.5 percent fat)
- HI = value of index at highest value of fat in our data set (19 percent fat).

This equation standardized the depletion ratios across indices, with one being the highest value attained for that particular index for the range of condition found in this study (no depletion) and zero being the lowest value attained for that index (complete depletion). Differences in depletion patterns among indices were then compared graphically. Steep slopes represented hypersensitivity (i. e., large changes in the index relative to small changes in condition); shallow slopes represented hyposensitivity (i. e., small changes in the index relative to large changes in condition); a slope of zero indicated no predictive capability.

Next, we compared variation associated with the indices relative to variation in the dependent variable (percent fat) for this set of models. We wanted to determine whether a seemingly good predictive model generated from data with a large range of condition (e. g., among seasons) could accurately assess

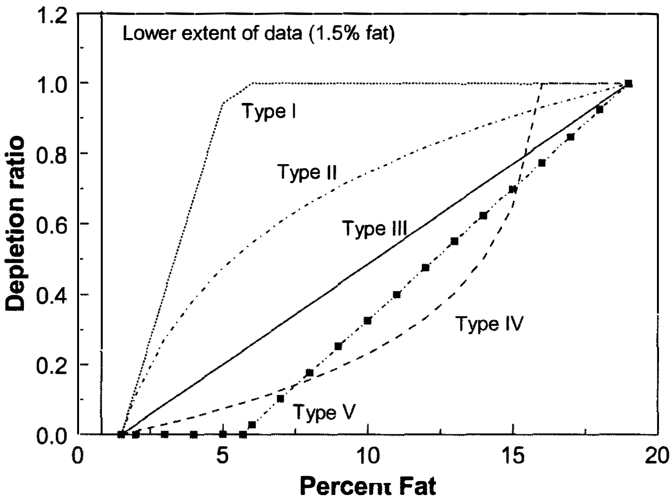


Figure 1. Depletion patterns of condition indices evaluated in the elk body composition study, 1998 to 1999. Each curve represents a different type of relation to body fat: Type I, almost asymptotic, Type II, power ($y = ab^x$), Type III, linear ($y = bx + a$), Type IV, logarithmic ($y = a[1 - e^{-bx}]$) and Type V, linear but truncated. Depletion ratios were standardized across indices (1.0 was the highest value attained for that index for the range of condition in this study [no depletion], and 0.0 was the lowest value attained for that index for the range of condition in this study [complete depletion]). Although we presented the curves with actual fat values, they should be used as relative values only. Individual curves vary due to different equation coefficients.

condition within smaller ranges typically found within seasons (see Hobbs 1987). We estimated within-season range of fat levels of wild elk to be 7 percentage-points from condition data collected during early November and late March (1998, 1999) from an elk herd in the Cascade Mountains, near Enumclaw, Washington. We randomly selected 26 subsets from our captive elk data, each with a 7 percentage-point range of body fat, and regressed percent fat on the index for each subset of data. Model performance was based on the average coefficient of determination of the 26 regressions and the percent of them that had 95 percent confidence levels that did not overlap zero.

Results

Total body fat ranged from 1.6 to 19.0 percent, and protein ranged from 16.6 to 24.8 percent of the ingesta-free body. Live mass ranged from 297 to 539 pounds (135–245 kg), and mass change ranged from plus 1.0 to minus 21.5 percent across the 2.5-month feeding period. These characteristics fell within ranges found in wild elk populations (Bender et al. undated, Cook, R. et al. 2004).

Of the total 50 single-variable models evaluated, most indices, particularly serum and urine, related poorly to percent fat ($r^2 < 0.25$). Twenty-four indices each accounted for greater than or equal to 25 percent of the variation in body fat. Of these, 12 were significantly correlated with body fat (see Cook et al. 2001a for predictive equations and coefficients of determination). Cow age influenced relations between percent fat and mandibular marrow fat (Figure 2), as did season, on relations between percent fat and two serum variables: insulin-like growth factor-1 and thyroxin (Figure 2).

For live animals, LIVINDEX (calculated from either the whole BCS or only the rump portion) accounted for the most variation in percent fat ($r^2 = 0.90$, Figure 2). Both BCS (using the entire score [$r^2 = 0.87$] or only the rump portion [$r^2 = 0.86$]) and rump fat thickness separately ($r^2 = 0.87$) were highly related to body fat (Figure 2). Body mass alone was poorly related to percent fat ($r^2 = 0.44$, Figure 2) and failed to increase the correlation of BCS when they were combined into a body reserve index ($r^2 = 0.79$). Thyroxine and insulin-like growth factor-1 were the only serum or urinary indices useful to predict body fat ($r^2 \leq 0.82$), but this predictive ability was restricted to early and late winter (Figure 2).

For dead elk, the modified Kistner score ($r^2 = 0.92$, Figure 2) and the Kistner subset score ($r^2 = 0.90$) using only the heart, pericardium and kidney scores (Figure 2) were most related to percent fat. The Wyoming index was moderately related to body fat ($r^2 = 0.69$, Figure 2) but is limited in use to when subcutaneous fat is present. Kidney fat mass ($r^2 = 0.86$, Figure 2) alone was superior to KFI_{full} ($r^2 = 0.77$, Figure 2), and KFI_{full} was superior to the traditional method of trimming ($r^2 = 0.74$, KFI_{trim}). Although CONINDEX worked moderately well ($r^2 = 0.70$), it was linear only at low to moderate levels of condition (less than 12.5 percent fat, Figure 2) but had no predictive ability at higher levels of condition. Femur marrow fat produced an r^2 of 0.89 using an inverse transformation of the dependent variable ($-1/y$). Mandible marrow fat was less curvilinear, but our estimate of the relationship for adults is tentative because of the confounding effect of age (Figure 2).

Range of Usefulness

We observed five types of depletion patterns (Figure 1), each with substantial differences in range of usefulness. Type I was derived from an almost asymptotic relation (femur marrow fat). Type II was derived from a power relation (mandibular marrow fat). Type III was derived from a linear relation

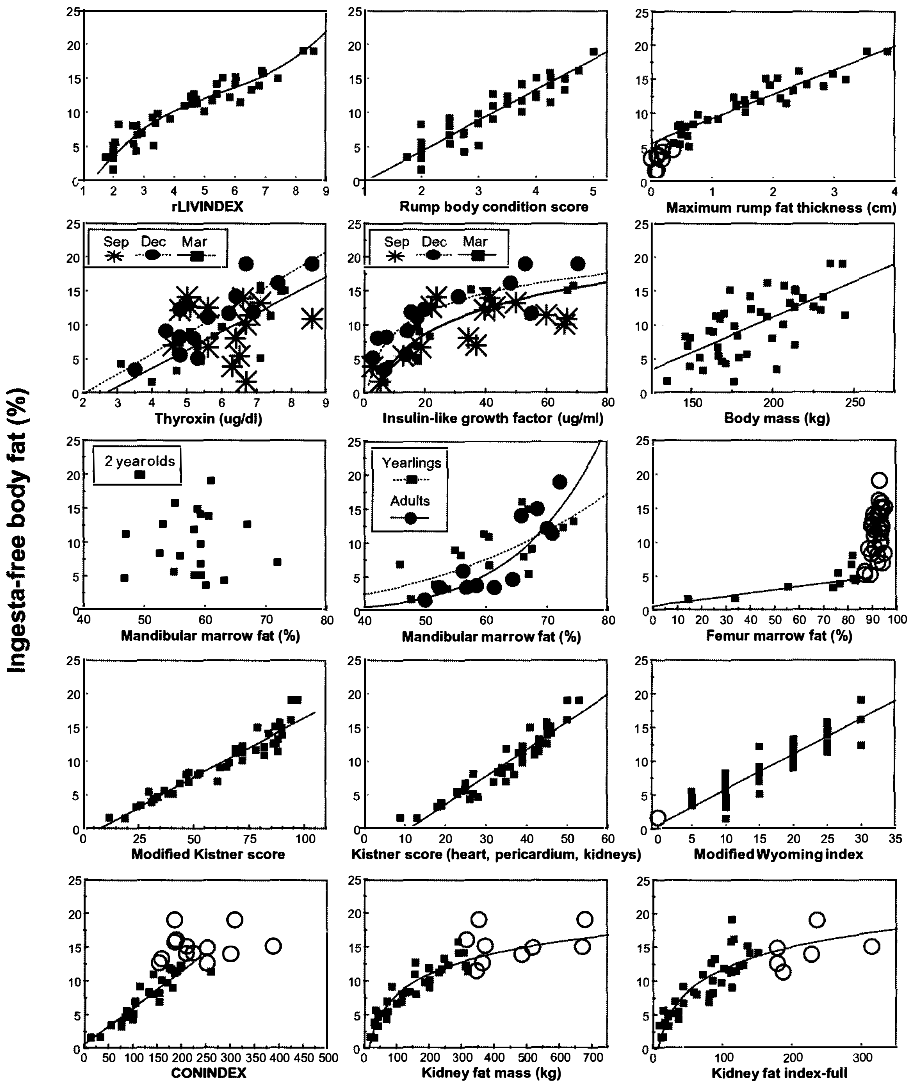


Figure 2. Relations of 14 nutritional condition indices with total body fat (percent) for 43 captive-raised cow elk. Seasonal or age trends are shown for thyroxin, insulin-like growth factor and mandibular marrow fat. Open circles represent where an index loses predictive ability (e. g., maximum rump fat thickness, femur marrow fat, Wyoming Index, CONINDEX and kidney fat indices).

(body condition scores, LIVINDEX, rLIVINDEX, Kistner scores, body reserve index, body mass, thyroxin). Type IV was derived from a logarithmic relation (kidney fat indices, insulin-like growth factor-1). Type V was derived from a linear relation but with an abruptly truncated range of usefulness (rump fat thickness, Wyoming index).

Model Sensitivity

When restricted to within-season ranges of condition, coefficients of determination generally were lower than for among-season ranges (Figure 3). Body condition score (Model 2), rump BCS (Model 3), MAXFAT (Model 4), LIVINDEX (Model 5), rLIVINDEX (Model 6), Kistner score (Model 10), Kistner subset score (Model 11) and kidney fat mass (Model 13) were significantly related to body fat ($P \leq 0.05$) for greater than 80 percent of the 26 data subsets. However, body mass (Model 1), body reserve index (Model 7), insulin-like growth factor-1 (Model 8), thyroxine (Model 9), Wyoming Index (Model 12), KFI_{full} (Model 14), KFI_{trim} (Model 15), femur marrow fat (Model 16), and CONINDEX (Model 17) were significantly related to body fat ($P \leq 0.05$) for less than or equal to 80 percent of the 26 data subsets. Body mass (Model 1), insulin-like growth factor-1 (Model 8), thyroxin (Model 9) and femur marrow fat (Model 16) were markedly insensitive; they were significantly related to body fat for less than 50 percent of the 26 data subsets.

Discussion

Past studies evaluating nutritional condition indices for ungulates have rarely addressed issues of reliability, sensitivity and applicability across space and time. Many have referenced these assessment limitations (e. g., Robbins 1983, Hobbs 1987, Cederlund et al. 1989, Harder and Kirkpatrick 1994) and have offered criteria that should be used to evaluate the value of an index (Robbins 1983, Hobbs 1987). Useful indices of condition should: (1) be linearly related to condition over the entire range of condition (Robbins 1983); (2) be insensitive to a variety of confounding influences such that specific relations developed for one area, time or diet are applicable to others without bias (Robbins 1983, Hobbs 1987); (3) share a biological relation with condition rather than just a significant statistical correlation (Robbins 1983); (4) exhibit low to moderate variation relative to the variation in the dependent variable (Hobbs 1987) and (5) be reasonably practical for field application. By using these criteria, our analyses indicate that many indices that were significantly related to percent fat exhibited

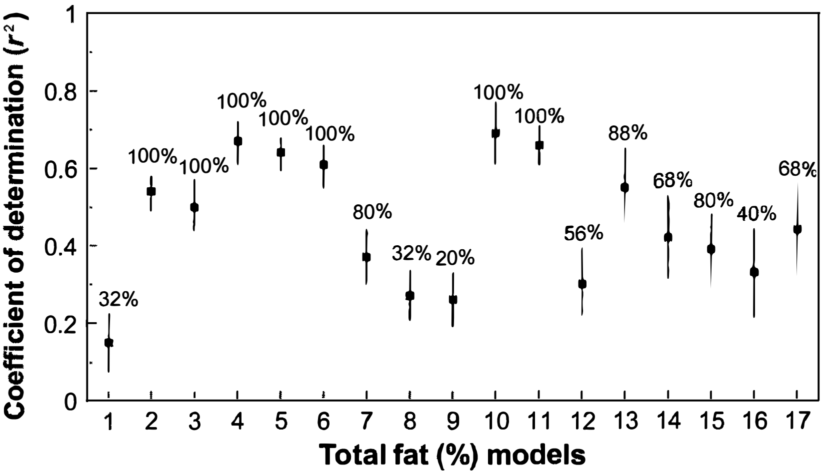


Figure 3. Sensitivity of 17 elk condition models was evaluated by subjecting each to 26 regressions within a restricted range of condition (7 percentage points of body fat representing within-season variation of condition of wild elk in western Washington; unpublished data, n = 50). The average coefficient of determination (± SE) and the percent of time the model was significant over the seven-point ranges are presented. Models used were (1) body mass; (2) body condition score; (3) rump body condition score; (4) maximum subcutaneous rump fat thickness; (5) an arithmetic combination of body condition score and maximum subcutaneous rump fat thickness (LIVINDEX); (6) an arithmetic combination of the rump body condition score and maximum subcutaneous rump fat thickness (rLIVINDEX); (7) body reserve index; (8) insulin-like growth factor-1; (9) thyroxine; (10) Kistner score; (11) Kistner subset score (heart, pericardium and kidney scores); (12) Wyoming index; (13) kidney fat mass; (14) kidney fat index$_{full}$; (15) kidney fat index$_{trim}$; (16) femur marrow fat; and (17) an arithmetic combination of kidney fat index$_{full}$ and femur marrow fat (CONINDEX).

nonlinear relations and were insensitive to small changes in condition. These indices often are those most utilized in the field today.

Range of usefulness generally is a function of linearity of relations between indices and nutritional condition. Transforming such data to make them linear is a common approach that facilitates analysis with linear statistical models. However, attempting to produce good statistical fit via this approach masks important biological attributes and shortcomings of indices (Robbins 1983:222). Nonlinearity of many condition indices often results from sequential patterns of fat mobilization across various areas of the body (Harder and Kirkpatrick 1994). As animals decline in condition, fat mobilization is believed to occur in subcutaneous depots first, viscera—including the kidneys—next and finally in the marrow (Cederlund et al. 1989).

The different type curves of Figure 1 generally reflect this sequence of fat mobilization and, in turn, identify patterns of range of usefulness. Indices exhibiting Types I and II curves have little sensitivity at high levels of condition and probably are of value only in winter and spring. Indices with Type V curves are marginally useful at low levels of condition and probably are of value only in summer and autumn. Indices with Type IV curves are most valuable at moderate levels of condition, and optimum season of use will depend on fat characteristics of the herd. Indices that are linear across the entire range of condition (Type III) greatly facilitate comparisons among herds, among seasons and across time. This analysis indicated that range of usefulness of indices based on only one fat depot will be limited to some extent, and that range of usefulness will be greatest for indices that include measurements of more than one fat depot or muscle.

Our sensitivity analysis revealed that models with even small differences in coefficients of determination differed in their ability to predict across within-season ranges of percent fat (Figure 3). In general, indices with moderate relations to body fat, curvilinear relations or indices based on a relatively small number of categories provided poor predictive capability when restricted in this manner. With these conclusions in mind, we rank the condition indices available to biologists from good to ugly.

The Good

This category includes indices that can be used with high precision across all seasons, ages and ranges of condition. For live animals, rLIVINDEX (displaying a slightly curvilinear relation, Figure 2) was the most correlated to percent fat of any live animal index. Combining the rump BCS and rump fat thickness reduces potential subjectivity over moderate and high levels of condition where rump fat is more effective, and it relies solely on BCS only on the low end of condition (the range where BCS appears to be least subjective; Cook, unpublished data 2000). Despite its precision, two potential drawbacks may limit the use of this technique: (1) extensive training is necessary for both the ultrasound technique and the body condition score to ensure consistency in data collection, and (2) it may be expensive if live animal capture has to be done with helicopters.

Carcass and musculature scores have produced strong correlations to condition, particularly the Kistner score (Kistner et al. 1980, Watkins et al. 1991). We also found a tight linear relationship between the modified Kistner score and percent fat. In addition, by using only the heart, pericardium and kidney scores (the most easily attainable and identifiable parts of the Kistner score), we were able to predict percent fat almost as well as the whole score across the entire range of condition.

The Less than Good

This category includes indices that can be reliably used only for limited ranges of body fat. Thus they may be of poor or no value during certain seasons or even for certain herds in any season if their fat levels exist outside the range of usefulness. Kidney fat indices have a long history of use in elk studies (e. g., Trainer 1971, Kohlmann 1999) despite a well-recognized nonlinear relation to body fat (e. g., Finger et al. 1981, Depperschmidt et al. 1987). Problems with prediction typically were believed to be mostly at the low end of condition (Harder and Kirkpatrick 1994). However, our data indicate that kidney fat indices also are of marginal value above moderate levels of condition (greater than 13 percent body fat). In addition, though kidney fat indices result from quantitative measurements of fat mass (versus more subjective visual scores, e. g., Kistner scores), consistency and accuracy may be compromised, particularly when samples are collected by untrained personnel or hunters, because complete removal of only the fat associated with kidneys can be subjective.

Combining femur fat and kidney fat indices into the CONINDEX may correct for poor predictive ability of kidney fat at low levels of condition for deer (Connolly 1981), but this index failed to predict higher levels of body condition of elk accurately (Figure 2).

The subcutaneous rump fat index covered the greatest range and the highest range in condition (more than 6 to 19 percent body fat) of the fat-based indices we examined, supporting the contention that the last depot of fat accretion is subcutaneous. Unlike the other fat indices, rump fat thickness declines linearly as body fat decreases until it is depleted, and, thus, its range of usefulness is two to three times greater. Even so, the value of this index may be limited, particularly during winter and spring.

The Bad

This category includes indices that can be used as measures of condition, but have the most restricted range of usefulness or display season and age effects. Femur fat was the most nonlinear index we assessed. It demonstrated good predictive capability when body fat was below 6 percent, but it had no predictive power above this value. However, when we did a linear transformation, femur fat had one of the highest correlations with body fat of any index we assessed, illustrating vividly the danger of transformations to enhance statistical relations. Mandible fat has been offered as an easy-to-collect alternative to femur fat (Harder and Kirkpatrick 1994). Although mandible fat appears somewhat more linear, suggesting a greater range of usefulness, significant age effects and lower correlations cast doubt about its value for elk (Figure 2).

The Ugly

We found serum and urine indices to be of little value for assessing nutritional condition. Only thyroxin and insulin-like growth factor-1 produced significant correlations with body components, but both were greatly limited when restricted to evaluating small changes in body condition. They also displayed seasonal variations that limited their use as well (Figure 2). We suspect many of these serum and urine indices are rate variables, or more reflective of short-term nutritional status. Nutritional condition is a state variable, and it cannot be measured in terms of rates (Saltz et al. 1995).

Research and Management Implications

Despite over four decades of intensive research on western elk herds, very little is known about the nature and extent of nutrition's influences on most wild elk populations. Elk populations were generally increasing in much of their range during this period (Christensen et al. 1999), which probably helped foster a perception that nutrition was not very limiting (Cook, J. G. et al. 2004). Declines in productivity and population numbers in elk herds, a phenomenon that is becoming increasingly severe and widespread in the Northwest (Johnson et al. 2004), suggest that a new era with different challenges await elk biologists. Of those habitat attributes with the potential to influence productivity and demographics of large herbivore populations, nutrition is probably the most important (Parker et al. 1999, Cook, J. G. et al. 2004). Clearly, a better understanding of how nutrition influences elk populations is needed.

Suitable tools available to biologists for evaluating nutritional adequacy of habitat are limited (Harder and Kirkpatrick 1994, Cook 2002). General surveys of forage abundance and quality across landscapes are expensive and difficult to conduct, and interpreting their relevance to herd performance is problematic. A long history of serum and urine analyses has identified techniques that have potential at least for monitoring short-term nutritional status, but even the best and most studied of these remain controversial (Cook 2002). In contrast, estimates of nutritional condition are compelling because they reflect cumulative energy balance of the animal (thereby integrating the separate effects of nutritional adequacy of their environment with their nutrient demands) and because condition can strongly influence reproductive success and survival probability (Cook, J. G. et al. 2004). However, direct estimation of nutritional condition, either via dilution techniques (e. g. , Torbit et al. 1985) or laboratory assays of samples from homogenized carcasses, is largely impractical for evaluations of free-ranging animals. Various condition indices potentially offer a solution to problems of practicality, but our data indicate many pitfalls with their indiscriminate use, and those indices that require dead animals can limit research and monitoring designs and are increasingly less acceptable to society.

Our data identify several new techniques and infrequently used older techniques that are sensitive across wide ranges of nutritional condition, are robust across animal age and season, and are reasonably practical. The rLIVINDEX index, which combines a body condition score with ultrasound rump-fat measurements for live animals and the Kistner scores for dead animals, proved superior in our analysis. These in particular can help to open the door to a variety of research designs useful for evaluating nutrition's effect on populations. Foremost among potential applications may be an initial screening to evaluate the need for more detailed and expensive nutritional evaluations. Live-animal indices also provide opportunities for monitoring nutritional status among unhunted herds during seasons in which hunting is precluded or among unhunted segments (e. g., females) of populations. Thresholds linking nutritional condition with animal performance have now been developed for elk (Cook, J. G. et al. 2004) that provide criteria useful for relating animal condition to performance of elk cows, yearlings and calves. Additional applications for nutritional condition data on wild or captive elk populations include evaluations of: (1) relative degree of limitations among seasons and ranges by taking repeated measurements on the same animal across the yearly cycle, (2) relations between condition and productivity (Cook, J. G. et al. 2004), (3) predisposition to predation and starvation, (4) wildlife-habitat relations that are animal-productivity explicit (e. g., relations between home range characteristics and nutritional condition), (5) top-down versus bottom-up contributions to population dynamics and trends, and (6) the potential for herd augmentation (Bender et al. undated). Finally, modeling of carrying capacity and simulation modeling of population dynamics may require or benefit from estimates of nutritional condition (e. g., Hobbs et al. 1982, Hobbs 1989, DelGiudice et al. 2001).

Many of the above applications rely on use of live-animal indices for collecting nutritional condition data because they require flexible collection dates and, in some cases, sequential data on individual animals. In many situations, acquiring such data will require expensive helicopter time. Thus, the nature and extent of data required to address key management issues must be carefully considered in the context of costs of collecting data and interpretive value and in the context of reliability associated with each potential condition index and various research designs. Reduced costs up front are of little value if the data collected fail to address important issues. As stated by Hobbs (1987), the reliability should always precede applicability because unreliable predictions can be misleading.

A significant hurdle for using condition results of past studies emanates from the multitude of techniques used and, in particular, the units associated with each. For example, it is nearly impossible to link condition results from studies reporting kidney fat indices to those reporting body condition scores or femur fat and so on. Reporting nutritional condition in standard units whenever possible would greatly facilitate comparisons, and we suggest percent body fat of the ingesta-free body (Farley and Robbins 1994, Stephenson et al. 1998). For elk, our study provides equations (Cook et al. 2001a) to convert measures of nutritional condition from a variety of indices into estimates of percent body fat (though always accounting for limitations associated with each index). These equations facilitate comparisons among elk studies, and they provide a means to standardize data from long-term historical trends where different techniques were used or where changes to new and better techniques are being considered.

Reference List

Bender, L. C., M. A. Davison, J. G. Cook, R. C. Cook, and P. B. Hall. Undated. *Feasibility of population augmentation as a management strategy for the Nooksack elk herd, Washington. Final report*. Las Cruces, New Mexico: U. S. Geological Survey.

Cederlund, G. N., R. J. Bergstrom, and K. Danelli. 1989. Seasonal variation in mandible marrow fat in moose. *Journal of Wildlife Management*. 53:587–92.

Christensen, A. G., J. K. Lackey, and J. A. Rochelle. 1999. Elk in western forests: Too much success? *Transactions of the North American Wildlife and Natural Resources Conference*. 64:107–16.

Connolly, G. E. 1981. Assessing populations. In *Mule and black-tailed deer of North America*. ed. O. C. Wallmo, 287–345. Lincoln, Nebraska: University of Nebraska Press.

Cook, J. G. 2002. Nutrition and food habits. In *Elk of North America: Ecology and management, 2nd edition*, eds. , D. E. Toweill, and J. W. Thomas, 259–350. Washington, DC: Smithsonian Institution Press.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Summer-autumn nutrition and birth date influence on reproduction and survival of elk. *Wildlife Monographs*. 155:1–61.

Cook, R. C. 2000. *Studies of body condition and reproductive physiology in Rocky Mountain elk*. M. S. thesis, Moscow, Idaho: University of Idaho.

Cook, R. C., J. G. Cook, D. L. Murray, P. Zager, B. K. Johnson, and M. W. Gratson. 2001a. Development of predictive models of nutritional condition for Rocky Mountain elk. *Journal of Wildlife Management*. 65:973–87.

———. 2001b. Nutritional condition models for elk: Which are the most sensitive, accurate, and precise? *Journal of Wildlife Management*. 65:988–97.

Cook, R. C., J. G. Cook, and L. D. Mech. 2004. Nutritional condition of northern Yellowstone elk. *Journal of Mammalogy*. 85:714–22.

DelGiudice, G. D., R. A. Moen, F. J. Singer, and M. R. Riggs. 2001. Winter nutritional restriction and simulated body condition of Yellowstone elk and bison before and after the fires of 1988. *Wildlife Monographs*. 147:1–60.

Depperschmidt, J. D., S. C. Torbit, A. W. Alldredge, and R. D. Deblinger. 1987. Body condition indices for starved pronghorn. *Journal of Wildlife Management*. 51:675–8.

Farley, S. D., and C. T. Robbins. 1994. Development of two methods to estimate body composition of bears. *Canadian Journal of Zoology*. 72:220–6.

Finger, S. E., L. Brisbin, Jr., and M. H. Smith. 1981. Kidney fat as a predictor of body condition in white-tailed deer. *Journal of Wildlife Management*. 45:964–8.

Gerhardt, K. L., R. G. White, R. D. Cameron, and D. E. Russell. 1996. Estimating fat content of caribou from body condition scores. *Journal of Wildlife Management*. 60:713–18.

Harder, J. D., and R. L. Kirkpatrick. 1994. Physiological methods in wildlife research. In *Research and management techniques for wildlife and habitats, 5th edition*, ed., T. A. Bookhout, 275–306. Bethesda, MD: The Wildlife Society.

Hobbs, N. T., D. L. Baker, J. E. Ellis, D. M. Swift, and R. A. Green. 1982. Energy- and nitrogen-based estimates of elk winter range carrying capacity. *Journal of Wildlife Management*. 46:12–21.

Hobbs, N. T. 1987. Fecal indices to dietary quality: A critique. *Journal of Wildlife Management*. 51:317–20.

———. 1989. Linking energy balance to survival in mule deer: Development and test of a simulation model. *Wildlife Monographs*. 101:1–39.

Johnson, B. K., M. J. Wisdom, and J. G. Cook. 2004. Issues of elk productivity for research and management. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:551–71.

Kistner, T. P., C. E. Trainer, and N. A. Hartmann. 1980. A field technique for evaluating physical condition of deer. *Wildlife Society Bulletin*. 8:11–7.

Kohlmann, S. G. 1999. Adaptive fetal sex allocation in elk: Evidence and implications. *Journal of Wildlife Management*. 63:1,109–17.

Lanka, R., and J. Emmerich. 1996. *Fat deposition measurements as an index to mule deer body condition and spring/summer habitat conditions*. Cheyenne, Wyoming: Wyoming Game and Fish Department.

Parker, K. L., M. P. Gillingham, T. A. Hanley, and C. T. Robbins. 1999. Energy and protein balance of free-ranging black-tailed deer in a natural forest environment. *Wildlife Monographs*. 143:1–48.

Riney, T. 1955. Evaluating condition of free ranging red deer (*Cervus elaphus*), with special reference to New Zealand. *New Zealand Journal of Science and Technology*. 36:429–63.

Robbins, C. T. 1983. *Wildlife feeding and nutrition, 1st edition*. Orlando, FL: Academic Press.

Saltz, D., G. C. White, and R. M. Bartmann. 1995. Assessing animal condition, nutrition, and stress from urine in the snow: A critical review. *Wildlife Society Bulletin*. 23:694–704.

SAS Institute. 1988. *SAS/STAT user's guide, 6.03 edition*. Cary, North Carolina: SAS Institute.

Stephenson, T. R., K. J. Hundertmark, C. C. Schwartz, and V. Van Ballenberghe. 1998. Predicting body fat and body mass in moose with ultrasonography. *Canadian Journal of Zoology*. 76:717–22.

Torbit, S. C., L. H. Carpenter, D. M. Swift, and A. W. Alldredge. 1985. Mule deer-body composition—A comparison of methods. *Journal of Wildlife Management*. 49:86–91.

Trainer, C. E. 1971. *The relationship of physical condition and fertility of female Roosevelt elk (*Cervus canadensis roosevelti*) in Oregon*. M. S.

Nutrition and Parturition Date Effects on Elk: Potential Implications for Research and Management

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Rachel C. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Robert A. Riggs
Boise Cascade Corporation
La Grande, Oregon

Tim DelCurto
Eastern Oregon Agricultural Research Center
Union

Larry D. Bryant
U. S. Department of Agriculture, Forest Service
Washington, DC

Larry L. Irwin
National Council for Air and Stream Improvement
Stevensville, Montana

Introduction

Understanding and managing those mechanisms that affect population dynamics comprise, perhaps, the most fundamental aspect of wildlife management (Caughley 1977). Biologists generally categorize these mechanisms as either top-down (predator-driven) or bottom-up (habitat- or animal-density driven). Bottom-up influences involve imbalances between increasing animal density and key habitat resources. For large ungulates, abundance and nutritive value of forage are commonly thought to be the primary mediators of bottom-up regulation (Caughley 1979, McCullough 1984). Certainly, nutritional deficiencies can have extensive and often acute effects on reproduction, growth, development and survival (Verme and Ullrey 1984, Cook 2002).

Much research has focused on nutritional effects, particularly for livestock (National Research Council 1984) but also for deer (*Odocoileus* spp.) (Verme and Ullrey 1984, Parker et al. 1999), caribou (*Rangifer tarandus*) (Thomas 1982, Cameron et al. 1993, Cameron 1994) and moose (*Alces alces*) (Schwartz and Renecker 1998). A research emphasis on nutrition has been largely lacking for elk (Cook 2002), except to some extent in Colorado (e. g., Hobbs et al. 1982) and in relation to game-farming in Canada (Haigh and Hudson 1993). Moreover, nutrition has largely been discounted in national forest models used to reconcile elk habitat quality with other land management concerns (Edge et al. 1990). This reflects an apparent perception that nutrition is not a particularly important factor affecting most elk herds. It may also reflect uncertainty regarding how to evaluate nutritional resources across large landscapes in a manner relevant to ungulate populations.

Despite increased attention to nonforage aspects of habitat, such as roads and cover, productivity and population size are declining in a number of northwestern elk herds (Irwin et al. 1994, Gratson and Zager 1999, Ferry et al. 2001). This has spurred a reconsideration of the broader range of factors that may be contributing to herd demographics. Beyond nutritional limitations, low numbers of mature bulls (e. g., less than 5 bulls to 100 cows [Schommer 1991]) was considered a potentially important cause of declines. Noyes et al. (1996, 2002) demonstrated that such low numbers of mature bulls can delay breeding up to 2 or 3 weeks, thereby delaying and desynchronizing parturition. Ultimately, such effects might reduce calf survival due to a variety of proximate causes, primarily through winter mortality and perhaps through predation.

Despite a perception that nutrition on summer range is rarely limiting in the western United States (e. g., Marcum 1975, Wallmo et al. 1977, Lyon 1980, Nelson and Leege 1982, Leege 1984, Christensen et al. 1993, Unsworth et al. 1998), we opted, in our research, to investigate the potential for summer-autumn nutrition to contribute to bottom-up regulation. Nutritional requirements are appreciably elevated to support key summer-autumn life processes, such as lactation, juvenile growth and development, and recovery of mass lost during winter (Verme and Ullrey 1984, Oftedal 1985, Cook 2002). Forage quality and quantity may be greatest during the growing season, but it may nevertheless be insufficient to consistently satisfy high nutritional requirements during summer and autumn (e. g., Julander et al. 1961, Pederson and Harper 1978, Verme and Ullrey 1984, Merrill and Boyce 1991, Crête and Huot 1993, Parker et al. 1999, Alldredge et al. 2002).

This study was conducted from summer 1995 through spring 1998 using a captive herd of 57 cow elk to achieve three primary goals. First, we wanted to estimate the extent to which summer-autumn nutrition and parturition date could each influence reproduction and survival, and we wanted to explore the extent to which these two factors might exert interactive influence. Second, we wanted to quantify the nutritional requirements of lactating cows with calves. Third, we wanted to quantify the relationship between magnitude of nutritional restriction and magnitude of reduction in reproduction and survival.

We tested hypotheses regarding influences of summer-autumn nutrition and birth date, specifically those that pertain to direct effects of summer-autumn nutrition and birth date on reproduction and those that pertain to carry-over effects, across seasons, of nutrition and birth date on subsequent reproduction and survival. Herein, we briefly review findings of the study and discuss their implications.

Study Area and Elk Herd

The study site was 20 miles (30 km) west of La Grande in the Blue Mountains of northeast Oregon in forest zones at 4,200 to 4,400 feet (1,300–1,350 m). Facilities consisted of two pen complexes. The primary complex housed the cows year-round and consisted of six 1.85-acre (0.75-ha) pens. Small barns with stalls were built in each pen and were used for individualized feeding, weighing and handling. A smaller complex was used to hold the calves after they were weaned. It consisted of three 1-acre (0.4-ha) pens that were devoid of vegetation.

We used two cohorts of bottle-raised female elk captured from wild stock in northeast Oregon, the first born in 1991 ($n = 22$) and the second born in

1993 ($n = 35$) (Cook et al. 1996). All bulls used for breeding were from wild stock at the Starkey Experimental Forest and Range and were at least three years old.

Methods

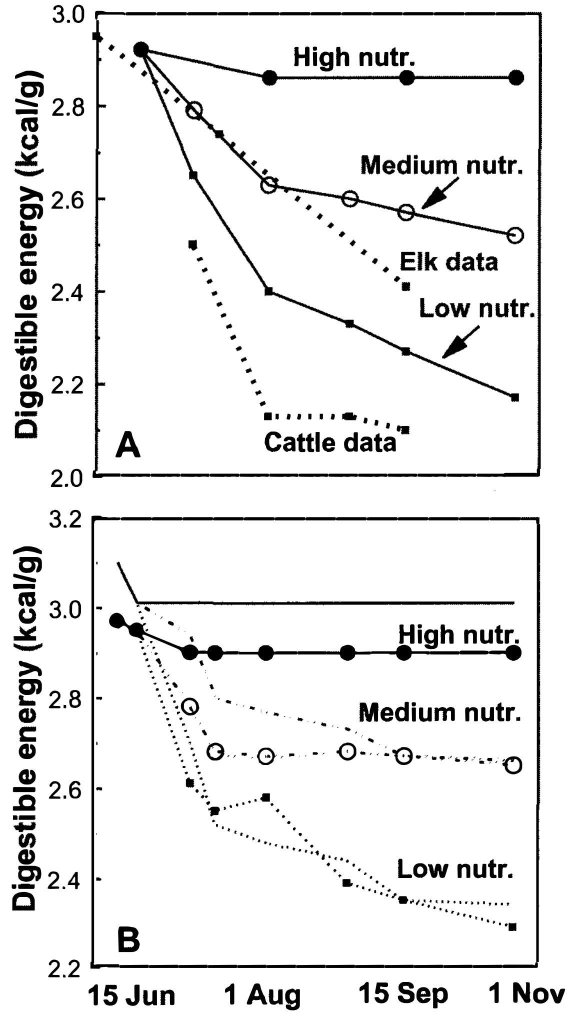
The study consisted of three primary experiments to evaluate relations between nutrition, birth date and reproduction-survival.

1. Cows and their calves were fed three levels of digestible energy (DE) ad libitum (Figure 1) from late June through early November in 1996 and 1997. These levels of nutrition were selected to represent dietary DE levels identified on elk summer ranges in northeast Oregon.
2. Early and late birthing treatments were induced in spring 1996 by placing a bull with half the cows in September and with the other half in October 1995. Early and late parturition was included in the summer experiments of 1997, based on postbirth stratification rather than by inducing breeding dates as during the rut of 1995.
3. Influences of summer-autumn nutrition levels and birth date on winter survival of calves were evaluated by feeding during winter a submaintenance ration, to mimic that likely to be realized under harsh winter conditions, and by monitoring number of days survived from early December through mid-March during winters 1996 to 1997 and 1997 to 1998.

Additionally, we conducted several supplementary experiments.

1. During summer 1997, cows that failed to breed the previous autumn were separated into two nutrition groups, one receiving the high nutrition, the other the low nutrition level, identical to those fed to lactating cows in the primary summer-autumn nutrition experiment of the same year. This provided insights of the potentially different effects of summer nutrition on lactating versus nonlactating cows.
2. From early December through early March 1997 to 1998, all pregnant cows were stratified into three winter nutrition levels, with the intention of comparing winter nutritional condition and late-autumn nutritional condition (i. e., ingesta-free body fat) influences on fetal survival. Results of the experiment also allowed an assessment of winter and late-autumn nutritional condition on cow survival.

Figure 1. In graph A, target digestible energy (DE) content of food offered to cow elk and their calves from late June through early November, 1996 and 1997. Dashed lines labeled "Elk" and "Cattle" show dietary DE levels of elk (J. G. Cook, unpublished data 1995) and cattle (Holechek et al. 1981) determined during dry years at moderate to low elevations in forest zones in the Blue Mountains of northeast Oregon. The average of these two DE levels set the target for the low nutrition treatment group. In graph B, actual DE content of food consumed by cows and calves from late June through early November, 1996 (lines without circles and squares) and 1997 (lines with circles and squares).



Response variables included mass change of cows, calves and yearlings, nutritional condition of cows, indexed by percent fat of the ingesta-free body (determined using the LIVINDEX score [Cook et al. 2001a]), pregnancy rates and timing of breeding of cows and yearlings, number of days of winter survived by calves during the calf winter survival experiments, and survival of cows and their fetus during the cow winter nutrition experiment. The winter experiments were designed to determine survival without allowing animals to die. Threshold criteria were established based on percent mass loss, body temperature, behavior and appearance from which to proclaim the animals "dead." They were then removed from the experiments and provided care and abundant food (a total of 5 elk of 110 died in these trials). During the two winters preceding both summer nutrition experiments, pregnant cows were fed a moderately submaintenance diet to induce about 10 percent body mass (BM) loss, to mimic mass loss similar to that during mild to normal winters. In March of both years, elk were placed on ad libitum diets of high quality food to eliminate nutritional restriction on fetal development during the third trimester.

In all cases, significant results refer to P values less than or equal to 0.05.

Results

Summer-autumn Nutrition, Parturition Date Experiments

The induced breeding periods of autumn 1995 provided 40 pregnant cows with two parturition pulses for the summer experiments: May 26 (range = May 12 to June 10) and June 19 (range = June 11 to June 29). Breeding in autumn 1996 provided 30 pregnant cows for the summer experiments. Postbirth stratification of birth date provided early-late parturition treatments of June 1 (range = May 20 to June 9) and June 20 (range = June 10 to July 8).

In summer-autumn 1996, parturition date and nutrition significantly affected cow BM changes, but to a markedly different degree. At the end of the experiment, cows on the high nutrition treatments averaged 5 and 10 percent heavier, approximately 25 pounds (12 kg) and 50 pounds (23 kg) heavier, than cows in the medium and low nutrition treatments, respectively. The parturition date increment amounted to 2 to 3 percentage points, a mass difference of about 10 pounds (5 kg). Mass dynamics followed a similar trend during the second summer experiment of 1997. Body fat also was significantly related to summer-autumn nutrition levels in both summer experiments, but was unrelated to parturition date (Figure 2). The marked effects of the high versus low nutrition levels evident for lactating cows was not evident for nonlactating cows fed the same two dietary levels (Figure 2), indicating a substantial interaction between lactation status and summer-autumn nutrition on body fat accretion in cow elk.

The low nutrition treatment effectively precluded pregnancy by most cows (about 80 percent of these cows failed to become pregnant), and the medium nutrition treatment significantly delayed timing of breeding. Parturition date failed to influence either pregnancy or timing of breeding in either year

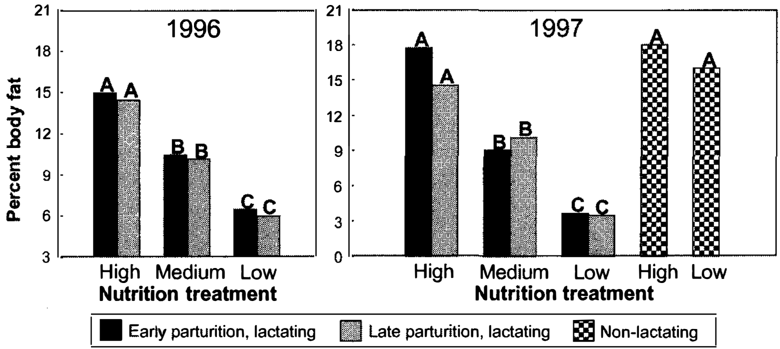


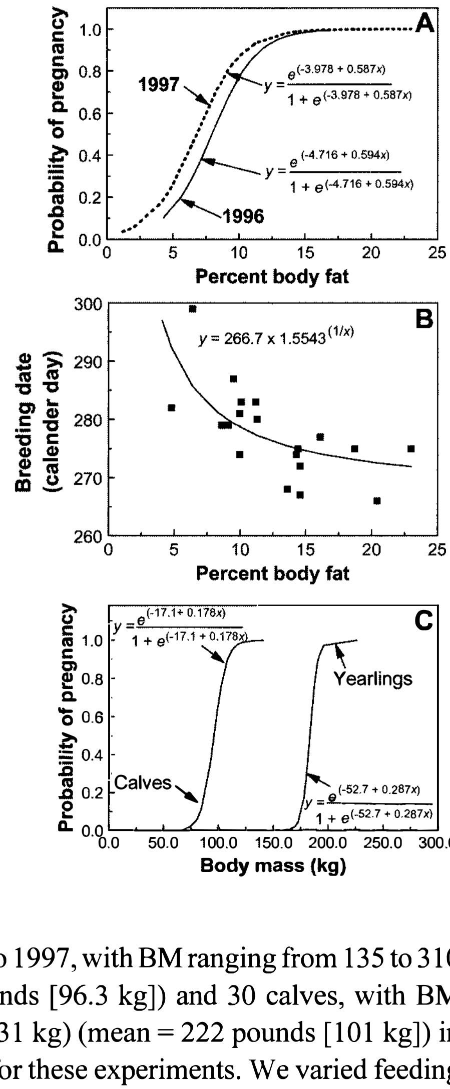
Figure 2. October ingesta-free body fat levels of cow elk across three levels of summer-autumn nutrition and two levels of parturition date. Within years, vertical bars with different letters differ significantly

(Figure 3A, B). Pregnancy failure resulted from a failure to enter estrus (Cook et al. 2001b). Body fat during the rut was significantly related to breeding probability and timing of breeding, and it provided a basis for prediction equations for both of these responses (Figure 3C).

Body size of calves by late June, when nutrition treatments were implemented, was significantly inversely correlated to their birth date and positively correlated to their birth mass. Both variables together accounted for about 90 percent of the variation in late-June BM. Results demonstrate the head-start advantage of early birth, and also indicated that daily growth rate of calves larger at birth was greater than that of smaller calves at birth. Growth starting at the time nutrition treatments were implemented in late June was profoundly influenced by nutrition (Figure 4) in both years. Body mass of calves at the end of autumn was a function of summer-autumn nutrition and, inconsistently, their birth date. Late-born calves in the low and medium nutrition groups overcame their late-start disadvantage, catching up with their early-born counterparts in their respective nutrition groups in 1996 (Figure 4A). In 1997, late-born calves in the high nutrition group also caught up with early-born counterparts (Figure 4C).

Pregnancy of yearling cows ($n = 21$) was significantly correlated to both their size as calves the previous autumn (1996) and to their size in autumn 1997 (Figure 3C). All five yearling cows in the previous-year high summer-autumn nutrition group, three of seven in the previous-year medium group and only one of seven in the previous-year low nutrition group bred (i. e., the summer-autumn nutrition treatment these yearlings received when they were calves).

Figure 3. In graph A, logistic relations between ingesta-free body fat during the breeding season and pregnancy probability of adult lactating cows in 1996 and 1997. In graph B, nonlinear relations between body fat during the breeding season and timing of breeding of lactating cows during 1997. In graph C, logistic relations between probability of breeding for yearlings in 1997, based on body mass during the rut in 1997 and body mass when these yearlings were calves the previous autumn (1996).



Calf Winter Survival Experiments

Forty calves in winter of 1996 to 1997, with BM ranging from 135 to 310 pounds (61–140 kg) (mean = 212 pounds [96.3 kg]) and 30 calves, with BM ranging from 125 to 290 pounds (57–131 kg) (mean = 222 pounds [101 kg]) in winter of 1997 to 1998 were available for these experiments. We varied feeding regimes of calves between the two winters. In the first winter, the magnitude of deficiency was increased gradually to about half of maintenance by mid-February. In the second, the magnitude of deficiency was increased relatively abruptly to half of maintenance by early January and held constant through the rest of winter.

In both winters, BM of calves was significantly correlated to the number of days of winter they were able to survive (Figure 5A, C). Smaller calves lost

Figure 4. Body mass of elk calves during summer and autumn 1996 (A, B) and 1997 (C, D) across three levels of summer-autumn nutrition and two periods of birth date. Actual body mass is presented in graphs A and C. In B and D, body mass was adjusted to remove effects of birth date and birth mass, by subtracting mass at the start of the time period (early July) from all subsequent mass estimates. Data values not connected by vertical lines differ significantly within weekly periods.

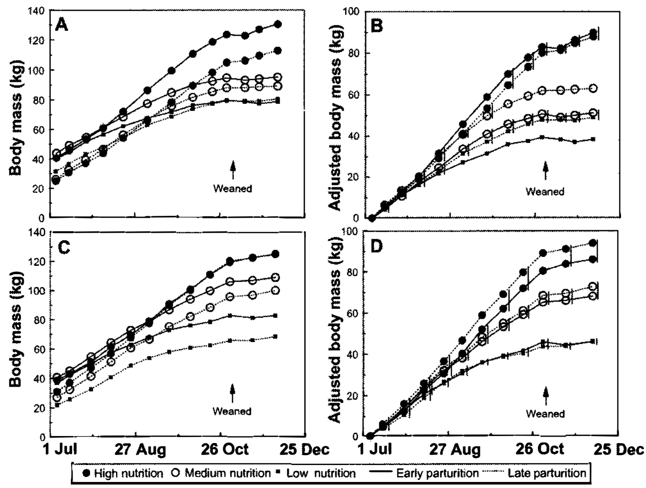
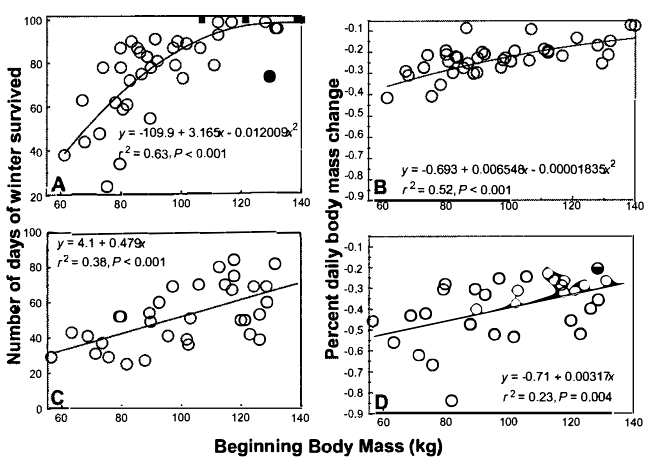


Figure 5. Number of days of winter survived (A, 1996–1997; C, 1997–1998) and rate of body mass loss (B, 1996–1997; D, 1997–1998) of elk calves as a function of their body mass at the start of winter. The four solid squares are data for calves that survived the entire winter experiment, and the solid circle indicates a data point treated as anomalous and excluded from the regression equation (but not the associated statistical parameters).



BM at a relatively faster rate than did larger calves (Figure 5B, D), indicating the mechanism accounting for reduced tolerance of smaller calves to winter undernutrition. Differences in feeding regime between the two winters probably accounted for the different functional responses of calves in the two winters (i.

e., nonlinear in 1996, linear in 1997, Figure 5A versus 5C). Additional analyses indicated that their nutrition level of the previous summer-autumn, but not their birth date, was significantly related to number of days that calves survived in winter (Cook et al. 2004).

Winter Survival of Cows and Fetuses

This experiment was conducted in 1998 with 40 pregnant cows divided equally among 4 treatment groups—high summer-autumn nutrition with high (45 percent of maintenance), moderate (55 percent of maintenance) and low (65 percent of maintenance) winter nutritional deprivation, and medium summer-autumn nutrition with low winter nutritional deprivation.

Summer-autumn nutrition and winter nutritional deprivation significantly influenced end-of-winter BM and fat levels (Figure 6A, B). Body fat of cows on the high winter nutritional deprivation treatment plummeted such that they ended winter with body fat equivalent to that of cows in the medium-summer-nutrition, low-winter-deprivation treatment group, indicating important interactive influences of summer and winter nutrition. One cow died and five particularly emaciated cows were removed from the study to prevent death. Four of these cows were in the high-summer-nutrition, high-winter-deprivation group; the other two were in the medium-summer-nutrition, low-winter-deprivation group. No cows lost their fetus prior to their removal from their study, but two aborted two weeks to two months after their removal and refeeding had been initiated. Both of these were in the medium-summer-nutrition, low-winter-deprivation group. Although limited, these data suggest elk cows may typically die before they abort. The mortality data provided an opportunity to develop a logistic model comparing effects of body fat in late autumn and the winter nutrition levels we implemented on survival probability (Figure 6C). This model probably is specific to our experimental setting and should not be considered robust across a wide variety of winter range conditions.

Discussion

Our results show that relatively small differences in DE content of food consumed by elk in summer and autumn can have very strong effects on fat accretion, timing of conception, probability of pregnancy of lactating cows, calf growth, yearling growth (see Cook et al. 2004) and yearling pregnancy rates.

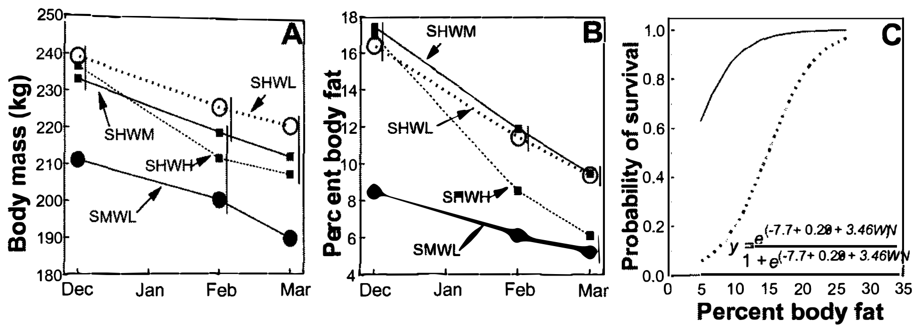


Figure 6. Body mass (A) and ingesta-free body fat (B) of pregnant cow elk during winter (1997–1998) across four summer-autumn and winter nutritional deprivation treatments: SHWL = high summer-autumn nutrition, low winter nutritional deprivation; SHWM = high summer-autumn nutrition, moderate winter nutritional deprivation; SHWH = high summer-autumn nutrition, high winter nutritional deprivation; and SMWL = medium summer-autumn nutrition, low winter nutritional deprivation. Within monthly time steps, data points not connected by vertical lines differ significantly. Data were used to calculate logistic relations (C) between probability of winter survival and late-autumn body fat ($P = 0.073$) at two levels of winter nutrition (WN) deprivation ($P < 0.05$) (low-to-moderate deprivation = solid line; high = dotted line). In the logistic equation, *WN* is an ordinal variable with values of 1 for high winter nutritional deprivation and 2 for low-to-moderate winter nutritional deprivation.

Effects of summer-autumn nutrition on fat accretion of cows and growth of calves significantly influenced their survival probability under harsh winter conditions.

Earlier birth usually resulted in larger size of calves in late autumn, but we were unable to document significant, consistent effects of parturition date on any other reproductive or survival attribute we evaluated. Despite a clear head-start growth advantage for calves due to early birth, our data suggest that delays in parturition, expected due to a very low ratio of mature bulls to cows (two to three weeks [Noyes et al. 1996, 2002]), are of insufficient magnitude to appreciably affect reproduction and survival in elk (see also Bender et al. 2002). To some extent, this finding may reflect accelerated growth of some (but not all) late-born calves that eliminated the head-start advantage of early birth. These faster-growing calves tended to be heavier at birth or were male, both of which contributed to faster growth and both of which occurred more frequently late in the parturition period. Additionally, Cook et al. (2004) reported some evidence indicating that delays in breeding did not necessarily result in similar delays in parturition, and Berger (1992) reported that late-breeding bison (*Bison bison*) in good condition shortened gestation by up to 15 days, synchronizing births with other females. Thus, much can occur to compensate for delayed breeding on the resulting calf crop by the time calves are 6 months old, just prior to their first winter, thereby obscuring the implications of moderately-delayed breeding. But, we note a potentially important caveat. Because our captive animals were not subjected to predation, this study could not evaluate the effect of parturition synchrony on predation-related mortality (see Keech et al. 2000); thus, it may be possible that delayed and desynchronized parturition effects are realized through a predation-mediated mechanism even where calf growth and survival is not constrained by nutrition directly.

That nutrition affects reproduction and survival was hardly surprising, but the extent to which relatively small differences in DE content of food (a maximum of 20 percent between the high and low nutrition levels, Figure 1) induced large differences in animal performance was surprising. This finding confirmed the multiplier-effect noted by White (1983); it occurred due to the combination of reduced DE concentration in food and the effect of reduced food quality on daily dry matter intake, evidently a result of reduced digesta passage rates. Ruminants cannot be expected to compensate appreciably for poor forage quality simply by eating more; they instead eat less as quality declines (see also Minson and Wilson 1994, Grey and Servello 1995). Thus, wild ungulates can be limited by nutritional quality of their forage, even in the face of what might appear to be abundant food quantity (Riggs et al. 1996).

In addition to marked effects on calf growth, probably the greatest effect of summer-autumn nutrition on animal performance during this season was reduction in pregnancy rates. Our data indicated a marked threshold effect of about 9 percent body fat, below which the probability of breeding declines precipitously. However, our study also demonstrated what has been supposed (e. g., Mautz 1978) and remains a suspicion of some (Parker et al. 1999: 38)—that effects of nutritional deficiencies occurring in summer-autumn often appear after summer-autumn, particularly during the subsequent winter and spring if winter conditions are sufficiently harsh. Although observations of winter mortality at first glance may implicate winter conditions as the limiting factor (e. g., the bottleneck of winter range [Wallmo et al. 1977]), the potentially predisposing contributions of summer-autumn nutrition to winter mortality should not be categorically discounted without reliable collaborative data.

Additionally, high pregnancy rates of cows should not necessarily be considered proof that summer-autumn nutrition is adequate and, thus, is of little concern for management. Summer-autumn nutrition that is just adequate to support high pregnancy rates is not necessarily adequate to avoid predisposition of cows to winter starvation, and it is inadequate (1) to support high growth rates of calves and yearlings, (2) to preclude predisposition of calves to winter starvation, and (3) to support high levels of breeding by yearling cows. Moreover, prime-aged animals in herds existing on marginal levels of summer nutrition just adequate to support high pregnancy rates in most years may be small-bodied (see Crête and Huot 1993), and such herds might be prone to substantial year-to-year variation in vital rates due to annual variation in weather. Under these marginal nutritional conditions, how habitats are managed may have particularly important influences on population dynamics.

A goal of this study was to identify nutritional requirements for various levels of performance. Our findings indicate DE levels of at least 82 kilocalories per ounce (2.9 kcal/g) of food over summer into early autumn are required by adult lactating cows for levels of performance approaching the genetic maximum for adult elk. Such a level resulted in daily DE intake of 162 to 172 kilocalories per pound of $BM^{0.75}$ (400–425 kcal/kg of $BM^{0.75}$). This level simultaneously satisfies requirements for lactation and recovery from previous-winter mass loss; although, it (82 kilocalories per ounce, 2.9 kcal/g) might be slightly deficient for supporting maximum growth of juveniles (Verme and Ozoga 1980) and yearlings (Cook et al. 2004). Elk in our medium nutrition treatment approximately maintained body fat levels at a constant level through early autumn (see Cook et al. 2004) and, thus, provided an estimate of maintenance for lactating cows during this period. Digestible energy content of their food ranged between 75 to 78 kilocalories per ounce (2.65–2.75 kcal/g) of food, resulting in daily DE intake of 131 to 152 kilocalories per pound of $BM^{0.75}$ (325–375 kcal/kg $BM^{0.75}$). This estimate agrees closely with that presented by Haigh and Hudson (1993), when converted to a $BM^{0.75}$ basis, and the $BM^{0.75}$ estimate calculated using a factorial approach by Cook (2002). However, it is markedly greater than previously estimated for elk by Nelson and Leege (1982, see Cook 2002). Their estimate (75–78 kilocalories per ounce, 2.65–2.75 kcal/g) may satisfy maintenance (i. e., constant body fat levels) of adult lactating cows, but it definitely will retard growth of calves and yearling cows (Cook et al. 1996, 2004) and probably will preclude recovery by adult cows of all mass lost the previous winter if the winter is harsh (Cook 2002). Occasional reproductive pauses (Cameron 1994) by individuals might occur in elk herds existing on such a maintenance plane of nutrition.

It is clear that nutritional requirement should be considered as a gradient, along which different levels of performance can be expected. Thus, nutritional requirements are most relevant in the context of animal performance targets, which of course is largely a function of what the public and wildlife managers expect or desire. One value of our research is to provide standards of performance (Table 1) with which to gauge the likelihood of nutritional limitation and to relate animal performance to nutritional resources in management settings. We mention several precautions for these guidelines. First, they are intended to apply to lactating cows because we assume that game managers are concerned primarily with productivity of their cows, not merely the maintenance of nonreproductive cows. Second, high mortality of juveniles, particularly in summer and early autumn, can mask inadequate nutrition because adults are spared to some extent the nutritional demands of raising an offspring (Verme and Ullrey 1984). It may be that overall fat levels of cows in herds experiencing high juvenile mortality in summer are greater than those in herds with low mortality, despite identical nutritional environments for both. Finally, our finding that seemingly small differences in DE content of forage have large effects on the performance of elk suggests the need for some caution for forage quality surveys. The ability of herbivores to select diets significantly greater in quality than generally available (Schwartz and Hobbs 1985) restricts the interpretive value of general forage quality surveys. Also, field and laboratory techniques that cause even a small bias of estimated DE in forage (just 10 percent) also might lead to important misinterpretations of nutritional adequacy.

Research and Management Implications

Our study did not directly test the hypothesis that forage conditions in summer and autumn do, in fact, exert strong limiting influences on free-ranging elk. The extent to which our findings are indeed relevant to management largely depends on how well our nutrition treatments represent the range of forage quality consumed by free-ranging elk. This caveat is particularly important for the low-nutrition treatment because its effect was so debilitating. This low level was based on the actual dietary quality of cattle (Holechek et al. 1981) and elk (J. G. Cook, unpublished data 1995) determined in low-to-moderate elevation in forest zones during dry years in the Blue Mountains of northeastern Oregon. Also, based on the review of Cook (2002), DE content of wild herbivore diets typically

Table 1. Estimated levels of performance expected for elk in temperate ecosystems as a function of dietary digestible energy (DE) from midsummer through midautumn. Animal performance estimates are based on late-October and November measurements. For adults, levels apply to prime-aged, roughly 3 to 12 years old, lactating cows.

Sum-aut nutritional status[b]	Dietary DE (kcal/g of food)	Calf mass (kg)	Yearling cow mass (kg)	Lactating adult cow fat (percent)	Yearling preg-nancy (percent)	Adult preg-nancy (percent)	Adult cow breeding date
Excellent	>2.90	125-145	195-230	16-25	≥90	≥90	≤30 Sep
Good	2.75-2.90	105-125	180-195	12-16	30-90	≥90	≤5 Oct
Marginal	2.40-2.75	90-105	160-180	8-12	5-30	70-90	≤10 Oct
Poor	<2.40	<90	<160	< 8	≤5	≤70	≥10 Oct

[a] Caveats and suggestions for proper use of these guidelines are described in detail by Cook et al. (2004).

[b] "Excellent" is defined as summer-autumn nutritional levels in which there are virtually no nutritional limitations. "Good" is defined as summer-autumn nutrition levels that exert minor limitations on performance, but the magnitude of this effect probably is too small to be of practical relevance. "Marginal" pertains to nutrition levels that may influence reproduction or survival (e.g., enhanced probability of death in winter, delayed breeding, delayed puberty). "Poor" pertains to nutrition levels that markedly affect reproduction and reduce survival probability.

ranges from 71 to 92 (2.5–3.25) in early summer, 64 to 85 (2.25–3.0) in midsummer, 62 to 71 (2.2–2.5) in late summer, 57 to 74 (2.0–2.6) in midautumn, and 36 to 57 kilocalories per ounce (1.25–2.0 kcal/g) in late autumn, based on western U. S. and Canadian studies ($n = 20$). If these studies provide reasonable estimates of actual diets for wild elk, then (1) the DE levels in our high nutrition group generally exceeded that of free-ranging elk by late summer, (2) our medium nutrition level generally mimicked the higher range of these estimates after midsummer and (3) our low nutrition level fell within these ranges by late summer. If so, it may be reasonable to speculate that marginally adequate to inadequate summer-autumn nutritional conditions prevail in many areas of the West. Where this is in fact the case, some of the consequences that can be expected include low or declining herd productivity, reduced hunting opportunity, increased severity of die-offs in relatively harsh winters and perhaps increased animal damage on agricultural land.

Elk herd declines in the Northwest are becoming a markedly contentious issue in the region, spawning (1) a plethora of newspaper articles decrying reduced hunting opportunity, calling for widespread predator control and accusing mismanagement by state wildlife departments, (2) new, expensive research to identify causes, and even (3) state-level legislative directives. These declines, along with the more widespread and recognized declines in mule deer populations (Carpenter 1998), are creating serious new challenges for large-ungulate biologists in the 21st century. Causes of declines are being debated (see Johnson et al. 2004), and contributions of nutrition (among other factors) have been postulated, acting through density-dependent mechanisms (McCullough 1984, Fowler 1987, Irwin et al. 1994) or via advancing forest succession in some areas (Gill et al. 1996; Bomar 2000; Peek et al. 2001, 2002) following years of fire suppression and, more recently, curtailment of timber harvest on federal land.

Despite a rich history of elk research over the last three decades, there has been little focus on influences of nutrition on elk herd abundance, productivity and demographics, nor has there been much focus on how management's influence on forage quality and quantity might affect these population attributes. Similarly, nutrition's role has been discounted in most of the habitat models and habitat evaluation procedures used by federal and state land and wildlife management agencies to manage vast areas on behalf of elk (Edge et al. 1990). Findings of our study and others indicate that nutritional attributes of habitat are at least as important for habitat modeling and planning as those habitat attributes typically included in these models and procedures (e. g., thermal cover, hiding cover, distance to roads). This is true whether elk herds are declining or not. Parker et al. (1999) noted that nutrient requirements, foraging and digestive efficiencies, and forage characteristics provide functional cause-and-effect relations that influence nutritional condition, body mass dynamics and, ultimately, reproduction and survival. And, most interrelations among them are quantitatively predictable. Thus, nutritional ecology, "offers the prospect of a quantitative, predictive and general theory of key relations between" large ungulates and their habitat (Parker et al. 1999:6). A number of efforts exist to model nutritional influences and nutrition-based carrying capacity for elk (e. g., Hett et al. 1978, Hobbs et al. 1982, Hobbs and Swift 1985, Roloff et al. 2001). But, the explicit objectives and approaches of these efforts to consider nutrition were never incorporated into the habitat effectiveness/habitat suitability approaches routinely applied on behalf of elk by most state and federal agencies in the West. We think that nutritional ecology provides a compelling basis for large-scale habitat evaluation procedures and that the need for incorporating nutrition into habitat evaluation procedures is heightened by new challenges presented by declining ungulate herds. We also think that management planning should begin explicitly accounting for nutritional values of management activities that occur on summer-autumn ranges, in addition to those on winter ranges. Reliable, nutrition-explicit, and large-scale habitat planning and management, however, undoubtedly will require a new research emphasis that links fine-scale nutritional attributes of habitat to population dynamics of elk herds.

Reference List

Alldredge, M. W., J. M. Peek, and W. A. Wall. 2002. Nutritional quality of forages used by elk in northern Idaho. *Journal of Range Management*. 55:253–9.

Bender, L. C., P. E. Fowler, J. A. Bernatowicz, J. L. Musser, and L. E. Stream. 2002. Effects of open-entry spike bull, limited-entry branched bull harvesting on elk composition in Washington. *Wildlife Society Bulletin*. 30:1,078–84.

Berger, J. 1992. Facilitation of reproductive synchrony by gestation adjustment in gregarious mammals: A new hypothesis. *Ecology*. 73:323–9.

Bomar, L. K. 2000. *Broad-scale patterns of elk recruitment in Idaho: Relationships with habitat quality and effects of data aggregation*. M.S. thesis. Moscow, Idaho: University of Idaho.

Cameron, R. D. 1994. Reproductive pauses by female caribou. *Journal of Mammalogy*. 75:10–3.

Cameron, R. D., W. T. Smith, S. G. Fancy, G. L. Gerhardt, and R. G. White. 1993. Calving success of female caribou in relation to body weight. *Canadian Journal of Zoology*. 71:480–6.

Carpenter, L. H. 1998. Deer in the West. In *Proceedings of the 1997 deer/elk workshop*. J. C. DeVos, Jr., ed . 1–10. Rio Rico, Arizona: Arizona Game and Fish Department.

Caughley, G. 1977. *Analysis of vertebrate populations*. New York, NY: John Wiley and Sons.

Caughley, G. 1979. What is this thing called carrying capacity? In *North American elk: Ecology, behavior, and management*, eds. M. S. Boyce, and L. D. Hayden-Wing, 2–8. Laramie, Wyoming: The University of Wyoming.

Christensen, A., L. J. Lyon, and J. W. Unsworth. 1993. *Elk management in the Northern Region: Considerations in forest plan updates or revisions, general technical report INT-303*. Ogden, Utah: U. S. Department of Agriculture, Forest Service.

Cook, J. G. 2002. Nutrition and food. In *North American elk: Ecology and management,* eds. D. E. Toweill, and J. W. Thomas, 259–349. Washington, DC: Smithsonian Institution Press.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs*. No. 155:1–61.

Cook, J. G., L. J. Quinlan, L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1996. Nutrition-growth relations of elk calves during late summer and fall. *Journal of Wildlife Management*. 60:528–41.

Cook, R. C., D. L. Murry, J. G. Cook, P. Zager, and M. J. Gratson. 2001b. Nutritional influences on breeding dynamics in elk. *Canadian Journal of Zoology*. 79:845–53.

Cook, R. C., J. G. Cook, D. L. Murry, P. Zager, B. K. Johnson, and M W. Gratson. 2001a. Development of predictive models of nutritional condition for Rocky Mountain elk. *Journal of Wildlife Management*. 65:973–87.

Crête, M., and J. Huot. 1993. Regulation of a large herd of migratory caribou: Summer nutrition affects calf growth and body reserves of dams. *Canadian Journal of Zoology*. 71:2,291–6.

Edge, W. D., S. L. Olson-Edge, and L. L. Irwin. 1990. Planning for wildlife in national forests: Elk and mule deer habitats as an example. *Wildlife Society Bulletin*. 18:87–98.

Ferry, M. L., T. S. Peterson, and J. C. Calhoun, editors. 2001. Status of elk populations on the Olympic Peninsula. *Olympic Natural Resources Center Conference Proceedings*. Forks, Washington: University of Washington, Olympic Natural Resources Center.

Fowler, C. W. 1987. A review of density dependence in populations of large mammals. In *Current mammalogy*, ed. H. C. Genoways, 401–41. New York, New York: Plenum Press.

Gill, R. M. A., A. L. Johnson, A. Francis, K. Hiscocks, and A. J. Peace. 1996. Changes in roe deer (*Capreolus capreolus*) population density in response to forest habitat succession. *Forest Ecology and Management*. 88:31–41.

Gratson, M. W., and P. Zager. 1999. *Study IV: Factors influencing elk calf recruitment, project W-160-R-25, subproject 31. 1998 progress report*. Lewiston, Idaho: Idaho Department of Fish and Game.

Grey, P. B., and F. A. Servello. 1995. Energy intake relationships for white-tailed deer on winter browse diets. *Journal of Wildlife Management*. 59:147–52.

Haigh, J. C., and R. J. Hudson. 1993. *Farming wapiti and red deer*. St. Louis, Missouri: Mosby-Year Book, Inc.

Hett, J., R. Taber, J. Long, and J. Schoen. 1978. Forest management policies and elk summer carrying capacity in the *Abies amabilis* forest, western Washington. *Environmental Management*. 2:561–6.

Hobbs, N. T., D. L. Baker, J. E. Ellis, D. M. Swift, and R. A. Green. 1982. Energy and nitrogen-based estimates of elk winter-range carrying capacity. *Journal of Wildlife Management*. 46:12–21.

Hobbs, N. T., and D. M. Swift. 1985. Estimate of habitat carrying capacity incorporating explicit nutritional constraints. *Journal of Wildlife Management*. 49:814–22.

Holechek, J. L., M. Vavra, and J. Skovlin. 1981. Diet quality and performance of cattle on forest and grassland range. *Journal of Animal Science*. 53:291–8.

Irwin, L. L., J. G. Cook, R. A. Riggs, and J. M. Skovlin. 1994. *Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystems, general technical report PNW-GTR-325*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Johnson, B. K., M. J. Wisdom, and J. G. Cook. 2004. Issues of elk productivity for research and management. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:551–71.

Julander, O., W. L. Robinette, and D. A. Jones. 1961. Relation of summer range condition to mule deer herd productivity. *Journal of Wildlife Management*. 25:54–60.

Keech, M. A., R. T. Bowyer, J. M. Ver Hoef, R. D. Boertje, B. W. Dale, and T. R. Stephenson. 2000. Life-history consequences of maternal condition in Alaskan moose. *Journal of Wildlife Management*. 64:450–62.

Leege, T. A. 1984. *Guidelines for evaluating and managing summer elk habitat in northern Idaho, wildlife bulletin number 11*. Boise, Idaho: Idaho Department of Fish and Game.

Lyon, J. L. 1980. Coordinating forestry and elk management. *Transactions of the North American Wildlife and Natural Resources Conference*. 45:278–87.

Marcum, C. L. 1975. *Summer-fall habitat selection and use by a western Montana elk herd*. Ph.D. thesis, Missoula, Montana: University of Montana.

Mautz, W. W. 1978. Sledding on a bushy hillside: The fat cycle in deer. *Wildlife Society Bulletin*. 6:88–90.

McCullough, D. R. 1984. Lessons from the George Reserve, Michigan. In *White-tailed deer: Ecology and management*, ed. L. K. Halls, 211–42. Harrisburg, Pennsylvania: Stackpole Books, Inc.

Merrill, E. H., and M. S. Boyce. 1991. Summer range and elk population dynamics in Yellowstone National Park. In *The Greater Yellowstone ecosystem: Redefining America's wilderness heritage*, eds. R. B. Keiter and M. S. Boyce, 263–73. New Haven, Connecticut: Yale University Press.

Minson, D. J., and J. R. Wilson. 1994. Prediction of intake as an element of forage quality. *In Forage quality, evaluation, and utilization*, ed. G. C. Fahey, 533–63. Madison, Wisconsin: ASA.

National Research Council. 1984. *Nutrient requirements of cattle, 6th edition*. Washington, DC: National Academy Press.

Nelson, J. R., and T. A. Leege. 1982. Nutritional requirements and food habits. In *Elk of North America: Ecology and management*, eds. J. W. Thomas, and D. E. Toweill, 323–67. Harrisburg, Pennsylvania: Stackpole Books.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management*. 66:1,301–7.

Noyes, J. H., B. K. Johnson, L. D. Bryant, S. L. Findholt, and J. W. Thomas. 1996. Effects of bull age on conception dates and pregnancy rates of cow elk. *Journal of Wildlife Management*. 60:508–17.

Oftedal, O. T. 1985. Pregnancy and lactation. In *Bioenergetics of wild herbivores*, eds. R. J. Hudson, and R. G. White, 215–38. Boca Raton, Florida: CRC Press.

Parker, K. L., M. P. Gillingham, T. A. Hanley, and C. T. Robbins. 1999. Energy and protein balance of free-ranging black-tailed deer in a natural forest environment. *Wildlife Monographs*. No. 143:1–48.

Pederson, J. D., and K. T. Harper. 1978. Factors influencing productivity of two mule deer herds in Utah. *Journal of Range Management*. 31:105–10.

Peek, J. M., B. Dennis, and T. Hershey. 2002. Predicting population trend of mule deer in south-central Oregon. *Journal of Wildlife Management*. 66:729–36.

Peek, J. M., J. J. Korol, D. Gay, and T. Hershey. 2001. Overstory-understory biomass changes over a 35-year period in southcentral Oregon. *Forest Ecology and Management*. 150:267–77.

Riggs, R. A., S. C. Bunting, and S. E. Daniels. 1996. Prescribed fire. In *Rangeland Wildlife*, ed. P. R. Krausman, 295–319. Denver, Colorado: Society for Range Management.

Roloff, G. J., J. J. Millspaugh, R. A. Gitzen, and G. C. Brundige. 2001. Validation tests of a spatially explicit habitat effectiveness model for Rocky Mountain elk. *Journal of Wildlife Management*. 65:899–914.

Schommer, T. 1991. *Analysis of big game statistics 1965–1990*. Baker City, Oregon: Wallowa-Whitman National Forest, U. S. Forest Service, Wallowa-Whitman National Forest.

Schwartz, C. C., and N. T. Hobbs. 1985. Forage and range evaluation. In *Bioenergetics of wild herbivores*, eds. R. J. Hudson and R. G. White, 25–52. Boca Raton, Florida: CRC Press.

Schwartz, C. C., and L. A. Renecker. 1998. Nutrition and energetics. In *Ecology and management of the North American moose*, eds. P. W. Franzman and C. C. Schwartz, 441–87. Washington, DC: Smithsonian Institution Press.

Thomas, D. C. 1982. The relationship between fertility and fat reserves of Peary caribou. *Canadian Journal of Zoology*. 60:597–602.

Unsworth, J. W., L. Kuck, E. O. Garton, and B. R. Butterfield. 1998. Elk habitat selection on the Clearwater National Forest, Idaho. *Journal of Wildlife Management*. 62:1,255–63.

Verme, L. J., and D. E. Ullrey. 1984. Physiology and nutrition. In *White-tailed deer: Ecology and management*, ed. L. K. Halls, 91–108. Harrisburg, Pennsylvania: Stackpole Books.

Verme, L. J., and J. J. Ozoga. 1980. Effects of diet on growth and lipogenesis in deer fawns. *Journal of Wildlife Management*. 44:315–24.

Wallmo, O. C., L. H. Carpenter, W. L. Regelin, R. B. Gill, and D. L. Baker. 1977. Evaluation of deer habitat on a nutritional basis. *Journal of Range Management*. 30:122–7.

White, R. G. 1983. Foraging patterns and their multiplier effects on productivity of northern ungulates. *Oikos*. 40:377–84.

Elk and Mule Deer Responses to Variation in Hunting Pressure

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Alan A. Ager
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

James H. Noyes
Oregon Department of Fish and Wildlife
La Grande

Norm Cimon
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

Hunting can exert a variety of effects on both targeted and nontargeted ungulates, and animals either run or hide in response to hunting pressure. If animals successfully elude hunters by running, the energetic cost may deplete fat reserves needed for survival during winter in temperate regions. If animals successfully elude hunters by hiding, there may be an energetic cost from lost foraging opportunities.

Most studies of ungulate responses to hunting have focused on changes in habitat selection. Ungulates typically respond to hunting by seeking areas of security (Irwin and Peek 1979, Knight 1980, Edge and Marcum 1985, Naugle et al. 1997, Millspaugh et al. 2000), by altering activity patterns (Naugle et al. 1997), by adjusting home ranges (Kufield et al. 1988, Root et al. 1988) or by moving long distances (Conner et al. 2001, Vieira et al. 2003). However, the difficulty of monitoring hunter density and elk and deer populations on large landscapes has prevented the collection of sufficient data to develop models of energetic costs associated with hunting or with other recreational activities. Variation in weather, hunter density, herd dynamics and seasonal conditions can likely bring about changes in the interactions between hunters and animals, making generalizations tenuous at best. Quantitative relationships between levels of hunting pressure and energy expenditure can be used to evaluate potential secondary effects of activities on nutritional condition of ungulates. For instance, frequent human disturbance that results in high energy expenditure by ungulates could adversely affect animal weight dynamics in winter, when forage is scarce, or in summer, when energy requirements are high for lactation and rebuilding body mass following winter (Cook et al. 2004).

In this study, we examined the effects of hunter density and associated motorized traffic on movement and habitat use by elk and mule deer over 10 years during 21 rifle and 2 archery hunting seasons at the Starkey Experimental Forest and Range (Starkey). Our goal was to quantify the relationship between levels of hunting pressure, as measured by hunter density, traffic counts, changes in movement and changes in habitat use patterns. These relationships were then used to estimate effects from variation in hunting pressure and type of hunt on daily and seasonal energy budgets of elk and mule deer.

Study Area

Starkey covers 40 square miles (101 km^2) on the Wallowa Whitman National Forest, 21 miles (35 km) southwest of La Grande, Oregon (45°15'N, 118°37'W). Our study was conducted in the Main Study Area (30 square miles [77.6 km^2]), which was enclosed with 7.9-feet (2.4-m) tall woven-wire fence (Bryant et al. 1993) and has been used for studies on Rocky Mountain elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*) and cattle since 1989 (Rowland et al. 1997). Starkey contained habitat for elk and mule deer that was typical of summer range conditions in the Blue Mountains. A network of drainages in the project area created a complex and varied topography. Vegetation at Starkey was a mosaic of coniferous forests, shrublands, wet meadows, riparian areas and grasslands.

Approximately 37 percent of the study area had forest canopy of more than 40 percent, and about 4 percent had forest canopy greater than 70 percent.

Traffic levels, recreational activities (including hunting), cattle grazing and timber management were representative of adjacent public land. About 500 cow-calf pairs of domestic cattle grazed the Main Study Area on a deferred rotation system through four pastures within the study area between June 15 and October 15 (Coe et al. 2001). Our study area at Starkey was three to four times larger than typical summer home ranges of elk in the Blue Mountains (7.7 to 11.2 square miles [20–29 km^2], Leckenby 1984), providing study animals with large-scale habitat choices commensurate with free-ranging herds. Details of the study area and facilities are available elsewhere (Wisdom et al. 1993, Noyes et al. 1996, Rowland et al. 1997).

Materials and Methods

Hunt Sample

We analyzed 13 rifle elk hunts, 8 rifle mule deer hunts and 2 archery elk hunts conducted between 1991 and 2000 (Table 1). Elk rifle hunts were 5 to 9 days long with 75 to 175 tags issued to hunters, and deer rifle hunts were 7 to 12 days long with 25 tags issued. Archery hunts were 30 days long with 85 tags issued. We staffed a hunter check station starting the day before the opening of each hunt through the end of the hunt; this was done for all elk rifle hunts, through the first three days of deer hunts and intermittently during the rest of the deer seasons. We staffed the archery check station during most days with project personnel or volunteers. For each hunter, we recorded number of days hunted and success. Prior to any hunts, the yearling and adult elk populations were estimated between 313 and 443 females and between 78 to 153 males (Noyes et al. 1996, 2002), and the mule deer population was estimated between 262 and 342 total males and females (Rowland et al. 1997). Densities of adult elk and deer in the study area were similar to those on adjacent public land (Johnson et al. 2000).

Animal Locations

We determined animal locations with an automated telemetry system that uses retransmitted long-range aid to navigating signals (LORAN-C) (Findholt et al. 1996, Rowland et al. 1997). Each radiocollared elk was used an average of 1.6 years, while each mule deer was used about 1.9 years. We monitored between 25 and 60 elk and 12 and 33 deer during the hunts. Locations

Table 1. Summary of elk (E) hunts and deer (D) hunts included in the study held in Main Study Area (30 square miles [77 km^2]) at the Starkey Experimental Forest and Range. H = habitat; V = velocity; T = traffic; Hu = hunter.

Hunt label	Date	Days of elk velocity data	Days of deer velocity data	Days of elk habitat data	Days of deer habitat data	Number of hunters	Traffic counts/ day	Elk locations	Deer locations	Available dependent variables for elk telemetry	Available dependent variables for deer telemetry	Available independent variables for both deer and elk
E3[1]	Aug. 15–23, 1992	6	Na[6]	8	Na	19–168	65–245	904	0	HV		HuT
E4[2]	Dec. 5–12, 1992	3	Na	4	Na	24–73	59–103	753	0	HV		HuT
E5[1]	Aug. 14–22, 1993	3	5	6	6	50–117	74–164	1,855	771	HV		HuT
E6[3]	Aug. 27–Sep. 25, 1994	30	20	30	30	2–53	26–66	6,368	4,146	HV	HV	HuT
E7[1]	Oct. 26–30, 1994	5	5	5	5	74–113	87–116	1,552	1,122	HV	HV	HuT
E8[4]	Nov. 5–13, 1994	9	9	9	9	25–130	42–168	3,407	2,054	HV	HV	HuT
E9[5]	Nov. 19–27, 1994	7	7	7	7	7–63	25–106	2,264	1,588	HV	HV	HuT
E10[1]	Aug. 18–25, 1996	9	1	9	4	15–134	Na	2,632	318	HV		Hu
E13[2]	Nov. 29–De. 5, 1997	5	5	5	5	12–69	Na	3,972	2,106	HV	HV	Hu
E15[2]	Dec. 5–11, 1998	4	5	4	5	13–46	2–25	1,496	723	HV	HV	HuT
E18[1]	Aug. 1–5, 2000	3	4	4	4	33–122	Na	859	712	HV	HV	Hu
E19[3]	Aug. 25–Sep. 24, 2000	29	28	30	30	12–52	1–33	9,709	5,993	HV	HV	HuT
E21[2]	Dec. 3–9, 1994	3	3	3	3	51–82	76–105	890	634	HV	H	HuT
D1[7]	Sep. 28–Oct. 4, 1991	5	Na	7	Na	Na	48–85	842	0	HV		T
D2[7]	Oct. 3–9, 1992	4	Na	4	Na	Na	58–145	1,042	0	HV		T
D3[7]	Oct. 2–8, 1993	7	4	7	6	2–16	33–71	2,272	878	HV	HV	HuT
D4[7]	Oct. 1–12, 1994	10	4	10	10	Na	14–54	2,040	1,322	HV	HV	T
D7[7]	Sep. 30–Oct. 11, 1997	8	7	8	7	3–12	Na	4,294	2,079	HV	HV	Hu
D8[7]	Oct. 3–14, 1998	12	7	12	12	0–24	10–100	4,225	1,615	HV	HV	HuT
D9[7]	Oct. 2–13, 1999	11	Na	12	12	Na	11–43	3,515	1,412	HV	HV	T
D10[7]	Sep. 30–Oct. 11, 2000	Na	10	12	12	Na	55–158	4,363	2,410	H	HV	T

[1] Rifle Spike-only elk hunt
[2] Rifle Antlerless elk hunt
[3] Archery any elk hunt
[4] Rifle any elk hunt
[5] Rifle antlered elk hunt
[6] Data not available due to sample size restrictions of 10 animals each with 10 locations, missing traffic counts, or missing hunter numbers
[7] Rifle antlered deer hunt.

were assigned to Universal Transverse Mercator coordinates of associated 98.4 by 98.4-feet (30 by 30-m) pixels containing habitat information stored in a geographic information system (GIS). Locations had a mean error of 175 feet ±19 feet (53 m ± 5.9 SE, Findholt et al. 1996).

Vehicular Traffic

We measured traffic throughout the year from a network of 71 traffic counters (Rowland et al. 1997, 2000). Preliminary analysis of traffic counter data showed that data collected at the counter located 0.15 miles (0.24 km) inside the main gate were highly correlated with data obtained at the other counters located throughout the study area. For this analysis, we used the daily counts of vehicles that passed over the Starkey main entrance counter as an index to total traffic activity. In addition, by using a single counter, the application of our results to areas with limited traffic data is made more feasible.

Habitat Variables

We selected 10 variables that were significant in resource selection models for elk or mule deer in previous studies at Starkey (Johnson et al. 2000, Table 1). However, we used distance to open roads rather than distance to roads of various traffic rates because, during hunting seasons, daily use of open roads was greater than the minimum value associated with roads with high traffic rates (more than 4 vehicles per 12 hours). These variables were significant in resource selection models for elk or mule deer in previous studies at Starkey. Additional details can be found in Johnson et al. (2000) and Rowland et al. (1998).

Data Analysis

We used day as our sampling unit and required a minimum of 10 locations from each of at least 10 animals of a species per day. Choice of minimum sample sizes was based on previous analyses (Ager et al. 2003). We calculated velocity between successive locations by dividing the horizontal distance moved by the elapsed time. We deleted velocity for any location if elapsed time to the previous location was fewer than 5 minutes or more than 240 minutes. Shorter elapsed times (fewer than 5 minutes) yielded velocities that were positively biased because of the random location error in the telemetry system. Velocities determined at longer elapsed times were negatively biased as a result of undetected movements between observations and home range effects (Ager et al. 2003). For habitat analysis, average sample size for elk during elk hunts was 254 observations per day on a total of 23 animals. Deer sample size during elk hunts was 171 observations per day on 15 animals. Sample sizes for velocity calculations were reduced approximately 15 percent as a result of the time filter placed on data.

Response Variables

We compared responses of elk and deer during elk and deer hunts to seven-day prehunt periods that started 2 weeks before the start of the hunting season. We calculated daily average velocity of elk and mule deer as dependent variables and used daily traffic counts and hunter density as independent variables in a regression analysis. We used the daily averages for the prehunt periods to set the intercept.

We calculated hourly velocities and habitats used for the prehunt and hunt periods and examined how normal daily patterns changed in relation to variation in hunter density. We pooled data across all animals and hunts to maximize the number of hunting days in the analysis. Although hunt-to-hunt variability was of interest, our goal was to obtain an overall estimate of animal response over a wide range of hunting conditions. Variation among hunts was high and number of hunts had one or more of the following conditions: (1) limited data for prehunt conditions, (2) limited variability in the hunter density and traffic counts, and (3) differences in the range of traffic and hunter density. Thus, using a single linear model containing a term for individual hunts would have reduced the possibility of obtaining useful information from several of the hunts, but the effects among the hunts would not have been comparable in many cases. We did not pool animals used in multiple years but rather considered them independent samples in each hunt because day was the sampling unit. We did not test for serial correlation in the data from successive days within a hunt after observing that elk and deer response to hunters at Starkey is dynamic and rapid in terms of changes in distribution and velocity.

Energetic Calculations

We estimated energy consumption as a function of velocity from equations provided by Robbins et al. (1979:449) and Parker et al. (1984:478). Robbins et al. (1979) estimated the cost of locomotion in kilocalories per kilogram per hour where Parker et al. (1984) estimated cost of locomotion in kilocalories per kilogram per minute. Robbins et al. (1979) estimated cost of location on slopes for uphill and downhill, but we did not factor energy consumption for slope because we could not accurately estimate animal paths and their respective slopes. The differential energy expenditure for uphill versus downhill travel results in an underestimation of energy expenditure. Average slope at Starkey was 17 percent.

Results

Daily density of hunters per square mile averaged 2.4 (0.91 hunters/km^2, range 0.03–1.66/km^2) for elk hunts, 0.1 (0.04 hunters/km^2, 0–0.31/km^2) for deer hunts, and 0.7 hunters per square mile (0.26 hunters/km^2, 0–0.67/km^2) for archery hunts. Daily traffic counts averaged 78 (6–272 range) for elk hunts, 56 (10–158 range) for deer hunts, and 55 (17–133 range) for archery hunts. During the prehunt periods, vehicles per day averaged 46 (18–86 range) before archery, 45 (1–165 range) before elk rifle, and 27 (1–65 range) before deer hunts. All deer hunts were excluded from further analysis because there was no measurable response of elk or mule deer during deer hunts, most likely because of the low hunter density and traffic rates.

Velocity

Velocity of elk movements increased during rifle and archery hunts (Figure 1A). During nonhunt periods, elk displayed daily patterns of velocity characterized by crepuscular peaks of about 11.5 feet per second (3.5 m/min) at 0600 to 0800 and 1700 to 1800 hours, and lower midday velocity of around 6.6 feet per second (2 m/min) during nonhunt periods. Elk velocity was greater in the early morning during rifle hunts than during archery hunts, but this pattern reversed during the afternoon (Figure 1A). Differences in timing of peaks during prehunt periods for archery and rifle hunts probably reflects differences in sunrise and sunset times for September versus December hunts.

During elk rifle hunts, the daily patterns of velocity were strongly affected by hunter density (Figure 2). At the lower density values (fewer than 0.8 hunters per square mile, [0.3 hunters/km^2]) the crepuscular peaks were broader and increased by around 3.3 feet per minute (1 m/min), and the midday velocity returned to prehunt levels. At high hunter density, more than 3.2 hunters per square mile (1.25 hunters/km^2), the daily pattern consisted of large velocity

Figure 1. Mean hourly velocities of elk (A) and mule deer (B) during and before elk rifle and archery hunts at Starkey.

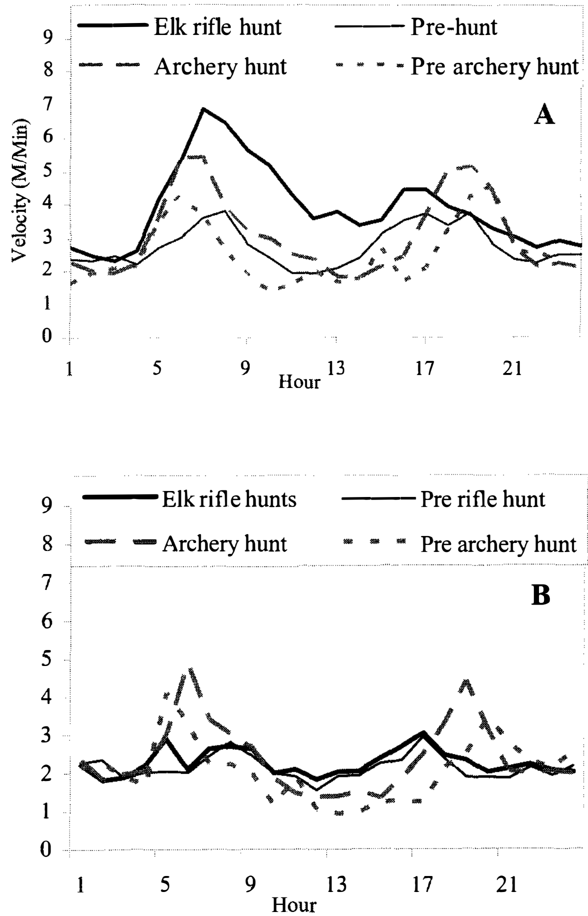
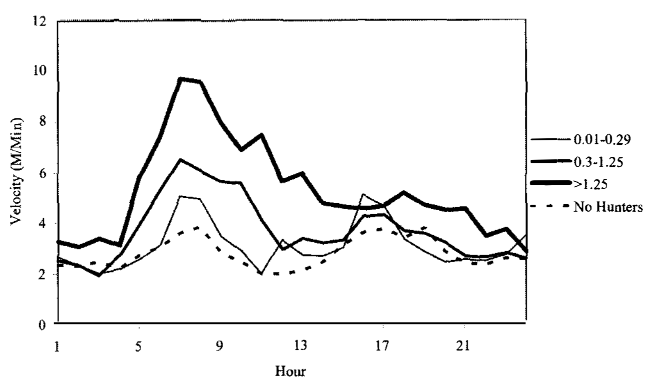


Figure 2. Hourly velocities of elk at four hunter densities (hunters/km²)for elk rifle hunts at Starkey.



increases throughout the day, with a peak at 0800 hours that was about 20 feet per minute (6 m/min) above the prehunt velocities (Figure 2). Only between 0000 to 0400 hours did the velocity at high hunter density return to values close to prehunt conditions.

Daily mean velocity of elk to rifle hunter density showed a linear relationship (Figure 3, $R^2 = 0.46$, $P < 0.001$), the velocity increasing 1.4 meters per min per hunter per square kilometer. The regression predicts that at the highest daily rifle hunter densities during the Starkey hunts, estimated mean velocity would increase from 6.9 feet per minute (2.7 m/min) during the prehunt period to around 16.5 feet per minute (5 m/min). Effects of archery hunters on elk appeared to be greater than that of rifle hunters, as velocity increased 2.2 meters per minute per hunter per square kilometer ($P < 0.001$) (Figure 4).

Figure 3. Daily mean elk and mule deer velocities versus hunter density for elk rifle hunts at Starkey.

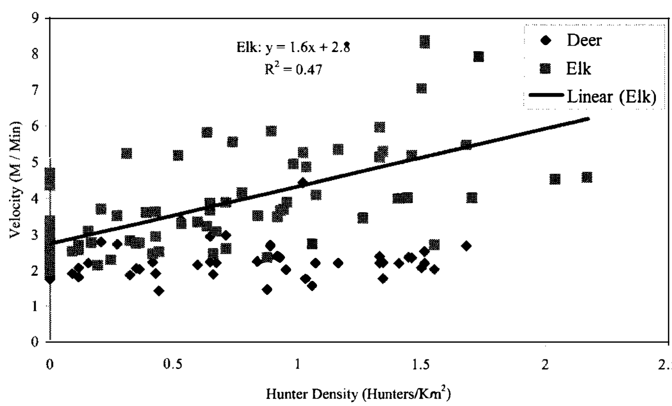
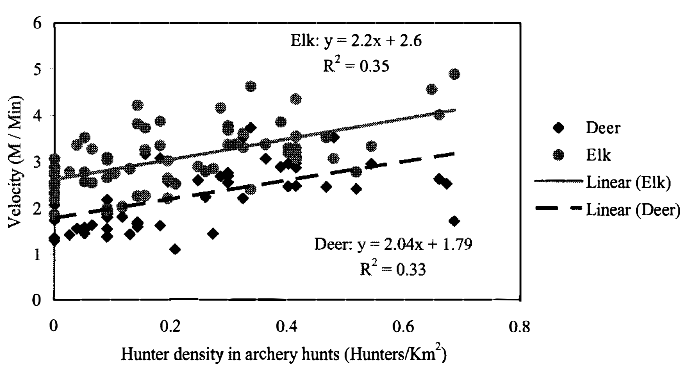


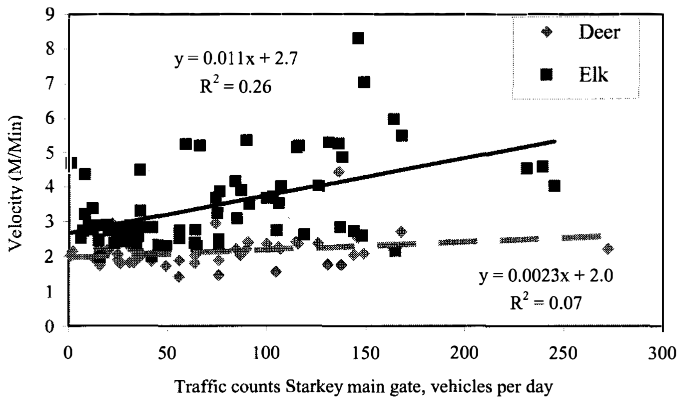
Figure 4. Daily mean elk and mule deer velocities versus hunter density for elk archery hunts at Starkey.



Mule deer showed little increase in hourly velocities during elk rifle hunting seasons (Figure 1B), except for a small increase in velocity around 0500 hours, but there was no significant increase in velocity as rifle hunter density increased (Figure 3). Mule deer velocity increased as archery hunter density increased (Figure 4, P <0.001), and the hourly velocities during archery season increased at sunrise and sunset (Figure 1B).

There was a positive relationship between traffic counts during elk rifle seasons and daily mean velocity of elk (Figure 5, $R^2 = 0.28$); although, this relationship was considerably weaker than that between hunter density and velocity. Also in contrast to rifle hunts, no relationship was observed between traffic counts and elk velocity during the archery hunts, likely due to the lower traffic levels. Deer showed a very low response to increased levels of traffic (Figure 5).

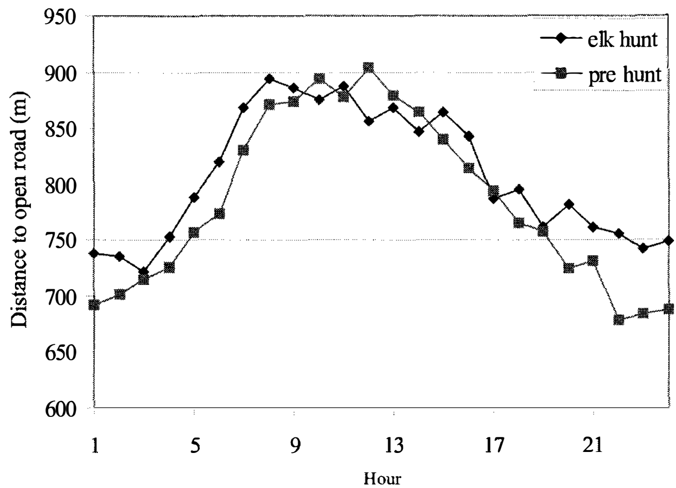
Figure 5. Elk and mule deer velocities versus traffic counts before and during elk rifle hunts at Starkey.



Habitat Variables

Distance of elk to open roads increased as both elk rifle hunter density ($y = 83x + 751$, $r^2 = 0.12$, where x = hunter/km^2) and traffic counts increased ($y = 0.8x + 740$, $r^2 = 0.11$, where x = daily traffic count and y is distance in m). During archery seasons, as hunter density increased, elk use of canopy cover decreased ($y = -7.3x + 38.5$, $r^2 = 0.18$, where x = hunter/km^2 and y = percent canopy cover). There were no other significant relations in traffic or hunter density during the archery seasons and any other habitat variables. Hourly habitat use at different levels of hunter density showed that prehunt daily patterns of habitat use for distance to open road (Figure 6) and all other habitat variables (not shown) were increasingly disrupted as hunter density increased. In addition to disruption, distance of elk to open roads increased especially in the nighttime hours, when elk did not move closer to roads, compared to prehunt distributions.

Figure 6. Hourly average distance of elk from an open road during rifle elk hunts at Starkey Experimental Forest and Range.



Energetic Cost of Hunting for Elk

At the average hunter density observed in this study, 2.3 hunters per square mile (0.91 hunters/km^2), velocity increased 4.3 feet per minute (1.3 m/min) over the background velocity for the nonhunt periods. Based on the relationship provided by Robbins et al. (1979) and Parker et al. (1984), this velocity increase translates to a 4-percent (Parker et al. 1984) to 10-percent (Robbins et al. 1979) increase in the normal daily energy budget. This estimate was slightly conservative because travel was assumed to be on level ground.

Discussion

This study represents our first attempt to measure the effects of variation in hunting pressure on elk and deer over a wide range of hunter densities, hunting conditions, traffic rates and rifle versus archery hunting. We found that elk responded by fleeing disturbance; whereas, deer eluded hunters by hiding. Our data indicate that both traffic count and hunter density have a positive linear relation with elk velocity. Moreover, hunter density appears to be a better indicator of animal disturbance than traffic counts, especially for archery hunts. We found differences between the archery and the rifle hunts in terms of animal responses; although, the study only included two archery hunts. Archery hunts appeared to affect animal movements for a longer portion of the day, suggesting that archers are actively pursuing elk throughout the day and evening.

In contrast to other previous studies (Irwin and Peek 1979, Edge and Marcum 1985, Millspaugh et al. 2000), we did not find major shifts in habitat use by elk or mule deer during the elk rifle hunting seasons. However, the daily patterns of habitat use were disrupted. We also did not observe any effects on elk or deer from the deer rifle hunts, most likely due to the low hunter densities associated with these hunts.

The results of this study suggest that energetic costs to elk from hunting may be significant in the context of both hunter density and the number of days over which hunting occurs; however, the energetic costs to deer may not be as great. In northeastern Oregon, many of the wildlife management units have 30 days of archery hunting, followed by 12 days of mule deer rifle hunting, then 14 days of rifle elk hunting and, finally, up to 9 days of antlerless elk hunting, totaling 56 to 65 days of hunting. In 1999, Oregon Department of Fish and Wildlife (2000) estimated that there were 61,804 and 49,063 recreational days for elk and mule deer hunting in the Starkey and Ukiah Wildlife Management Units (WMUs), respectively. There were 726 square miles (1,859 km^2) and 568 square miles (1,453 km^2) of deer and elk range in the Starkey and Ukiah WMUs, respectively (Oregon Department of Fish and Wildlife 1986) for a hunter density of 1.35 hunters per day per square mile (0.53 hunters/day/km^2).

Based on our estimate of increased velocity associated with rifle hunter density, and the energetic relationships identified in Parker et al (1984), this equated to an additional energy expenditure of 310 kilocalories per day for an adult cow elk assumed to weigh 510 pounds (232 kg). Assuming this additional energy comes from stored fat reserves and there are 9 kilocalories per gram of fat, this results in 1.2 ounces (35 g) of fat consumed per day or 4.4 pounds (2 kg) for the entire 63 days of hunting due to disturbance from hunters. Assuming the ingesta-free body mass of a lactating cow elk is 440 pounds (200 kg) and has 10-percent body fat, these 4.4 pounds (2 kg) of body fat represents about a 10-percent reduction in body fat. Cook et al. (2004) suggest that elk reproduction may be affected when body fat falls below 9 percent and is marginally affected at levels between 9 and 13 percent, conditions that occur in lactating cow elk in late autumn. Because most of the hunting disturbance occurs after the rut, the reduced fat levels may be more important during harsh winters and may carry over through the next year as cows rebuild tissues catabolized during winter.

Elk and mule deer at Starkey were closed populations; thus, animals were unable to escape from human harassment by moving to private land or other reserves. Without hunting, the movement rates we observed at Starkey were similar to those reported by Craighead et al. (1973). In northeastern Oregon, private land, wilderness, and roadless or road management areas provide areas where hunter density is lower. Thus, our estimates of energy expenditure for elk in our example may be high for those animals that are able to move to secluded areas where hunter density is low. However, animals that respond to hunting pressure by moving long distances from that pressure also would exert substantial energy that our study does not address.

Our energy calculations do not account for disruption of foraging cycles where foraging patterns are shortened or where animals move to more secluded areas with poorer forage resources. If those situations occur, then the effects of hunter density may be much more pronounced than we estimated. These potential effects deserve attention in future studies. Estimating absolute levels of energy loss will take more accurate measures of activity patterns, distributions and forage quality of ungulates in relation to variation in hunting pressure. For our analysis, we did not distinguish between flight movements and foraging which require more precise monitoring of elk to quantify the disruption of foraging patterns.

Our results may have implications for design and management of access and hunting seasons for mule deer and elk. First, the energetic costs of eluding hunters may be substantial under the combination of high hunter densities and long hunting seasons. The added energetic costs may have the potential to increase mortality of animals beyond those harvested in areas with severe winter conditions. Second, the motorized access provided to hunters, in combination with hunter density and season length, may affect the degree to which nonharvested animals are negatively affected by hunting. For example, it may be possible to reduce or to restrict human access, particularly motorized access, as part of hunting seasons but still accommodate higher hunter density without negative energetic costs on nonharvested animals. Alternatively, if motorized access is relatively unrestricted and landscape conditions facilitate ease of human movement (e. g., flat terrain and open environments), then managers may consider modifications in hunter densities and season lengths to meet population objectives for elk and mule deer. Such trade-offs deserve careful consideration in the integrated planning of human access with design of hunting seasons for management of elk and mule deer.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Bryant, L. D., J. W. Thomas, and M. M. Rowland. 1993. *Techniques to construct New Zealand elk-proof fence, general technical report PNW-GTR-313,* 1–17. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54: A51–A76.

Conner, M. M., G. C. White, and D. J. Freddy. 2001. Elk movement in response to early-season hunting in northwest Colorado. *Journal of Wildlife Management*. 65:926–40.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs*. 155:58–61.

Craighead, J. J., Craighead, F. C. Jr., Ruff, R. L., and B. W. O'Gara. 1973. Home ranges and activity patterns of nonmigratory elk of the Madison drainage herd as determined by biotelemetry. *Wildlife Monographs*. 33:1–50.

Edge, W. D., and C. L. Marcum. 1985. Movements of elk in reaction to logging disturbances. *Journal of Wildlife Management*. 49:926–30.

Findholt, S. L., B. K. Johnson, L. D. Bryant, and J. W. Thomas. 1996. Corrections for position bias of a Loran-C radio telemetry system using DGPS. *Northwest Science*. 70:273–80.

Irwin, L. L., and J. M. Peek 1979. Relationships between road closures and elk behavior in northern Idaho. *North American elk: Ecology, behavior and management*, eds. M. S. Boyce, and L. D. Hayden-Wing, 199–204. Laramie, Wyoming: University of Wyoming.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of elk and mule deer in spring. *Journal of Wildlife Management*. 64:685–97.

Knight, R. R. 1980. The Sun River elk herd. *Wildlife Monographs*. 23.

Kufield, R. C., D. C. Bowden, and D. L. Schrupp. 1988. Influence of hunting on movements of female mule deer. *Journal of Range Management*. 41:70–2.

Leckenby, D. A. 1984. Elk use and availability of cover and forage habitat components in the Blue Mountains, northeast Oregon, 1976–1982. *Oregon Department of Fish and Wildlife, Wildlife Research Report*. 14:1–40.

Millspaugh, J. J., G. C. Brundige, R. A. Gitzen, and K. J. Raedeke. 2000. Elk and hunter space-use sharing in South Dakota. *Journal of Wildlife Management*. 64:994–1,003.

Naugle, D. E., J.A. Jenks, B.J. Kernohan, and R. R. Johnson. 1997. Effects of hunting and loss of escape cover on movements and activity of female white-tailed deer, *Odocoileus virginianus*. *Canadian Field-Naturalist*. 111:595–600.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management*. 66:1,301–7.

Noyes, J. H., B. K. Johnson, L. D. Bryant, S. L. Findholt, and J. W. Thomas. 1996. Effects of bull age on conception dates and pregnancy rates of cow elk. *Journal of Wildlife Management*. 60:508–27.

Oregon Department of Fish and Wildlife. 1986. *Management objectives and related benchmark data for Oregon's Rocky Mountain Elk*. Portland, Oregon: Oregon Department of Fish and Wildlife.

———. 2000. *2000 Big Game Statistics*. Portland, Oregon: Oregon Department of Fish and Wildlife.

Parker, K. L., C. T. Robbins, and T. A. Hanley. 1984. Energy expenditures for locomotion by mule deer and elk. *Journal of Wildlife Management*. 48:474–88.

Robbins, C. T., Y. Cohen, and B. B. Davitt. 1979. Energy expenditure by elk calves. *Journal of Wildlife Management*. 43:445–53.

Root, B. G., E. K. Fritzell, and N. F. Giessman. 1988. Effects of intensive hunting on white-tailed deer movements. *Wildlife Society Bulletin*. 16:145–51.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396:1–62*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–85.

Rowland, M. M., P. K. Coe, R. J. Stussy, A. A. Ager, N. J. Cimon, B. K. Johnson, and M. J. Wisdom. 1998. *The Starkey habitat database for ungulate research: Construction, documentation, and use, general technical report PNW-GTR-430:1–48*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Vieira, M. E. P., M. M. Conner, G. C. White, and D. J. Freddy. 2003. Effects of archery hunter numbers and opening dates on elk movement. *Journal of Wildlife Management*. 67:717–28.

Wisdom, M. J., J. G. Cook, M. M. Rowland, and J. H. Noyes. 1993. *Protocols for care and handling of deer and elk at the Starkey Experimental Forest and Range, general technical report PNW-GTR-311:1–49*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Movements and Habitat Use of Rocky Mountain Elk and Mule Deer

Alan A. Ager
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Haiganoush K. Preisler
U. S. Department of Agriculture, Forest Service,
Pacific Southwest Research Station
Albany, California

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

Understanding how ungulates use large landscapes to meet their daily needs for food, security and other resources is critical to wildlife management and conservation practices (Johnson et al. 2002). For ungulates like Rocky Mountain elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*), landscapes are a mosaic of different resources that are exploited in well-defined seasonal and daily cycles (e. g., Green and Bear 1990). Complex movement patterns emerge when the cyclical behaviors are realized on landscapes that are heterogeneous in space and time (Gross et al. 1995, Etzenhouser et al. 1998). Both the juxtaposition and the grain of habitat patches within a home range are strong determinants of movement patterns, and the overall habitat suitability as well (Etzenhouser et al. 1998). The influence of patch arrangement on habitat quality was recognized in early elk habitat models (Leckenby 1984); although, the linkage between movement patterns and habitat arrangements had yet to be studied.

In this paper, we describe a progression of studies on the Starkey Experimental Forest and Range (Starkey) that concerned Rocky Mountain elk and mule deer movements and habitat use. The work focused on interpatch movements associated with crepuscular habitat transitions and did not consider finer-scale movements associated with foraging activities (Gross et al. 1995, Johnson et al. 2002). We first used the Starkey data to describe the linkage between movements and habitat use at Starkey (Johnson et al. 2000, Rowland et al. 2000) and elsewhere (Mackie 1970, Craighead et al. 1973, Collins and Urness 1983, McCorquodale et al. 1986, Beier and McCullough 1990, Green and Bear 1990, Unsworth et al. 1998). This work motivated a subsequent investigation about spatial patterns of movements. We then explored ways to build a behavioral model of movement that encapsulated both habitat use and spatial organization of movements. The paper concludes with a discussion of the importance of understanding movement patterns in the management of elk and mule deer.

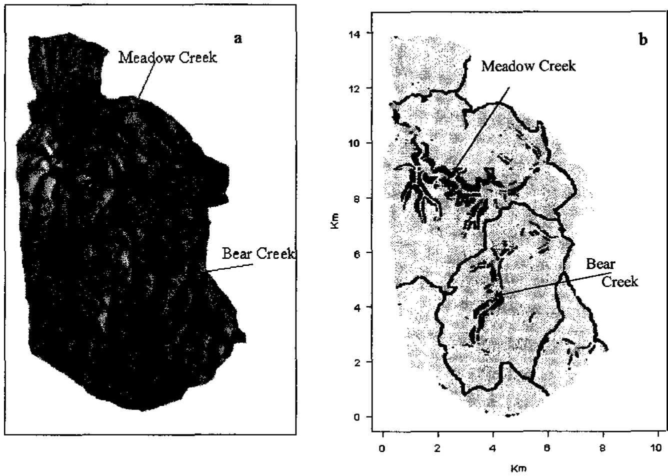
Study Area and Data Collection

Starkey covers 63 square miles (101 km^2) on the Wallowa-Whitman National Forest, about 25 miles (40 km) southwest of La Grande, Oregon. Starkey is a mosaic of coniferous forest (ponderosa pine [*Pinus ponderosa*], Douglas fir [*Pseudotsuga menziesii*], grand fir [*Abies grandis*]) and bunchgrass meadows (*Pseudorogneria* spp.) dissected by numerous small drainages (Figure 1), creating a complex array of topography and vegetation. The project area was enclosed within 8-foot- (2.4-m-) tall woven-wire fence and has been used for studies on Rocky Mountain elk, mule deer and cattle since 1989 (Wisdom et al. 2004). A loran-C telemetry system is used to monitor locations of about 50 elk, 50 mule deer and 50 cows from April to December, obtaining locations every 1 to 2 hours on each animal. The work described here used 400,000 elk and mule deer locations collected over 6 years (1991–1996) within the 48 square mile (77.6 km^2) Main Study Area. We used locations from a total of 144 elk and from 58 mule deer. Habitat variables studies were selected based on their importance in previous work at Starkey (Johnson et al. 2000).

Cycles of Movement and Habitat Use

We studied the daily cycles of elk and mule deer movements and seasonal changes in these cycles by fitting periodic functions to the 1991 to 1996

Figure 1. Map of Starkey, showing topography and major drainages (a), and roads open to vehicular traffic (b). Shaded areas in b are areas where slopes area greater than 40 percent.



location data and by testing for differences among seasons at specific hourly intervals. The key findings of this work are summarized as follows and are reported in detail by Ager et al. (2003). Elk showed pronounced 24-hour cycles with crepuscular transitions for many habitat variables, including canopy cover, distance to hiding cover, cosine of aspect (Figures 2a-c), herbage, and distance to open roads (Figures 3a-b). Habitat transitions appeared to be closely linked to rapid changes in elk movements (Figures 4a-b) for most habitat variables but not all (Figure 5a). Morning movements were uphill (Figure 4b), towards more convex topography (Figure 5b) and at increasing distance to streams (Figure 5c). Afternoon movements were directed towards easterly aspects (Figure 5a), steeper slopes (Figure 4c) in valley landforms (Figure 5b) and towards streams (Figure 5c). At dusk, movements were strongly upslope (Figure 4b), out of drainages and towards foraging areas (lower canopy cover, greater distance to hiding cover, increased herbage production, closer to roads and more southerly and westerly aspects) (Figures 2a-c, 3a-b, 5a). For mule deer, these cycles were largely absent from the data (cf. Figures 2a, 2d), and considerable variation was observed among the individual deer in terms of their habitat use patterns.

The daily patterns of habitat use and movements changed among monthly intervals for elk and, to a lesser extent, mule deer. Canopy, distance to hiding cover (Figures 2a-b) and distance to open roads (Figure 3b) changed across monthly intervals in terms of daily amplitudes and average value. Seasonal differences were most evident between late spring (15 April to 14 June) and early summer (15 June to 14 August). The changes were best explained in terms of

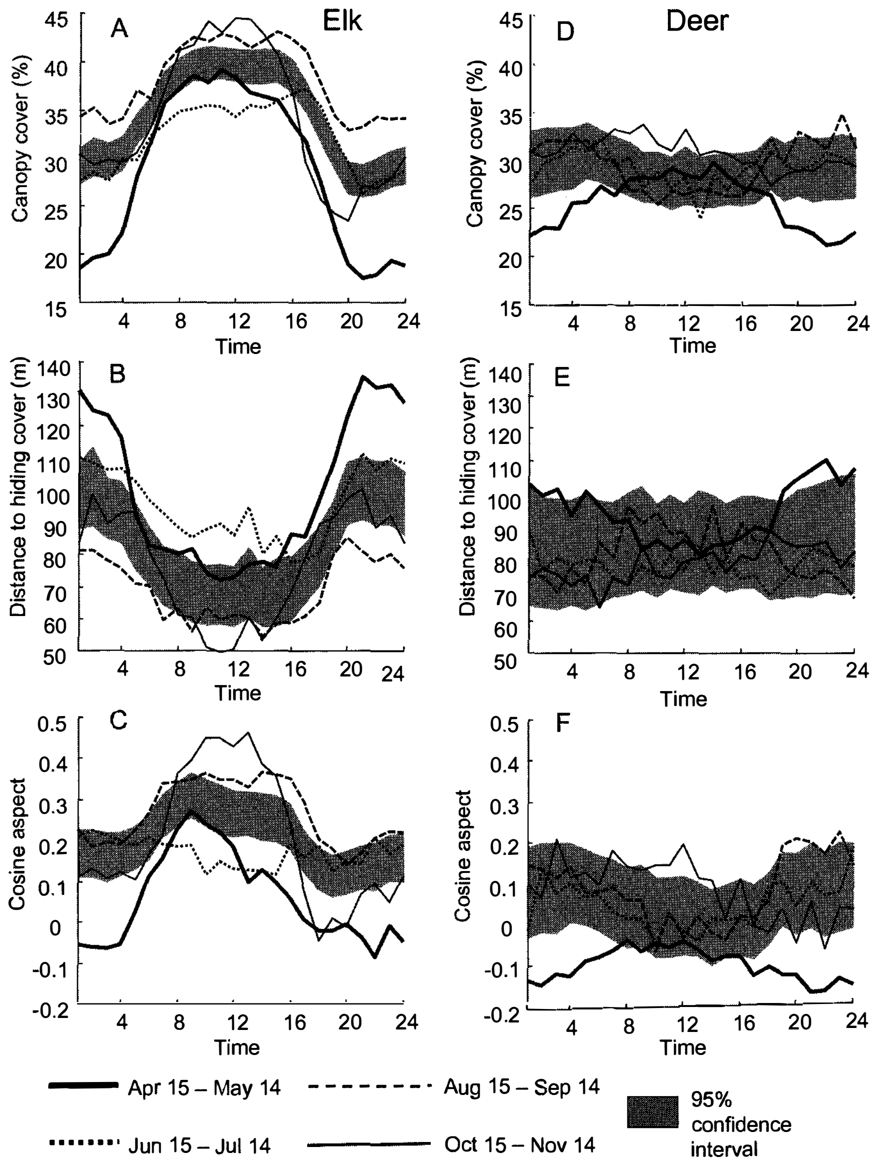


Figure 2. Means of habitat variables canopy cover, distance to cover and cosine aspect by hour and monthly interval for elk (left column) and mule deer (right column). Values plotted are means across animals. For clarity, only 4 of the 71-month time intervals are shown (intervals 1, 3, 5 and 7). Shaded area is the grand mean bounded above and below by 2 pooled, within-interval standard deviation (SD). Pooled within-interval SDs represent the average SD within all 7 monthly intervals studied. Figure is from Ager et al. (2003).

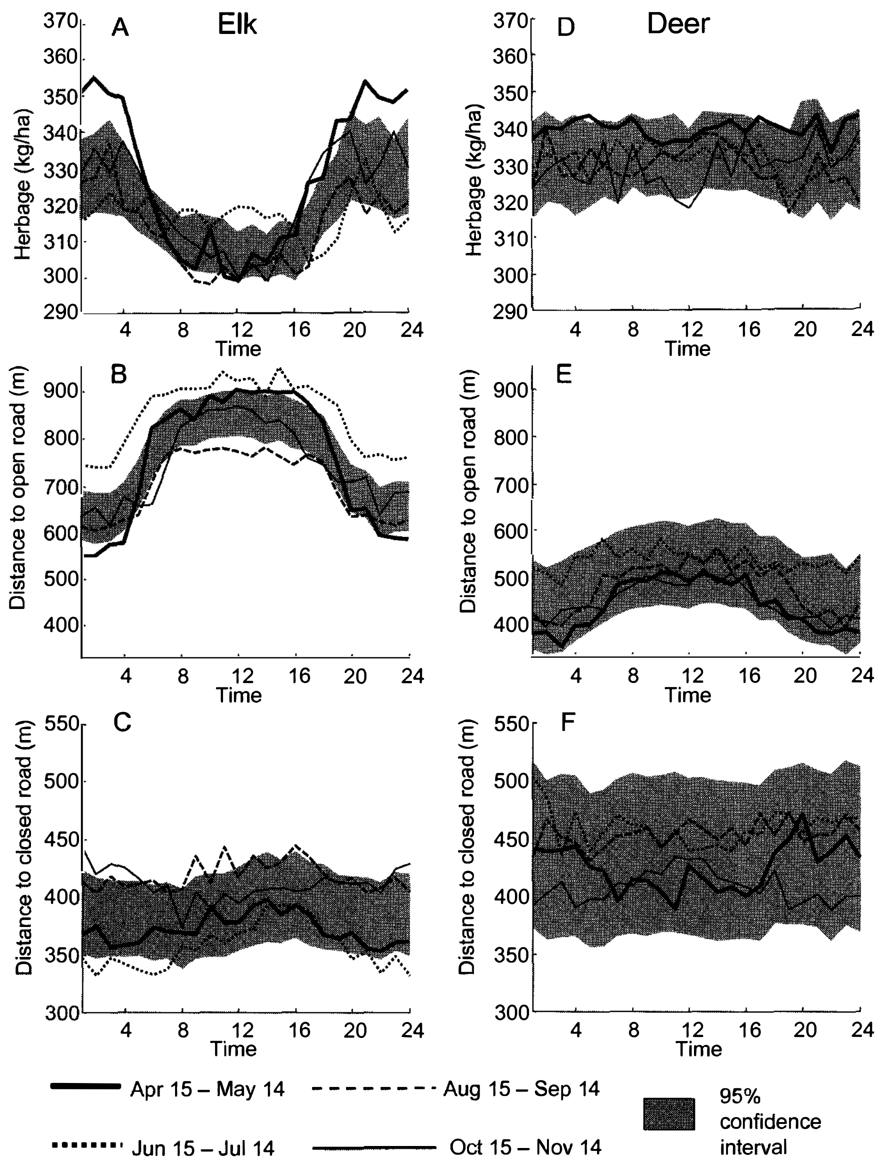


Figure 3. Means of habitat variables herbage production, distance to open road and distance to closed road by hour and montly interval for elk (left column) and mule deer (right column). See Figure 2 for additional explanation. Figure is from Ager et al. (2003).

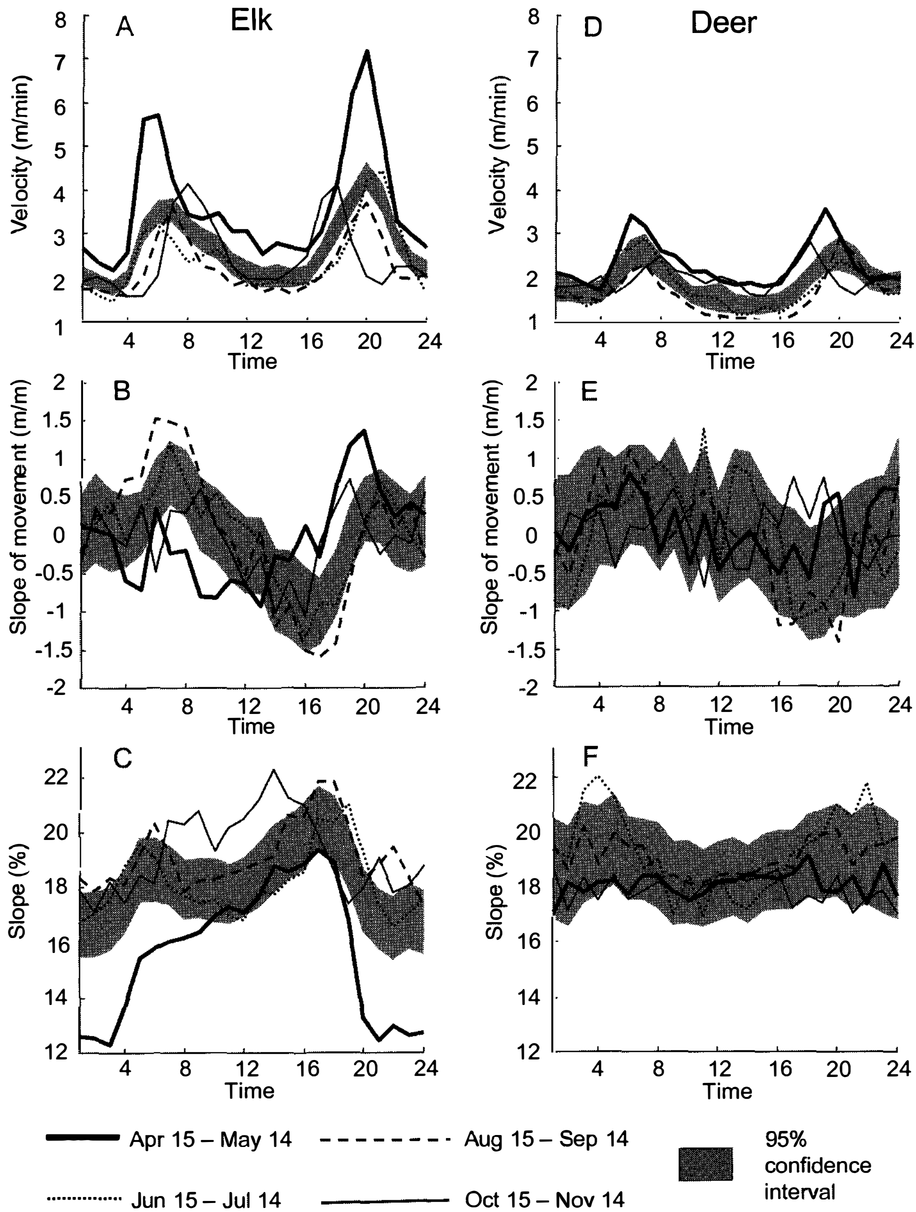


Figure 4. Means of habitat and movement variables velocity, slope of movement and percentage slope by hour and montly interval for lek (left column) and mule deer (right column). See Figure 2 for additional explanation. Figure is from Ager et al. (2003).

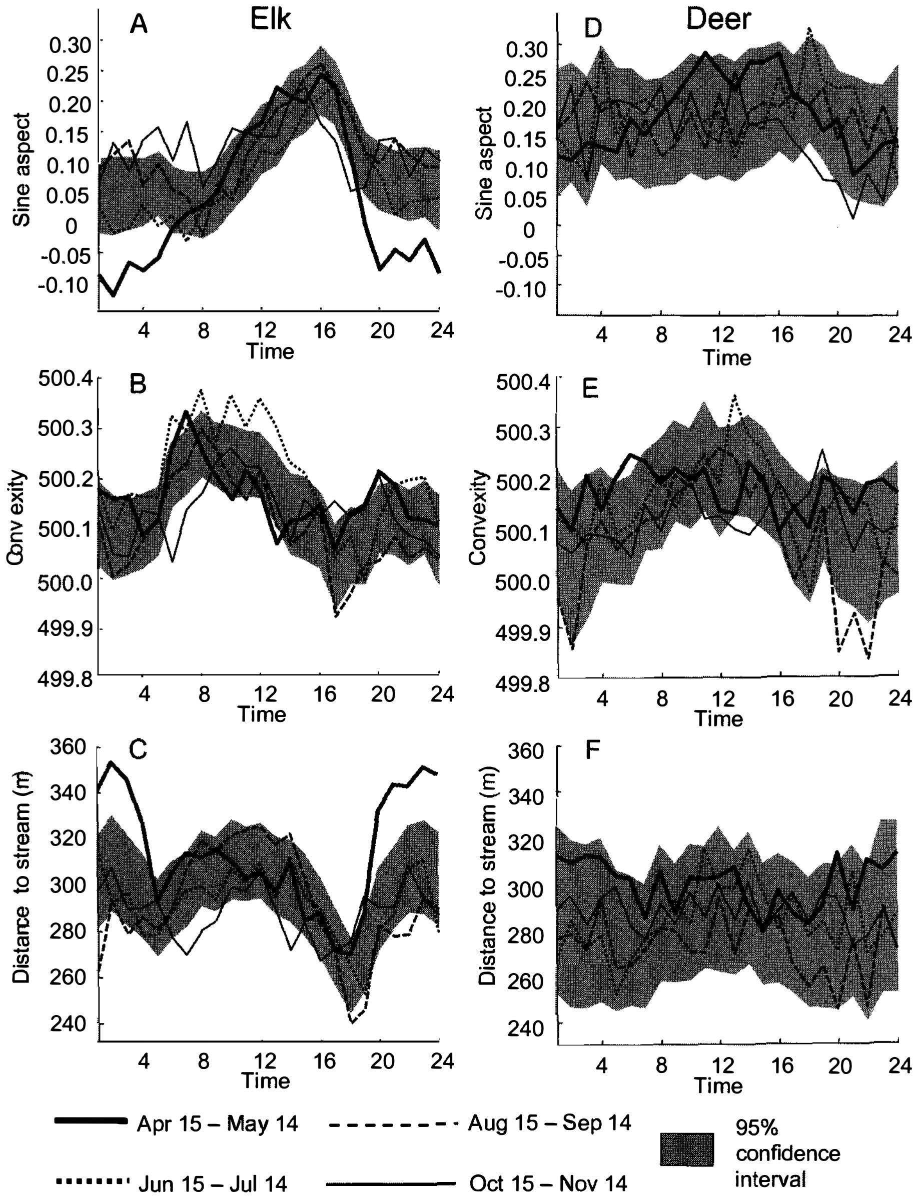


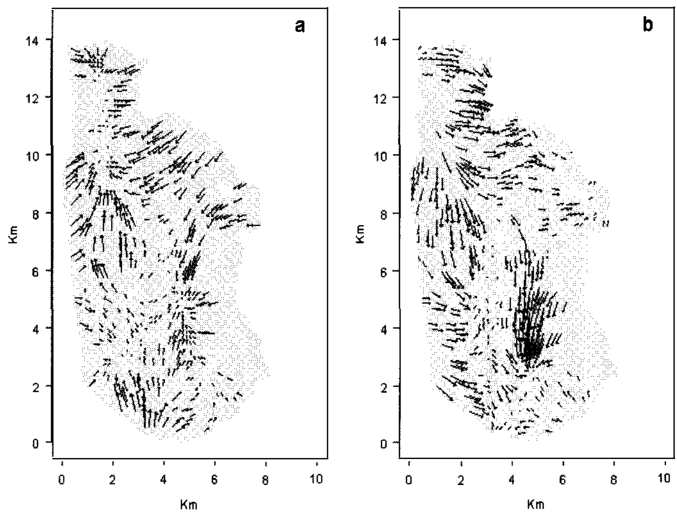
Figure 5. Means of habitat variables sine aspect, convexity and distance to stream by hour and monthly interval for elk (left column) and mule deer (right column). See Figure 2 for additional explanation. Figure is from Ager et al. (2003).

forage phenology at Starkey (Skovlin 1967). High daily amplitudes for velocity and habitat variables in the spring (15 April to 14 May) and autumn (15 October to 14 November) reflected rapid movements to highly preferred meadows at Starkey that produce desired forage in early spring and in autumn after the first substantial rains (Skovlin 1967). The lower velocity and dampened daily cycles during summer reflected a higher use of forested areas throughout the day and night, which are preferred due to their higher midsummer forage production (Edgerton and Smith 1971, Holechek et al. 1982, Unsworth et al. 1998).

Spatial Patterns of Movements

The results from Ager et al. (2003) motivated a number of questions about how elk and mule deer movements are spatially organized on the Starkey landscape. For instance, are there movement corridors between the different habitats? What is the effect of edge on movements? Are movements organized after topography or after other features in Starkey? Are there areas of high and low speed? Are there habitat features that impede movements? Are the dusk and dawn movements reciprocal? To address these questions we explored movement patterns and found that, by smoothing movement vectors with nonparametric regression (Brillinger et al. 2002, 2004; Preisler et al. 2004) and by plotting these on a rendered terrain of Starkey, we could address a number of questions related to the spatial component of movement. First, we observed that mule deer movement vectors appeared mostly random and, thus, had little or no spatial organization, perhaps due to the spatial resolution of the telemetry data and the sampling frequency (Ager et al. 2003). Elk movement vectors for the midsummer (15 July to 15 August) season also appeared weak, reflecting the lower movement rates during this season (Ager et al. 2003). In contrast, elk movement vectors for spring (15 April to 15 June) showed a strong directional component, especially for crepuscular periods (Figures 6a, b). Vector fields also revealed a dendritic pattern of movement (Forman 1995:270) in areas where there is significant topographic relief (Figure 1a). The consequence of the dendritic movement behavior appeared to split the elk into discrete movement cohorts in the project area. Subsequent analyses of movements relative to drainage directions showed a statistically significant association (Kie et al. in press). The effect of Meadow Creek Canyon (Figure 1b) on movement vectors is readily apparent, where movement vectors do not cross the canyon (Figures

Figure 6. Gradient vector fields for 0500 (a) and 1900 (b) hours estimated from the spring telemetry data. Arrows were plotted where at least one elk was observed. The unit vector corresponds to 0.8 kilometers per hour. The plots illustrate the crepuscular transitions between daytime and nighttime habitat. Figure is from Preisler et al. (2004).

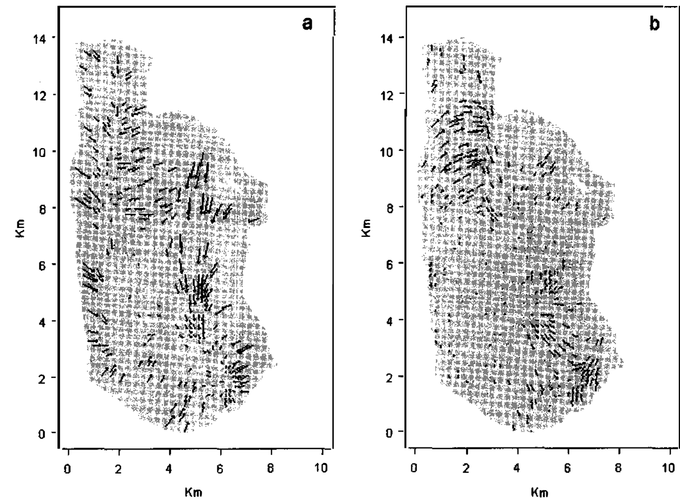


6a, b). Bear Creek had similar effects. Movements also appeared to be reciprocal between dawn and dusk, i. e., the direction of arrows at dawn were opposite of those at dusk at most spatial locations. The dusk movements to grasslands (Figure 6b) appeared stronger and spatially focused as compared to the dawn movements (Figure 6a). The seasonal changes in movement noted in Ager et al. (2003) were also apparent in the estimated vector fields (cf. Figures 6, 7). The plots for summer showed markedly diminished movement vectors; although, there still is some evidence of the elk's avoidance of steep terrain.

Linking Spatial Movement Patterns with Habitat Preferences

Movement vectors were related to habitat variables to explain the spatial cycles of movement on the basis of elk behavior. This work was motivated by the concept of potential fields applied to animal movement (Brillinger et al. 2001) and considers both the stochastic and correlated components of animal movement behavior. A potential field may be visualized by imagining a ball rolling (the animal) around in the interior of a bowl (potential surface), as the bowl is being shaken (random component). In our case, the potential field is a complex surface representing attraction and repulsion to specific habitat features at different times of the day. The potential field is built from a set of additive potential functions for each habitat variable that affects movement. The individual, potential functions

Figure 7. Gradient vector fields for 0500 (a) and 1900 (b) hours estimated from the summer telemetry data. Movement vectors show diminished velocities in summer, as compared to spring, and qualitatively different movement patterns. Figure is from Preisler et al. (2004).



describe movements (attraction versus repulsion) as a function of distance to habitat features at specific times of day. Movements that are seemingly random, like foraging paths in a meadow, or movements that cannot otherwise be explained with environmental covariates are included as stochastic terms in the model. The reader is referred to Brillinger et al. (2004) and Preisler et al. (2004) for details.

For the initial model, we used the data from Ager et al. (2003) for elk in spring (15 April to 15 May), and we focused on the crepuscular movements between foraging and resting areas. We tested a number of habitat covariates and found four that had significant influence on movement vectors, these being distance to security areas, distance to foraging meadows, distance to steep slopes and distance to streams (Preisler et al. 2004). A composite potential field was estimated for the crepuscular periods (Figures 8a, 9a) and, when plotted, showed areas of attraction corresponding to peak foraging and resting areas. When compared to the elk distributions two hours later (Figures 8b, 9b), there is a general agreement between the potential field and the elk densities. The potential functions for each habitat variable also showed specific relationships between the level of attraction versus repulsion and distance (see Preisler et al. 2004). Thus, a model showing habitat selection and habitat transitions was developed from movement vectors and habitat covariates. This work is a first step towards building an empirically based stochastic movement model that accounts for both the spatial and temporal dynamics of movement and habitat preferences. Work

Figure 8. (a) Estimated potential surface for 0500 hours during spring. (b) Kernel density estimate (elk/km^2) for observed elk locations between the hours of 1000 and 1300. Most of the areas of low potential (attraction regions) at dawn correspond with regions of high elk density around noon. Figure is from Preisler et al. (2004).

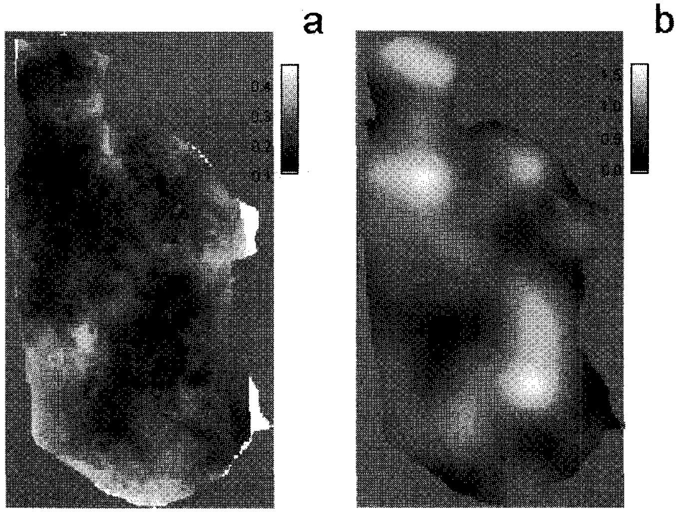
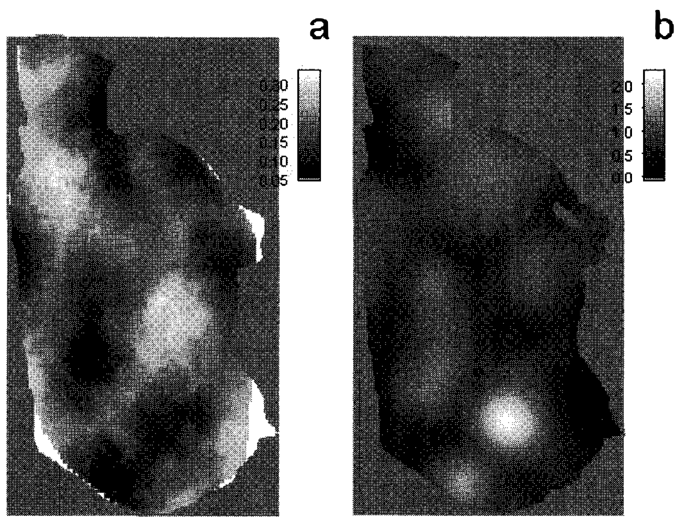


Figure 9. (a) Estimated potential surface for 1900 hours during spring. (b) Kernel density estimate (elk/km^2) for observed elk locations between the hours of 2200 and 0100. Areas of low potential (attraction regions) at dusk correspond with regions of high elk density around midnight. Figure is from Preisler et al. (2004).



is ongoing to incorporate finer-scale navigational cues that elk use during the crepuscular transitions that are relative to specific animal positions. This includes modeling the dendritic movement patterns (Forman 1995) that ungulates exhibit on steep terrain. Here, the potential function might change relative to the animal's position. Using potential fields to model fine-scale foraging movements like those among swards and adjacent feeding stations presents further challenges.

Management Implications

Our description of daily and seasonal cycles of habitat use, spatial patterns of movement vectors, and linkage between habitat use and movements adds to the interpretation of previous studies at Starkey (Johnson et al. 2000, Rowland et al. 2000). The daily and seasonal cycles of habitat use exhibited by elk and, to a lesser extent, mule deer show the dynamic nature of habitat-use patterns and underscore the importance of movement studies. The movement analyses were limited to summer range conditions, and the consideration of movement on larger landscape, including seasonal migrations offer additional challenges and insights into ungulate behaviors (Bergman et al. 2000, Johnson et al. 2002). Due to the sampling interval of the Starkey data (1–2 hours), telemetry location error and perhaps the scale of the habitat data, we could not explain finer scale movements outside of the crepuscular periods. Modeling movements during the foraging periods would require additional consideration of variables, like forage intake rate, forage biomass and other foraging behavior factors (Shipley and Spalinger 1995).

Current elk and mule deer habitat models do not consider interpatch landscape movements in their measurement of habitat quality. The influence of topographic pattern on crepuscular movements between preferred resting versus ruminating and foraging habitats was manifested in movement patterns that were aligned with drainages. Thus, the observed elk density in a given foraging habitat was dependent on the ability of elk to connect to suitable security areas using movements that parallel the topographic pattern of drainages. We hypothesize that the fit of Starkey resource selection functions on other landscapes will be influenced by the spatial arrangement of habitats on the landscape and the presence of suitable movement corridors to link daytime and nighttime habitats. For instance, a canyon that lies between highly desirable foraging and security areas will degrade the resource value of these two habitats due to the lack of a suitable connection between them. Further work is needed to better understand how movement patterns might influence elk and mule deer densities in specific habitats and whether these considerations are problematic in the extrapolation of resource selection functions.

The dynamic nature of habitat use and movements by elk and, to a lesser extent mule deer, has important implications for the development and application of habitat suitability models. Diel changes in habitat use needs to be considered when telemetry data are used to estimate resource selection functions.

The need for empirical methods to analyze movements will rapidly grow as the rapid advances in automated telemetry systems materialize and as large telemetry data sets are generated. New GPS telemetry with higher accuracy and sampling frequency will enable significant advances in our ability to build movement models that represent a broad range of ungulate behavior and spatio-temporal scales.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use of Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:687–707.

Beier, P., and D. R. McCullough. 1990. Factors influencing white-tailed deer activity patterns and habitat use. *Wildlife Monographs*. 109:1–51.

Bergman, C. M., J. A. Schefer, and S. N. Luttich. 2000. Caribou movement as a correlated random walk. *Oecologia*. 123:364–74.

Brillinger, D. R., H. K. Preisler, A. A. Ager, and J. G. Kie. 2001. The use of potential functions in modeling animal movement. In *Data Analysis from Statistical Foundations*, ed. A. K. Md. E. Saleh, 369–86. Huntington, New York: Saleh Nova Science Publishers, Inc.

———. 2004. An exploratory data analysis (EDA) of the paths of moving animals. *Journal of Statistical Planning and Inference*. 122:43–63.

Brillinger, D. R., H. K. Preisler, A. A. Ager, J. G. Kie, and B. S Stewart. 2002. Employing stochastic differential equations to model wildlife motion. *Bulletin of the Brazilian Mathematical Society*. 33:93–116.

Collins, W. B., and P. J. Urness. 1983. Feeding behavior and habitat selection of mule deer and elk on northern Utah summer range. *Journal of Wildlife Management*. 47:646–63.

Craighead, J. J., F. C. Craighead, Jr., R. L. Ruff, and B. W. O'Gara. 1973. Home ranges and activity patterns of nonmigratory elk of the Madison drainage herd as determined by biotelemetry. *Wildlife Monographs*. 33:1–50.

Edgerton, P. J., and J. G. Smith. 1971. *Seasonal forage use by deer and elk on the Starkey Experimental Forest and Range, Oregon, research paper PNW-RP-112:1–12*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Forman. R. T. T. 1995. *Land mosaics*. Cambridge, United Kingdom: Cambridge University Press.

Etzenhouser, M. T., M. K. Owens, D. E. Spalinger, and S. B. Murden. 1998. Foraging behavior of browsing ruminants in a heterogeneous landscape. *Landscape Ecology*. 13:55–64.

Green, R. A., and G. D. Bear. 1990. Seasonal cycles and daily activity patterns of Rocky Mountain elk. *Journal of Wildlife Management*. 54:272–9.

Gross, J. E., C. Zank, N. T. Hobbs, and D.E. Spallinger. 1995. Movement rules for herbivores in spatially heterogeneous environments: Responses to small scale pattern. *Landscape Ecology*. 10:209–17.

Holechek, J. L., M. Vavra, J. Skovlin, and W. C. Krueger. 1982. Cattle diets in the Blue Mountains of Oregon II. Forests. *Journal of Range Management*. 35:239–42.

Johnson, C. J., K. L. Parker, D. C. Heard, and M. P. Gillingham. 2002. Multiscale behavioral approach to understanding the movements of woodland caribou. *Ecological Applications*. 12:1,840–60.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of elk and mule deer in spring. *Journal of Wildlife Management*. 64:685–97.

Kie, J. G., A. A. Ager, H. K. Preisler, and R. T. Bowyer. In press. Landscape-level movements of North American elk (*Cervus elaphus*): Effects of habitat patch structure and topography. *Landscape Ecology*.

Leckenby, D. A. 1984. Elk use and availability of cover and forage habitat components in the Blue Mountains, northeast Oregon, 1976–1982. *Oregon Department of Fish and Wildlife, Wildlife Research Report*. 14:1–40.

Mackie, R. J. 1970. Range ecology and relations of mule deer, elk, and cattle in the Missouri River Breaks, Montana. *Wildlife Monographs*. 20:1–79.

McCorquodale, S. M., K. J. Raedeke, and R. D. Taber. 1986. Elk habitat use patterns in the shrub steppe of Washington. *Journal of Wildlife Management*. 50:664–9.

Preisler, H. K., A. A. Ager, B. K. Johnson, and J. G. Kie. 2004. Modeling animal movements using stochastic differential equations. *Environmetrics*. 15:643–57.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Shipley, L. A., and D. E. Spalinger. 1995. Influence of size and density of browse patches on intake rates and foraging decisions of young moose and white-tailed deer. *Oecologia*. 104:112–21.

Skovlin, J. M. 1967. *Fluctuations in forage quality on summer range in the Blue Mountains, research paper PNW-RP-44:1–20*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Unsworth, J. W., L. Kuck, E. O. Garton, and B. B. Butterfield. 1998. Elk habitat selection on the Clearwater National Forest, Idaho. *Journal of Wildlife Management*. 62:1,255–63.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:455–74.

Spatial and Temporal Interactions of Elk, Mule Deer and Cattle

Priscilla K. Coe
Oregon Department of Fish and Wildlife
La Grande

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Kelley M. Stewart
University of Alaska, Fairbanks
Fairbanks

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Introduction

Elk (*Cervus elaphus)*, mule deer (*Odocoileus hemionus*) and cattle share millions of acres of public and private forests and rangelands across the western United States and Canada. These three species have important social, ecological and economic values. Understanding their interspecific interactions may clarify two recurring issues in their management: competition for food and competition for space, both may result in decreased animal fitness. Animal unit equivalents (AUEs) among these three species have been based on equivalent body mass (Society for Range Management 1989), whereby one cow is equivalent to two and one-half elk or to six mule deer. Hobbs and Carpenter (1986) argue that AUEs should be based on dietary overlap, and the argument can be extended to include spatial overlap. Consequently, the ecological impact of one species on the landscape may not be equivalent to another species. Furthermore, allocating forage becomes challenging if managers do not clearly understand the spatial and dietary overlap among these three species. Accurate predictions of ungulate distributions over time and space may help managers regulate densities and understand effects of specific ungulates on ecosystem processes.

Many factors may influence the seasonal distribution of domestic and native ungulates, including vegetation composition, topography and distance to water (Mueggler 1965, Leckenby 1984, Peek and Krausman 1996, Wisdom and Thomas 1996). Ungulates also distribute themselves in response to disturbances, such as traffic (Rowland et al. 2000), hunting (Johnson et al. 2004) and logging (Pederson et al. 1980). In addition, there may be inter- and intraspecific influences on animal distribution. These interactions may produce different patterns of distribution at different scales of investigation (Bowyer et al. 1997). This paper summarizes studies of ungulate interactions at Starkey Experimental Forest and Range (Starkey) in terms of how interactions among ungulate species may affect animal distributions over space and time.

Past studies of interspecific interactions among elk, mule deer and cattle have indicated potential competition (Skovlin et al. 1968, Mackie 1970, Dusek 1975, Knowles and Campbell 1982, Nelson 1982, Austin and Urness 1986, Wallace and Krausman 1987, Loft et al. 1991, Peek and Krausman 1996, Wisdom and Thomas 1996, Lindzey et al. 1997), while others have inferred commensalism (Anderson and Scherzinger 1975, Frisina and Morin 1991, Peek and Krausman 1996). Competition occurs when individuals or species use similar resources that are in short supply. Inadequate forage quality or quantity may decrease nutritional planes such that population performance of one or more species decreases (Birch 1957, Putnam 1996). In contrast, commensalism occurs when one species benefits from association with another species, while the other species is unaffected (Martin 1990).

Simple descriptive approaches to interactions among large herbivores result in inherent difficulties (Painter 1980). Overlapping distributions could be evidence for competition or dependence. Nonoverlap could be an expression of active avoidance or ecological separation, which occurs when two species evolved together. Although sexual segregation, or spatial separation of sexes outside the mating season, is nearly ubiquitous among polygamous ungulates (Bowyer 1984, McCullough et al. 1989, Scarbrough and Krausman 1998, Kie and Bowyer 1999), our paper concentrates on distribution of females. Comparisons of distribution with and without one ungulate species present during the same season and on the same ground should help to illuminate whether competition is occurring. Diet studies of ungulates during different seasons, both with and without prior grazing also should help to establish competitive interactions.

Studies of ungulate interactions began at Starkey in 1954 (Skovlin et al. 1968). Deer were summer-long residents while elk were spring (May to June) and fall migrants through the area. The investigators monitored use by deer and elk in two replicates of pasture systems supporting light (40 acres per animal unit [16 ha/animal unit]), moderate (30 acres per animal unit [12 ha/animal unit)]) and heavy (20 acres per animal unit [8 ha/animal unit]) cattle grazing. Over a period of 11 years (1954–1964), elk and mule deer use was measured from pellet groups and plant utilization surveys. They found that both elk and mule deer used pastures not grazed by cattle more than any of the cattle-grazed pastures, with use declining as cattle stocking rate increased. They found less of an effect by cattle on mule deer than on elk, indicating possible competition between elk and cattle only.

Methods

In 1989 an ungulate-proof fence was built around Starkey for long-term studies of elk, mule deer and cattle (Rowland et al. 1997). Three studies of spatial interactions among elk, mule deer and cattle took place in the enclosed areas. The largest-scale study was conducted in main study area (19,026 acres [7,700 ha]) during spring, when only elk and mule deer were present (Johnson et al. 2000). In a smaller (5,930-acre [2,400-ha]) subpasture of Main Study Area, Smith-Bally, responses of elk and mule deer to cattle were investigated during early and late summer, and elk and mule deer distributions were analyzed with and without cattle present (Coe et al. 2001). Finally, in the 3,459-acre (1,400-ha) Northeast Study Area, spatial and temporal competitive interactions among all three species were documented (Stewart et al. 2002). Scale of analysis was defined by spatial extent (the size of the study area), spatial grain (the smallest spatial unit used in analysis), temporal extent (the time span of the study) and temporal grain (the smallest unit of time used in analysis; Table 1).

In the Main Study Area, we investigated interactions of elk (n = 88) and mule deer (n = 45) during spring, when cattle were not present (Johnson et al. 2000). Resource selection functions were estimated for both species. A resource selection function is a value for a resource unit that is proportional to the probability of the unit being used by an animal (Manly et al. 1993). Resource units

Table 1. Scales of species interaction analyses at Starkey Experimental Forest and Range, northeast Oregon

Measure of scale	Elk and mule deer[a]	Elk, mule deer and cattle[b]	Elk, mule deer and cattle[c]
Spatial extent acre (ha)	19,012 (7,700)	19,012 (7,700[d]), 5,926 (2,400)[e], 5,926 (2,400)[f],	3,457 (1,400)
Spatial grain acre (ha)	0.22 (0.09)	5,926 (2,400)[d], 19 (7.7)[e], 0.22 (0.09)[f]	5.55 (2.25)
Temporal extent (yrs)	4	2	3
Temporal grain (mean number of days	56.5	27	7[g], 0.25[h]

[a] Spring (Johnson et al. 2000)
[b] Summer (Coe et al. 2001)
[c] Spring, summer and fall (Stewart et al. 2002)
[d] Pasture level analysis
[e] Plant community level analysis
[f] Pixel level analysis (resource selection functions)
[g] Seven-day model
[h] Six-hour model

were represented as 98.4 by 98.4 feet (30 x 30 m) cells. A resource selection function may be mapped as a probability of use by the species across a landscape. To investigate interspecific interactions between mule deer and elk, the probability of use for one species was used as a variable to estimate a resource selection function for the other species.

In Smith-Bally pasture we investigated responses of elk and mule deer to cattle at several spatial grains (Coe et al. 2001; Table 1). We analyzed counts of animal locations (n = 25–55 elk, 12–36 mule deer and 35–42 cattle) at the pasture and habitat level within the pasture. We estimated resource selection functions at the pixel level. To examine species use at the pasture level, we used relative counts of elk and mule deer locations within the pasture versus the rest of main study area during years when cattle were present and the same days during years when cattle were absent. A temporally correlated Poisson regression accounted for autocorrelation among days and nonnormally distributed count data. The same process was used to investigate whether elk and mule deer changed their use of four major habitat types within the Smith-Bally pasture when cattle were present, compared to when they were absent. Finally, resource selection functions were estimated for both elk and mule deer at the pixel level when cattle were present and when they were absent in Smith-Bally pasture. Cattle resource selection functions were also estimated for the same time periods and at the same spatial grain.

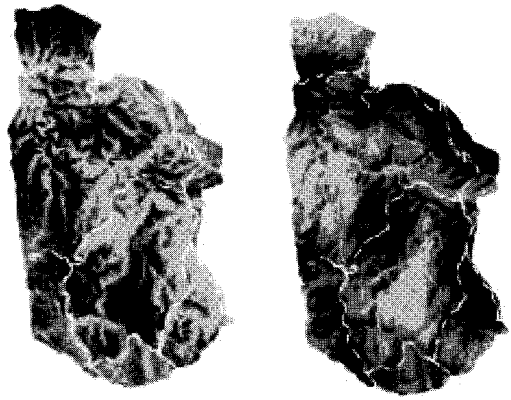
In the Northeast Study Area, Stewart et al. (2002) investigated the relative influence of interference versus exploitive competition among elk, mule deer and cattle, after accounting for niche partitioning. Stewart et al. (2002) used multivariate analysis of variance (MANOVA) to examine seasonal niche partitioning among these three species of large herbivores by examining the interactions of animal locations with random locations ($n = 465$) for independent variables associated with habitat selection (e. g. habitat type, distance to water, distance to roads, slope and aspect). Habitat variables included in MANOVA models had been selected previously from species-specific logistic regression to determine which variables were important to that species (Stewart et al. 2002). Multiple regression was used to examine competition among the three species while accounting for niche partitioning, by including habitat variables that had been selected from logistic regression being held in the model, and the number of sympatric species and conspecifics within a 5.55-acre (2.25-ha) area surrounding a focal animal location. Stewart et al. (2002) used two temporal windows to examine the relative effects of interference and exploitive competition in those multiple-regression models. One window was a 6-h temporal window to investigate interference competition, based on the number of sympatric animals that were present within the 5.55-acre (2.25-ha) window plus or minus 3 hours of a focal animal location. And, the other window was a 7-day temporal window to examine effects of exploitive competition, based on the number of animals that were present 7 days prior to the focal animal location. Finally, Stewart et al. (2002) compared movements of mule deer and elk 2 weeks before and after cattle were introduced to the study area (early summer) and removed (autumn) to examine potential competitive displacement of mule deer and elk by cattle.

Results

Elk and Mule Deer—Spring

In Main Study Area during spring, elk were found on flatter and more westerly aspects than mule deer, and they were found further from roads with high (more than four vehicles per day) and medium (one to four vehicles per day) traffic (Figure 1). Several of the habitat attributes that mule deer selected were opposite from those elk selected; for example, mule deer selected steeper and northeast-facing aspects, and they selected sites closer to high and medium

Figure 1. Resource selection function values for elk and mule deer during spring at Starkey Experimental Forest and Range. Light to dark shading indicates increasing proportion of use by each species (from Johnson et al. 2000).



traffic (Johnson et al. 2000). When the mule deer resource selection function was incorporated into the elk resource selection function, and vice versa, the resulting coefficients for the incorporated resource selection functions were negative and significant, indicating that each species selected resources that the other did not. The magnitude of the elk resource selection function in the mule deer model was greater, however, indicating that mule deer were more strongly affected by elk than elk were by mule deer. Further investigation revealed that mule deer use of five habitat types ranked according to elk resource selection function (RSF) was inverse of elk selection. Elk habitat selection within the ranked mule deer resource selection function, however, displayed no pattern. This is further indication that mule deer may have been avoiding elk more than elk were avoiding mule deer.

Elk, Mule Deer and Cattle—Summer

At the pasture level, elk occurred less frequently in Smith-Bally pasture when cattle were introduced, both in early summer and late summer (Figure 2, top). The elk that stayed in the pasture when cattle were present shifted their use of the ponderosa pine/Douglas fir type as a result of cattle (Figure 2, bottom). In early summer, elk were displaced from the ponderosa pine/Douglas fir habitat by cattle. In late summer, elk were displaced into the ponderosa pine/Douglas fir habitat by cattle (Coe et al. 2001). Forage in this habitat was most palatable and nutritious in early summer; consequently, elk likely were negatively affected by this displacement. At the pixel level, early summer resource selection functions for elk when cattle were present were significantly different from elk resource

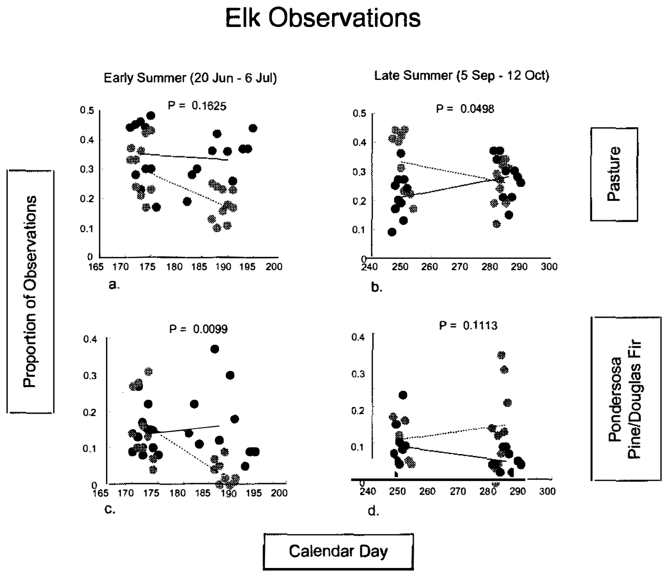


Figure 2. Comparison of the proportion of elk locations in Smith-Bally pasture and in the ponderosa pine/Douglas fir plant community within that pasture, when cattle were present (grey circles, n = 20 days) versus when cattle were absent (black circles, n = 20 days), Starkey Experimental Forest and Range, northeastern Oregon, 1993–1996 (from Coe et al. 2001).

selection functions when cattle were absent. Conversely, late summer resource selection functions for elk when cattle were present were similar to elk resource selection functions when cattle were absent (Coe et al. 2001). In early summer, elk selection for five habitat variables differed if cattle were absent; they selected for sites with gentler slopes, less convex topography, lower canopy, closer to edge of forest stand and further from roads with low traffic rates. In late summer, elk selected denser canopy when cattle were absent; otherwise, elk resource selection functions did not differ based on presence of cattle (Coe et al. 2001). When we included the cattle resource selection function in the elk models, we found that cattle could be used as a predictor of elk distribution in some conditions. With cattle absent, elk selected some of the same resources that cattle select in early spring. When cattle were present, however, elk selected different resources and were spatially separate from cattle. Conversely, in late summer elk resource selection functions were similar to those of cattle, regardless of cattle presence.

Mule deer reduced use of the Smith-Bally pasture in late summer when cattle were introduced but did not change use in early summer with regard to cattle presence. Mule deer use of the ponderosa pine/Douglas fir habitat type was opposite that of elk; mule deer were probably responding to elk movements rather than cattle movements. We observed mule deer changes in habitat use to be opposite those of elk in three out of four season/year combinations. Mule deer resource selection functions were not affected by the presence of cattle.

Elk, Mule Deer and Cattle—Spring, Summer and Fall

In Northeast Study Area, habitat selection differed among seasons for elk, mule deer and cattle. Wilks' lambda ($P = 0.015$) revealed a species-by-location (used, random) interaction that indicated differences in selection of some habitat variables among species (Stewart et al. 2002). Bivariate plots of 95-percent confidence intervals indicated that cattle differed from mule deer and elk by avoiding steeper slopes and high elevations, particularly during spring and summer; by contrast, mule deer and elk overlapped in use of slope and elevation, but they partitioned use of vegetative communities (Stewart et al. 2002, Figure 3).

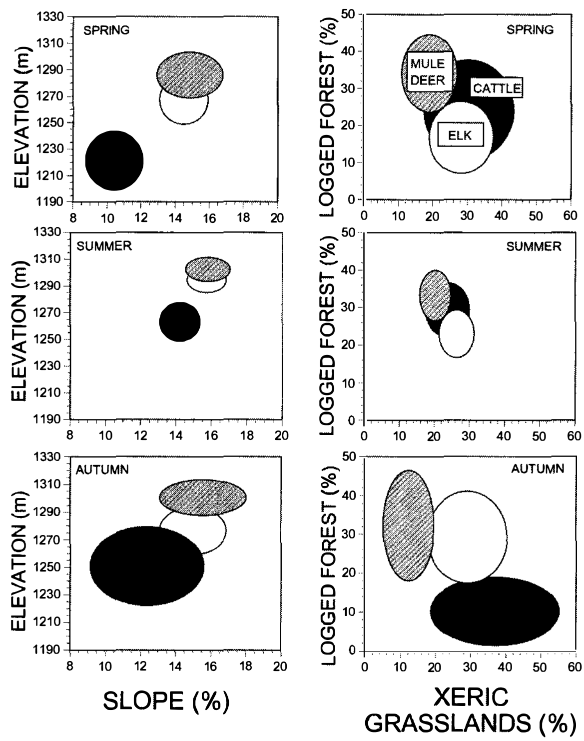


Figure 3. Bivariate plots of niche partitioning based on elevation and slope (left) and on logged forest and xeric grasslands (right). Ellipses are 95 percent confidence interval for cattle, elk and mule deer across seasons on the Starkey Experimental Forest and Range, northeastern Oregon, 1993–1995 (from Stewart et al. 2002).

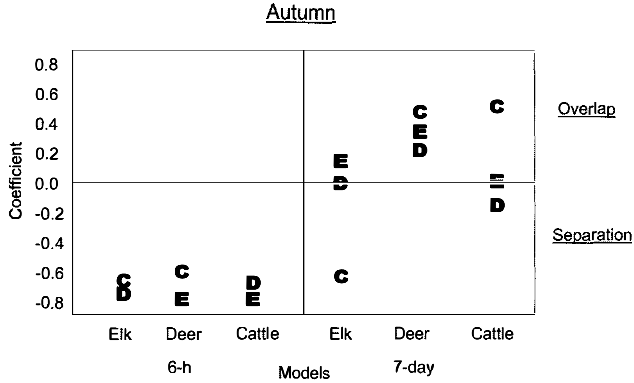
Spatial avoidance among mule deer, elk and cattle was stronger for the 6-hour models than for the previous 7 days; coefficients of association from multiple regressions were strongly negative for the 6-hour models, indicating strong avoidance among the three species during all seasons (Stewart et al. 2002). Stewart et al. (2002) observed an interaction of season by species by treatment (Wilks' lambda, $P = 0.046$) and an interaction of species by treatment (Wilks' lambda, $P = 0.002$) for use of slope and elevation by elk and mule deer following introduction and removal of cattle during spring and autumn. Because those interactions were significant, the authors analyzed species (elk and mule deer) and seasons separately. Elk moved to higher elevations following introduction of cattle during spring, and they returned to lower elevations following removal of cattle in autumn (Stewart et al. 2002). Conversely, mule deer moved to lower elevations following introduction of cattle during spring, possibly in response to displacement of elk following introduction of cattle (Stewart et al. 2002). The addition of cattle to the northeast did not affect the slope of habitats used by mule deer during spring; although, deer moved to more level ground following removal of cattle in autumn (Stewart et al. 2002).

Resource partitioning for the 6-hour models was interpreted as interference competition, and 7-day partitioning was interpreted as possible interference or exploitive competition. In autumn (September 15 to October 15), coefficients were strongly positive, compared to spring and summer, in all of the 7-day models, indicating spatial overlap among all species (Figure 4). The exception, during autumn, was the elk model, where elk continued to avoid cattle. Mule deer more strongly avoided elk than elk avoided mule deer as evidenced by nonsignificant mule deer variables in the elk models, but highly significant elk variables in the mule deer models. This occurred for all seasons except during the earliest period (June 15 to June 30).

Discussion

Three separate investigations yielded similar information about spatial relationships of elk, mule deer and cattle within Starkey at three different scales. Spatial separation was noted for elk and mule deer and for elk and cattle at all scales analyzed during spring and early summer. At the largest scale, the Main Study Area, remarkable spatial separation was seen for elk and mule deer (no cattle present) in spring, so much so that maps of resource selection functions for each species were nearly mirror images. In Smith-Bally pasture spatial

Figure 4. Standardized competition coefficients, determined from weighted multiple regressions for elk, mule deer and cattle during autumn (September 15 to October 15) in Starkey Experimental Forest and Range. Number of conspecifics (C = cattle, D = mule deer and E = elk) plus the focal animal,was the dependent variable for 6-hour and 7-day models (Stewart et al. 2002).



separation of elk and mule deer was evidenced in that mule deer response was opposite to that of elk in their use of plant communities. In Northeast Study Area, spatial separation between elk and mule deer was maintained in the 5.55-acre (2.25-ha) neighborhood surrounding each focal animal in early summer for both temporal scales analyzed.

Elk and cattle spatial separation occurred in the two studies where cattle were present. Both studies concluded that elk avoid cattle during summer. Both studies also noted more overlap among all ungulates during late summer; spatial overlap of all species during late summer and fall occurred in the two studies that encompassed these seasons. This overlap is indicative of possible exploitive competition between the three species of ungulates as forage resources become depleted later in the grazing season, especially in light of other findings at Starkey. Other Starkey studies have found nutritional deficits of both elk and cattle in late summer (Cook et al. 2004, Holechek et al. 1982). Spatial overlap, indications of nutritional deficits and diet overlap in grand fir (*Abies grandis*) habitats during August (Findholt et al. 2004) implicate competition for resources as a potential limiting factor in ungulate productivity during late summer and fall.

All of the analyses are consistent with the hypothesis of a cascading effect of larger ungulates displacing smaller ones. In both the Smith-Bally and in the Northeast Study Area analyses, cattle displaced elk, and all three studies cited evidence of elk displacing mule deer. If this hypothesis is true (i. e., if larger ungulates choose habitat first), elk could suffer nutritional deficits sooner than cattle, and deer could suffer sooner than elk in a limited-forage situation.

Policy Implications

Following is a list of the implications of this research:

- Cascading effects of larger herbivores choosing resources before smaller herbivores do imply that decisions that change distribution of cattle will likely change distributions of elk and mule deer.
- Cattle are the most easily manipulated and are the largest herbivore in northeastern Oregon; thus, they can be a tool in managing spatial distributions of elk and mule deer.
- Management of ungulate density in late summer and fall (e. g., stocking reductions in areas of high ungulate overlap) could ensure high productivity of both wild and domestic ungulates as forage resources become limited.
- Resource selection functions, which account for interspecific interactions of elk, mule deer and cattle, can predict animal distributions over a landscape and can act as part of a larger model to predict forage removal and animal productivity.
- Estimating animal unit equivalents is dependent on two basic factors—distributional overlap and dietary overlap. Animal unit equivalents cannot be based strictly on body weight; results from these studies indicate spatial separation occurs, effectively discounting the animal unit equivalencies for these three species.

Reference List

Anderson, C. C., and R. J. Scherzinger. 1975. Improving quality of winter forage for elk by cattle grazing. *Journal of Range Management*. 28:120–5.

Austin, D. D., and P. J. Urness. 1986. Effect of cattle grazing on mule deer diet and area selection. *Journal of Range Management*. 39:18–21.

Birch, L. C. 1957. The meaning of competition. *American Naturalist*. 91:5–18.

Bowyer, R. T. 1984. Sexual segregation in southern mule deer. *Journal of Mammalogy*. 62:574–82.

Bowyer, R. T., V. V. Ballenberghe, and J. G. Kie 1997. *The role of moose in landscape processes: Effects of biogeography, population dynamics, and predation*, ed. J. A. Bissonette. Springer, New York: Wildlife and Landscape Ecology.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:A51–76.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monograph*. 155:58–61.

Dusek, G. L. 1975. Range relations of mule deer and cattle in prairie habitat. *Journal of Range Management*. 39(3):605–16.

Findholt, S. L., B. K. Johnson, L. L. McDonald, J. W. Kern, A. Ager, R. J. Stussy, and L. D. Bryant. 2002. *Adjusting for radiotelemetry error to improve estimates of habitat use, general technical report PNW-GTR-555*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Findholt, S. L., B. K. Johnson, D. Damiran, T. DelCurto, and J. G. Kie. 2004. Diet composition, dry matter intake and diet overlap of mule deer, elk and cattle. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:670–86.

Frisina, M. R., and F. G. Morin. 1991. Grazing private and public land to improve the Fleecer Elk Winter Range. *Rangelands*. 13:291--4.

Hobbs, N. T., and L. H. Carpenter. 1986. Viewpoint: Animal-unit equivalents should be weighted by dietary differences. *Journal of Range Management*. 39:470.

Holechek, J. L., M. Vavra, J. Skovlin, and W. C. Krueger. 1982. Cattle diets in the Blue Mountains of Oregon II. Forests. *Journal of Range Management*. 35:239--42.

Johnson, B. K., M. J. Wisdom, and J. G. Cook. 2004. Issues of elk productivity for research and management. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:551–71.

Johnson, B. K, J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of elk and mule deer in spring. *Journal of Wildlife Management*. 64:685–97.

Kie J. G., and R. T. Bowyer. 1999. Sexual segregation in white-tailed deer: Density-dependent changes in use of space, habitat selection and dietary niche. *Journal of Mammalogy*. 80:1,004–20.

Knowles, C. J., and R. B. Campbell. 1982. Distribution of elk and cattle in a rest-rotation grazing system. In *Proceedings of Wildlife-Livestock Relationships Symposium*. Moscow, Idaho: University of Idaho, Forestry, Wildlife and Range Experimental Station.

Leckenby, D. A. 1984. *Elk use and availability of cover and forage habitat components in the Blue Mountains, northeast Oregon, 1972–1982, wildlife research report 14*, Salem, Oregon: Oregon Department of Fish and Wildlife.

Lindzey, F. G., W. G. Hepworth, T. A. Mattson, and A. F. Reeve. 1997. *Potential for competitive interactions between mule deer and elk in the Western United States and Canada*. Laramie, Wyoming: Wyoming Cooperative Fish and Wildlife Research Unit.

Loft, E. R., J. W. Menke, and J. G. Kie. 1991. Habitat shifts by mule deer: The influence of cattle grazing. *Journal of Wildlife Management*. 55(1):16–26.

Mackie, R. J. 1970. Range ecology and relations of mule deer, elk, and cattle in the Missouri River Breaks, Montana. *Wildlife Monographs*. 20:1–79.

Manly, B. F. J., L. L. McDonald, and D. L. Thomas. 1993. *Resource selection by animals: Statistical analysis and design for field studies*. London, United Kingdom: Chapman and Hall.

Martin, E. 1990. *A Concise Dictionary of Biology*. Oxford, United Kingdom: Oxford University Press.

McCullough, D. R., D. H. Hirth, and S. J. Newhouse. 1989. Resource partitioning between sexes in white-tailed deer. *Journal of Wildlife Management*. 53(2):266–83.

Mueggler, W. F. 1965. Cattle distribution on steep slopes. *Journal of Range Management*. 18(5):255–7.

Nelson, J. R. 1982. Relationships of elk and other large herbivores. In *Elk of North America, ecology and management*, eds. J. W. Thomas and D. E. Toweill, 415–41. Harrisburg, Penn: Stackpole Books.

Painter, G. G. 1980. *Elk and cattle spatial interaction*. M.S. thesis, Pullman, Washington: Washington State University.

Pedersen, R. J., A. W. Adams, and J. M. Skovlin. 1980. *Elk habitat use in an unlogged and logged forest environment, research report no. 9*. Salem, Oregon: Oregon Department of Fish and Wildlife.

Peek, J. M., and P. R. Krausman. 1996. Grazing and mule deer. In *Rangeland Wildlife*, ed. P. R. Krausman, 183–92. Denver, Colorado: Society of Range Management.

Putnam, R. J. 1996. *Competition and resource partitioning in temperate ungulate assemblies*. London, United Kingdom: Chapman and Hall.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64(3):672–84.

Scarbrough, D. L., and P. R. Krausman. 1988. Sexual segregation by desert mule deer. *The Southwest Naturalist*. 33(2):157–65.

Skovlin, J. M., P. J. Edgerton, and R. W. Harris. 1968. The influence of cattle management on deer and elk. *Transactions of the North American Wildlife and Natural Resources Conference*. 33:169–81.

Society for Range Management. 1989. *A glossary of terms used in range management*. Denver, Colorado: Edison Press.

Stewart , K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83(1):229–44.

Wallace, M. C., and P. R. Krausman. 1987. Elk, mule deer, and cattle habitats in central Arizona. *Journal of Range Management*. 40(1):80–3.

Wisdom, M. J., and J. W. Thomas. 1996. Elk. In *Rangeland Wildlife*, ed. P. R. Krausman, 158–81. Denver, Colorado: Society for Range Management.

Diet Composition, Dry Matter Intake and Diet Overlap of Mule Deer, Elk and Cattle

Scott L. Findholt
Oregon Department of Fish and Wildlife
La Grande

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande

Daalkhaijav Damiran
Eastern Oregon Agricultural Research Center
Union

Tim DelCurto
Eastern Oregon Agricultural Research Center
Union

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*) and cattle share rangeland throughout much of interior western North America. Considerable debate exists about the degree to which facilitation or competition occurs for forage between these three species (Nelson 1982, Wisdom and Thomas 1996, Miller 2002).

Prior cattle grazing can have beneficial effects on elk nutrition. The removal of forage by cattle can improve forage quality by enhancing regrowth of forage or by changing ratios of live to dead plant material (Cook 2002), and forage quality on elk winter ranges in the interior Northwest can be improved by cattle grazing in the spring (Anderson and Scherzinger 1975, Clark 1996). To date, however, studies have not shown enhancement of forage quality in the summer following late spring or early summer grazing by elk or cattle.

Competition for forage between mule deer, elk and cattle is probably greatest on winter and spring-autumn ranges (Nelson 1982, Wisdom and Thomas 1996) and is minimal between wild ungulates and cattle during summer (Miller 2002). Most summer ranges for mule deer and elk are on large areas of public land containing a diversity of habitats and high potential for forage production (Miller 2002). On some summer ranges, however, competition for forage may exist in late summer and early fall because forage quality can be poor and not meet nutritional requirements of wild ungulates and cattle (Hanley et al. 1989, Cook 2002). This is especially evident in regions where summer drought is the normal part of the climatic regime (Vavra and Phillips 1980, Svejcar and Vavra 1985).

On the Starkey Experimental Forest and Range (Starkey), in northeastern Oregon, competition for forage may occur in late summer among all three species, especially between elk and cattle (Coe et al. 2001, Stewart et al. 2002). Conditions at Starkey typify those on summer ranges shared by mule deer, elk and cattle in forests of the interior western United States. Consequently, we conducted a manipulative experiment on Starkey summer range to evaluate the potential for competition or facilitation for forage among mule deer, elk and cattle in grand fir (*Abies grandis*) forests. Our specific objectives were to determine diet composition, dry matter intake rates and percent dietary overlap of all three species in response to prior grazing by elk and cattle. In this paper we focus on results obtained from the bite count data. A detailed analysis of the nutritional consequences of previous grazing by elk or cattle on subsequent diets of mule deer, elk or cattle will be presented elsewhere (Damiran in prep.). Results from our study could improve range management for mule deer, elk and cattle on public land, especially with regard to allocating forage among these three species when ranges are shared in forested habitats (Ager et al. 2004).

Study Area

Starkey is located on the Wallowa-Whitman National Forest, 22 miles (35 km) southwest of La Grande, Union County, northeastern Oregon. Vegetation is a mosaic of coniferous forests and open areas containing shrubland and grassland. Forest stands are dominated by ponderosa pine (*Pinus ponderosa*), with grand fir, and Douglas fir (*Pseudotusga menziesii*) occurring on northern aspects. Elevation ranges from 3,675 to 4,922 feet (1,120–1,500 m).

We conducted our research in four, 5.56-acre (2.25-ha) enclosures located in grand fir forests logged 15 to 20 years ago. We chose the grand fir vegetation type because of its dominance on summer and fall ranges in the Blue Mountains and the interior western United States. Moreover, grand fir forests support a high level of forage production, particularly after logging or burning. In addition, other research at Starkey indicated that mule deer, elk and cattle concentrated much of their foraging activity after mid summer on early successional stages of logged grand fir vegetation types (Coe et al. 2001).

Methods

Grazing Treatments

Each enclosure contained three, 1.85-acre (0.75-ha) pastures (Figure 1). In 1998 and 1999, during late June and again in mid- to late July, one pasture of each enclosure was grazed by four to five steers and another pasture was grazed by four elk. The remaining pasture of each enclosure was not grazed (control pasture). Our goal during grazing treatments was to have 40 percent of the current year's vegetation production utilized by elk and steers on each pasture. This is the standard used by the U. S. Department of Agriculture, Forest Service for cattle allotments on upland sites in good condition in northeastern Oregon. Assignment of grazing treatments to pastures was done randomly and remained the same in 1998 and 1999.

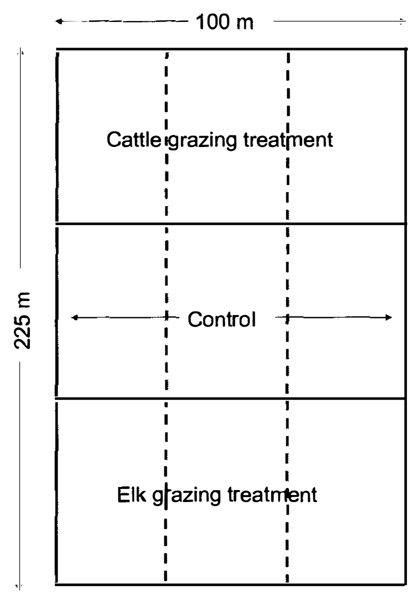


Figure 1. Layout of pastures used to determine diet composition, bite rates, dry matter intake and diet overlap among cattle, elk and mule deer in response to previous grazing by cattle or elk on the Starkey Experimental Forest and Range, northeastern Oregon.

Feeding Trials

After completion of the grazing treatments, each 1.85-acre (0.75-ha) pasture was subdivided into three 0.62-acre (0.25-ha) pastures. In August 1998 and 1999 we used bite-counts to obtain information on diet composition of tractable mule deer, elk and cattle from each pasture as described by Wickstrom et al. (1984). Each 0.62-acre (0.25-ha) pasture was grazed by four mule deer, four elk or four steers. The elk and steers were the same animals used during grazing treatments. We conducted four feeding trials with each animal in each 0.62-acre (0.25-ha) pasture, which resulted in 16 feeding trials per species in each treatment at the four enclosures, totaling 192 feeding trials per species per year.

During each 20-minute feeding trial, we used a small hand-held tape recorder to record the number of bites of each plant species consumed. We randomly assigned animal species to pastures for the feeding trials at the start of each year.

Dry Matter Intake

After feeding trials were completed in each 0.62-acre (0.25-ha) pasture, we clipped at least 25 samples of the most common plant species (greater than 5 percent of the diet) to simulate plant parts and sizes of bites recorded during feeding trials. Samples were oven-dried at 122 degrees Fahrenheit (50° C) to constant weight. We calculated mean weights of bites of each plant species found in the diet in each 0.62-acre (0.25-ha) pasture. Mean weights were multiplied by the number of bites of each plant species to determine total dry matter intake.

Data Analysis

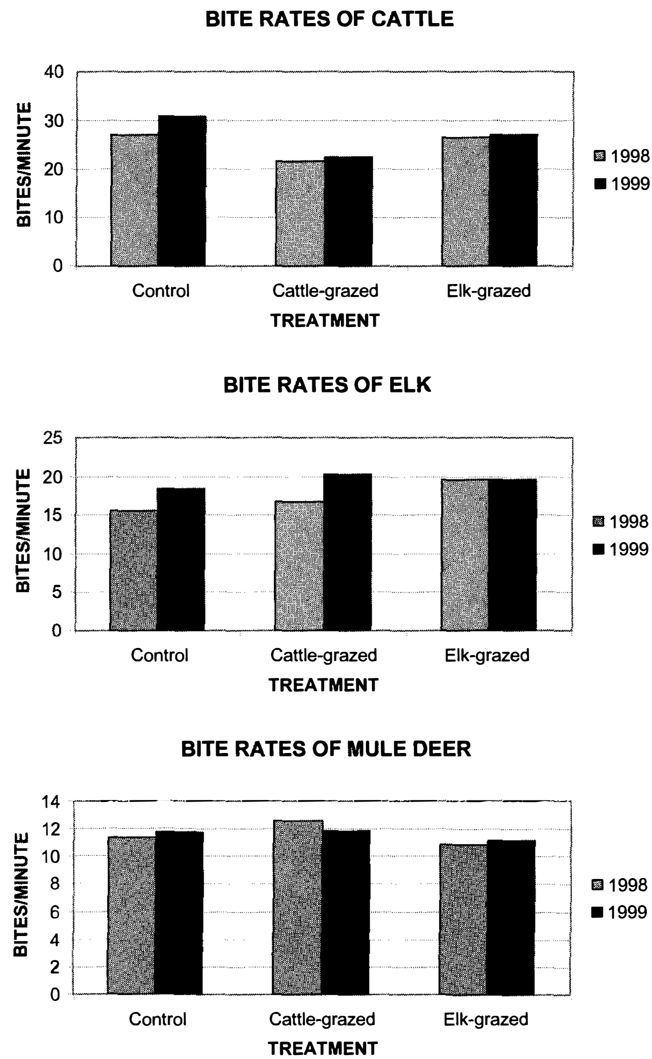
Data were analyzed as a split-plot design with the proc-mixed procedure in statistical analysis software (SAS) (Littell et al. 1996) to determine whether diets of mule deer, elk and cattle differed among pastures previously grazed by cattle or elk and among control (ungrazed) pastures. We used the least-squares (LS) means procedure of SAS to determine if treatment means were different. Bite count data (measured in bites per minute) were used to calculate percent overlap in diet. Percent overlap of major forage classes composing the diets of mule deer, elk and steers was determined with Horn's index of similarity (Horn 1966).

Results

Diet Composition and Bite Rates

Cattle. Total bites per minute of steers were less in pastures previously grazed by cattle in 1998 and 1999, compared to ungrazed pastures and pastures previously grazed by elk ($P \leq 0.03$, Figure 2). Graminoids composed the bulk of the steer diets both years (Figure 3). Steers consumed less graminoids and more

Figure 2. Bites per minute of cattle, elk and mule deer in response to previous grazing by cattle or elk on the Starkey Experimental Forest and Range, northeastern Oregon.



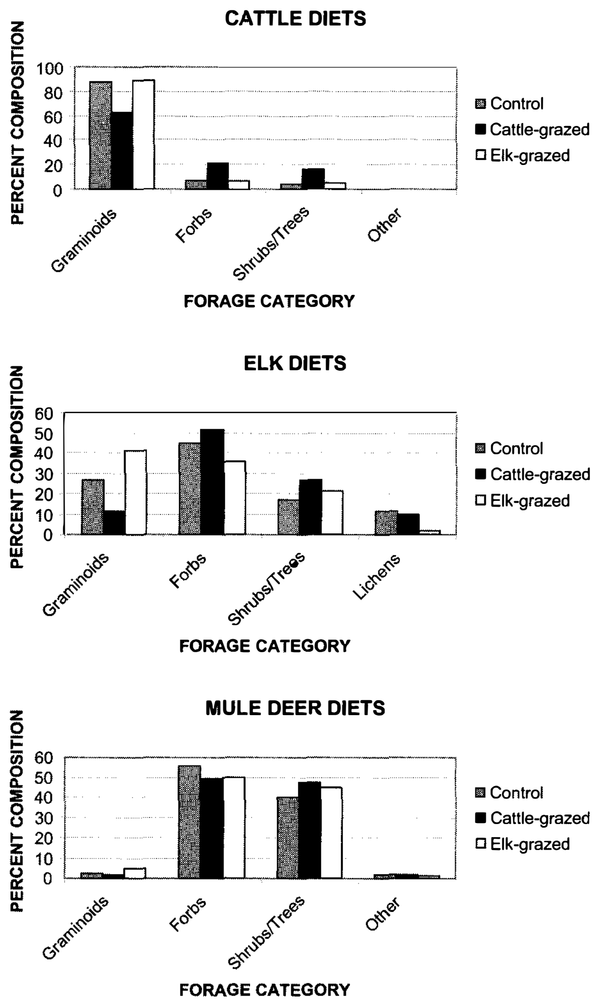


Figure 3. Diet composition of cattle, elk and mule deer by major forage category on the Starkey Experimental Forest and Range, northeastern Oregon. Diet composition was derived from bite count data.

shrubs and trees in pastures previously grazed by cattle however, compared to ungrazed pastures or pastures previously grazed by elk in 1998 and 1999 ($P \leq 0.04$). Also, steers consumed more forbs in pastures previously grazed by cattle in 1999 ($P \leq 0.02$) but not in 1998 ($P \geq 0.11$). Intake of major forage categories in steer diets did not change in response to prior elk grazing ($P > 0.10$).

Elk. Total bites per minute of elk was greater in pastures previously grazed by elk compared to ungrazed pastures in 1998 ($P = 0.02$) but not in 1999 ($P = 0.58$, Figure 2). Although elk consumed mostly forbs, graminoids, shrubs and trees, lichens were also prevalent in their diets (Figure 3). Bites per minute of elk on graminoids was greater in pastures previously grazed by elk, compared to pastures previously grazed by cattle during both years ($P \leq 0.01$). Elk diets also contained more graminoids in pastures previously grazed by elk compared to control pastures in 1998 ($P = 0.05$). Bites per minute of elk on forbs were higher in pastures previously grazed by cattle, compared to control pastures and pastures previously grazed by elk but less in pastures previously grazed by elk compared to control pastures ($P \leq 0.03$). In 1998, bites per minute of elk did not vary with treatment ($P \geq 0.43$). However, this same year elk consumed more shrubs and trees in pastures previously grazed by cattle compared to control pastures ($P = 0.02$). In 1998 and in 1999, elk consumed less lichen in pastures previously grazed by elk ($P \leq 0.05$, Figure 3).

Mule deer. Total bites per minute of mule deer did not vary with treatment either year ($P > 0.10$, Figure 2). Mule deer consumed mostly forbs, shrubs and trees (Figure 3). Intake of graminoids, forbs, shrubs, trees and other food did not differ in those pastures previously grazed by cattle or elk compared to control pastures (Figure 3).

Dry Matter Intake

In 1998 and 1999, total dry matter intake (measured in grams per minute) of steers was less in pastures previously grazed by cattle, compared to pastures previously grazed by elk or ungrazed pastures ($P \leq 0.03$, Figure 4). Total dry matter intake did not vary among treatments either year for elk or mule deer ($P \geq 0.26$, Figure 4).

Percent Diet Overlap

Mean diet overlap of major forage classes in control pastures (ungrazed pastures) was 49 percent between cattle and elk, 59 percent between mule deer and elk but only 19 percent between cattle and deer when data from 1998 and 1999 were combined (Figure 5). During both years, diet overlap between cattle and deer increased in pastures previously grazed by cattle ($P \leq 0.01$) but not in pastures previously grazed by elk ($P \geq 0.67$). Percent diet overlap between cattle and elk was higher in pastures previously grazed by elk, compared to control pastures both years ($P \leq 0.006$), and also was higher in pastures previously grazed by cattle in 1999 ($P \leq 0.001$). Percent diet overlap between mule deer and elk did not vary with treatment either year ($P \geq 0.17$).

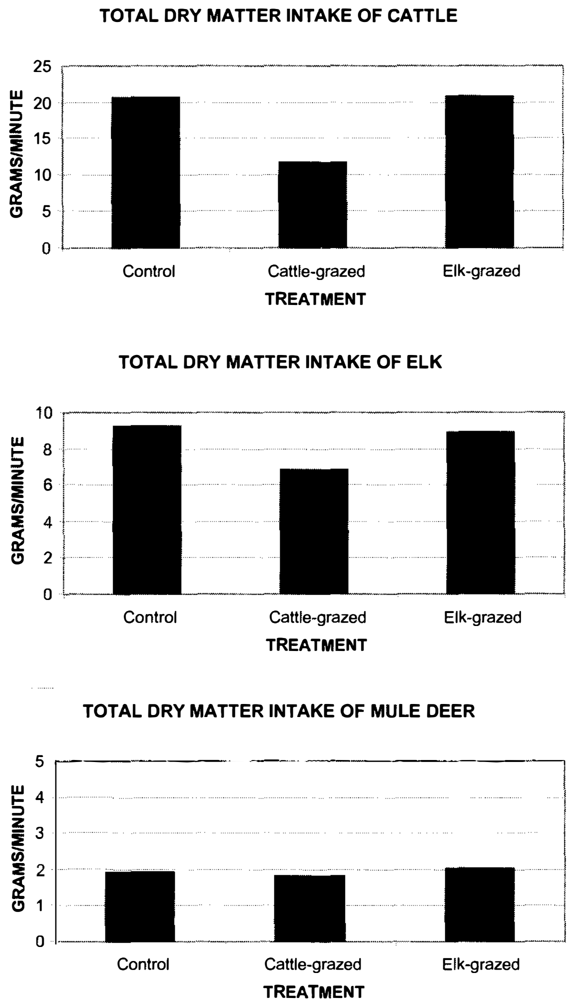
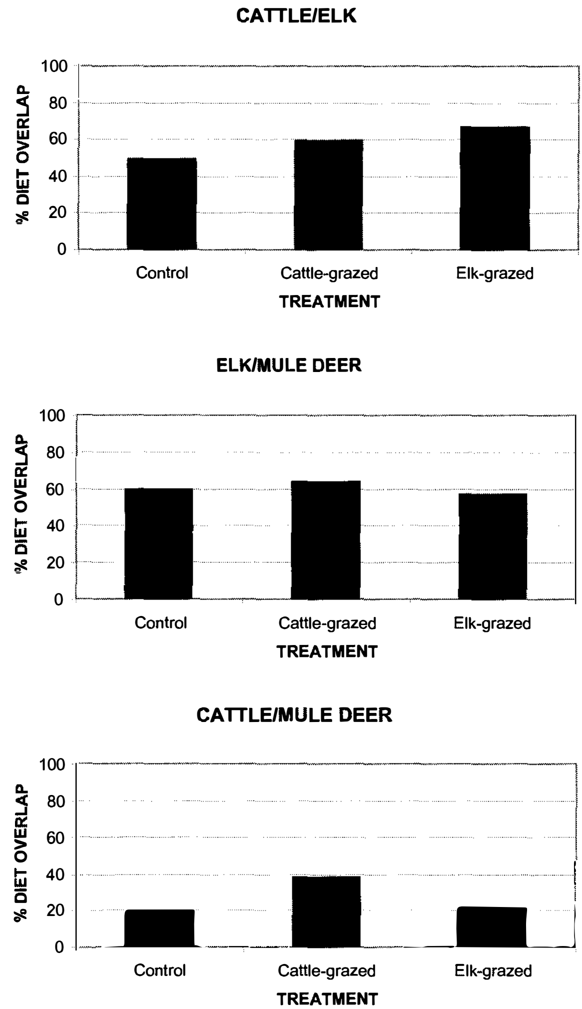


Figure 4. Total dry matter intake (grams/minute) of cattle, elk and mule deer in response to previous grazing by cattle or elk on the Starkey Experimental Forest and Range, northeastern Oregon.

Discussion

Numerous studies have evaluated the food habits of mule deer, elk and cattle (Kufeld 1973, Kufeld et al. 1973, Holechek 1980, Cook 2002). These studies show large variation by year of study, location, specific herbivore and research technique. Thus in this discussion, we focus on the results of our study in relation to other studies in northeast Oregon where habitats are generally similar to those in our study area.

Figure 5. Percent diet overlap among mule deer, elk and cattle by major forage category calculated with Horn's index of similarity at the Starkey Experimental Forest and Range, northeastern Oregon. Bite count data were used to determine percent diet overlap.



Forage intake estimated from clipping or handpicking simulated bites that represent bites ingested can be biased, especially with wild ungulates (Parker et al. 1993). In our study, results from bite counts (measured in bites per minute) and dry matter intake rates (measured in grams per minute) showed similar effects of previous grazing by cattle or elk. This similarity suggests that both methods can be used to estimate diet composition and to evaluate the potential for competition for forage among mule deer, elk and cattle.

Holechek et al. (1982) found that cattle grazed mostly on grasses but also consumed large quantities of forbs and shrubs in ponderosa pine or in Douglas fir forest types at Starkey in late summer. In our study, cattle also consumed mostly grasses and sedges. In response to previous cattle grazing, however, cattle reduced their consumption of graminoids and increased their use of forbs and shrubs. The potential for competition for forage between cattle and elk and cattle and mule deer increased when cattle changed their diets as preferred cattle forage was consumed. Holechek et al. (1982) and Miller and Vavra (1981) also reported that cattle readily switch their diets from one forage type to another. Elk are an intermediate feeder compared to mule deer or cattle (Kufeld 1973, Hofmann 1988). Edgerton and Smith (1971) found that elk and mule deer diets consisted of 58 percent grasses and sedges, 27 percent forbs, and 15 percent shrubs during summer at Starkey. In their study it was not possible to separate elk from mule deer diets because forage production and utilization were estimated on summer range shared by the two species. Korfhage (1974) and Korfhage et al. (1980) found that elk diets in the Blue Mountains, northeastern Oregon were evenly balanced among all major vegetative components, but forbs and shrubs were more important dietary components than graminoids during late summer. In our study, elk diets consisted of a more even percentage of all major forage classes in contrast to the more concentrated diets of mule deer and cattle. Interestingly, elk consumed a substantially higher percentage of lichens, with no consumption of this forage by cattle and only a minor percentage consumed by mule deer. Like cattle, elk also switched their diets in response to previous grazing. The most dramatic change in elk diets was in pastures previously grazed by elk. Elk appeared to increase their consumption of graminoids in response to a decline of available lichens. This increased the potential for competition between cattle and elk.

Other research at Starkey suggests that competition may exist between cattle and elk during late summer. In ponderosa pine-bunchgrass summer range at Starkey, Skovlin et al. (1968) found that elk use decreased as rate of cattle stocking increased. In a more recent study at Starkey, Coe et al. (2001) discovered that, when cattle were present, elk use of the same pastures decreased and use of the ponderosa pine-Douglas fir plant community in the same pasture increased in late summer. In Colorado, however, Hobbs et al. (1996) found that, at high densities, elk were in direct competition with cattle, but, at low density, elk had a facilitative effect on cattle diets.

Because of their anatomical and digestive attributes, mule deer were expected to have a more selective diet and choose higher quality forages than elk or cattle (Hofmann 1988). Mule deer switched their diet the least of the three herbivores in response to previous grazing by cattle or elk. This lack of flexibility in their diet could result in increased competition with elk or cattle in areas that have high ungulate or cattle densities or that have low forage production. Kie et al. (1991) concluded that competition between cattle and mule deer was highest during years of below-average precipitation. Austin and Urness (1986), however, concluded that deer and cattle did not compete for forage in their study area as deer use increased.

In our study we determined the crude protein (CP), the acid detergent fiber (ADF), the neutral detergent fiber (NDF) and the *in vitro* dry matter digestibility (IVDMD) of plant samples collected that represented those parts of species common in the diets of mule deer, elk and cattle (Damiran et al. 2003). The most dramatic effect of prior grazing on subsequent diets of mule deer, elk or cattle was the influence of previous cattle grazing on diets of steers. The grams per minute of CP and IVDMD consumed by cattle in pastures previously grazed by cattle was 32.4 and 43.0 percent less, respectively, compared to control (ungrazed) pastures. This suggests that the potential for intraspecific competition among cattle in secondary succession grand fir vegetation types is high during late summer.

Our results support Holechek's et al. (1981) findings that, overall, the diets of cattle at Starkey did not meet the National Research Council (NRC) standard for nutrient (CP and digestible energy [DE]) requirements during late summer. By contrast, nutrient intake of mule deer and elk did not change in response to previous grazing by cattle or elk (Damiran et al. 2003).

One measure of the potential for competition between species is the amount of dietary overlap. Dietary overlap or lack of dietary overlap, however, does not necessarily equate to high or low levels of competition. One would expect that the distinct differences in ruminant physiology among mule deer, elk and cattle would result in low dietary overlap, especially between mule deer and cattle (Hanley 1982, Baker and Hobbs 1987, Hofmann 1988). Willms et al. (1980) observed that mule deer diets in British Columbia changed in response to varying levels of forage utilization by cattle. When forages were abundant, cattle and deer diets were similar, but, as forage utilization increased, dietary overlap decreased with deer preferring shrubs and cattle preferring grasses. In our study, dietary overlap was the lowest between cattle and mule deer in ungrazed pastures. The potential for competition between these two species, however, increased greatly in response to previous cattle grazing. Steers consumed less graminoids and more forbs and shrubs in response to prior cattle grazing. This switching of cattle diets from graminoids to forbs and shrubs nearly doubled the dietary overlap between cattle and deer. By contrast, Damiran et al. (2003) found that intake of CP and IVDMD for mule deer did not change in response to prior grazing by cattle or elk, suggesting that competition was not occurring in our study even though dietary overlap increased.

The greatest potential for competition in ungrazed secondary succession grand fir was between mule deer and elk. Mule deer and elk consumed many of the same forbs and shrubs. Dietary overlap of mule deer with elk, however, did not change in response to previous grazing by cattle or elk. Coe et al. (2001) found that mule deer use of pastures at Starkey declined when elk were present, which suggests that interference competition may be occurring between these two species.

Dietary overlap was also high between cattle and elk, especially in pastures previously grazed by steers or elk. As others have suggested, the dietary choices of elk can overlap closely with dietary selection by cattle, making the two species potential competitors for available forage. Based on interactions between cattle and elk at Starkey, Coe et al. (2001) concluded that dietary competition for forage could occur between elk and cattle in late summer. At the level of grazing in our study, CP and IVDMD in elk diets were not affected by previous grazing, but these two nutrients declined in steer diets (Damiran et al. 2003). As mentioned previously, however, the decline in CP and IVDMD in cattle diets was a result of intraspecific competition for forage among cattle, not interspecific competition with elk.

In contrast to results from our study, Stewart et al. (2003) determined diet composition of free-ranging mule deer, elk and cattle at Starkey using microhistological analyses of fecal samples and found little overlap in dietary niche among these three herbivores. This study was conducted later in the summer during July and August, however, and it was conducted on free-ranging animals with access to a variety of habitat types. Also, cattle feces were collected from the Main Study Area, and feces from mule deer and elk were collected from the Northeast Study Area where cattle were not present. Finally, results from microhistological analyses of fecal samples are not comparable to bite count data because of inherent biases associated with fecal samples that cannot control different rates of digestion of different forage classes (McInnis et al. 1983).

Our results suggest that inter- and intraspecific competition for forage may exist among all three species during summer in previously logged grand fir forests. However, realization of competition depends on the densities of the various herbivores and annual net primary production, and it depends on dietary overlap resulting in negative nutritional consequences. The probability of competition would increase during years of low forage production, heavy herbivore stocking or both. Moreover, demonstration of true competition between two or more species requires a documented reduction in animal or population performance by one species as a result of sharing a limited resource with the other species (Wisdom and Thomas 1996).

Policy Implications

1. Our results suggest that inter- and intraspecific dietary competition may exist among mule deer, elk and cattle during the summer in secondary succession grand fir vegetation types. Consequently, it is important for managers to monitor the forage base in this habitat type and, if necessary, reduce livestock use to maintain plant vigor and to leave adequate forage for mule deer and elk. Alternatively, it may be necessary to reduce densities of mule deer and elk to maintain adequate nutrition for cattle. Which approach is taken (reduction of livestock or wild ungulate use) depends on the multispecies objectives set for the allotment, as evaluated from analyses of the trade-offs of varying stocking rates among the species (Wisdom and Thomas 1996). Although the fitness of mule deer and of elk may not be affected during the summer, their ability to survive severe winters and to reproduce may be negatively affected by poor summer nutrition (Cook et al. 2004).
2. We found no strong evidence that previous elk or cattle grazing improved forage quality of mule deer, elk or cattle that grazed the same pastures later that year. During the summer in secondary succession grand fir, it appears that prior grazing by cattle or elk does not improve the forage quality for those herbivores that subsequently graze the same pastures. In our study, however, forage quality did not improve subsequent to cattle and elk grazing; soil moisture may not have been adequate in late June

and July for vegetation regrowth. The potential for prescribed cattle grazing, however, as a management tool to maintain or to improve mule deer and elk forage during late summer, needs more exploration.

3. High dietary similarity among these three species and an increase in dietary overlap in response to previous grazing by cattle and elk indicate that cattle and elk readily shift their diets during late summer. This knowledge will help managers optimize use of grand fir vegetation types for grazing by wild ungulates and livestock.
4. Based on our findings and on research by Holechek et al. (1981), the diets of cattle may not meet the NRC standard for nutrient requirements during summer in forested vegetation types at Starkey. This finding further suggests that competition for forage is more likely during summer, and it deserves careful consideration in allotment management planning for areas with similar environmental conditions as Starkey.
5. Our project and other studies indicate a need for additional research on inter- and intraspecific competition for forage among mule deer, elk and cattle. More questions remain on how the timing and intensity of grazing affects the nutritional condition and fitness of all three species. Likewise, it would be helpful to have more information on their habitat selection and distributional overlap. This will help to contribute to the continuation of the multiple use concept mandated on public land, and it will guarantee proper management of the forage resource.

Reference List

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:687–707.

Anderson, E. W., and R. J. Scherzinger. 1975. Improving quality of winter forage for elk by cattle grazing. *Journal of Range Management*. 28: 120–25.

Austin, D. D., and P. J. Urness. 1986. Effects of cattle grazing on mule deer diet and area selection. *Journal of Range Management.* 39:18–21.

Baker, D. L., and N. T. Hobbs. 1987. Strategies of digestion: Digestive efficiency and retention time of forage diets in montane ungulates. *Canadian Journal of Zoology*. 65:1,978–84.

Clark, P. E. 1996. Use of livestock to improve the quality of elk winter range forage in northeastern Oregon. Ph. D. dissertation, Oregon State University, Corvallis.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:A51–A76.

Cook, J. G. 2002. Nutrition and food. In *North American elk: Ecology and management,* eds. D. E. Toweill, and J. W. Thomas, 259–349. Washington, DC: Smithsonian Institution Press.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Erwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monograph*. 155:1–61.

Damiran, D. (In prep.) *Inter- and intraspecific summer foraging dynamics of mule deer, elk, and cattle*. Ph.D. dissertation, Oregon State University, Corvallis.

Damiran, D., T. DelCurto, S. L. Findholt, G. D. Pulsipher, and B. K. Johnson. 2003. Influence of previous cattle and elk grazing on the subsequent quality and quantity of diets for cattle, deer, and elk grazing late-summer mixed-conifer rangelands. In *Proceedings, Western Section, American Society of Animal Science*. 54:320–24.

Edgerton, P. J., and J. G. Smith. 1971. *Seasonal forage use by deer and elk on the Starkey Experimental Forest and Range, Oregon, research paper PNW-RP-112*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Hanley, T. A. 1982. The nutritional basis for food selection by ungulates. *Journal of Range Management*. 35:146–9.

Hanley, T. A., C.T. Robbins, and D. E. Spalinger. 1989. *Forest habitats and the nutritional ecology of Sitka black-tailed deer: A research synthesis with implications for forest management, general technical report PNW-GTR-230*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Hobbs, N. T., D. L. Baker, G. D. Bear, and D. C. Bowden. 1996. Ungulate grazing in sagebrush grassland: Mechanisms of resource competition. *Ecological Applications*. 6:200–17.

Hofmann, R. R. 1988. Anatomy of the gastro-intestinal tract. In *The ruminant animal digestive physiology and nutrition*. ed. D.C. Church, 14–43. Englewood Cliffs, New Jersey: Prentice Hall.

Holechek, J. L. 1979. The effects of vegetation type and grazing systems on the performance, diet and intake of yearling cattle. Ph.D. thesis, Oregon State University, Corvallis.

Holechek, J. L., M. Vavra, and J. Skovlin. 1981. Diet quality and performance of cattle on forest and grassland range. *Journal of Animal Science*. 53: 291–8.

Holechek, J. L., M. Vavra, J. Skovlin, and W. C. Krueger. 1982. Cattle diets in the Blue Mountains of Oregon II. Forests. *Journal of Range Management*. 35:239–42.

Horn, H. S. 1966. Measurement of "overlap" in comparative ecological studies. *American Naturalist*. 100:419–24.

Kie, J. G., C. J. Evns, E. R. Loft, and J. W. Menke. 1991. Foraging behavior by mule deer- the influence of cattle grazing. *Journal of Wildlife Management*. 55:665–74.

Korfhage, R. C. 1974. *Summer food habits of elk in the Blue Mountains of northeastern Oregon based on fecal analysis*. M. S. thesis, Washington State University, Pullman.

Korfhage, R. C., J. R. Nelson, and J. M. Skovlin. 1980. Summer diets of Rocky Mountain elk in Northeastern Oregon. *Journal of Wildlife Management*. 44:746–50.

Kufeld, R. C. 1973. Foods eaten by the Rocky Mountain elk. *Journal of Range Management*. 26:106–13.

Kufeld, R. C., O. C. Wallmo, and C. Feddema. 1973. *Foods of the Rocky Mountain mule deer, research paper RM-RP-111*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996. *SAS system for mixed models*. Cary, North Carolina: SAS Institute, Inc.

McInnis, M. L., M. Vavra, and W. C. Krueger. 1983. A comparison of four methods used to determine the diets of large herbivores. *Journal of Range Management*. 36:302–7.

Miller, W. 2002. Elk interactions with other ungulates. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 435–47. Washington, DC: Smithsonian Institution Press.

Miller, R. F., and M. Vavra. 1981. Deer, elk and cattle diets on northeastern Oregon rangelands. In *Proceedings of the wildlife-livestock relationships symposium*, eds. J. M. Peek, and P. D. Dolke. 10:500–8

Nelson, J. R. 1982. Relationships of elk and other large herbivores. In *Elk of North America, ecology and management*, eds. J. W. Thomas, and D. E. Toweill, 415–41. Harrisburg, Pennsylvania: Stackpole Books.

Parker, K. L., M. P. Gillingham, and T. A. Hanley. 1993. An accurate technique for estimating forage intake of tractable animals. *Canadian Journal of Zoology*. 71:1,462–5.

Stewart, K. M., R. T. Bowyer, J. G. Kie, B. L. Dick, and M. Ben-David. 2003. Niche partitioning among mule deer, elk, and cattle: Do stable isotopes reflect dietary niche? *Ecoscience*. 10:297–302.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83:229–44.

Svejcar, T., and M.Vavra. 1985. The influence of several range improvements on estimated carrying capacity and potential beef production. *Journal of Range Management*. 38:395–9.

Vavra, M., and R. L. Phillips. 1980. Drought effects on cattle performance. In *Proceedings, Western Section, American Society of Animal Science*. 31:157–60.

Wickstrom, M. L., Charles T. Robbins, Thomas A.Hanley, Donald E. Spalinger, and Steven M. Parish. 1984. Food intake and foraging energetics of elk and mule deer. *Journal of Wildlife Management*. 48:1,285–301.

Willms, W., A. McLean, R. Tucker, and R. Ritcey. 1980. Deer and cattle diets on summer range in British Columbia. *Journal of Range Management*. 33:55–59.

Wisdom, M. J., J. G. Cook, M. M. Rowland, and J. H. Noyes. 1993. *Protocols for care and handling of deer and elk at the Starkey Experimental Forest and Range, general technical report PNW-GTR-311*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Wisdom, M. J. and J. W. Thomas. 1996. Elk. In *Rangeland Wildlife*, ed. P. R. Krausman, 157–82. Denver, Colorado: Society of Range Management.

Landscape Simulation of Foraging by Elk, Mule Deer and Cattle on Summer Range

Alan A. Ager
*U. S. Department of Agriculture, Forest Service,
Umatilla National Forest
Pendleton, Oregon*

Bruce K. Johnson
*Oregon Department of Fish and Wildlife
La Grande*

Priscilla K. Coe
*Oregon Department of Fish and Wildlife
La Grande*

Michael J. Wisdom
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon*

Introduction

Cattle, mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) share more area of spring, summer and fall range than any other combination of wild and domestic ungulates in western North America (Wisdom and Thomas 1996). Not surprisingly, conflicts over perceived competition for forage have a long history, yet knowledge about actual competition is limited (Van Dyne et al. 1984b, Hobbs et al. 1996, Johnson et al. 1996). One of the first studies of the Starkey Project was designed to address the issue of whether mule deer and elk compete with cattle for available forage on summer range. A component of this study was to build a forage allocation model that could be used to analyze forage allocation problems on summer range in the Blue Mountains. This model would use data on animal spatial distributions, resource selection patterns, behavioral interactions and diet selection of cattle, elk and mule deer that was collected as part of the Starkey Project at the Starkey Experimental Forest and Range (Starkey) (Johnson et al. 2000; Coe et al. 2001, 2004; Findholt et al. 2004).

Modeling the forage removal and animal performance for multiple species of ungulates across large landscapes is a complex problem (Weisberg et al. 2002). The high degree of temporal and spatial variability in ungulate distributions, forage production and nutritional value of forage contribute to the problem (Wisdom and Thomas 1996). Several early forage allocation models built for western rangelands were never widely used, owing to insufficient data, model complexity and institutional barriers (Van Dyne et al. 1984a, McInnis et al. 1990). A prototype forage allocation model built from Starkey data (Johnson et al. 1996) suffered from similar problems but did provide a framework for further discussions and model development (Vavra et al. 2004). This model used linear programming with a weighted objective function that contained terms for forage production, forage energy content and resource selection coefficients. Animal foraging behavior could be optimized with respect to each of these three variables or some weighted combination. The Johnson et al. (1996) model generated reasonable predictions of species distributions and forage consumption patterns at monthly time steps. However, the linear programming framework was cumbersome and had limited capability to analyze the temporal dynamics of ungulate foraging behavior.

Using many of the parameters from the earlier work, we built a more detailed, spatially explicit, individual animal foraging model, called the Starkey foraging model (SFM). Initial testing of this model was described in Vavra et al. (2004). In this paper, we describe additional developments and testing, and we demonstrate the model's capability to predict forage removal and animal performance at Starkey. Ultimately, the model or subsequent outgrowths are intended for use in allotment management planning on summer ranges shared by cattle, mule deer and elk.

Methods

The SFM uses empirical data on habitat preferences, forage production, forage quality and energy dynamics of cattle, mule deer and elk. These data are coupled with information on foraging behavior to simulate forage consumption by the three ungulates on the Starkey landscape. The SFM was developed in Object Pascal, using the Delphi 6 (Borland Inc.) integrated development environment.

Data sources used for the SFM are described in detail by Vavra et al. (2004) and are summarized here.

Habitat preferences for each species were incorporated using resource selection functions developed at Starkey (Johnson et al. 2000, Coe et al. 2001). These resource selection functions (RSFs) were estimated from Starkey telemetry data, collected between 1993 and 1996, and were estimated for monthly time steps, from April through October (Tables 1, 2). The RSF's represent the probability of an animal visiting a particular pixel over the monthly interval, as described by Johnson et al. (1996, 2000).

Forage production was estimated using several empirical models built from Starkey data (clipped plots from 1993–2000) and other sources (Vavra et al. 2004). We built functions to predict herbage production as a function of calendar day for grasslands, ponderosa pine *(Pinus ponderosa*) and riparian ecotypes. The equations for these ecotypes were extrapolated to the seven plant-association groups in the model (moist meadows, dry meadows, bunchgrass and shrub lands, warm dry forests with grass understory, warm dry forests with shrub understory, cool moist forest with grass understory, cool moist forests with shrub understory). The forage production was partitioned into forbs, grass and shrubs, using scaling factors developed by Hall (1973) and Johnson and Hall (1990). The growth functions were also adjusted for canopy closure on a pixel basis, using relationships developed at four grazing exclosures at Starkey and the data of Pyke and Zamora (1982). Forage growth was represented in the model on a daily time step, and we used the same growth functions for forage regrowth as those used for initial forage growth.

Forage quality, as measured by in vitro digestible energy (IVDDM) of forage, was obtained from the literature (Holechek et al. 1981, Svejcar and Vavra 1985, Sheehy 1987, Westenskow 1991) and data from Starkey. Digestible energy (DE) was calculated from IVDDM using methods of McGinnis et al. (1990), with estimates made on a monthly time step.

The spatial dynamics of animal foraging were modeled as a multiscale process that involved the selection of foraging patches and the subsequent selection of forage within the patch. We used concepts and data from a variety of sources for the foraging component of the model (Spalinger and Hobbs 1992; Gross et al. 1993, 1995; Shipley and Spalinger 1995) as well as observations on elk and mule deer movements at Starkey (Ager et al. 2003). Foraging patches were defined at the same scale as the Starkey spatial database, that is, each 30-

Table 1. Coefficients of resource selection functions for mule deer and elk during six monthly time steps in Main Study Area 1993 to 1996, Starkey Experimental Forest, northeastern Oregon. Seasons 1 to 6 correspond to May 16 to June 15, June 16 to July 15, July 16 to August 15, August 16 to September 15, September 16 to October 15, and October 16 to November 15. Coefficients are standardized (top) and nonstandardized (bottom). Coefficients for elk when cattle were not present were estimated in Smith-Bally pasture (seasons 2 and 5) and Bear pasture (seasons 3 and 4).

	Season	Intercept	Distance to edge of patch	Forage prod-uced	Shape of patch	Distance to traffic zero	Distance to traffic low	Distance to traffic medium	Distance to traffic high	Percnt slope	Aspect east west	Aspect north south	Topo-graphic convex	Soil depth	Distance to cover	Percent can. cover	Distance to cattle fence	Elk distance to water
Deer	1	-3.4588						-0.4284	-0.3431	0.2505	0.2159		0.2346					
		-25.2478						-0.0005	-0.0003	0.0200	0.2951		0.0449					
	2	-3.8910							-0.5615	0.2326	0.1344		0.1250					
		-15.4409							-0.0003	0.0186	0.1830		0.0239					
	3	-4.4878							-0.5473	0.3817	0.1723	0.2151	0.0818					
		-12.1049							-0.0006	0.0305	0.2353	0.3210	0.0156					
	4	-3.7869							-0.4496	0.2767	0.1254		0.0936		-0.2073			
		-12.3734							-0.0007	0.0221	0.1703		0.0179		-0.0016			
	5	-3.9353						-0.3948	-0.5563	0.2355			0.1174					
		-14.3405						-0.0003	-0.0006	0.0188			0.0224					
	6	-3.9259									0.0811		0.1466		-0.1514			
		-17.8403									0.1097		0.0280		-0.0012			
Elk	1	-2.4546						0.1191		-0.1119			0.1181	0.1470		0.0552		
		-14.0412						0.0001		-0.0089			0.0226	0.0121		0.0025		
	2	-2.8329	-0.0378	-0.0568	-0.0455		-0.2775	0.0741		0.1075	-0.0442	0.1034	0.1944	0.1384				
		-21.2643	-0.0008	-0.0003	-0.2897		-0.0004	0.0001		0.0086	-0.0607	0.1543	0.0371	0.0114				
	No	-2.9761			-0.1288		-0.2905			-0.2912	-0.1899		0.2856	0.1601	-0.2750	0.2510	0.2891	
	cattle	-26.8541			-0.7677		-0.0004			-0.0223	-0.2701		0.0477	0.0126	-0.0021	0.0015	0.0007	
	3	-3.6208	0.1038	0.0377	-0.0681			0.1237	0.2306	0.1190		0.2491	0.1617	0.1851	-0.1919	0.1776		
		-20.3917	0.0022	0.0002	-0.4333			0.0002	0.0002	0.0095		0.3722	0.0309	0.0153	-0.0015	0.0081		
	No	-3.3056			-0.1649		0.3010	0.3256	0.5570	-0.1375	-0.1520	0.1803	0.3226	0.3783	-0.4220			0.1815
	cattle	-32.3572			-0.9822		0.0004	0.0003	0.0008	-0.010	-0.2119	0.2587	0.0534	0.0300	-0.0032			0.0011
	4	-3.0575	0.0992			0.1182	0.0984		0.1946			0.1946	0.1706	0.1527	-0.1558	0.1709		
		-20.4503	0.0021			0.0005	0.0002		0.0003			0.2900	0.0326	0.0126	-0.0012	0.0078		
	No	-2.6522		-0.1112								0.1301	0.1945	0.1697		0.2209		0.1823
	cattle	-19.5567		-0.0005								0.1867	0.0323	0.0134		0.0098		0.0011

Table 1 (continued). Coefficients of resource selection functions for mule deer and elk during six monthly time steps in Main Study Area 1993 to 1996, Starkey Experimental Forest, northeastern Oregon. Seasons 1 to 6 correspond to May 16 to June 15, June 16 to July 15, July 16 to August 15, August 16 to September 15, September 16 to October 15, and October 16 to November 15. Coefficients are standardized (top) and nonstandardized (bottom). Coefficients for elk when cattle were not present were estimated in Smith-Bally pasture (seasons 2 and 5) and Bear pasture (seasons 3 and 4).

	Season	Intercept	Distance to edge of patch	Forage prod-uced	Shape of patch	Distance to traffic zero	Distance to traffic low	Distance to traffic medium	Distance to traffic high	Percnt slope	Aspect east west	Aspect north south	Topo-graphic convex	Soil depth	Distance to cover	Percent can. cover	Distance to cattle fence	Elk distance to water
Elk	5	-3.1617	0.0463				0.0822					0.2379	0.1598	0.1212	-0.1813	0.1874		
		-19.0188	0.0010				0.0001					0.3556	0.0305	0.0100	-0.0014	0.0085		
	No cattle	-2.2976		-0.2736	-0.1136							0.2324	0.0907		-0.3904			0.2638
		-9.4488		-0.0012	-0.6781							0.3325	0.0151		-0.0030			0.0016
	6	-3.2960								0.0978		0.1396	0.1757	0.0915	-0.1612	0.0580		
		-20.4223								0.0078		0.2073	0.0336	0.0075	-0.0012	0.0026		

Table 2. Coefficients of resource selection functions for cattle during four monthly time steps in cattle pastures 1993 to 1996 at Starkey Experimental Forest, northeastern Oregon. Seasons 2 to 5 correspond to June 16 to July 15, July 16 to August 15, August 16 to September 15, and September 16 to October 15. Coefficients are standardized (top) and nonstandardized (bottom).

	Season	Intercept	Distance to edge	Forage prod-uced	Distance to road	Distance to fence	Percent slope	Aspect east west	Topo-graphic convex	Soil depth	Distance to cover	Percent can. cover	Distance to water
Cattle	2	-2.4895	-0.0613		-0.1756	0.3043	-0.4726	-0.1063	-0.0526		-0.2089	-0.2743	0.1252
		2.8039	-0.0014		0.0008	0.0008	-0.0365	-0.1489	-0.0088		-0.0016	-0.0123	0.0008
	3	-3.0240	-0.1597	0.0452	-0.9849		-0.1370	-0.0917	-0.0660	0.0943		0.0563	-0.1300
		4.0840	-0.0033	0.0005	-0.0007		-0.0120	-0.1217	-0.0139	0.0078		0.0028	-0.0007
	4	-2.8177	-0.1728	0.0747			0.0747	-0.0584					-0.7470
		-2.4244	-0.0036	0.0007			0.0007	-0.0754					-0.0009
	5	-2.7450	-0.2228	0.0864		0.1516			-0.4536	0.0711		-0.1650	
		0.9900	-0.1710	0.0004		0.0004			-0.0075	0.0056		-0.0013	

by-30 meter pixel. Selection of foraging patches was modeled by using a neighborhood search algorithm that searched a 10 by 10 pixel neighborhood and that subsequently chose the pixel that maximized an index of preference according to:

$$PREF_p = (RSF_{spm} * W_{rsf}) + (DE_{pm} * W_{qual}) + (F_{pm} * W_{mass}) \quad (1)$$

where $PREF_p$ equals pixel preference score for pixel p; RSF_{spm} equals resource selection function score $(0 < RSF < 1)$ for pixel p, species s, and month m; DE_{pm} equals digestible energy in megacalories per kilogram forage for pixel p and month m; and F_{pm} equals forage (in kilograms per hectare) present on pixel p and month m.

Here, W_{rsf}, W_{qual} and W_{mass} are weighting coefficients that control the relative importance of habitat selection, forage quality (DE) and standing forage biomass in the foraging process, respectively. The formulation recognized that both resource selection functions and forage characteristics need to be considered in the selection of foraging areas. Initially, we used a product of RSF_{spm}, DE_{pm} and F_{pm} to calculate the preference score and included the weighted coefficients W_{qual}, W_{rsf} and W_{mass} as exponents. This method created some scaling issues that led to the current formulation. Although the weighted coefficients could be species-specific, we used the same values for each species in the present simulations. Pixels were selected for foraging by randomly sampling the pixels and respective preference scores in each 10 by 10 pixel neighborhood 90 times (90 percent of the total number of pixels) to reflect the fact that animals have a less than perfect knowledge of the surrounding forage conditions. The pixel with the highest preference score was selected, and foraging was initiated. A range of values was used for the weighting coefficients in equation (1) as well as the spatial search parameters as part of the model building process. Values used in the simulations for equation (1) are described later. To prevent animals from foraging on high RSF pixels with very low or nonexistent forage biomass, we added a constraint that required a selected pixel to contain 80 percent of the forage biomass of the previously selected pixel. Although foraging areas still could be selected based primarily on their RSFs, this constraint also had the effect of moderating the rate of forage depletion of the pixels with the highest RSF scores and allowed the simulation of RSF-driven foraging without resulting in infinite pixel searches.

To allow for selection of foraging areas outside the animal's sensory detection range, we nested the neighborhood search within a low-frequency metaneighborhood search that allowed simulated animals to move (i. e. Levy flight, Marell et al. 2002) to another neighborhood if larger values for equation (1) were found. We experimented with a range of values for the search neighborhood size, the metaneighborhood size, and the "jump" frequency and distance. In the current simulations, we set values for the metaneighborhood at 100 by 100 pixels, the jump frequency at 0.1 and the jump distance at 3,281 feet (1,000 meters).

Once a foraging pixel was selected, consumption of forage (grass, forbs and shrubs) was modeled with simulated individual bites. Bite size was estimated using data from foraging trials conducted at Starkey (Findholt et al. 2004) and elsewhere (J. Cook, personal communication), and it was 0.04 ounces (1.1 g) for cows, 0.007 ounces (0.20 g) for mule deer and 0.02 ounces (0.55 g) for elk. It should be noted that we did not constrain intake rate by bite size or other bite-dependent variable (Gross et al. 1993); hence, the bite process served primarily as a mechanism to sample the three types of vegetation data in the pixel over successive bites. Bite selection in the pool of simulated forage at each pixel was modeled as a Monte Carlo process that simulated successive bites that removed forage types in proportion to the sum of total forage available multiplied by simulated forage DE at the pixel, quantified as:

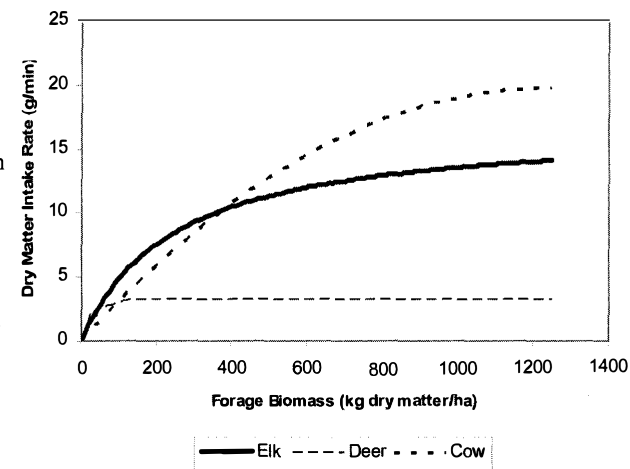
$$P_{ts} = \frac{(F_{pdt} * WB_{mass}) + (DE_{pmt} * WB_{qual})}{\sum_{1}^{t} [(F_{pdt} * WB_{mass}) + (DE_{pmt} * WB_{qual})]} \tag{2}$$

where P_{ts} equals probability of removing forage type t for species s $(0 < P_{ts} < 1)$; F_{pdt} equals forage (in kilograms per hectare) of type t on pixel p at day d; DE_{pmt} equals digestible energy (in megacalories per kilogram) for forage type t, pixel p, and month m; WB_{mass} equals weighting factor for forage biomass; and WB_{qual} equals weighting factor for forage quality.

This foraging process simulated removal of vegetation in proportion to biomass and energy content, or some weighted combination. It also recognized that, while animals can focus their foraging on specific forage types, other nonpreferred types are also depleted at a lesser rate. Initially, we used WB_{mass} of 1.0 and WB_{qual} equal to metabolic body weight (body weight$^{-0.75}$), with the idea that mule deer would select for high forage DE, and cattle would select for forage bulk (Findholt et al. 2004). Elk, with their intermediate body weight, were simulated as having a foraging behavior intermediate to that of mule deer and cattle (Findholt et al. 2004). Initial simulations showed that stronger weighting of the energy component was needed to significantly influence the forage composition.

Using the foraging rules described above, simulated animals were allowed to forage until they consumed 4.8 ounces (135 g) of forage dry weight per kilogram of metabolic body weight (Cook et al. 2004) or until the total foraging time per day exceeded 12 hours (Cook 2002), whichever condition came first. The foraging time was calculated using relationships between standing biomass and intake rate from Wickstrom et al. (1984:1,291) for elk and for mule deer, and from data from Starkey for cattle (Figure 1). For elk, we used the relationship for mixed forest conditions presented by Wickstrom et al. (1984) and combined the grass and mixed-forest data to develop a relationship for mule deer. Intake rates could also have been predicted using relationships between bite size and plant size (Spallinger and Hobbs 1992), but the latter data were not available for conditions at Starkey.

Figure 1. Relationship between standing forage biomass and dry matter intake rate for elk, mule deer and cattle. Functions for elk and deer were developed from data in Wickstrom et al. (1984). The elk relationship was developed from the Wickstrom et al. (1984) mixed-forest type relationship. The function for cattle was developed from grazing trials on Starkey and the bison data in Spallinger and Hobbs (1992).



Energy balance and weight change was updated daily using prorated monthly energy requirements (Table 3) obtained from a number of sources (Hudson and White 1985a, b; Cook 2002). Daily energy generated by consumed forage was calculated, using the energy conversion equation as:

$$Me = 1000 \times (F \times (0.038 \times \%DE + 0.18)/1.22) \quad (3)$$

where DE equals digestible energy (megacalories per kilogram of forage) and F equals forage biomass (dry matter in kilograms per hectare) consumed on a given day of forage.

Negative energy balances were translated into a weight loss by using a conversion rate of 6 megacalories per kilogram. Positive daily energy balances

Table 3. Daily energy demands of adult female mule deer, cow and elk, in millicalories per day, by month. Data from Hudson and White (1985a, b), Sheehy (1987) and Cook (2002).

Species	Month						
	Apr.	May	June	July	Aug.	Sept.	Oct.
Cattle	23.0	23.0	23.0	22.0	21.0	19.0	18.0
Elk	10.0	10.5	16.0	15.9	13.2	12.0	11.0
Deer	3.0	3.0	6.3	4.3	4.3	3.6	3.1

were translated into a weight gain by using the conversion rate of 12 megacalories per kilogram.

Most simulations used herd sizes of 500 cows, 450 elk and 250 mule deer under a summer deferred-rotation grazing system (April 15 to November 15, 210 days). These are the approximate stocking rates and the summer range foraging season at Starkey. In other simulations, the stocking rates varied, depending on the objective of the simulation. On each day, cattle foraging was simulated first, followed by elk and then mule deer, which gave cattle preference over elk and mule deer and which gave elk preference over mule deer for the available forage (Coe et al. 2004). Initial weights were set at 992 pounds (450 kg), 507 pounds (230 kg) and 132 pounds (60 kg) per animal for cows, elk and mule deer, respectively, based on data from Starkey. Typical execution time for the model was about one minute. We first ran simulations to examine the effects of different weights in the pixel preference equation on animal performance, foraging patterns and movements. This involved 125 simulations where each weight was varied by a factor of 10 between 1 and 100,000. We selected a set of weights where the model outputs appeared to be not overly influenced by the values and replicated observed animal performance at Starkey. The effects of different weights in equation (2) were then tested in a similar process in an additional 25 simulations. We then ran additional simulations to test how incremental changes in the number of cattle, mule deer and elk (2–2,500) and in forage production (10–100 percent of normal) affected animal performance. The latter simulations were intended to represent varying drought intensities. Reductions in forage quality from drought (Vavra and Phillips 1980, Weisberg et al. 2002) were not modeled due to limited data.

Results

Simulations using a range of values (1–100,000) for the W_{rsf} W_{mass} and W_{qual} coefficients in equation (1) produced reasonable weight gains for cattle, elk

Figure 2. Simulated weight change in cattle for a range of values for W_{rsf} and W_{mass} in equation (1). X axis contains values for the W_{mass}, (forage biomass) weights for the first equation. Legend entries are the values for W_{rsf} in equation (1). Animal populations were 500 cattle, 450 elk and 250 mule deer.

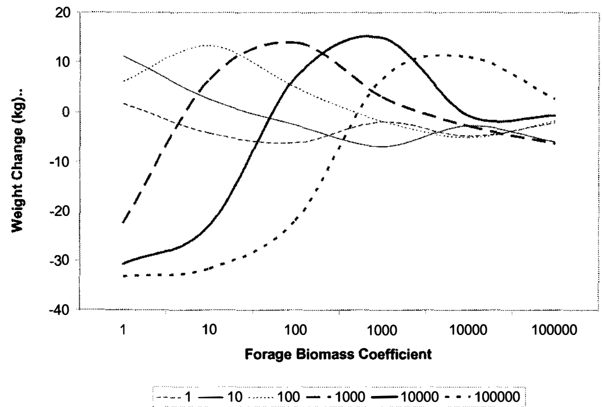


Figure 3. Simulated weight change in mule deer for a range of values for W_{rsf} and W_{mass} in equation (1). X axis contains values for the W_{mass}. Legend entries are the values for the W_{rsf}. Animal populations were 500 cattle, 450 elk and 250 mule deer.

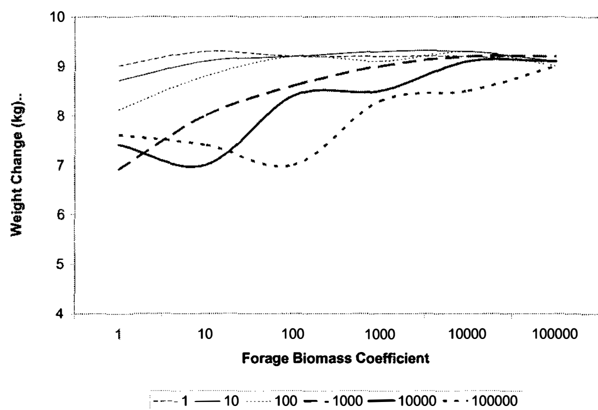
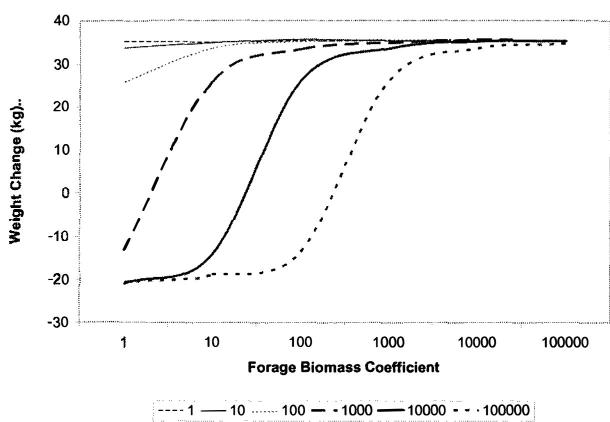


Figure 4. Simulated weight change in elk for a range of values for W_{rsf} and W_{mass} in equation (1). X axis contains values for the W_{mass} weights for equation (1). Legend entries are the values for the W_{rsf} weights in equation (1). Animal populations were 500 cattle, 450 elk and 250 mule deer.



and mule deer (Figures 2–4). For instance, mule deer, which generally gain around 11 to 22 pounds (5–10 kg) per animal at Starkey, showed simulated weight gains of 15.4 to 19.8 pounds (7–9 kg) for the range of coefficients tested. Cattle and elk showed more pronounced changes in animal weights (Figures 2, 4); although, a wide range of coefficients replicated the weight changes observed for cattle 0 to 22 pounds (0–10 kg) and elk 22 to 44 pounds (10–20 kg) at Starkey. For all species, increasing W_{rsf}, relative to Wmass, forced simulated animals to forage in areas of high RSF values (Figure 5) and generally resulted in decreased animal weights. The effect of increasing Wrsf on weight reductions was dampened as the F_{mass} coefficient was increased to values above 1000.

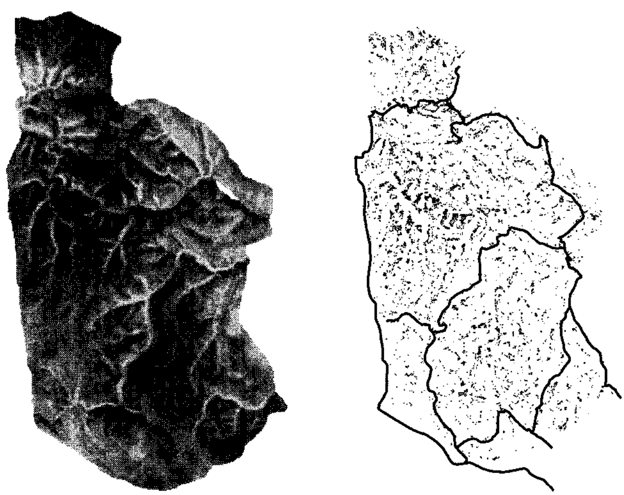


Figure 5. (A) Plot of resource selection functions for elk developed for Starkey data (Coe 2004). Values plotted were the sum of the monthly RSF scores, described in Coe (2004), and range from near 0.15 (white) to 1.5 (black). (B) Results of simulation showing relative forage removal by elk within the Starkey area using W_{rsf} of 10,000 and W_{mass} of 1,000. Dark areas correspond to areas of highest forage removal.

Changes in average cattle weights ranged from -72.6 to 26.4 pounds (-33 to 12 kg) (Figure 2), the negative weight changes being associated with a high values of W_{rsf} and low values of W_{mass}. Cattle showed an intermediate optimal weight gain of 22 pounds (10 kg) when the W_{rsf} was increased by a factor of 10 over the W_{mass}. This trend was not found for elk or mule deer (Figures 3, 4). The most plausible explanation for this is that a higher forage quality is realized at this combination of W_{mass} and W_{rsf}; although, this was not tested.

Results of simulations for elk showed weight changes between -44 pounds (-20 kg) and 77 pounds (35 kg), with weight gains over a wide range of W_{rsf} and W_{mass}. However, when the W_{rsf} became 1,000 times the W_{mass}, negative weight changes were observed. Unlike cattle, weight changes with different combinations of W_{rsf} and W_{mass} were asymptotic with the maximum values at about 77 pounds (35 kg). Compared to the W_{mass} and W_{rsf} coefficients, changing the forage quality (W_{qual}) coefficient had a very minor effect, producing weight differences less than 2.2 pounds (1 kg) over the entire range (1–10,000) of values simulated.

Simulated animal distributions were compared with the maps of the RSF scores to examine how well the model replicated observed animal distributions at Starkey (Figure 5). For space reasons, we limit the comparison to elk and note that the findings for elk are typical for the cattle and mule deer. The comparison is made difficult by the fact that the RSF maps represent a long-run probability of animal use during presumed periods of peak foraging based on 6 years of telemetry data; whereas, the outputs from a simulation run represent animal use for one season and represent only foraging activities. We did not perform statistical testing of the differences in simulated versus observed distributions; although, this would have provided more definitive comparison. The maps show that simulations with high values of Wrsf generated animal distributions that were compatible with the RSF maps (Figure 5). In contrast, simulations with a relatively high weighting for W_{mass} generated markedly different animal distributions that reflected high levels of foraging on productive grassland meadows (Figure 6a).

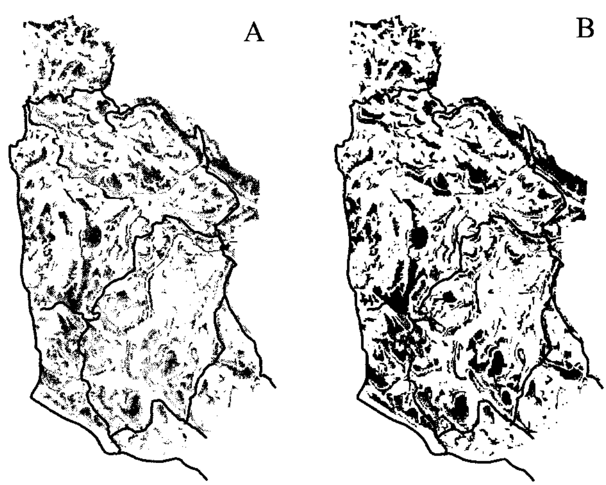


Figure 6. Results of simulation showing relative forage removal by elk within the Starkey area using weights of W_{rsf} of 1,000 and W_{mass} of 10,000. Dark areas correspond to areas of highest forage removal. (B) Same as (A), with forage production reduced to 10 percent of normal.

The effect of changing W_{rsf} and W_{mass} weights on the relative use of pixels with different RSF scores was examined by assigning the RSF probabilities to integer classes from 1 to 40 and then measuring the forage removal for each class. The integer classes were generated by rescaling the RSF scores by 100 times. Values above 0.4 were assigned the integer class 40. Simulations were run with W_{rsf} of 10,000 and W_{mass} of 1, and W_{rsf} and W_{mass} both equal 1. The results (Figure 7) showed that a significant amount of forage was removed from higher RSF class pixels when Wrsf was weighted at 10,000 versus 1. The difference is somewhat magnified, however, by the overall higher total forage removal in the simulations, where both the W_{rsf} and W_{mass} coefficients are set at one.

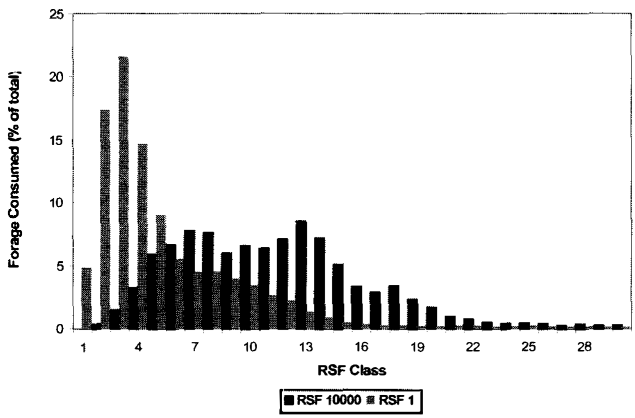


Figure 7. Forage consumption by two simulated elk using W_{rsf} weights of 1 and 10,000 and W_{mass} of 1. Data plotted are the percent of total forage consumed in each RSF class. RSF classes were calculated as RSF times 100. The figure shows that increasing the RSF weight for selecting foraging pixels results in a larger percentage of forage removal from the higher RSF pixels.

To choose a set of coefficients for further simulations we looked for values that resulted in weight changes that approximated those observed at Starkey using the highest possible values of W_{rsf}. In this way we could simulate the approximate animal performance at Starkey while replicating animal distributions to the extent possible. We also were interested in finding coefficients where the simulated weight gains did not change sharply with small changes in the coefficients. Using these criteria we selected a W_{mass} of 1,000 and W_{rsf} of 10,000, with W_{qual} of 1. Then, we simulated a range of values for the WB_{mass} and WB_{qual} coefficients in equation (2). These simulations were to examine how selecting for forage biomass versus energy within a pixel would affect animal performance. The results of this simulation showed that a wide range of coefficients generated the same results for all three species, when the WB_{qual} coefficient was reduced to less than 10. In the latter case, weights dropped by a maximum of 22 pounds (10 kg) for elk and lesser amounts for the other species. Accordingly, we set both WB_{qual} and WB_{mass} at 10 for the remaining simulations.

In a subsequent set of simulations, the forage production was varied from 10 to 100 percent of normal, using the model coefficients selected above. These simulations examined the effect of disturbances, like drought, on animal performance. The results showed that, as forage production was decreased, weights for cattle and elk were markedly reduced, while mule deer were not affected (Figure 8). The effect of reduced forage production on weight change was nonlinear and started when forage production was about 60 percent of normal for cattle and 50 percent for elk (Figure 8). For all species, the response resembled the intake rate functions incorporated into the model (Figure 1). Most likely, the weight reductions resulted from lower intake rates associated with reduced standing forage biomass. Some slight differences were noted in the simulated animal distributions for between normal and 10 percent forage production, the latter showing more area foraged (Figure 6a,b).

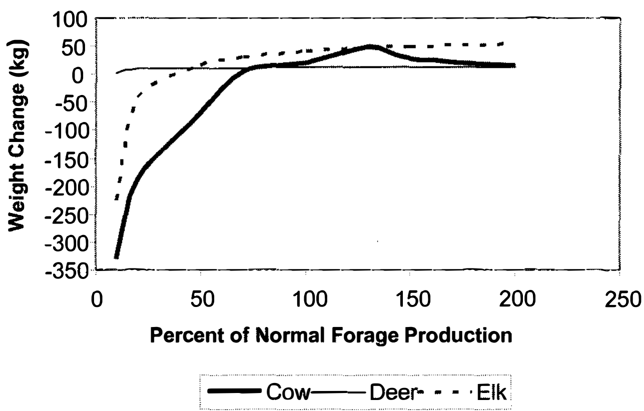
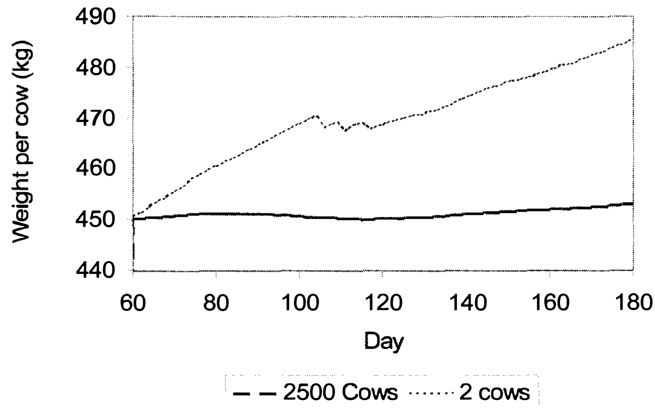


Figure 8. Results of simulations to examine the effect of reductions in forage production on average animal weight change for cattle, elk and mule deer. Simulations used 500 cows, 60 mule deer and 450 elk. Forage production was reduced by a constant percentage of the normal growth rate throughout the growing season.

Simulations to examine how animal performance varied under different population levels showed intraspecific effects for all three species. Simulations where the cattle herd was varied between 2 and 2,500 animals did not result in changes in elk or mule deer weights. However, average weight change per cow was reduced from 76.8 to 7 pounds (34.9 to 3.2 kg) when the herd size was increased (Figure 9). Likewise, when the mule deer population was increased from 2 to 2,500 animals, mule deer weights decreased from 17 to 4 pounds (7.8–1.7 kg) per animal. Elk population increases from 2 to 2,500 animals resulted in elk weights decreasing from 74 to 15 pounds (33.7–6.8 kg) per animal. Interspecific effects on animal weights were negligible, except in the elk

Figure 9. Changes in average animal weight for cattle over the grazing season for a cattle population of 2 and 2,500.



simulations where cattle weights declined from 44 to 35.9 pounds (20.0–16.3 kg) per animal when elk were increased from 2 to 2,500 animals. Elk weight decreased by only a fraction (73.7 versus 73.3 pounds [33.5 vs. 33.3 kg]) per animal and mule deer weights were unchanged when the cattle population was increased from 2 to 2,500.

Discussion

Foraging behavior by free ranging ungulates on large landscapes over time is a complex process that can only be approximated with models (Turner and Wu 1994, Moen et al. 1997, Weisberg et al. 2002). The current work illustrates the inherent complexity of the problem for summer range conditions in the Blue Mountains. While our model does not consider many of compensatory mechanisms in the foraging process, it can replicate animal weight dynamics observed at Starkey as well as provide reasonable predictions of animal distributions. The model demonstrated that both forage biomass and RSF scores need to be included in a simulation model to replicate observed animal distributions and weight changes and that some balance between the two best summarizes actual foraging behavior at the landscape scale. We found that modeling forage site selection based on RSF scores resulted in significant weight loss for cattle and elk and, to a lesser extent, mule deer. Forage depletion on high RSF pixels probably reduced forage intake rates and led to the lower weight gains. In addition, RSF scores for elk and mule deer did not always reflect selection of the most productive foraging areas, due to other habitat considerations, like distance to open roads. When forage site selection was based primarily on standing biomass, the simulated animal distributions were not representative of Starkey telemetry data. Simulations showed that, by weighting the RSF about 100 times less than forage biomass to calculate pixel preference scores, the model would produce reasonable animal weights and select high RSF pixels as well.

Comparing empirical animal distribution with the simulations was difficult because the former developed from six grazing seasons of data and showed more diffuse spatial patterns of animal use, compared to the latter. Although the RSF values used for the model were estimated for peak foraging periods, they likely include observations when animals were not foraging as well. Thus without consideration of these other activities in the model there will always be some discrepancy between RSF values and simulated animal foraging patterns. The two data sources could be made more comparable if the animal distributions generated by the forage model were compared with the same number of animal locations simulated directly from the RSF probabilities.

When we measured forage removal, with respect to RSF probabilities on the Starkey landscape, and changed the RSF weights in the pixel preference equation, we found that the model did indeed lead simulated animals to spend more time foraging in areas with higher RSF scores. Using these methods, additional simulations could be performed to measure the loss of foraging opportunities as a result of selecting foraging pixels on the basis of distance to roads or other human influences. In this way, the effect of human disturbance on animal performance could be examined.

We were also able to quantify changes in animal performance, resulting from a reduction in forage production at the landscape scale. Reductions in forage production might result from drought or natural disturbance. Changes in animal weight with decreasing forage production closely resembled the functional response of intake rate to decreasing forage biomass for the three species (Figure 1) and shows the importance of forage intake dynamics in the context of modeling animal performance (Gross et al. 1993). Simulating animal performance under a range of forage production values should also consider increased movements (Wickstrom et al. 1984) and, in the case of drought-limited forage production, a reduction in forage quality (Vavra and Phillips 1980, Weisberg et al. 2002). The latter relationship could easily be incorporated into the SFM; although, there is little data from which to develop a quantitative relationship. Vavra and Phillips (1980) observed a 20- to 30-percent reduction in digestible dry matter during a drought year when precipitation was 39 percent of normal. Reductions in forage quality of this magnitude would have a significant impact on simulated animal weights.

We observed negligible interspecific effects on animal weight when population levels of each species were varied between 2 and 2,500 animals. However intraspecific effects were observed for all three species, manifested in reduced weight gain, compared to simulations where population levels replicated those at Starkey. Weisberg et al. (2002) also found stronger intraspecific than interspecific competition for forage when they modeled cattle and elk on shared range. Hobbs et al. (1996) in their study of elk and cattle competition found significant reductions in calf weights while cow weights were not significantly unchanged. Competitive effects among the species might be better studied with our model by examining changes in forage intake rates over the season instead of animal weights. Adding calves to the model might also provide a means to study the competition question in more detail. In any event, additional model refinements and a battery of simulations are needed to carefully examine questions of competition among the three species.

The major challenge to refine the current model is to determine what mechanisms in the foraging process are the most important determinants of landscape scale foraging behavior and animal performance. Environmental heterogeneity (Shipley and Spalinger 1995, Etzenhouser et al. 1998, WallisDeVries et al. 1999, Johnson et al. 2002), movement rules (Gross et al. 1995) and cognitive abilities all influence the foraging behavior of ungulates on large landscapes. For the purposes of analyzing stocking on summer range in the Blue Mountains, some of the finer details of the foraging process may not be needed in the current model. One important gap in the model is the lack of local data on the functional response of intake rate for cattle, elk and mule deer for conditions at Starkey. Development of these relationships should be a high priority since these functions are strong determinants of animal performance for scenarios where forage biomass is limited, due to high stocking rates or low forage production. Modeling intake rate at the bite level, rather than using standing biomass, may provide different results than obtained here since intake rate is poorly correlated with standing biomass for highly selective foragers, like mule deer (Spallinger and Hobbs 1992).

Considerable detail could be added to the energetic component of our model by building on previous work (Wickstrom et al. 1984; Hudson and White 1985a,b). For instance, we did not change energy budgets to reflect increased daily movements at lower levels of standing forage biomass. We also did not consider the energy requirements as a function of animal age. Another important addition would be the growth and development of calves for all three species.

Our ultimate goal is to use the SFM to evaluate different grazing management strategies on summer range landscapes, in areas like the forest types of the interior western United States, and to test various hypotheses about the effects of alternative stocking rates for ungulates. In this regard, the objective might be to identify the existence of key stocking thresholds that correspond to changes in animal performance at the species level (Hobbs et al. 1996). Such a tool is currently not available for use in allotment management planning on lands administered by the U. S. Department of Agriculture, Forest Service and the U. S. Department of Interior, Bureau of Land Management, the two largest federal land managers in the United States. Moreover, the mechanistic structure of our model, based on individual foraging behavior, could help managers and public interests improve their understanding of how ungulates use the landscape to meet their foraging needs.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern, and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy.* 84:1,076–88.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, andM. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management.* 54:A51–A76.

Coe, P. K., B. K. Johnson, K. M. Stewart, and J. G. Kie. 2004. Spatial and temporal interactions of elk, mule deer and cattle. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:656–69.

Cook, J. G. 2002. Nutrition and food. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 259–349. Washington, DC: Smithsonian Institution Press.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. Delcurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs.* 155:1–61.

Etzenhouser, M. T., M. K. Owens, D. E. Spalinger, and S. B. Murden. 1998. Foraging behavior of browsing ruminants in a heterogeneous landscape. *Landscape Ecology.* 13:55–64.

Findholt, S. L., B. K. Johnson, D. Damiran, T. DelCurto, and J. G. Kie. 2004. Diet composition, dry matter intake and diet overlap of mule deer, elk and cattle. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:670–86.

Gross, J. E., L. A. Shipley, T. N. Hobbs, D. E. Spalinger, and B. Wunder. 1993. Functional response of herbivores in food-concentration patches: Tests of a mechanistic model. *Ecology.* 74:778–91.

Gross, J. E., C. Zank, N. T. Hobbs, and D. E. Spalinger. 1995. Movement rules for herbivores in spatially heterogeneous environments: Responses to small scale patterns. *Landscape Ecology.* 10(4):209–17.

Hall, F. C. 1973. *Plant communities of the Blue Mountains in eastern Oregon and southeastern Washington, R-6 area guide 3–1.* Portland, Oregon: U. S. Department of Agriculture, Forest Service Pacific Northwest Region.

Hobbs, N. T., D. L. Baker, G. D. Bear, and D. C. Bowden. 1996. Ungulate grazing in sagebrush grassland: Effect of resource competition on secondary production. *Ecological Applications* 6:218–27.

Holechek, J. L., M. Vavra, and J. Skovlin. 1981. Diet quality and performance of cattle on forest and grassland range. *Journal of Animal Science.* 53:291–8.

Hudson, R. J., and R. G. White. 1985a. Computer simulation of energy budgets. In *Bioenergetics of wild herbivores*, eds. R. J. Hudson, and R. G. White, 261–90. Boca Raton, Florida: CRC Press.

Hudson, R. J., and R. G. White. 1985b. *Bioenergetics of wild herbivores*. Boca Raton, Florida: CRC Press.

Johnson, B. K., A. Ager, S. A. Crim, M. J. Wisdom, S. L. Findholt, and D. Sheehy. 1996. Allocating forage among wild and domestic ungulates—A new approach. In *Proceedings: Sustaining rangeland ecosystems symposium*, *SR 953*, eds. W. D. Edge, and S. L. Olson-Edge, 166–9. Corvallis, Oregon: Oregon State University.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection of mule deer and elk during spring. *Journal of Wildlife Management.* 64:685–97.

Johnson, C. G., Jr., and F. C. Hall. 1990. *Plant associations of the Blue Mountains, R6 ecolological area 3*. Portland, Oregon: U. S. Department of Agriculture, Forest Service Pacific Northwest Region.

Johnson, C. J., K. L. Parker, D. C. Heard, and M. P. Gillingham. 2002. Multiscale behavioral approach to understanding the movements of woodland caribou. *Ecological Applications*. 12:1,840–60.

Marell, A., J. P. Ball, and A. Hofgaard. 2002. Foraging and movement paths of female reindeer: Insights from fractal analysis, correlated random walks, and Levy flights. *Canadian Journal of Zoology*. 80:854–65.

McInnis, M. L., T. M. Quigley, M. Vavra, and H. R. Sanderson. 1990. Predicting beef cattle stocking rates and live weight gains on Eastern Oregon rangelands: Description of a model. *Simulation*. 1990:137–45.

Moen, R., J. Pastor, and Y. Cohen. 1997. A spatially explicit model of moose foraging and energetics. *Ecology*. 78:505–21.

Pyke, D. A., and B. A. Zamora. 1982. Relationships between overstory structure and understory production in the grand fir/myrtle habitat type of northcentral Idaho. *Journal of Range Management*. 35:769–73.

Sheehy, D. P. 1987. *Grazing relationships of elk, deer, and cattle on seasonal rangelands in Northeastern Oregon*. Ph. D dissertation. Oregon State University, Corvallis.

Shipley, L. A., and D. E. Spalinger. 1995. Influence of size and density of browse patches on intake rates and foraging decisions of young moose and white-tailed deer. *Oecologia*. 104:112–21.

Spalinger, D. E., and N. T. Hobbs. 1992. Mechanisms of foraging in mammalian herbivores: New models of functional response. *American Naturalist*. 140:325–48.

Svejcar, T., and M. Vavra. 1985. *Seasonal forage production and quality on four native and improved plant communities—Agricultural experiment station technical bulletin 149*. Corvallis, Oregon: Oregon State University.

Turner, M. G., and Y. Wu. 1994. Simulating winter interactions among ungulates, vegetation, and fire in Northern Yellowstone Park. *Ecological Applications*. 4:472–96.

Van Dyne, G. M., W. Burch, S. K. Fairfax, and W. Huey. 1984a. Forage allocation on arid and semiarid public grazing lands: Summary and recommendations. In *Developing strategies for rangeland* *management*, ed. National Research Council and National Academy of Sciences, 1–26. Boulder, Colorado: Westview Press.

Van Dyne, G. M., P. T. Kortopates, and F. M. Smith. 1984b. Quantitative frameworks for forage allocation. In *Developing strategies for rangeland management*, eds. National Research Council/National Academy of Sciences, 289–416. Boulder, Colorado: Westview Press.

Vavra, M., and R. L. Phillips. 1980. Drought effects on cattle performance, diet quality and intake. *Western Section, American Society of Animal Science.* 31:157–60.

Vavra, M., A. A. Ager, B. K. Johnson, M. J. Wisdom, M. A. Hemstrom, and R. R. Riggs. 2004. Modeling the effects of large herbivores. In *The Interior Northwest Landscape Analysis System, general technical report PNW-GTR-610*, eds. J. Hayes, A. Ager, and R. Barbour, 82–103. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station.

WallisDeVries, M. F., E. Laca, M. W. Demment. 1999. The importance of scale of patchiness for selectivity in grazing herbivores. *Oecologia.* 121:355–63.

Weisberg, P. J., N. T. Hobbs, J. E. Ellis, and M. B. Coughenour. 2002. An ecosystem approach to population management of ungulates. *Journal of Environmental Management.* 65:181–97.

Westenskow, K. J. 1991. *Conditioning bunchgrass on elk winter range*. M. S. Thesis. Oregon State University, Corvallis.

Wickstrom, M. L., C. T. Robbins, and T. Hanley. 1984. Food intake and foraging energetics of elk and mule deer. *Journal of Wildlife Management.* 48:1,285–301.

Wisdom, M. J., and J. W. Thomas. 1996. Elk. In *Rangeland Wildlife*, ed. P. R. Krausman, 157–81. Denver, Colorado: Society For Range Management.

Thermal Cover Needs of Large Ungulates: A Review of Hypothesis Tests

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Larry L. Irwin
National Council for Air and Stream Improvement
La Grande, Oregon

Larry D. Bryant
U. S. Department of Agriculture, Forest Service
Washington, DC

Robert A. Riggs
Boise Cascade Corporation
La Grande, Oregon

Jack Ward Thomas
University of Montana
Missoula

Introduction

A great deal of big game research occurred in western North America during the 1960s through the 1980s, and many advances in our knowledge occurred as a result. Timber harvest increased during this period in many localities, and this trend was often perceived to threaten ungulate populations (Hieb 1976). Thus, it is not surprising that appreciable research in this era focused on relations between forestry and elk (*Cervus elaphus*). Logging's most apparent immediate effect is modification of the forest overstory, and a concept arose that was new, at least in the West, for elk—dense forest cover moderates the effects of harsh weather sufficiently to confer survival and reproductive advantages in the winter, in particular, and during the summer. Forest stands that confer such advantages are referred to as thermal cover.

To some degree, support for the thermal cover concept arose from observations that forest cover can moderate weather. During winter, temperature can be several degrees warmer under forest canopies at night (Reifsnyder and Lull 1965, Parker and Gillingham 1990) due to long-wave radiation emitted from the forest canopy (Moen 1968, Beall 1974, Grace and Easterbee 1979). Long-wave radiation can be absorbed by animals, potentially enhancing energy balance (Grace and Easterbee 1979). Forest canopies also reduce wind speed (Grace and Easterbee 1979) and, theoretically, reduce convective heat loss. During summer, shade from forest cover reduces diurnal, ambient temperature fluctuations and provides opportunity to avoid elevated heat loads that can result from direct solar radiation (Demarchi and Bunnell 1993). A number of habitat selection studies demonstrated that ungulates use dense forest stands disproportionately to their availability at certain times (e. g., Irwin and Peek 1983, Leckenby 1984). Selective use of dense forests under several circumstances was consistent with the prediction that thermal cover was operative; thus, this was assumed to be in response to thermal protection provided by cover (Beall 1974, Armstrong et al. 1983, Leckenby 1984, Zahn 1985, Ockenfels and Brooks 1994). Such an interpretation can be supported by modeling using energy balance equations (e. g., Grace and Easterbee 1979, Parker and Gillingham 1990, Demarchi and Bunnell 1993).

Thus, over time a fairly broad base of literature was developed that supported the thermal cover hypothesis. However, arguments were made that the evidence was not sufficient to demonstrate significant biological benefits. In particular, it was unclear if observed selection for forest cover, when it occurs, was related to thermal protection from harsh weather or if it was due to some other reason (Peek et al. 1982). Evidence of support from habitat selection studies is only inferential (Riggs et al. 1993), and there is virtually no support for the thermal cover hypothesis from experimental research specifically designed to establish cause-and-effect relations. Using simulation models, Swift et al. (1980) and Hobbs (1989) concluded that thermal cover had negligible influences on ungulates during winter. Hobbs (1989) indicated that forage conditions, either during or prior to winter, exerted greater effects on winter survival of mule deer (*Odocoileus hemionus*) than did thermal cover. Although thermal cover benefits were certainly real in theory, their influences in the field may be too infrequent or inconsistent to be meaningful under many circumstances (Riggs et al. 1993).

Nevertheless, the preponderance of original research results supported the concept of thermal cover. As these results became more widely accepted, they were incorporated into a variety of habitat-evaluation models. For elk, variables that measure the abundance and, in some cases quality, of thermal cover have been widely incorporated into habitat evaluation procedures (e. g., Wisdom et al. 1986, Thomas et al. 1988a, Christensen et al. 1993). These models were used extensively to develop national forest plans (Edge et al. 1990) and often were used to make site-specific decisions regarding timber harvest or other management activities across a variety of scales.

Over the last half-century, there have been four explicit experimental tests of thermal cover's influence on well-being of big game (Robinson 1960; Gilbert and Bateman 1983; Freddy 1984, 1985, 1986; Cook et al. 1998). Herein, we briefly review these studies and discuss their management implications in the context of new problems facing the game-management community in the 21st century.

Oregon Elk Study: Cook et al.

Of the studies attempting to directly evaluate the influence of thermal cover on animal performance, the Cook et al. (1998) study was the most intensive and extensive. Moreover, it seems the only study that directly evaluated the effects of thermal cover on big game in summer.

Study Area and Methods

This elk study consisted of a series of controlled experiments conducted from 1991 to 1995 using tractable elk. The study area was located between 4,200 to 4,400 feet (1,300–1,350 m) on a northeast-facing slope in the grand fir (*Abies grandis*) forest zone in the Blue Mountains of northeastern Oregon, land typically used by elk as summer range in the area. As such, the site afforded colder and more mesic winter conditions than typically occur on elk winter ranges in the region, and it provided weather conditions typically encountered by elk during summer.

The study area was created from a contiguous, mature, uneven-aged forest by prescriptive logging in nine 5-acre (2.3-ha) cover treatment units. By random selection, three units were clearcut, three were partially harvested to reduce overstory canopy density (providing "marginal" cover as defined by

Thomas et al. 1988a) and three were left unharvested (providing "satisfactory" cover as defined by Thomas et al. 1988a). This layout provided three replicates of four levels of cover for the experiment: (1) clear-cut habitat with no overstory cover, (2) partially cut habitat averaging 30 percent overstory canopy cover, (3) uncut habitat averaging 70 percent canopy cover and (4) units set up to provide a combination of clear-cut and dense forest habitat to the elk. In the first three treatments, elk were sequestered at the center of the units in a 0.25-acre (0.1-ha) pen. In the combination treatment units, elk were held in a pen that extended 65 feet (20 m) into the clearcut and 165 feet (50 m) into the dense forest, providing elk with the ability to choose among cover types as they desired. All understory vegetation (potential forage) was removed from the pens. The 12 pens were designed to hold 3 elk each, providing a total capacity of 36 elk in the layout.

Experiments consisted of placing elk in the pens for 4-month periods in winter (early December through mid-March) and summer (late May through late September) and of documenting differences in body mass dynamics, nutritional condition and activity. All elk received identical diets fed on a metabolic mass basis ($BM^{0.75}$); they received submaintenance rations in winter to induce loss of 5 to 10 percent of body mass, and they received good quality diets that were adequate for growth in summer. Calves were used in the winter experiment of 1991 to 1992, yearlings in summer of 1992 and yearlings in winter 1992 to 1993. A new cohort of calves were raised in 1993 and were used in the winter experiment of 1993 to 1994, the summer experiment of 1994 and the winter experiment of 1994 to 1995. Elk were randomly reassigned to new pens at the beginning of each seasonal experiment.

Elk were weighed twice weekly, body composition (percent fat, protein, water) was determined at the start and end of each experiment using dilution techniques with deuterium oxide, and activity was recorded automatically with leg-mounted activity sensors.

Findings

Microweather characteristics measured during the study demonstrated that forest canopy reduced wind speed, reduced solar radiation flux during the day, and increased net radiation flux at night. They indicated little to no effect of forest canopy on ambient temperature or relative humidity.

No positive effects of thermal cover on elk were documented during any of the four winter experiments. But, there were significant differences in body mass and body condition dynamics among cover treatments. Generally, elk in the dense forest stands lost most mass and fat; elk in clearcuts lost least mass and fat, whereas mass and fat loss of elk in the moderate cover and combination cover units were intermediate (Figure 1). Each winter of the study, elk in clearcuts lost significantly less mass and body fat than did elk in the thermal cover treatment units. No relevant differences in activity patterns were documented among treatment units during any of the winters. These patterns were consistent across all winters, despite substantial differences in winter weather conditions. They ranged from abnormally cold (monthly temperature averaging as low as minus 10 degrees Fahrenheit (-15°C) and snowy in one winter, cool with substantial rain falling in another winter and moderate to normal temperature and precipitation in the other winters). During the winter 1993 to 1994, substantial rain fell and nine elk calves either died or were removed from the study to prevent death. Of these, five were from the dense cover treatment, three were from the moderate cover treatment, and one was from the zero cover treatment.

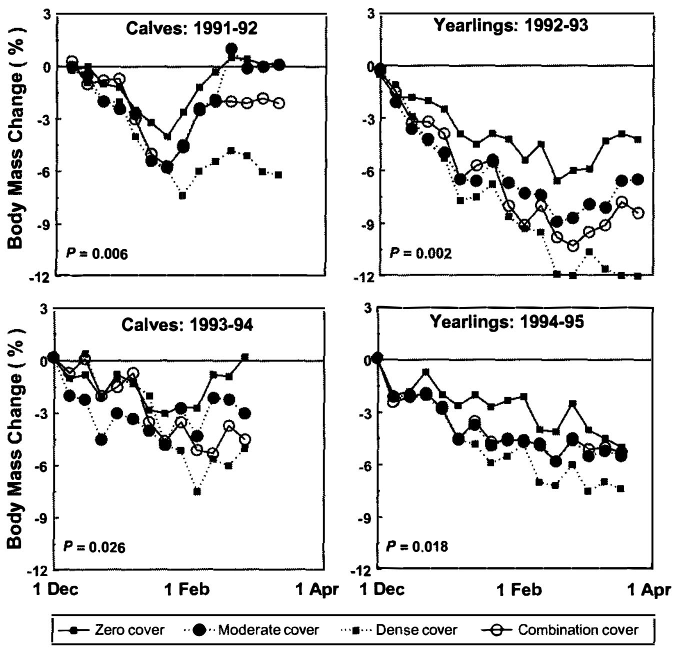


Figure 1. Body-mass dynamics of female elk calves and yearlings in four forest cover treatments during winter 1991 to 1995 in northeastern Oregon (adapted from Cook et al. 1998). P-values indicate statistical significance level of the forest cover treatments on body mass change during each winter experiment.

During the two summer experiments, no significant differences were found in body mass, fat gain or activity patterns among the four cover treatments

(Figure 2). Elk in clearcuts and moderate cover treatment units consumed more water than did elk in dense cover units, however. Both summers were warmer and drier than normal for this region of eastern Oregon.

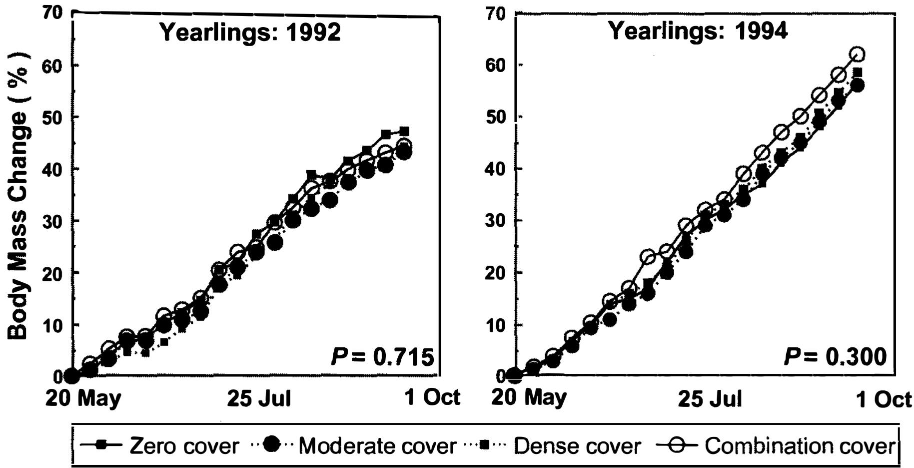


Figure 2. Body-mass dynamics of yearling elk cows in four forest cover treatments during summer 1992 and 1994 in northeastern Oregon (adapted from Cook et al. 1998). P-values indicate statistical significance level of the forest cover treatments on body mass change during each summer experiment.

Colorado Mule Deer Study: Freddy

The thermal cover research reported by Freddy (1984, 1985, 1986) evaluated the effects of thermal cover on mule deer during winter from 1982 to 1985, with the majority of data collected in winters of 1983 to 1984 and 1984 to 1985. It was unique in that thermal cover was provided by human-made structures designed to reduce the effects of heat loss on deer. Deer had free choice among different cover structures that mitigated wind-chill heat loss, radiant heat loss at night and conductive heat loss to snow-packed frozen soil while bedded. They had full access to solar radiation during the day.

Study Area and Methods

The study was conducted at the Colorado Division of Wildlife Junction Butte Research Center near Kremmling, an area located in the sagebrush (*Artemisia* spp.) vegetative zone in a high mountain basin (7,300 feet [2,226 m]) of northcentral Colorado. Twelve individual deer pens were constructed on the site with six equipped with artificial cover and the other six providing no cover. All vegetation was removed within and adjacent to pens, and snow was packed to prevent deer from using snow as a source of cover. Pens were 33 by 33 feet (10 x 10 m), were constructed of large mesh woven wire that was 7 feet (2.2 m) high and were contained within a 5-acre (2.2-ha) fenced pasture that excluded neighboring wild deer.

Deer of various age classes (fawns to adults) were paired by body mass and randomly assigned to either the cover or no-cover treatments in experiments lasting from mid-December through mid-March. In 1983 to 1984, deer were fed pelleted rations ad libitum. Response variables included changes in weight measured at 2.5-week intervals, amount of food consumed at 3-day intervals, and activity determined during 6 behavioral trials during the winter. During the winter of 1984 to 1985, deer were fed a lower quality pellet offered in amounts equaling 60 percent of maintenance, adjusted to declining body mass every 2.5 weeks, such that the deer would be nutritionally stressed over the winter. During this second winter, cover was enhanced by incorporating a three-sided roofed wooden hut that was 4 feet (1.2 m) tall and improved wind-breaks. And, more extensive darkened soil was added to enhance solar absorption and warming.

Findings

The winter of 1983 to 1984 was unusually severe with periods of high winds and very cold ambient temperature to about minus 30 degrees Fahrenheit (-35°C) (minimum temperature averaged below minus 5 degrees Fahrenheit (-20°C) during much of the winter). Mortality in wild deer in the surrounding area represented 50 to 55 percent of the population (D. Freddy, personal communication 2004). Deer with cover lost on average 5.0 percent of their body mass over the entire winter (from December 9 through March 17), whereas deer without cover lost 4.3 percent of their mass. There were no consistent patterns of mass loss within 2.5-week intervals of the winter period (Figure 3A). Amount of food consumed and activity budgets did not differ between deer with and without cover across the entire winter period or within any 2.5-week interval.

Weather during the winter of 1984 to 1985 was milder; although, high winds and cold temperatures (less than minus 5 degrees Fahrenheit [-20°C]) again were common. The restricted feeding regime imposed this year of the study resulted in all deer consuming virtually identical amounts of food per pound of

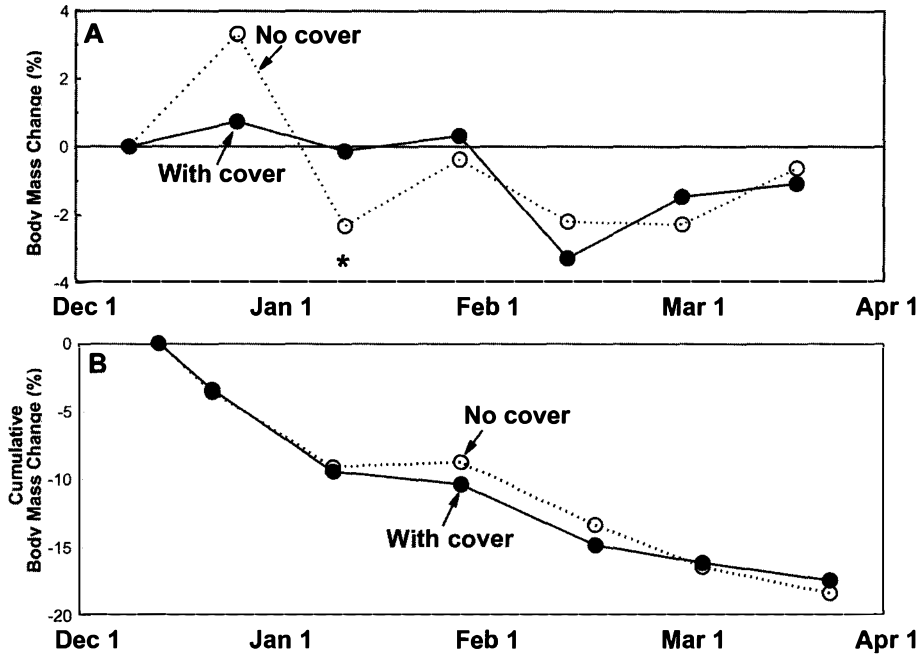


Figure 3. Body-mass dynamics of mixed-age mule deer with and without access to thermal cover during winter 1983 to 1984 (A) and 1984 to 1985 (B) in northcentral Colorado (adapted from Freddy 1984, 1985). In A, data values indicate percent change in body mass during each 2.5-week period of the winter; the asterisk denotes statistically different body-mass change during the 2.5-week period of late December to early January. In B, data values indicate cumulative body-mass change over the entire winter (no significant differences due to cover availability detected).

body mass. Between December 13 and March 23, no significant differences in body mass loss were detected between treatment groups (Figure 3B). At least during the first half of winter, deer without cover spent 5 to 10 percent more of each day attempting to forage than did deer with cover (attempts at foraging were on aspen logs present in the pens only in 1984 to 1985).

Overall, findings of the study indicated that thermal cover failed to significantly improve the ability of mule deer to survive in winter. It was noted that cover in the form of roofed huts with three covered sides along with additional wind breaks provided high-quality thermal protection, at least as good as cover provided by natural forest conditions. Thus, these experiments likely provided a "100-percent test" (Freddy 1985:17) of thermal cover's influence on deer in winter.

Maine White-tailed Deer Study: Gilbert and Bateman

Study Area and Methods

The Gilbert and Bateman study of 1983 was conducted at the D. B. Demeritt Forest at Orono, Maine. The area received 76 inches (193 cm) of snowfall each winter and mean January temperature averaged 18.5 degrees Fahrenheit (-7.5°C). Eleven pens, each 0.25 acres (0.10 ha), were constructed to provide three to four replicates of three overstory conditions: (1) natural tree cover averaging 72 percent canopy cover, (2) absolute clear-cut averaging 6 percent cover and (3) clear-cut with vertical wall windbreaks, averaging 8 percent cover. Each pen contained a feeder in which pellets were provided ad libitum. Brush and tree limbs within reach of deer were removed; snow cover eliminated access to herbaceous vegetation. Eleven mixed-gender white-tailed deer (*Odocoileus virginianus*) fawns were used, with one deer per pen. This study was conducted over a single winter from December 1970 through March 1971.

Response variables included behavior, food intake, changes in body mass and changes in fat levels. Behavior was estimated by direct observation. Food intake was measured at weekly intervals. Body mass was measured at the start and at the end of the experiment. At the end of the experiment, the deer were euthanized and kidney fat and femur fat indices were calculated.

Findings

Ambient temperatures were not clearly defined but ranged as low as minus 30 degrees Fahrenheit (-34°C) in this study. Deer in the natural forest and absolute clear-cut pens consumed equal amounts of food; whereas, deer in the clear-cut pens with windbreaks tended to consume less food. Similarly, body mass loss was virtually identical for deer in the natural forest pens (6.1 percent) and deer in the absolute clear-cut pens (6.2 percent); whereas, deer in the clear-cuts with windbreaks lost 9.6 percent of their starting mass. Femur and kidney fat indices tended to be higher for deer in the natural forest pens than for deer in either of the clear-cut treatments. However, apparent differences in food intake, fat indices and body mass changes failed to vary significantly among deer in any of the three cover treatments. Activity patterns were affected by changes in weather, but no differences were detected as a function of cover treatments.

Despite finding no effect of forest cover on deer, including on food consumption, the authors held to a conservative conclusion that, "if fawns have access to an adequate food supply, they can withstand severe winter weather even in the absence of shelter" (Gilbert and Bateman 1983:399). But, the authors also indicated that, "increasing attention should be given to the quality of food available on winter range," (Gilbert and Bateman 1983:399) and they referenced the importance of physical condition of fawns in late autumn.

Maine White-tailed Deer Study: Robinson

Study Area and Methods

The Robinson study (1960) was conducted at Orono, Maine, during the winters of 1957 to 1958 and 1958 to 1959. Forests in three 1.5-acre (0.6-ha) pens were modified to produce sparse, moderate and dense coniferous cover. The sparse cover pen consisted of trees no larger than 5 inches (12.7 cm) diameter at breast height (d.b.h.) and supported canopy cover of 34 percent. The moderate pen contained an all-ages mixture of hardwood and conifers with a coniferous canopy cover of about 55 percent. The dense cover pen was described as an early-mature stand of several species of conifers with canopy cover of 73 percent. Natural browse within access of the deer was removed from the pens. Temperature during the study ranged as low as minus 22 degrees Fahrenheit (-30°C) up to about 34 degrees Fahrenheit (1°C), and snow accumulations seldom exceeded 2 feet (0.6 m).

During the first winter experiment, one male and one female fawn were placed in each pen; three males and two females were placed in each pen during the second winter experiment. Deer were fed a pelleted ration, the quantity of which was restricted to induce a submaintenance level of nutrition. At the start and at the end of the experiments, body mass, girth and hind foot length were estimated, and blood samples were collected. A condition factor was calculated based on body mass and hind foot length. A visual body condition score also was used to evaluate status of the deer. At the end of the experiment, deer were euthanized, and femurs were used to estimate femur fat levels. Meteorological measurements, including nocturnal minimum temperatures, humidity and wind speed, were collected during the study.

Findings

Minimum temperature at night was lower and wind speed was greater in the sparse cover unit than in either of the other two treatment units.

Nevertheless, decline in condition based on the condition factor was virtually identical among the three treatment areas in both winters of the study, and no differences in condition based on their body condition score were detected. Femur fat levels indicated no differences among treatments at the end of the first winter. Femur fat tended to be lower in deer in the sparse cover unit than in the moderate and dense cover units after the second winter experiment, but this was attributed to generally poorer condition of deer in the sparse unit at the start of this experiment.

Conclusions were conservative in that the failure to find significant benefits of dense forest cover was attributed to flaws in the study. In particular, the sparse cover pen contained a small patch of relatively dense forest cover. The author indicated this patch provided sufficiently moderate microclimatological conditions to preclude differences in animal response among the three treatment units.

Discussion

Observations that dense forests can moderate harsh weather in summer and winter intuitively support the thermal cover hypothesis and provide the bioenergetic basis of models that illustrate a significant benefit of thermal cover (e. g., Grace and Easterbee 1979, Demarchi and Bunnell 1993). But, what has never been clear was whether or not the magnitude of the thermal cover effect, relative to the anatomical and physiological adaptations of large mammals, was sufficient to induce a biologically relevant improvement in energy balance over sufficient time to improve survival and reproduction (Freddy 1984, Riggs et al. 1993). The preponderance of experimental evidence indicates that the weather-moderating effects of thermal cover are probably insufficient to be of much biological value, at least under the conditions reflected in the experimental studies. Furthermore, findings that thermal cover could actually be detrimental (Cook et al. 1998) indicate that energetic benefits of greater solar radiation levels in the clearcuts can far outweigh whatever benefits may arise from thermal cover. Solar radiation effectively warms the animal (Parker and Robbins 1984, Parker and Gillingham 1990), thereby reducing the amount of food or endogenous energy required for thermal stasis. In the Colorado deer study, failure to show a positive effect of solar radiation would be expected because solar radiation was equally available in all pens. Reasons are unclear for failure to show a positive effect of solar radiation in the two Maine deer studies, but overcast skies typical of the northeastern United States would reduce the frequency of direct solar input.

Although forests can moderate harsh winter weather, substantially in the case of wind, Cook et al. (1998) showed that the magnitude of the modification may be negligible, depending on short- and long-term weather patterns. For example, Cook et al. (1998) reported that elevated winds normally occurred during the day when temperature was moderate; thus, the potentially deleterious effect of wind was reduced (see Blaxter et al. 1963, Chappel and Hudson 1978). However, wind was typically calm at night when cooler temperatures prevailed. Additionally, the contribution to energy balance of long-wave radiation from forest canopies is only relevant at night under clear skies because clouds also radiate long-wave radiation (Reifsnyder and Lull 1965). Clear skies at night are a relatively infrequent occurrence in maritime ecosystems of the Northwest (Cook et al. 1998). Similarly, because long-wave radiation accounts for temperature differences between forested and nonforested areas, it is only on clear nights when ambient temperature differences occur between forested and nonforested stands (Reifsnyder and Lull 1965). Even on clear nights, however, gentle breezes and cold airflows across unlevel terrain mix air masses and reduce temperature differences (Cook et al. 1998)

The study by Cook et al. (1998) was the only study to directly test the effect of thermal cover in summer. In both summer experiments, the yearling elk were fed diets that were suboptimal, that is, adequate to support growth but below growth rates which these animals are capable. Thus, this diet should have induced moderate energetic stress and would be expected to heighten sensitivity of these elk to their energetic environment. In both summers, maximum ambient temperatures were greater than or equal to 77 degrees Fahrenheit (25°C) during at least 30 percent of the days. Parker and Robbins (1984) reported that upper critical temperature (the point at which metabolic rate increases to dissipate heat) of yearling elk while standing was 77 to 86 degrees Fahrenheit (25–30°C) (operative temperature, which integrates effects of wind, solar radiation and long-wave radiation). Thus, elk in this study, particularly those in the no-cover treatments, should have been heat-stressed because operative temperature ranges markedly higher than ambient temperature during sunny days (Demarchi and Bunnell 1993).

Studies with livestock indicate severe heat stress can affect performance, particularly of high producing breeds (e. g., milk cows) (National

Research Council 1984, Young 1988). This occurs because high ambient temperature reduces feed intake (Young 1988) and may induce panting and elevate respiration, thus increasing maintenance requirements (National Research Council 1984). Heat-stress effects are greatest when temperature and humidity are both high and when nocturnal cooling is minimal, such as in the southeastern United States (National Research Council 1984, Young 1988). Despite observations that large ungulates often use shade in summer (Zahn 1985, Demarchi and Bunnell 1993, but see Merrill 1991), results of the Oregon experiments show that regimes of ambient temperature, humidity and solar radiation occurring on western montane summer ranges in forest zones fail to exert sufficient heat stress to affect performance of elk.

Parker and Robbins (1984) showed that elk in summer pelage are well-adapted to high operative temperature, even that in excess of their upper critical temperature, mostly due to a relatively great concentration of cutaneous sweat glands. Merrill (1991) calculated that heat loss of elk standing in openings during hot summer days was sufficient to compensate for solar radiation flux and high ambient temperature in the Cascades of western Washington. McCorquodale and Eberhardt (1993) concluded that elk colonizing the Arid Lands Ecology Reserve, a low-elevation, hot, shrub-steppe area in Washington, did so with no appreciable detriment to their fitness (e. g., population growth rate was among highest ever reported for elk) despite the lack of any forest cover.

The reductionist approach used in the four studies directly testing the thermal cover hypothesis certainly is open to some criticism because these studies cannot possibly account for all of the complex interactions among habitat, weather, animal behavior and animal performance inherent to free-ranging settings. Nevertheless, the value of such manipulative studies is that they can indeed tease out possible cause-and-effect relations and test their veracity. In contrast, most empirical support for the thermal cover hypothesis arose from observations of habitat selection. Linking animal survival and reproduction to availability of thermal cover, based on observations of animal choice for dense forests, was first questioned over two decades ago (Peek et al. 1982). Since then, recognition has increased that habitat selection studies as typically conducted must infer explanations for observed selection patterns because they generally are ineffective for identifying cause-and-effect mechanisms, particularly mechanisms that link habitat characteristics to animal performance and carrying capacity (Hobbs and Hanley 1990, Parker et al. 1999, Morrison 2001). Thus, the practice of attributing effect (in this case, observed habitat use) to cause (in this case, enhancing energy balance) amounts to hypothesis generation, not hypothesis testing.

Management Implications

The experimental studies outlined above evaluated the weather-moderating influences of forest cover (i. e., influences on wind speed, ambient temperature, and long- and short-wave radiation fluxes). They did not evaluate other potentially beneficial aspects of forest cover, which under some circumstances could include enhanced security, reduced snow depth and a better foraging environment. Thus, results of these experimental studies cannot be used to categorically reject all potential benefits of forest cover to elk. Together, however, they indicate the thermal cover benefit attributed to dense forest cover is probably not operative across a considerable range of climate, including climates in boreal ecosystems of the northeastern United States, maritime ecosystems of the inland Pacific Northwest, and in cold, dry ecosystems of the central Rocky Mountains. We see little justification in these ecological settings for retaining thermal cover as a primary component of habitat evaluation models for elk.

Wildlife management's focus on thermal cover has had an important influence on the profession's thinking in that it implicitly directed attention to the importance of energy balance to reproductive success and survival. Regardless of the thermal cover hypothesis' lack of veracity, the central theme it addressed remains credible. Among habitat attributes that can be managed, these remain fundamental to energy balance: forage quality and forage quantity, due to their effect on energy intake, and structural attributes of habitat that mediate energy expenditures associated with travel and harassment (e. g., snow intercept, security cover).

Reducing human disturbance via minimizing unrestricted road traffic, reducing attendant harassment and providing security cover have hardly been controversial among biologists. Along with providing thermal cover, these goals formed the primary emphasis and focus of current management paradigms embodied in habitat planning models and procedures used on behalf of elk (Christensen et al. 1993). Meanwhile, attention to nutrition was low and, in some cases, was deemphasized (Edge et al. 1990, Cook et al. 1998). This occurred despite a growing body of science demonstrating nutrition's importance to animal performance (e. g., National Research Council 1984, Vernne and Ullrey 1984, Cook 2002).

Continued attention to cover management may be warranted in many circumstances where security is low or where snow accumulations limit animal performance. However, it may be time to shift our attention toward relationships between herd productivity and nutrition-based attributes of habitat. The inverse relation between overstory canopy density and forage abundance (McConnell and Smith 1970, Klinka et al. 1996) indicates an important tradeoff between providing dense forest cover and providing forage resources that affect carrying capacity (e. g., Hett et al. 1978). Although research has certainly demonstrated the importance of energy balance to animal performance, the last 30 years of research has paid little attention to developing coarse-scale, management-level habitat models and planning procedures that address nutritional issues, and it has paid little attention to the effects of specific silvicultural systems that may influence the nutritional value of large landscapes to large herbivores. Defining overstory-understory relationships and their relation to animal nutritional status, particularly in the context of specific silvicultural systems, remains a substantial hurdle to modeling elk-production relations in western forest landscapes.

The biopolitical setting has changed considerably over the last two decades, creating new challenges to elk managers and researchers. First, gone are the days of comparatively high timber harvests on federal land and, with them, the concerns that emerged 30 years ago (Hieb 1976). The declines in timber harvest, in turn, may increase concern about reductions in the benefits from timber harvest, namely increased forage abundance. Where forestry practices have intensified on private land, more articulate understanding of how intensive forestry practices influence elk forage values would certainly be useful.

Second, in many areas, particularly in the northwestern United States, elk herds are declining (Irwin et al. 1994, Gratson and Zager 1999, Ferry et al. 2001), like mule deer herds across much of the West (Carpenter 1998). Thus, to a greater degree than in the past, if habitat models are to be relevant, they must integrate those aspects of habitat that significantly influence productivity and demographics of elk herds. Most extant elk-habitat models, "predict only effectiveness of the habitat to facilitate elk use. They are neither designed nor expected to predict elk numbers or productivity" (Thomas et al. 1988b:6). Yet, numbers and productivity are indeed the variables ultimately of interest to elk managers and the public. Thus, it seems to us that we should refocus our attention on these variables and their relationship to forestry to a greater extent in the 21st century than we have in the past.

Reference List

Armstrong, E., D. Euler, and G. Racey. 1983. Winter bed-site selection by white-tailed deer in central Ontario. *Journal of Wildlife Management*. 47:880–4.

Beall, R. C. 1974. Winter habitat selection and use by a western Montana elk herd. Ph. D. dissertation, Missoula, Montana: University of Montana.

Blaxter, K. L., J. P. Joyce, and F. W. Wainman. 1963. Effect of air velocity on the heat losses of sheep and cattle. *Nature*. 197:1,115–6.

Carpenter, L. H. 1998. Deer in the West. In *Proceedings of the 1997 deer/elk workshop*, ed. J. C. DeVos, Jr., 1–10. Rio Rico, Arizona: Arizona Game and Fish Department.

Chappel, R. W., and R. J. Hudson. 1978. Winter bioenergetics of Rocky Mountain bighorn sheep. *Canadian Journal of Zoology*. 56:2,388–93.

Christensen, A. G., L. J. Lyon, and J. W. Unsworth. 1993. *Elk management in the northern region: Considerations in forest plan updates or revisions*, *general technical report INT-GTR-303*. Ogden, Utah: U. S. Department of Agriculture, Forest Service.

Cook, J. G. 2002. Nutrition and food. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 259–350. Washington, DC: Smithsonian Institution Press.

Cook, J. G., L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1998. Relations of forest cover and condition of elk: A test of the thermal cover hypothesis in summer and winter. *Wildlife Monographs*. No. 141:1–61.

Demarchi, M. W., and F. L. Bunnell. 1993. Estimating forest canopy effects on summer thermal cover for Cervidae (deer family). *Canadian Journal of Forestry Research*. 23:2,419–26.

Edge, D. W., S. L. Olson-Edge, and L. L. Irwin. 1990. Planning for wildlife in national forests: Elk and mule deer habitats as an example. *Wildlife Society Bulletin*. 18:87–98.

Ferry, M. L., T. S. Peterson, and J. C. Calhoun, editors. 2001. Status of elk populations on the Olympic Peninsula. *Olympic Natural Resources Center conference proceedings*. Forks, Washington: University of Washington, Olympic Natural Resources Center.

Freddy, D. J. 1984. Quantifying capacity of winter ranges to support deer-Evaluation of thermal cover used by deer. In *Colorado Division of Wildlife*, 21–25. Denver, Colorado: Wildlife Research Report.

———. 1985. Quantifying capacity of winter ranges to support deer-Evaluation of thermal cover used by deer. In *Colorado Division of Wildlife*, 13–36. Denver, Colorado: Wildlife Research Report.

———. 1986. Quantifying capacity of winter ranges to support deer-Evaluation of thermal cover used by deer. In *Colorado Division of Wildlife*, 9–18. Denver, Colorado: Wildlife Research Report.

Gilbert, F. F., and M. C. Bateman. 1983. Some effects of winter shelter conditions on white-tailed deer, *Odocoileus virginianus*. *Canadian Field Naturalist*. 97:391–400.

Grace, J., and N. Easterbee. 1979. The natural shelter for red deer (*Cervus elaphus*) in a Scottish glen. *Journal of Applied Ecology*. 16:37–48.

Gratson, M. W., and P. Zager. 1999. *Study IV: Factors influencing elk calf recruitment, project W-160-R-25, subproject 31, 1998 progress report*. Lewiston, Idaho: Idaho Department of Fish and Game.

Hett, J., R. Taber, J. Long, and J. Schoen. 1978. Forest management policies and elk summer carrying capacity in the *Abies ambilis* forest, western Washington. *Environmental Management*. 2:561–6.

Hieb, S. R., editor. 1976. *Proceedings of the elk-logging-road symposium*. Moscow, Idaho: Forest, Wildlife, and Range Experiment Station, University of Idaho.

Hobbs, N. T. 1989. Linking energy balance to survival in mule deer: Development and test of a simulation model. *Wildlife Monographs*. No. 101:1–39.

Hobbs, N. T., and T. A. Hanley. 1990. Habitat evaluation: do use/availability data reflect carrying capacity? *Journal of Wildlife Management*. 54:515–22.

Irwin, L. L., J. G. Cook, R. A. Riggs, and J. M. Skovlin. 1994. *Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystems, general technical report PNW-GTR-325,* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Irwin, L. L., and J. M. Peek. 1983. Elk habitat use relative to forest succession in Idaho. *Journal of Wildlife Management*. 47:664–72.

Klinka, K., H. Y. H. Chen, Q. Wang, and L de Montigny. 1996. Forest canopies and their influence on understory vegetation in early-seral stands on West Vancouver Island. *Northwest Science*. 70:193–200.

Leckenby, D. A. 1984. *Elk use and availability of cover and forage habitat components in the Blue Mountains, Northeast Oregon, 1976–1982. Wildlife Research Report 14*. Portland, Oregon: Oregon Department of Fish and Wildlife.

McConnell, B. R., and J. G. Smith. 1970. Response of understory vegetation to ponderosa pine thinning in eastern Washington. *Journal of Range Management*. 23:208–12.

McCorquodale, S. M., and L. E. Eberhardt. 1993. The ecology of elk in an arid environment: An overview of the Hanford elk project. In *Proceedings of the Western States and Provinces Elk Workshop*, eds. R. L. Callas, D. B. Kock, and E. R. Loft, 56–63. Sacramento, California: California Department of Fish and Game.

Merrill, E. H. 1991. Thermal constraints on use of cover types and activity time of elk. *Applied Animal Behavioral Science*. 29:251–67.

Moen, A. N. 1968. Energy exchange of white-tailed deer, western Minnesota. *Ecology*. 49:676–82.

Morrison, M. L. 2001. A proposed research emphasis to overcome the limits of wildlife-habitat relationship studies. *Journal of Wildlife Management*. 65:613–23.

National Research Council. 1984. *Nutrient requirements of beef cattle. 6th edition*. Washington, DC: National Academy Press.

Ockenfels, R. A., and D. E. Brooks. 1994. Summer diurnal bed sites of Coues white-tailed deer. *Journal of Wildlife Management*. 58:70–5.

Parker, K. L., and M. P. Gillingham. 1990. Estimates of critical thermal environments for mule deer. *Journal of Range Management*. 43:73–81.

Parker, K. L., M. P. Gillingham, T. A. Hanley, and C. T. Robbins. 1999. Energy and protein balance of free-ranging black-tailed deer in a natural forest environment. *Wildlife Monographs*. No. 143:1–48.

Parker, K. L., and C. T. Robbins. 1984. Thermoregulation in mule deer and elk. *Canadian Journal of Zoology*. 62:1,409–22.

Peek, J. M., M. D. Scott, L. J. Nelson, D. J. Pierce, and L. L. Irwin. 1982. The role of cover in habitat management for big game in northwestern United States. *Transactions of the North American Wildlife and Natural Resources Conference*. 47:363–73.

Reifsnyder, W. F., and H. W. Lull. 1965. *Radiant energy in relation to forests. technical bulletin 1344*. Washington, DC: U. S. Department of Agriculture, Forest Service.

Riggs, R. A., J. G. Cook, L. L. Irwin, and J. L Spicer. 1993. Relating timber management to cover for big game on interior northwest winter range: Some thoughts on reducing conflict. In *Proceedings of the western states and provinces elk workshop*, eds. R. L. Callas, D. B. Kock, and E. R. Loft, 18–36. Sacramento, California: California Department of Fish and Game.

Robinson, W. L. 1960. Test of shelter requirements of penned white-tailed deer. *Journal of Wildlife Management*. 24:364–71.

Swift, D. M., J. E. Ellis, and N. T. Hobbs. 1980. Nitrogen and energy requirements of North American cervidae in winter: A simulation study. In *Proceedings of the second international reindeer/caribou symposium*, eds. E. Reimers, E. Gaare, and S. Skjennegert, 244–51. Roros, Norway: Direktoratet for vilt og ferskvannsfisk, Trondheim.

Thomas, J. W., D. A. Leckenby, L. J. Lyon, L. L. Hicks, and C. L. Marcum. 1988b. *Integrated management of timber-elk-cattle: Interior forests of western North America*, *general technical report PNW-GTR*-225. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Thomas, J. W., D. A. Leckenby, M. Henjum, R. J. Pedersen, and L. D. Bryant. 1988a. *Habitat-effectiveness index for elk on Blue Mountain winter ranges, general technical report PNW-GTR-218*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Verme, L. J., and D. E. Ullrey. 1984. Physiology and nutrition. In *White-tailed deer: Ecology and management*, ed. L. K. Halls, 91–108. Harrisburg, Pennsylvania: Stackpole Books.

Wisdom, M. J., L. R. Bright, C. G. Carey, W. W. Hines, R. J. Pederson, D. A. Smithey, J. W. Thomas, and G. W. Witmer. 1986. *A model to evaluate elk habitat in western Oregon, publication R6-F&WL-216-1986*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Young, B. A. 1988. Effect of environmental stress on nutrient needs. In *The ruminant animal: Digestive physiology and nutrition*, ed. D. C. Church, 456–67. Englewood Cliffs, New Jersey: Prentice Hall.

Zahn, H. M. 1985. Use of thermal cover by elk (*Cervus elaphus*) on a western Washington summer range. Ph. D. thesis, Seattle, Washington: University of Washington.

Cattle and Elk Responses to Intensive Timber Harvest

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Bruce K. Johnson
Oregon Department of Fish and Wildlife
La Grande, OR

Martin Vavra
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Jennifer M. Boyd
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Priscilla K. Coe
Oregon Department of Fish and Wildlife
La Grande

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Alan A. Ager
U. S. Department of Agriculture, Forest Service,
Umatilla National Forest
Pendleton, Oregon

Norman J. Cimon
*U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station*
La Grande, Oregon

Introduction

Forested habitats for cattle and elk (*Cervus elaphus*) in the western United States have changed substantially in response to intensive timber management during the latter half of the 20th century. Consequently, the subject of how elk and other ungulates respond to timber management is a high-profile, long-standing issue that continues to be studied and debated (Lyon and Christensen 2002). The need for additional knowledge about effects of timber management on cattle and elk remains high, given the fact that timber management continues to affect nearly all cattle and elk ranges on national forests in the western United States (Wisdom and Thomas 1996, Lyon and Christensen 2002).

Accordingly, we conducted a landscape experiment regarding cattle and elk responses to timber harvest and associated human activities at the Starkey Experimental Forest and Range (Starkey) (Figure 1) in northeastern Oregon. Our specific objectives were to (1) summarize knowledge regarding cattle and elk responses to timber harvest; (2) document changes in spatial distributions and weight gains by these ungulates; (3) document changes in elk vulnerability to hunting, as measured before, during and after timber harvest at Starkey; and (4) describe management implications of our findings related to timber harvest planning.

Measuring Effects of Timber Management on Ungulates

We define timber harvest as "logging" activities that extract merchantable wood from forest stands, that is, tree felling and bucking and the subsequent yarding, decking and hauling of logs for subsequent manufacture of wood products. By contrast, we define timber management as timber harvest and all other activities implemented in direct support of sustainable production of merchantable wood. Timber management therefore includes field layout of harvest units (cut units), such as marking trees and harvest boundaries, site

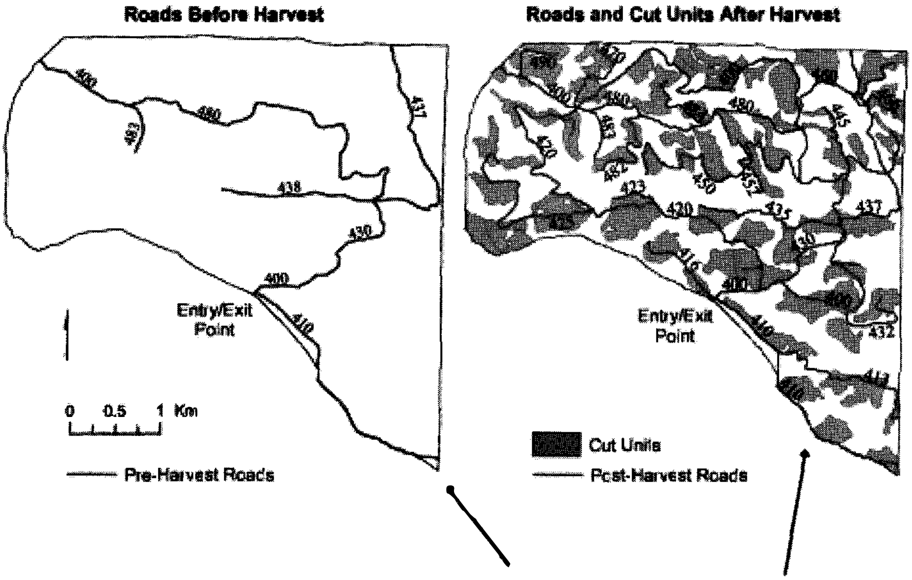
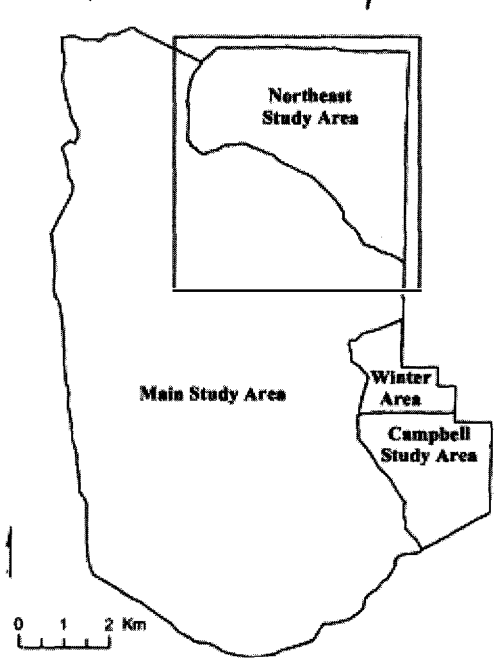


Figure 1. Northeast Study Area of Starkey, a 3,590-acre ungulate-proof enclosure, with roads before timber harvest (top left) versus roads and cut units after timber harvest (top right). The main haul routes used by log trucks during timber harvest (1992) included the 400, 420, 430, 437, 440, 460 and 480 roads that formed a set of interconnected loops spanning the inner portions of the study area (top right) and that led to the entry/exit point of the enclosure on its southwest boundary.



preparation after timber harvest, such as prescribed burning, windrowing and mechanical removal of fine fuels, seeding or planting of commercially-desirable tree seedlings after site preparation, and subsequent mechanical thinning of trees and other silvicultural treatments designed to enhance wood production.

Past research indicates that cattle and elk responses to timber harvest can change substantially in relation to four factors: (1) the types of response variables measured, (2) the spatial scales of measurement, (3) the time period following timber harvest over which the response variables are measured (temporal scale), and (4) the confounding effects of other human activities associated with timber harvest, such as increased human activities from increased road access. Types of response variables can include estimates of change in ungulate behavior (Ward 1976), resource selection (Lyon 1976), or population performance (Leege 1976). Changes in habitat condition brought about by timber harvest, such as modification of biomass and quality of forage in relation to each ungulate's requirements, also have been studied extensively (e.g., Hershey and Leege 1976, Lyon 1976, Miller and Krueger 1976, Svejcar and Vavra 1985). Such differences in response variables in relation to timber harvest have contributed to differences in results and conclusions among various studies, leading to confusion about potential effects on ungulates. For example, documenting a change in ungulate habitat condition or habitat use in response to timber harvest says nothing about the effect on ungulate populations or their nutritional status (see discussion by Garton et al. 2001).

Similarly, the spatial scales at which ungulate responses are measured directly affect the interpretation of results (Boyce et al. 2002, Parker 2002). Spatial scale refers to the extent (boundaries and size of an area evaluated) over which an evaluation is conducted and the mapping resolution (accuracy of each mapping unit, such as a pixel or polygon) at which a response is measured at a given extent (Turner et al. 2001, Gutzwiller 2002). For example, ungulate responses can be measured within a given stand that is subjected to timber harvest, or measured among a mosaic of stands that surround the area of harvest. Measurements of ungulate responses at the stand-level, versus a mosaic of stands encompassing the area of timber harvest, yield different but complementary insights (Boyce et al. 2002, Parker 2002).

One of the most important factors affecting ungulate responses to timber harvest is the temporal scale at which effects are measured. We define near-term effects as ungulate responses measured 1 to 10 years after timber harvest.

We consider long-term effects as ungulate responses measured more than 10 years after harvest. Confounding the long-term responses of ungulates to timber harvest is the frequency of subsequent harvest activities. For example, if timber harvest occurs every 10 years in a given watershed, it may not be possible to measure a long-term response to any one set of harvest units beyond that of individual stands.

Perhaps the most challenging aspect to studies of timber harvest is the confounding effect of other human activities associated with harvest, such as road construction and the subsequent changes in human access. Increased human access can lead to increased hunting pressure on elk, owing to the larger network of roads established as part of timber management (Christensen et al. 1991, Unsworth et al. 1993). Although these factors can be mitigated, such as with road obliterations or closures, the effects of such management activities are difficult to evaluate separately from the effects of other harvest activities, such as tree felling and bucking, during the time of harvest. By contrast, changes in human access resulting from timber harvest can be studied as a separate effect after harvest is completed.

Overview of Current Knowledge

Elk and other ungulates typically thrive in early-seral forests, owing to the high biomass of palatable forage produced under these open-canopy sites (see Wisdom and Cook 2000). Consequently, the increased forage produced by timber harvest could be perceived as a positive event to ungulates. Several factors however, provide confounding influences, both short- and long-term, complicating this perception.

Short-term disturbances by the act of timber harvest (Ward 1976, Edge and Marcum 1985), concomitant road building, and resultant traffic (Hershey and Leege 1976, Leege 1976, Perry and Overly 1976, Ward 1976) affect elk behavior. From a review of several studies, Lyon and Christensen (2002) found that elk are sometimes displaced from harvest areas by as much as 5 miles (8 km). Most often, however, the distance elk moved appeared to be the minimum required to avoid contact with people and equipment. Continual logging within an individual watershed (5 consecutive years) may impose learned behavior that delays return to previously used habitats (Lyon 1979). Edge et al. (1985) reported that home ranges of individual animals were not altered when areas of extensive cover remained available within their home range. The authors speculated that where cover is limited, harvest activities increase home-range size and reduce home-range fidelity.

Many studies support the concept that timber harvest is beneficial to forage production for elk and other wild ungulates (Hershey and Leege 1976, Leege 1976, Lyon 1976, Schroer et al. 1993, Unsworth et al. 1998). In the Coast Range of western Oregon, Crouch (1974) found that clearcutting and slash burning were the most practical means of maintaining black-tailed deer (*Odocoileus hemionus columbianus*) habitat, at no cost to wildlife managers. Continued harvest was required as forage production declined rapidly post-harvest because of the rapid regeneration rate of coniferous trees in the Coast Range.

Size of harvest units, however, has a substantial influence on subsequent forage use by ungulates. Scott et al. (1982) reported that small areas of disturbance are used more heavily than large ones. In that light, optimum size and arrangement of timber harvest units have been identified to maximize ungulate use of resulting forage areas. Lyon (1976) reported that harvest units of 10 to 40 acres (4–16 ha) were optimum. Reynolds (1966) found elk use was heavy on clearcuts more than 20 acres (more than 8 ha), but use declined as opening size increased. Lyon and Jensen (1980) suggested that elk are more prone to use large harvest units in regions where large natural openings occur. Edge and Marcum (1991) suggested that both topography and forest cover should be considered in the development of logging operations and road placement. Wisdom et al. (1986) and Thomas et al. (1988) summarized knowledge of elk use in relation to edges between forage and cover areas, showing that highest use occurs within 100 yards (91 m) of such edges, decreasing with distance from the edges. These summaries supported the earlier guidelines established by Black et al. (1976) and Thomas et al. (1979), which identified a ratio of 40-percent cover to 60-percent forage areas as an optimal mix of habitats for elk.

If timber harvest changes human access within and near the harvest units, elk distributions can be expected to shift. Elk will avoid areas with increased access, selecting areas with little or no access (Wisdom and Cook 2000). Specifically, high road densities negatively influence elk distribution, in that elk avoid habitats near roads open to traffic (Rowland et al. 2000, Wisdom et al. 2004). The influence of open roads is not uniform, however, in that elk show increasing avoidance of areas near roads with increasing rates of motorized traffic (Wisdom et al. 2004). Cole et al. (1997) found that road management areas where access was restricted to administrative uses reduced home-range size and increased the survival of Roosevelt elk. Additionally, Lyon (1976) found that elk used habitats with greater canopy closure in areas of higher road density.

Timber harvest also affects the vulnerability of elk to hunting. Lyon and Christensen (1992:3) defined elk vulnerability as a "measure of elk susceptibility to being killed during the hunting season." Elk vulnerability to hunting can be affected by two aspects of timber harvest (Christensen et al. 1991). First, the removal of timber opens up the landscape, making elk more visible and therefore more vulnerable to harvest by hunters. Second, the increased number and extent of roads established to facilitate removal of logs greatly enhances the opportunity for more hunters to access the landscape, increasing the likelihood of hunter contact with elk. These effects were documented by Leptich and Zager (1991), Unsworth et al. (1993) and Hayes et al. (2002), and they were discussed at length in the compilation of papers by Christensen et al. (1991).

In contrast to elk, no major changes in cattle distribution during timber harvest are likely because domesticated animals typically are confined to pastures and generally tolerate human activities. Following timber harvest and the concomitant decrease in overstory canopy, however, a release of understory vegetation occurs that alters forage biomass, quality and phenology, often leading to changes in cattle distribution. Forage biomass can increase from two to eight times that of preharvest forage production (Svejcar and Vavra 1985) depending on intensity of the cut, site potential and soil disturbance (Hedrick et al. 1968). Miller and Krueger (1976) reported that 60 percent of the forage consumed in a given pasture by cattle was from areas logged and reseeded. Road construction to facilitate harvest also provides improved distribution for cattle (Hedrick et al. 1968). Consequently, timber harvest generally provides new grazing areas for cattle.

Harris (1954) also reported that cattle seldom use dense overstory canopies except during conditions of extreme heat or intense insect harassment. Hedrick et al. (1968) found it more difficult to obtain moderate or heavy forage use by cattle under dense overstory canopies than under open canopies. Consequently, cattle distribution and use of new forage areas can be expected to increase substantially after timber harvest, until such time that overstory canopies again become dense.

Depending on season of use, livestock production may or may not be improved by timber harvest. Svejcar and Vavra (1985) found that forage quality on timber-harvested sites declined more rapidly than on unharvested sites. Consequently, weight gain per individual animal could actually decline in late summer on harvested sites. However, weight gain per acre could improve dramatically if stocking rates were increased to match the improvement in forage production in early summer and midsummer (Svejcar and Vavra 1985).

Timber Harvest Experiment at Starkey

Methods of Implementing the Timber Harvest Activities

We studied cattle and elk responses to timber harvest in the 3,590-acre (1,454-ha) Northeast Study Area of Starkey (Figure 1) from 1989 to 1996, encompassing periods before, during, and after harvest. The area is enclosed with an ungulate-proof fence, allowing direct measurements of ungulate responses to controlled, landscape experiments, such as timber harvest, traffic, hunting and other public land uses (Rowland et al. 1997, Wisdom et al. 2004). To study cattle and elk responses to our timber harvest experiment, a mosaic of units was harvested across the study area over a short time period, and no other human activities beyond those associated with timber management and hunting were allowed.

Timber sale planning and layout of harvest units occurred from 1989 to 1991, timber harvest and log hauling occurred during 1992, and conifer regeneration activities (site preparation, planting of tree seedlings and stocking surveys) occurred from 1993 to 1996. Throughout our analyses and paper, we refer to the period of timber sale planning and layout as before harvest, the period of timber harvest and log hauling as during harvest and the period of conifer regeneration activities as after harvest. Details about each period were described by Rowland et al. (1997)

Timber harvest during 1992 consisted of approximately 7 million board feet of commercial tree species that were logged from 1,207 acres (488 ha) of the study area from November 1991 through 1992 (referred to here as 1992 or during harvest period). Timber harvest encompassed 50 percent of forested lands in the Northeast Study Area (Figure 1). Harvest was a salvage sale that focused on removal of grand fir (*Abies grandis)* and Douglas-fir (*Pseudotsuga menziesii)* that had been killed by the combined effects of western spruce budworm (*Choristoneura occidentalis*), Douglas-fir tussock moth (*Orgyia pseudotsugata*), and drought during 1988 to 1990.

Most timber harvest took the form of shelterwood and seed tree regeneration cuts, with some commercial thinning and individual tree selections. The 63 harvest units ranged in size from 3 to 55 acres (1.2–22.0 ha). Harvest units were dispersed throughout the study area, denying ungulates the opportunity to find large areas of escape cover (Figure 1). Moreover, management guidelines for elk cover, such as maintenance of dense cover for presumed hiding and thermoregulatory benefits (Thomas et al. 1979, 1988), were intentionally ignored as part of the experimental design. Finally, the relatively small size of the study area (3,590 acres [1,454 ha]), smaller than many summer ranges used by elk (Leckenby 1984, Edge et al. 1985), combined with the ungulate fence, did not allow animals the option of avoiding the experiment by leaving the area or moving to locations far from harvest activities. Instead, the experiment was intentionally designed to measure cattle and elk responses to changes in the environment brought about by timber harvest, in the absence of options for ungulates to avoid the experimental area.

Extensive road construction also took place to facilitate log removal (Figure 1). Approximately 24 miles (39 km) of new roads were constructed. Another 4 of the 10 miles (6.5 of the 16 km) of roads present before timber harvest were renovated. The study area, however, was closed to public access, with the exception of hunting seasons each fall, when hunters were allowed entry for hunting purposes only (Rowland et al. 1997).

Despite no public entry, the study area received substantial road traffic in relation to timber sale planning and layout, timber harvest and log hauling, and conifer regeneration activities. Motorized traffic entering the study area during timber sale planning and layout (1989–1991) averaged 10 vehicles per day and was composed mostly of U. S. Forest Service vehicles and some vehicles associated with reconnaissance work by logging crews. During timber harvest and log hauling (1992), traffic entering the study area increased to an average of 29 vehicles per day; this traffic was composed almost entirely of log trucks that were hauling logs from the harvest units and of minor traffic from logging crews and U. S. Forest Service contract administrators.

When harvest was completed (1993–1996), traffic entering the study area declined substantially to an average of three vehicles per day, composed mostly of U. S. Forest Service vehicles and some contractor's vehicles associated with conifer regeneration activities. An even lower rate of traffic (less than 1 vehicle per day) occurred during the hunting seasons each fall, from 1989 to 1995, when hunters were allowed vehicle entry only to retrieve game brought to roads. Traffic rate increased to more than 10 vehicles per day during hunting seasons in 1996, when vehicle access by hunters was allowed on established roads.

For all periods of study, estimates of traffic rate were based on automated traffic counters installed and monitored at or near the entry/exit point to the study area (Figure 1), using methods described by Rowland et al. (1997) and Wisdom et al. (2004). An automated, 16-millimeter camera also was installed and monitored at the entry/exit point (Rowland et al. 1997), which verified that traffic was dominated by U. S. Forest Service vehicles before harvest, by log trucks during harvest and again by U. S. Forest Service vehicles after harvest.

Methods of Measuring Cattle and Elk Responses to Timber Harvest

We assessed the short-term effects of timber harvest on cattle and elk by evaluating their spatial distributions before (1989–1991), during (1992) and after (1993–1996) harvest. We also evaluated annual weight gains of each species before, during and after harvest across those same years. Finally, we estimated the vulnerability of elk to hunting before, during and after timber harvest. The experiment ended in 1996; thus, we did not measure long-term (more than 10 years postharvest) responses of ungulates to timber harvest.

We maintained approximately 50 cow elk, their calves and 12 adult bull elk in our study area each year, from spring through fall, throughout all periods of study. Elk entered the study area during early to mid-April of each year from an adjacent Winter Area (Rowland et al. 1997). By late fall (early to mid-December), elk typically were trapped and moved or baited from the study area back to the adjacent Winter Area (see Rowland et al. 1997). Winter 1991 to 1992 was extremely mild, and we were unable to remove or entice many of the elk from the study area that winter, which reduced sample sizes of animals during 1991 to 1992 that were used to estimate spatial distributions with radio-telemetry (Table 1) and to estimate weight gains. We stocked 50 cow-calf pairs of cattle in the study area from mid-June through mid-October of each year as part of a season-long summer grazing system described by Rowland et al. (1997).

Estimating Changes in Cattle and Elk Distributions. To estimate changes in spatial distributions of cattle and elk, we monitored the movements of radio-collared adult females of each species (Table 1) with an automated telemetry

Table 1. Number of radio-collared elk and cattle for each year before (1989-1991), during (1992), and after (1993-1996) timber harvest in the Northeast Study Area of Starkey Experimental Forest and Range, northeast Oregon.

Year	No. Elk	No. Cattle
1989	10	9
1990	18	15
1991	11	13
1992	4	12
1993	12	11
1994	20	13
1995	13	9
1996	13	10

system described by Findholt et al. (1996, 2002), Rowland et al. (1997) and Kie et al. (2004). We used random samples consisting of 1,000 locations selected from the radio-collared animals that were monitored during each of the three periods: before, during and after harvest (Table 1). Samples of locations therefore were a random composite taken from all radio-collared animals that were monitored in a given period.

We used the animal locations to estimate the distribution of each species during each period under a fixed-kernel analysis (Worton 1989). We used a bandwidth set at 0.5 of the reference bandwidth and plotted volume contours at increments of 0.05 from 0.05 to 0.95 . Kernel methods provide a "probability density estimate of a distribution based on a sample of points" (Seaman et al. 1998:95). The kernel method of estimating and mapping distributions has been used extensively in wildlife research because it is a nonparametric estimator that requires few assumptions in contrast to parametric methods (Kernohan et al. 2001; Marzluff et al. 2001, 2004). Moreover, the kernel method is an unbiased estimator of the underlying, true animal distribution when more than 300 locations of unbiased animal locations are used (Garton et al. 2001).

We initially mapped the kernel distributions of animal locations, by the 0.05-contour intervals, for each species and time period in the study area. These maps provided an overall picture of spatial use by each species for each time period. For all subsequent analyses, we used the portion of each species' distribution occurring within the upper 50 percent of kernel volume, or 50 percent contour, for each time period. The upper 50 percent of kernel volume highlights areas of concentrated use by elk or cattle. Such areas often are referred to as "core areas" in analyses of spatial use (Kernohan et al. 2001). Throughout our paper, we refer to these areas as the "upper 50 percent of kernel volume," "50-percent core area" or "core area." These areas also are referred to as centers of a "utilization distribution," described by Millspaugh et al. (2000) and Marzluff et al. (2001).

Because core areas or centers of utilization distributions are based on a substantially higher number of animal locations than the outer contours of such distributions, the core areas typically provide a more robust estimate, in contrast to estimates made at the periphery of the kernel volume (Seaman et al. 1998, 1999; Marzluff et al. 2001). Moreover, our interest was in monitoring changes in concentrated areas of use (the most frequently used areas) across time periods, as could be done with analysis of the core areas.

We used maps of the upper 50 percent of kernel volume to estimate the change in spatial use by cattle and elk in four ways. First, we calculated the percent area within the upper 50 percent of kernel volume for each species and time period that occurred in each of four regions, or quadrants, in our study area. For this analysis, we subdivided the study area into northwestern, northeastern, southwestern and southeastern quadrants, each of equal size. We then expressed the percent area of the upper 50 percent of kernel volume occurring within each of the quadrants as a percentage, for each species and time period. This analysis allowed us to further quantify the shift in species' distributions across periods. Millspaugh et al. (2000) employed similar concepts in their application of utilization distributions for assessing hunter-elk interactions.

Second, we calculated the percent overlap of the cut units (top right, Figure 1) within the upper 50 percent of kernel volume, by species and time period. This analysis was intended to portray the degree to which cattle or elk distributions may have shifted away from or toward the harvest units during or after harvest. For example, if percent overlap of the harvest units with the upper 50 percent of kernel volume was 70 percent before timber harvest but 35 percent during harvest, it would suggest the species was avoiding the units during harvest activities.

Third, we calculated the percent area within the 50-percent core area for each species and time period across the entire study area. Under this analysis, if a higher percentage of the study area occurred under the core area, it would indicate that distributions of cattle or elk were more diffuse, or spread out, across a larger area. A lower percentage would indicate that distributions were more concentrated in smaller portions of the study area. The degree to which distributions are more diffuse or concentrated has been used as an index of habitat quality in home-range analyses. Larger core areas or home ranges suggest lower habitat quality, whereas smaller areas indicate higher habitat quality, owing to like differences in the area over which animals must meet their needs (see Carey et al. 1992, Zabel et al. 1995, Cole et al. 1997).

Fourth, we assessed the degree to which cattle and elk selected or avoided the mainline roads, which received highest frequency of motorized traffic (N. Cimon, unpublished data 2004) that entered the study area (see traffic rates stated earlier). Mainline roads were identified as the 400, 430 and 480 roads before harvest (top left, Figure 1) and the 400, 420, 430, 437, 440, 460 and 480 roads during and after harvest (top right, Figure 1). For this analysis, we calculated the mean distance of 30- by 30-meter pixels to the closest mainline road, for those pixels within each species' core area during each period; this constituted our estimate of the area used by each species in relation to distance from the mainline roads. We then divided this number by the mean distance of the entire study area's 30- by 30-meter pixels (n = 16,133) to the closest mainline road, which constituted our estimate of the area available to each species in relation to distance from the mainline roads. A ratio of 1.0 of used versus available pixels suggests neither selection nor avoidance of roads. A ratio of greater than 1.0 indicates avoidance; a ratio of less than 1.0 suggests selection toward roads.

We also calculated the mean percent slope and mean percent overhead canopy closure of all 30- by 30-meter pixels within the species' 50-percent core for each time period, and compared these estimates across time and with overall estimates for the study area. We did this to gain further insight as to whether ungulates might have been seeking areas of greater security (Christensen et al. 1991) in proximity to roads or cut units during and after harvest. These variables were available from spatial layers estimated and used in prior ungulate research at Starkey (Johnson et al. 2000, Rowland et al. 2000).

A minor effect on our distribution analyses resulted from our not restricting the kernel estimation to the specific boundaries of the study area. That is, elk and cattle movements were restricted to the enclosure boundaries that formed our study area (Figure 1), but kernel estimation was not. This resulted in a small portion of kernel volume being mapped along or just outside the study area boundaries. Consequently, the core areas used in our analyses would have changed slightly if we had constrained the kernel estimation to the specific enclosure boundaries.

Estimating Changes in Weight Gains of Cattle and Elk. To evaluate potential effects of changes in nutritional status of ungulates following timber harvest, we estimated annual weight gains of each species and compared these annual changes over time. To account for annual variation in ungulate weight gains due to annual differences in weather and associated forage conditions, we also calculated weight gains for cattle and elk in the Main Study Area of Starkey (Figure 1) as a background reference for conditions of no timber harvest. The Main Study Area is immediately adjacent to the Northeast Study Area, is subject to the same weather conditions and was not subjected to timber harvest activities during the period of our study (1989–1996) (see Rowland et al. 1997). Thus, we used the weight gains from the Main Study Area as a control, against which changes in ungulate weight gains in the Northeast Study Area could be compared.

To estimate annual weight gain, we weighed adult cow elk (more than 2 years old) during late winter or early spring (late February to early April), before animals entered each study area, and during late fall or early winter (late November to early January), after these same animals left that study area. We weighed calf elk after they left the study areas in early winter. Similarly, we weighed adult cattle (more than 2 years old) and calves as they entered the study areas in mid-June and as they left the study areas in mid-October of each year.

Annual weight gains were expressed as an average daily gain for elk cows and for beef cows and calves. Specifically, we computed the difference between each animal's weight before it entered the study area and the weight after it left. We divided that difference by the number of days over which the difference was computed. We then calculated a mean daily gain and a 95-percent confidence interval about the mean for each species and age class for each year.

Standardizing each animal's weight gain by the number of days over which the gain was measured accounted for the fact that some animals were weighed earlier or later than other animals before entering or after leaving the study areas. Only elk cows and beef cows and calves with weights measured both before entering the study areas and after leaving the study areas were included in our analysis of a given year's weight gains. Thus, we used a repeated measures approach in calculating weight gains, in that gains were computed for each animal, based on its weight before entering and after leaving the study area, which increased the precision of our estimates.

For elk calves, all of which were born in the study areas, we calculated the weight gain in absolute amount for each animal, based on their fall weights after they left the study area. We calculated a mean weight of elk calves and a 95-percent confidence interval about the mean for each year and study area.

Under our analysis, if timber harvest did not affect weight gains, we would expect that variation in gains observed in our study area would remain consistent with gains calculated for these species and age classes each year in the Main Study Area, and that annual variation in gains in both study areas was due to variation in weather or other factors. By contrast, if timber harvest caused an increase or decline in weight gains, we would expect gains in our study area to increase or decrease in relation to such gains observed for the same species and age class in the Main Study Area.

Monitoring Changes in Elk Vulnerability to Hunting. We evaluated the vulnerability of elk to being killed by hunters before, during and after timber harvest with data from hunts of branch-antlered bulls that occurred in the study area from 1989 through 1996 (Rowland et al. 1997). For each year, we calculated hunter success rates and the number of days required for hunters to harvest an animal, using information from Starkey's hunter check station (Rowland et al. 1997). From 1990 through 1995, the branch-antlered bull hunt was foot-access only, with permission to use vehicles to retrieve dead animals. In 1996, vehicle access during the hunting season was allowed. Under this analysis, if hunter success increased or the number of days required to kill an elk decreased during or after timber harvest, this would suggest that the increased openness and road access resulting from timber harvest caused elk to be more vulnerable to hunting.

Changes in Cattle and Elk Distributions

Before timber harvest (1989–1991), elk were concentrated in the western portion of the Northeast Study Area (Figure 2), particularly in the southwest quadrant (Figure 3). Elk distribution shifted substantially during timber harvest (1992), with use concentrated along portions of the study area's outer boundaries (Figure 2). Elk distribution shifted most to the southeastern and northeastern quadrants during harvest, which had received little use beforehand (top right, Figure 3). Elk distributions also became more diffuse during timber harvest. Nearly twice as much of the study area was within the upper 50 percent of kernel volume compared to the period before harvest (bottom right, Figure 3).

Distribution shifts by elk during harvest were not related to elk avoidance of the cut units, as areas where timber cutting occurred overlapped more with the 50-percent core area for elk during harvest than before harvest (bottom left,

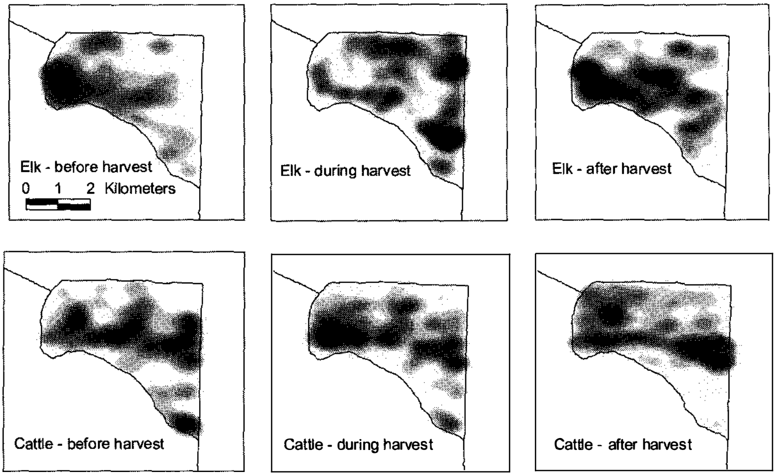


Figure 2. Spatial distributions of elk and cattle before (1989–1991), during (1992), and after (1993–1996) timber harvest, using fixed-kernel analysis of animal locations for each time period in the Northeast Study Area of Starkey, northeastern Oregon.

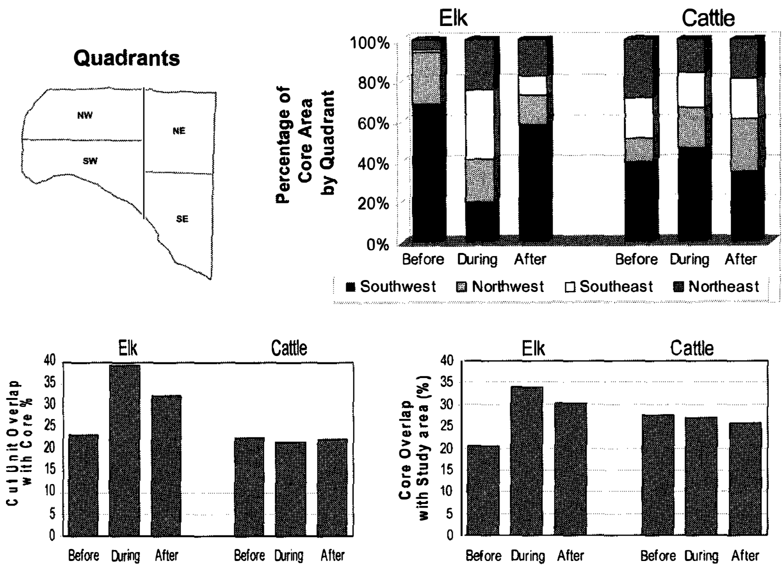


Figure 3. Percentage of the upper 50 percent of kernel volume, or 50-percent core area, of cattle and elk distributions occurring in each of four quadrants (top map) by time period (top right bar charts), the spatial overlap of timber harvest units or cut units, within the 50-percent core area for each species (bottom left bar charts) and the percent area of the entire Northeast Study Area within the upper 50 percent of kernel volume, by time period and species (bottom right bar charts).

Figure 3). Distribution shifts by elk during timber harvest also were not related to elk avoidance of the mainline roads used most frequently by log trucks (400, 420, 430, 437, 440, 460 and 480 roads, Figure 1)—the 50-percent core area for elk during harvest was closer to these roads than overall habitat available to elk. Specifically, the mean distance of all pixels in the 50-percent core area for elk was 655 yards (599 m) from these roads, but the overall area available to animals had a mean distance of 916 yards (838 m) to the same roads, for a selection ratio of 0.72 during harvest. Elk selection toward roads, however, was even stronger before harvest, when the selection ratio was 0.42, indicating that elk decreased their use near roads during harvest when traffic rate increased substantially (per traffic rates stated in methods).

In addition, the 50-percent core area for elk had a steeper average slope (21%) during harvest compared to the average slope (17%), suggesting that elk selected areas of greater security while close to roads. By contrast, the core area for elk had an average slope of 16 percent before harvest, similar to the average slope of 17 percent available to animals. Contrary to the pattern of elk seeking steeper slopes during timber harvest, the core area for elk had a mean overhead canopy closure (33%) similar to its availability (32%). This also was the pattern before harvest, when the core area for elk had a mean overhead canopy closure of 40 percent compared to available canopy closure of 41 percent.

In 1993 to 1996, after timber harvest was completed, elk again concentrated in the western half of the study area, as well as in the interior portions of the study area that were largely unused during timber harvest (Figure 2). As with the period before harvest, elk use was concentrated in the southwestern quadrant (top right, Figure 3). Elk use of the southeastern quadrant also diminished substantially, as occurred during harvest. Elk distribution also was more diffuse after timber harvest compared to before harvest, but not as diffuse as during harvest (bottom right, Figure 3).

After timber harvest, elk also continued to select areas closer to the mainline roads than the area available for use. The mean distance of pixels in the 50-percent core area for elk was 332 yards (304 m) from these roads, but overall habitat available to animals had a mean distance of 916 yards (838 m) to the same roads, for a selection ratio of 0.36 after harvest. This selection ratio was twice as strong as that observed during harvest (0.72), further suggesting that elk moved closer to roads after log hauling was completed and traffic subsided (per traffic rates stated in methods). The average slope of pixels within the 50-percent core area for elk after harvest (17%) also was similar to that before harvest

(16%), indicating that elk moved back to more gentle slopes once timber harvest had ended.

In contrast to elk, cattle showed little change in distribution during all periods of study (Figures 2 and 3). The areas of highest concentration were generally consistent before, during, and after timber harvest, with little change in the percentage of the upper 50 percent of kernel volume by quadrant across time (top right, Figure 3) or across the study area (bottom right, Figure 3). Harvest unit overlap with the 50-percent core area of cattle remained between 20 and 25 percent across all three periods (bottom left, Figure 3). The focal point of cattle distribution was along the Syrup Creek drainage—the east-west continuum of uncut area running east-west through the middle of the study area between the 420 and 480 roads (top right, Figure 1)—coinciding with darkest areas of cattle use during all three periods in Figure 2. Interestingly, overlap of the cut units with the 50-percent core area for cattle was less than that for elk, both during and after timber harvest (bottom left, Figure 3), as was the percent area under the 50-percent core for cattle versus elk during these two periods (bottom left, Figure 3).

Cattle distribution, like that of elk, was consistently closer to the mainline roads than available habitat. The mean distance of pixels in the 50-percent core area for cattle was 1,052 yards (962 m) from these roads, but overall habitat available to animals had a mean distance of 1,206 yards (1103 m) to the same roads, for a selection ratio of 0.87 before harvest. The 50-percent core area for cattle was progressively closer to the mainline roads during and after harvest, with road selection ratios of 0.69 and 0.35, respectively.

Unlike elk, cattle showed no evidence of selection of areas with characteristics of greater security from humans. Average slopes under the core area used by cattle did not appear to change across time periods (before—17%, during—18%, after—16%) and was the same or similar to the average slope of 17 percent in the study area. Moreover, average overhead canopy closure of pixels in the core areas of cattle was highest before harvest (43%) but declined during and after harvest (34% for both periods), a pattern that corresponded to the change in average overhead canopy closure for the study area (42% before harvest, 32% during and after harvest).

Changes in Weight Gains of Elk and Cattle

Annual weights gains of adult female elk and calf elk in the Northeast Study Area were not different from gains for these same age classes in the Main

Study Area, based on overlapping 95-percent confidence intervals between mean gains within each year and age class (Figure 4). Moreover, no trend in weight gains in either study area was evident (Figure 4). Instead, weight gains of elk cows and calves were highly variable across years in the Northeast Study Area, but consistent with the variability in weight gains observed across years in the Main Study Area, where timber harvest did not occur (Figure 4). That is, the direction and degree of variability in annual weight gains was generally consistent between the two study areas, suggesting that weight gains in elk were largely affected by annual variability in weather patterns that affect annual changes in forage biomass and quality.

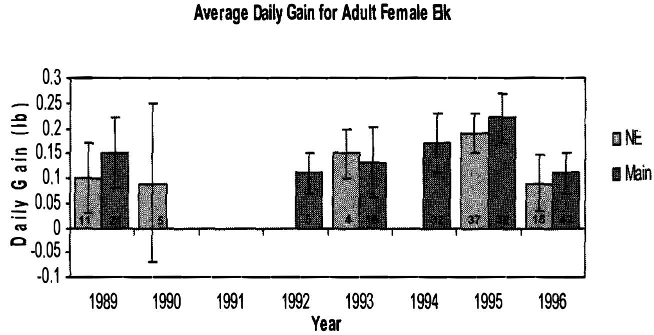
In contrast to elk, mean weight gains for beef cows and calves in the Northeast Study Area were mostly higher (nonoverlapping 95-percent confidence intervals) than those documented in the Main Study Area within each year and age class (Figure 5). In 1992 and 1993, however, the 95-percent confidence intervals overlapped between mean weight gains of beef cows in the two study areas. And, in 1995, mean weight gain of beef cows was higher in the Main Study Area (Figure 5). The pattern of higher weight gains in the Northeast Study Area versus those in Main Study Area was more consistent for beef calves, except for one year (1992) where confidence intervals overlapped.

As with elk, no trend in weight gains in either study area was evident for either age class of cattle (Figure 5). Also similar to elk was the high annual variability in weight gain observed for beef cows and calves and the strong consistency in the degree and direction of this variability between Main and Northeaststudy areas across years (Figure 5). In contrast to elk, however, cattle weight gains were more precise for each year, and the annual variability appeared to be more similar between the two study areas (compare Figure 4 with Figure 5).

Changes in Elk Vulnerability to Hunter Harvest

Compared to the period before timber harvest, hunter success improved and the number of hunter days per harvested animal declined during and after timber harvest (Table 2). The highest hunter success and the lowest number of hunting days required to take an animal occurred in 1996, when postharvest, open conditions existed together with unlimited vehicle access (Table 2).

For the years before timber harvest (1989–1991), hunter success averaged 22 percent, requiring an average of approximately 19 days to achieve this level of success. During timber harvest (1992), hunter success increased to



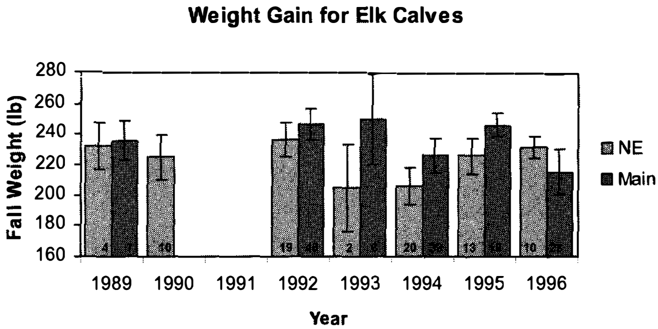


Figure 4. Mean weight gain by adult female elk and calf elk before (1989–1991), during (1992) and after (1993–1996) timber harvest in the Northeast Study Area, compared to mean weight gain for the same years and age classes in the Main Study Area, Starkey, northeastern Oregon. Error bars are the 95-percent confidence intervals about each mean. No weight data were available for cows or calves during 1991 in either study area or for cows in Northeast Study Area during 1994. Data on the insufficient sample size of one cow elk weighed during 1992 in Northeast Study Area also was not included. Sample size (number of animals weighed) is shown at the bottom of each bar.

35 percent, with hunters spending an average of 9 days to achieve this success (number of days required to harvest an animal). For the years after timber harvest (1993–1996), hunter success remained higher and similar to success during the year of timber harvest, with an average success of 32 percent. Moreover, an average of 14 days were required for hunters to take an animal postharvest.

Management Implications

Elk responded to the period of timber harvest by making a substantial shift in distribution, while cattle did not. Interestingly, the shift in distribution by elk and the lack of change in distribution by cattle did not appear to change animal

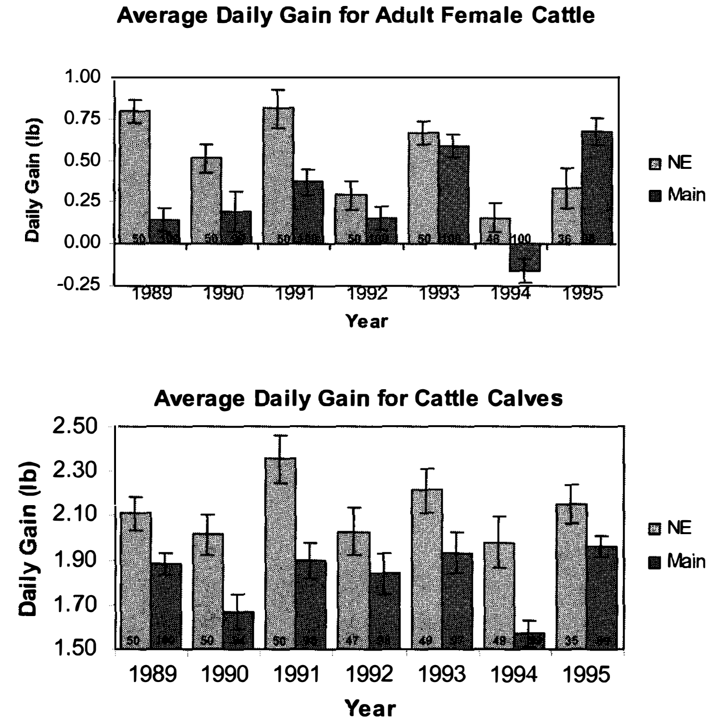


Figure 5. Mean weight gain by cattle, for adult cows and calves, before (1989–1991), during (1992) and after (1993–1995) timber harvest in the Northeast Study Area, compared to mean weight gain for the same years and age classes in the Main Study Area, Starkey, northeastern Oregon. Error bars are the 95-percent confidence intervals about each mean. Weight data for 1996 were not available. Sample size (number of animals weighed) is shown at the bottom of each bar.

performance for either species. Our study is one of the few cases in which a measure of animal or nutritional performance (weight gain) was evaluated in combination with distributional responses of animals under a landscape experiment. Garton et al. (2001) strongly emphasized the need to evaluate the population or nutritional consequences of landscape choices made by wildlife under studies of resource selection, habitat use and spatial distribution. Garton et al. (2001) also highlighted the few studies where the demographic or nutritional consequences of landscape choices made by a wildlife species were documented, and they particularly noted that changes in animal selection or distribution do not always result in changes in population or animal performance.

In our study, the nutritional consequences of each species' spatial response to timber harvest suggest that each species was able to maintain animal

Table 2. Success rates and number of days hunted by elk hunters during branch-antlered bull hunts that occurred before (1989–1991), during (1992) and after (1993–1996) timber harvest in Northeast Study Area, Starkey. Data on number of days hunted and number of days hunted per animal harvested were not available for 1989.

Year	No. hunters	No. elk harvested	No. days hunted	Hunter success (percent)	No. days hunted per animal harvested	No. animals available for harvest	Type of hunter access allowed
1989	10	3	na	30	na	6	foot only
1990	23	5	127	22	25.4	10	foot only
1991	14	2	63	14	31.5	8	foot only
1992	26	9	78	35	8.7	13	foot only
1993	15	3	66	20	22.0	10	foot only
1994	27	5	95	19	19.0	12	foot only
1995	23	8	81	35	10.1	12	foot only
1996	24	13	69	54	5.3	14	vehicle access

performance during and after harvest. Yet, elk substantially changed distribution during the experiment, while cattle did not. These results serve as a demonstration that studies of animal behavior and distribution in relation to human disturbances may not provide strong inference about demographic or nutritional effects.

Our results must be viewed with caution, however, as the relatively small sample sizes on which weight gains were estimated suggests that we had low statistical power and a higher likelihood of committing Type II errors, that is, of falsely concluding that weight gains did not change in relation to timber harvest when, in fact, such changes did occur but were not detected. Moreover, we did not evaluate the long-term (more than 10 years postharvest) effects of timber harvest on weight gains, which could be substantially different than short-term weight gains.

The substantial increase in percent area under the upper 50 percent of kernel volume for elk during and after timber harvest also serves as a cautionary note. This finding indicates that the elk population became more dispersed during and after timber harvest, suggesting longer movements over larger areas by elk to meet their needs. Larger areas used by a species indicate lower habitat quality and reduced population performance when these patterns exist over extended periods (see Carey et al. 1992, Zabel et al. 1995).

We also did not assess other response variables related to animal and nutritional performance of ungulates, such as those studied and discussed by Cook (2002) and Cook et al. (2004). Examples include estimates of pregnancy rates and body fat, which provide complementary insights on ungulate performance beyond measures of weight gain. Moreover, we did not evaluate changes in forage biomass, quality and phenology that could have provided additional insights about changes in potential carrying capacity. We also did not assess diet selection or diet quality of ungulates, which provide direct estimates of the nutritional plane of animals (Cook 2002, Cook et al. 2004). Density of both species in the study area may have been low enough that weight gains for both species were high before timber harvest and, therefore, did not change substantially following the presumed change in forage conditions after harvest.

Results from our analysis of cattle and elk distributions complement an earlier analysis of ungulates in our study area by Stewart et al. (2002), who found that cattle and elk were spatially separated during summer. Our results demonstrated a substantial shift in elk distributions during timber harvest, while cattle distributions were nearly unchanged. Our results, therefore, indicate that elk were responding more strongly to timber harvest activities than to cattle distributions.

Our results also are surprising in that we found no evidence that elk avoided the cut units or the mainline roads during and after timber harvest. The mainline roads received a high frequency of log-truck traffic throughout the harvest period of 1992, and it is possible that elk became habituated to this form of predictable, consistent traffic. This contrasts with the less predictable and diverse forms of motorized traffic that occur when roads are open to the public and that presumably contribute to elk avoidance of these open roads (e.g., Wisdom et al. 2004). Importantly, our study area was not open to public access and motorized traffic, with the exception of highly restricted public traffic that was allowed as part of timber harvest and elk hunting. Model predictions of elk avoidance of roads open to traffic (e.g., Thomas et al. 1979, Rowland et al. 2000) do not account for specialized, restricted forms of high-frequency traffic, such as that associated with timber harvest in areas that otherwise are closed to public access. This aspect of elk response to motorized traffic deserves more attention in future research.

Our results on weight gain for elk also complement an earlier analysis by Rinehart (2001), who found no difference in early winter weights of elk before versus after timber harvest in our study area. Rinehart (2001) also found no difference between annual weights of elk in our study and those in the adjacent, Main Study Area where timber harvest did not occur. Finally, Rinehart (2001) also noted that changes in resource selection and home-range size after timber harvest did not equate to a reduction in animal performance of elk, as indicated by the lack of a trend in early winter weights before versus after harvest.

The high variability in weight gains across years for both ungulate species and all age classes strongly suggests that annual gains were largely affected by annual variability in weather patterns that affect annual changes in forage biomass and quality. This suggestion is supported by work of Vavra and Phillips (1979, 1980), who found that variation in summer precipitation directly affected the diet quality and subsequent weight gains of cattle on northeastern Oregon summer ranges having similar weather and forage conditions as those in our study area. During a drought year (1977), diet quality of cattle was substantially lower than years of higher summer moisture (1975 and 1976), resulting in substantially lower weight gains during the drought year.

These patterns of high variability in weight gains of domestic ungulates in response to year-to-year variation in weather, precipitation and subsequent forage conditions were summarized by Holechek et al. (1989, 1998); these authors concluded that growing season variability in precipitation exerted a strong effect on ungulate performance on rangelands across the western United States, particularly during drought years. Similar inferences were made by Cook et al. (2002) regarding annual variation in elk performance, as affected by annual variation in summer weather that appears to have a strong influence on summer nutrition of animals.

Although animal performance of elk did not appear to change in response to timber harvest, the post-harvest landscape in our study area clearly increased hunter success and reduced the number of hunting days required to harvest an animal. These results suggest that increased visibility and access associated with timber harvest increased the vulnerability of elk to hunters. These results further suggest that timber harvest may have the strongest and most enduring effects on elk vulnerability to hunting, in contrast to other effects we measured, such as changes in distribution and potential avoidance of human disturbances. Consequently, our results suggest that the potential for intensive timber harvest to substantially increase elk vulnerability to hunting is a key issue that deserves careful attention as part of timber harvest planning.

Those points as context, consideration of our results, combined with results from previous studies, could focus on timber sale designs that minimize elk vulnerability to hunting and that provide a relatively even and continuous stream of forage availability over space and time. Example considerations at the watershed scale could include the following:

- Manage for and retain security areas for elk in watersheds when planning the layout of harvest units in time and space. Security areas serve primarily to mitigate any increase in elk vulnerability to hunting when timber harvest activities result in increased visibility and human access in a watershed. A security area for elk was defined by Hillis et al. (1991:38) as a nonlinear block of hiding cover at least 250 acres (101 ha) and at least one-half mile (0.8 km) from roads open to motorized traffic. Hiding cover was defined by Thomas et al. (1979:109) as vegetation capable of hiding 90 percent a standing adult deer or elk from the view of a human at a distance equal to or less than 200 feet (61 m). In particular, Hillis et al. (1991) suggested that security areas are most effective in minimizing elk vulnerability to hunting when such areas compose at least 30 percent of a watershed. However, other management options, such as minimizing road access, accounting for elk security provided by steep slopes and convex topography, and changes in hunting season regulations, also can be considered in tandem with management of security areas. These additional management options were outlined and discussed by Thomas (1991).
- Restrict motorized and mechanized forms of hunter access in watersheds after timber harvest, to prevent an increase in elk vulnerability to harvest by hunters. Maintain such restrictions on motorized and mechanized forms of hunter access until such time that vegetative growth in timber harvest units provides sufficient hiding cover to help reduce vulnerability. In watersheds with flat terrain and large areas subjected to timber harvest over a short time period, restrictions on human access will be especially effective in minimizing an increase in elk vulnerability to hunting. In watersheds with steep terrain and large areas of hiding cover, restrictions on hunter access after timber harvest may be neither necessary nor effective, in that elk may have ample areas in which to hide or find security from hunters. In particular, plan the layout of harvest units to retain security areas (particularly areas with steeper slopes and greater convexity) near the cut units to facilitate animal escapement from these focal points of hunter access (per discussion by Hillis et al. 1991).
- Plan timber harvest activities in time and space such that a mosaic of seral stages are maintained to provide a variety of foraging conditions for

cattle and elk. Timber harvest is likely to cause an immediate but short-term (1- to 3-year) decline in forage availability in the harvest units, followed by a large increase in forage that may last 10 years or longer. The biomass and phenology of forage plants is likely to be more diverse and stable with the implementation of a mosaic of timber harvest activities in time and space. Security areas for elk can also be sustained consistently under a strategy of timber harvest that produces a mosaic of seral stages in time and space.

Reference List

Black, H., R. J. Scherzinger, and J. W. Thomas. 1976. Relationships of Rocky Mountain Elk and Rocky Mountain Mule Deer habitat to timber management in the Blue Mountains of Oregon and Washington. In *Proceedings, Elk-logging-roads symposium,* 11–31. Moscow: University of Idaho.

Boyce, M. S., M. G. Turner, J. Fryxell, and P. Turchin. 2002. Scale and heterogeneity in habitat selection by elk in Yellowstone National Park. *Ecoscience*. 10:421–31.

Carey, A. B., S. P. Horton, and B. L. Biswell. 1992. Northern spotted owls: Influence of prey base and landscape character. *Ecological Monographs*. 62:223–50.

Christensen, A. G., L. J. Lyon, and T. N. Lonner, compilers. 1991. *Proceedings, Elk Vulnerability Symposium*, 1–330. Bozeman, Montana: Montana State University.

Cole, E. K., M. D. Pope, and R. G. Anthony. 1997. Effects of road management on movement and survival of Roosevelt elk. *Journal of Wildlife Management*. 61:1,115–26.

Cook, J. G. 2002. Nutrition and food. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 259–349. Washington, DC: Smithsonian Institution Press.

Cook, J. G., B. K. Johnson, R. C. Cook, R A. Riggs, T. Delcurto, L. D. Bryant, and L. L. Irwin. 2004. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs*. 155:1–61.

Crouch, G. L. 1974. Interaction of deer and forest succession on clearcuttings in the Coast Range of Oregon. In *Proceedings, wildlife and forest* *management in the Pacific Northwest*, ed, Hugh C. Black, 133–8. Corvallis, Oregon: Oregon State University.

Edge, W. D., and C. L. Marcum. 1985. Movements of elk in relation to logging disturbances. *Journal of Wildlife Management.* 49(4):926–30.

———. 1991. Topography ameliorates the effects of roads and human disturbance on elk. In *Proceedings, Elk Vulnerability Symposium*, compilers, A. G. Christensen, L. J. Lyon, and T. N. Lonner, 132–7. Bozeman, Montana: Montana State University.

Edge, W. D., C. L. Marcum, and S. L. Olson. 1985. Effects of logging activities on home-range fidelity of elk. *Journal of Wildlife Management.* 49(3):741–4.

Findholt, S. L., B. K. Johnson, J. W. Kern, A. A. Ager, L. L. McDonald, and R. J. Stussy. 2002. *Adjusting for radiotelemetry error to improve estimates of habitat use, general technical report PNW-GTR-555.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Findholt, S. L., B. K. Johnson, L. D. Bryant, and J. W. Thomas. 1996. Corrections for position bias of a LORAN-C radio-telemetry system using DGPS. *Northwest Science.* 70:273–80.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. In *Radio tracking and animal populations*, eds., J. Millspaugh, and J. Marzluff, 1–42. Sand Diego, California: Academic Press.

Gutzwiller, K. J., editor. 2002. *Applying landscape ecology in biological conservation.* New York, New York: Springer-Verlag.

Hayes, S. G., D. J. Leptich, and P. Zager. 2002. Proximate factors affecting male elk hunting mortality in northern Idaho. *Journal of Wildlife Management.* 66:491–9.

Harris, R. W. 1954. Fluctuations in forage utilization of ponderosa pine range in eastern Oregon. *Journal of Range Management.* 7:250–5.

Hedrick, D. W., J. A. Young, J. A. B. McArthur, and R. F. Kenniston. 1968. *Effects of forest and grazing practices on mixed coniferous forests in northeastern Oregon, technical bulletin 103.* Corvallis, Oregon: Agricultural Experimental Station, Oregon State University.

Hershey, T. J., and T. A. Leege. 1976. Influences of logging on elk on summer range in north-central Idaho. In *Proceedings, Elk-logging-roads symposium*, 73–80. Moscow, Idaho: University of Idaho.

Hillis, J. M., M. J. Thompson, J. E. Canfield, L. J. Lyon, C. L. Marcum, P. M. Dolan, and D. W. McCleerey. 1991. Defining elk security: The Hillis paradigm. In *Proceedings, elk vulnerability symposium*, compilers, A. G. Christensen, L. J. Lyon, and T. N. Lonner, 38–43. Bozeman, Montana: Montana State University.

Holechek, J. L., R. D. Pieper, and C. H. Herbel. 1989. *Range management principles and practices, second edition*. Englewood Cliffs, New Jersey: Prentice-Hall.

———. 1998. *Range management principles and practices, third edition*. Englewood Cliffs, New Jersey: Prentice-Hall.

Johnson, B. K, J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk in spring. *Journal of Wildlife Management*. 64:685–97.

Kernohan, B. J., R. A. Gitzen, and J. J. Millspaugh. 2001. Analysis of animal space use and movements. In *Radio tracking and animal populations*, eds., J. Millspaugh, and J. Marzluff, 126–66. San Diego, California: Academic Press.

Kie, J. G., A. A. Ager, N. J. Cimon, M. J. Wisdom, M. M. Rowland, P. K. Coe, S. L. Findholt, B. K. Johnson, and M. Vavra. 2004. The Starkey database: Spatial-environmental relations of North American elk, mule deer and cattle at the Starkey Experimental Forest and Range in northeastern Oregon. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:475–90.

Leckenby, D. A. 1984. *Elk use and availability of cover and forage habitat components in the Blue Mountains, northeast Oregon, 1976–1982, wildlife research report Number 14*, Portland, Oregon: Oregon Department of Fish and Wildlife.

Leege, T. A. 1976. Relationship of logging to decline of Pete King Elk herd. In *Proceedings, elk-logging-roads symposium*, 6–10. Moscow, Idaho: University of Idaho.

Leptich, D. J., and P. Zager. 1991. Road access management effects on elk mortality and population dynamics. In *Proceedings, elk vulnerability symposium*, compilers, A. G. Christensen, L. J. Lyon, and T. N. Lonner, 126–37. Bozeman, Montana: Montana State University.

Lyon, L. J. 1976. Elk use as related to characteristics of clearcuts in western Montana. In *Proceedings, elk-logging-roads symposium*, 69–72. Moscow, Idaho: University of Idaho.

———. *Influences of logging and weather on elk distribution in western Montana, research paper INT-RP-236.* Ogden, Utah: U. S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Lyon, J. L., and A. G. Christensen. 1992. *A partial glossary of elk management terms, INT-GTR-288.* Ogden, Utah: U. S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

———. 2002. Elk and land management. In *North American elk: Ecology and management*, eds., D. E. Toweill and J. W. Thomas, 557–82. Washington DC: Smithsonian Institution Press.

Lyon, L. J., and C. E. Jensen. 1980. Management implications of elk and deer use of clear-cuts in Montana. *Journal of Wildlife Management.* 44(2):352–62.

Marzluff, J. M., J. J. Millspaugh, P. Hurvitz, and M. S. Handcock. 2004. Relating resources to a probabilistic measure of space use: Forest fragments and Steller's jays. *Ecology.* 85:1,411–27.

Marzluff, J. M., S. T. Knick, and J. J. Millspaugh. 2001. High-tech behavioral ecology: Modeling the distribution of animal activities to better understand wildlife space use and resource selection. In *Radio tracking and animal populations*, eds., J. Millspaugh, and J. Marzluff 310–26. San Diego, California: Academic Press.

Miller, R. F., and W. C. Kruerger. 1976. Cattle use on summer foothill rangelands in northeastern Oregon. *Journal of Range Management.* 29:367–71.

Millspaugh, J. J., G. C. Brundige, R. A. Gitzen, and K. J. Raedeke. 2000. Elk and hunter space-use sharing in South Dakota. *Journal of Wildlife Management.* 64(4):994–1,003.

Parker, K. L. 2002. Advances in nutritional ecology of cervids at different scales. *Ecoscience.* 10:395–411.

Perry, C., and R. Overly. 1976. Impact of roads on big game distribution in portions of the Blue Mountains of Washington. In *Proceedings, elk-logging-roads symposium*, 62–8. Moscow, Idaho: University of Idaho.

Reynolds, H. G. 1966. *Use of openings in spruce-fir forests by deer, elk, and cattle, Note RM-63.* Fort Collins, Colorado: U. S Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station Research.

Rinehart, J. M. 2001. Effects of intensive salvage logging on Rocky Mountain elk at the Starkey Experimental Forest and Range. M.S. thesis, University of Montana, Missoula.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396.* Portland, Oregon: U. S Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management.* 64(3):672–84.

Schroer, G. L., K. J. Jenkins, B. B. Moorhead. 1993. Roosevelt elk selection of temperate rain forest seral stages in western Washington. *Northwest Science.* 67(1):23–9.

Scott V. E., G. L. Crouch, and J. A. Whelan. 1982. *Responses of birds and small mammals to clearcutting in a subalpine forest in Central Colorado, research note RM-422.* Fort Collins, Colorado: U. S Department of Agriculture, Forest Service, Rocky Mountain Station,

Seaman, D. E., B. Griffith, and R. A. Powell. 1998. KERNELHR: A program for estimating animal home ranges. *Wildlife Society Bulletin.* 26:95–100.

Seaman, D. E., J. J. Millspaugh, B. J. Kernohan, G. C. Brundige, K. J. Raedeke, and R. A. Gitzen. 1999. Effects of sample size on kernel home range estimates. *Journal of Wildlife Management.* 63(2): 139–47.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy.* 83:229–44.

Svejcar, T., and M. Vavra. 1985. The influence of several range improvements on estimated carrying capacity and potential beef production. *Journal of Range Management.* 38:395–9.

Thomas, J. W. 1991. Elk vulnerability—A conference perspective. In *Proceedings, elk vulnerability symposium*, compilers, A.G. Christensen, L. J. Lyon, and T. N. Lonner 318–9. Bozeman, Montana: Montana State University.

Thomas, J. W., D. A. Leckenby, M. Henjum, R. J. Pedersen, and L. D. Bryant. 1988. *Habitat-effectiveness index for elk on Blue Mountain winter*

ranges, PNW-GTR-218. Portland, Oregon: U. S Department of Agriculture, Forest Service, Pacific Northwest Region.
Thomas, J. W., H. Black, Jr., R. J. Scherzinger, and R. J. Pedersen. 1979. Deer and elk. In *Wildlife habitats in managed forests: The Blue Mountains of Oregon and Washington, agricultural handbook no. 553,* ed. J. W. Thomas, 104–27. Portland, Oregon: U. S. Department of Agriculture, Forest Service.
Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. *Landscape ecology in theory and practice, pattern and process*. New York, New York: Springer-Verlag.
Unsworth, J. W., L. Kuck, E. O. Garton, and B. R. Butterfield. 1998. Elk habitat selection on the Clearwater National Forest, Idaho. *Journal of Wildlife Management.* 62:1,255–63.
Unsworth, J. W., L. Kuck, M. D. Scott, and E. O. Garton. 1993. Elk mortality in the Clearwater drainage of northcentral Idaho. *Journal of Wildlife Management.* 57(3):495–502.
Vavra, M., and R. L. Phillips. 1979. Diet quality and cattle performance on forested rangeland in northeastern Oregon. *Proceedings, western section, American society of animal science*. 30:170–3.
———. 1980. Drought effects on cattle performance. *Proceedings, western section, American society of animal science.* 31:157–60.
Ward, A. L. 1976. Elk behavior in relation to timber harvest operations and traffic on the Medicine Bow Range in South-Central Wyoming. In *Elk-logging-roads symposium proceedings*, 32–57. Moscow, Idaho: University of Idaho.
Wisdom, M. J., and J. G. Cook. 2000. North American elk. In *Ecology and management of large mammals in North America,* eds. S. Demarais, and P. R. Krausman, 694–735. Upper Saddle River, New Jersey: Prentice-Hall, Inc.
Wisdom, M. J., and J. W. Thomas. 1996. Elk. In *Rangeland wildlife,* ed. P. R. Krausman, 157–82. Denver, Colorado: Society for Range Management.
Wisdom, M. J., L. R. Bright, C. G. Carey, W. W. Hines, R. J. Pedersen, D. A. Smithey, J. W. Thomas, and G. W. Witmer. 1986. *A model to evaluate elk habitat in western Oregon, publication no. R6-F&WL-216–1986*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region.

Wisdom, M. J., N. J. Cimon, B. K. Johnson, E. O. Garton, and J. W. Thomas. 2004. Spatial partitioning by mule deer and elk in relation to traffic. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:509–30.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. *Ecology*. 70:164–8.

Zabel, C. J., K. McKelvey, and J. P. Ward, Jr. 1995. Influence of primary prey on home-range size and habitat-use patterns of northern spotted owls (*Strix occidentalis caurina*). *Canadian Journal of Zoology*. 73(3):433–9.

Management Implications of Ungulate Herbivory in Northwest Forest Ecosystems

Robert A. Riggs
Boise Cascade Corporation
La Grande, Oregon

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Larry L. Irwin
National Council for Air and Stream Improvement
Stevensville, Montana

Introduction

Understanding is now emerging that wild and domestic ungulates are not merely products that compete for resources but are among the agents controlling ecosystem processes and properties (McNaughton 1984, 1992; Pastor and Naiman 1992; Molvar et al. 1993; Hobbs 1996; Pastor and Cohen 1997). With this understanding has come recognition that greater attention should be given to understanding how these ungulate herbivores function together in forests of the Northwest (Irwin et al. 1994, Schreiner et al. 1996, Riggs et al. 2000).

In the Interior Northwest, forest zones receive greater precipitation, have deeper soil profiles, and have higher soil moisture-holding capacity than adjacent nonforested uplands. Thus, they can contribute substantially to landscape potential for herbivore production. In coastal forests and in many interior forests, infrequent but intense natural disturbances (e. g., fire) and episodic management activities (e. g., logging) initiate potentially food-rich secondary successions and are, therefore, often the primary determinants of herbivore carrying capacity (e. g., Hett et al. 1978). Seasonal concentration of ungulates on some forest sites can have profound influences on forest vegetation and on site characteristics (Paine 1969, McShea and Rappole 1992, Molvar et al. 1993, Waller and Alverson 1997). Better knowledge of such relationships may help ungulate managers in the Northwest to define consequences of alternative ungulate management strategies and to define a more integrated vision for forestry-ungulate management systems.

Forestry and game management have been viewed as separate, if albeit potentially complimentary, endeavors. Attempts have been made to integrate game management objectives with forestry through regulation of forest practices. However, these attempts have not sought integration of the influences of herbivory with those of silviculture. If integration is to be achieved, models will have to be used to organize the integration.

Brief Summary of Herbivory Effects

Scientists know precious little about how herbivores influence northwestern forests. Most of what is known is based on studies in boreal and eastern deciduous forests. Nevertheless, results of work in these ecosystems have been more or less borne out in northwestern studies to the extent experimental designs have allowed comparison. Consequently, this literature can certainly draw a generalized picture of how herbivores function in forests of this region.

We know that foraging habits are strongly linked to the chemistry and structure of plant tissues (Hobbs 1996) and to the morphological adaptations of herbivores (Hofmann 1973, Shipley 1999). Palatable plant tissues tend to contain lower amounts of secondary metabolites that suppress palatability and digestibility, and contain lower amounts of indigestible components than do unpalatable plant tissues. Thus, palatable tissues are more easily biodegraded, assimilated and recycled. Palatable plants also tend to have less formidable structural defenses and are more easily cropped. Where choices are available, large herbivores tend to select those plants and tissues that are most efficiently cropped and biodegraded.

Selective foraging can have a host of direct and indirect influences on forest communities. These include changing understory and overstory plant composition and structure (Alverson and Waller 1997, McShea 1997, Waller and Alverson 1997), acceleration of conifer growth (Bryant et al. 1983, DeToit et al. 1991, Pastor et al. 1993, Pastor and Cohen 1997), modification of forest structure (e. g., Stromayer and Warren 1997), modification of associated fauna (DeCalesta 1994, McShea 1997) and development of alternate stable states (Starfield and Chapin 1996, Stromayer and Warren 1997, Augustine et al. 1998).

In the short-term, selective herbivory may enhance forest productivity simply because of the accelerated flow of nutrients that herbivores facilitate via their consumption, degradation and cycling of plant-bound nutrients (Pastor et al. 1993, Hobbs 1996, Pastor and Cohen 1997). Over the long-term, however, the suppression of palatable plants conveys a competitive advantage to unpalatable plants, particularly to conifers. This advantage facilitates an acceleration of the rate at which conifers become dominant, and—as forest nutrients become progressively cycled and bound in the less palatable and less degradable conifer tissues—forest productivity ultimately declines. There has been some suggestion that moose may be able to facilitate elevated stable states of plant production in subartic forests (Molvar et al. 1993), but it remains unclear whether such facilitated equilibria are either frequent or persistent in other ecosystems.

Western temperate forests are characterized by substantial variations in climate, in nutrient relations and in water relations that ultimately limit production potential on any given forest site. These forests are also characterized by extreme variation in the frequency, intensity and extent of episodic disturbances (e. g., fire, silviculture) that arrest and initiate successions while simultaneously repartitioning nutrients. Thus, even in the absence of herbivores these forests have tremendous potential for spatially explicit dynamism in vegetation. Adding multiple species of herbivores only complicates the picture.

Although northwestern forests are dotted with grazing exclosures, knowledge of how herbivores interact with other disturbance agents to influence these forests is not highly detailed. Long-term data that describes grazing history and succession have been collected at relatively few sites. Where long-term data sets do exist, differences in succession are readily apparent, and, in a few cases, the stable end-point of succession evidently has been affected (e. g., Peek et al. 1978, Schreiner et al. 1996, Riggs et al. 2000, Alldredge et al. 2001). Selective herbivory during the growing season can suppress plant taxa that are palatable at that time of year (Figure 1, Riggs et al. 2000), but dormant-season browsing can enhance some plant taxa (Peek et al. 1978). In grand fir (*Abies grandis*) and in Douglas fir (*Pseudotsuga menziesii*), accumulation of understory biomass and accumulation of understory and forest-floor nutrients (nitrogen, phosphorous, sulfur, calcium, magnesium and potassium) is greater where ruminant herbivores are excluded than where they are allowed to graze (Table 1, Riggs et al. 2000). Selective herbivory probably enhances conifer production in the short-term (Monfore 1983, Weigand et al. 1993), but it may reduce conifer production over

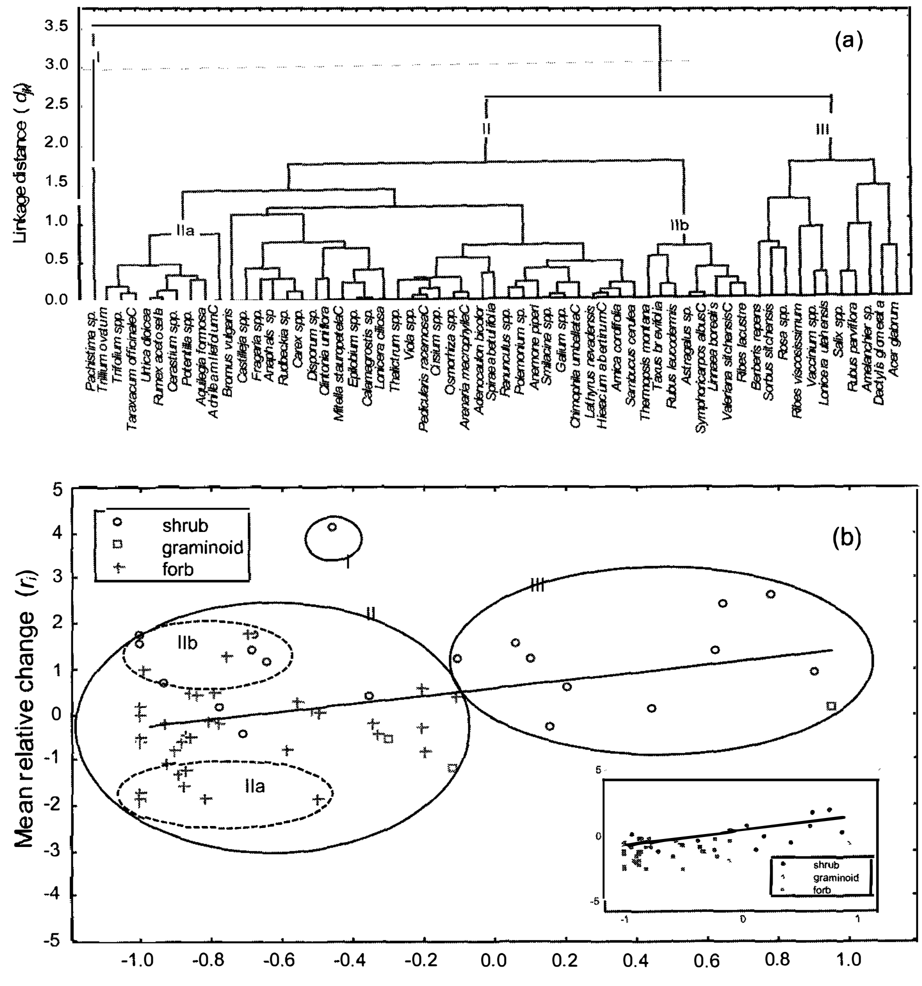


Figure 1. Joining tree (a) and scatter plot (b) for cluster analysis of secondary succession in grazed and ungrazed areas. In (b), e_i represents elk dietary electivity (after Ivlev 1961), and r_i represents relative change of plant taxa (inside relative to that outside of exclosures) over 27 years of secondary succession. Positive values of r_i indicate greater development inside exclosures than outside; for example, the shrub in group I (*Pachystima myrsinites*) developed four times more canopy inside exclosures than it did outside exclosures over the 27-year period. Elk dietary preferences explained from 15 percent (large graph, n = 59 taxa) to 37 percent (inset graph, n = 51 taxa) of the difference in succession. Nearly all species exhibiting positive dietary electivity are shrubs, and all but 1 of these developed to a greater extent on sites protected from ungulates. Figure adapted from Riggs et al. (2000).

Table 1. Mean biomass and nutrient accumulation in understory vegetation and forest floor, outside and inside seven herbivore exposures on forest sites in northeastern Oregon. Differences in nutrient accumulations are largely attributable to reduced accumulation of shrub biomass, which in turn can be attributed largely to selective foraging by wild herbivores. Time lapsed to produce these effects ranged from 27 to 30 years (see Riggs et al. 2000 for details).

Component	Outside	Inside	p
	Pounds per acre (kg/ha)		
Understory vegetation			
Graminoids	210 (187)	149 (133)	NS[a]
Forbs	458 (409)	477 (426)	NS
Shrubs	1,168 (1,042)	3,203 (2,858)	0.05
Total, understory	1,836 (1,639)	3,829 (3,417)	0.05
Forest Floor	9,971 (8,896)	15,394 (13,735)	0.05
Nutrients:			
Nitrogen	213 (190)	351 (313)	0.01
Phosphorus	36 (32)	50 (45)	0.01
Sulfur	11 (10)	16 (14)	NS
Calcium	306 (273)	500 (446)	0.01
Magnesium	31 (28)	56 (50)	0.01
Potassium	59 (53)	91 (81)	0.05
Total, nutrients	656 (586)	1,064 (949	0.01

[a]NS = not significant; difference between outside and inside the exposure was not significant ($P > 0.10$)

the long-term (Weigand et al. 1993, Riggs et al. 2000). The capacity of seral vegetation to support herbivores can be reduced by selective suppression of palatable plants early in succession (Irwin et al. 1994, Riggs et al. 2000, Alldredge et al. 2001).

Different species or combinations of herbivores can have dramatically different influences on the composition of forest understory, provided that foraging selectivity is accommodated. In Douglas fir and in grand fir forests of the interior, early-season grazing by cattle suppresses herbaceous plants, while season-long grazing by big game suppresses shrubs (Erickson 1974, Krueger and Winward 1974, Hedrick et al. 1968, Edgerton 1987, Riggs et al. 2000; see Peek et al. 1978).

To a large extent, herbivores operate in sequence with episodic disturbances. Fire, logging, herbicides and fertilizer applications each catalyze changes in plant composition and in nutrient pools, and these changes may be either abrupt or subtle, depending on the severity of the episode. Herbivores then modify the subsequent secondary succession by altering plant composition and nutrient dynamics. Influences of herbivory on a given forest site are regulated by foraging selectivity, by nutrient and dry-matter demand, and by productivity and grazing resistance of plants in the secondary succession (Riggs et al. 2000). Prescribed fire may initially produce shrub-dominated seres, but repeated fire—coupled with chronic browsing by big game—produces alternate seres in which the dominant plants are herbs (Peek 1989). Aspen (*Populus tremuloides*) regeneration can be suppressed by chronic post-fire herbivory (Bartos et al. 1994). Nutrient dynamics and site productivity may be modified when large herbivores selectively suppress nitrogen-fixing plants in postfire successions as well (Riggs et al. 2000). Such interactions involving herbivores and episodic disturbance can strongly influence the effectiveness of habitat management that is based on episodic disturbance (Riggs et al. 1996).

The Concept of Regime

The term "regime" has been used by fire ecologists to place fire behavior and effect in common context (e. g., Heinselman 1981, Kilgore 1981, Woodmansee and Wallach 1981, Agee 1993). In efforts to understand effects of livestock grazing on ecosystems, range scientists have characterized livestock grazing regimes in terms of the type of livestock, the livestock density, the seasonal timing, and the intensity and frequency of grazing in systems that integrate these factors (e. g., Valentine 1990). Other vegetation controlling agents, such as forest site, forestry practices and herbivory by big game, can be characterized in similar terms. Each vegetation-controlling agent can be characterized by a multifactor regime (Figure 2). Regimes for individual agents may be independent of one another or interdependent in the disturbance sequence (Figure 3). Spatial context can be a key to understanding interactions as well because interspersion of plant communities influences seed dispersal and animal use patterns.

Many different combinations of disturbance agents can be hypothesized. Only some constitute practical management strategies; among these, even fewer are functionally different from one another. The salient point is that understanding the role of herbivory in controlling forest vegetation necessarily involves understanding how herbivory functions in an interactive sequence with other disturbance agents (Riggs et al. 2000).

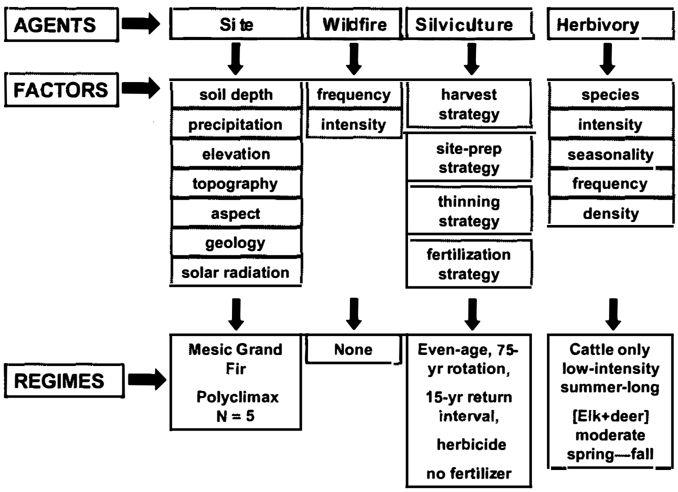


Figure 2. Schematic of selected factor regimes for vegetation-controlling agents in northwestern forests. The more likely combinations can be represented with rather simple descriptions as shown in boxes at bottom.

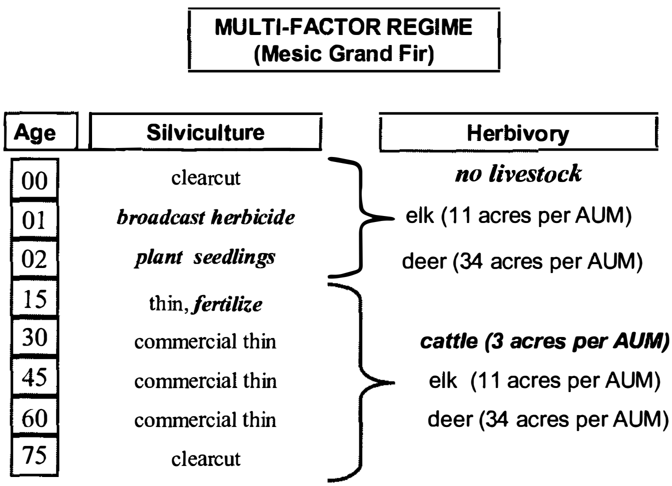


Figure 3. Seventy-five-year disturbance chronology for a mesic grand fir site in the Blue Mountains of northeastern Oregon. In this scenario, timber harvest and planting strategies are fixed, fire is categorically precluded, and italics indicate those disturbance agents over which the manager has discretion (herbicides, fertilization and livestock). Herbivory operates in a sequence with episodic agents. Grazing regimes illustrated here are derived from historical estimates for sites studied by Riggs et al. (2000).

Some Conceptual Models to Consider

Management disciplines necessarily employ models that describe how management systems are organized and how they function (Westoby et al. 1989), a number of which might integrate herbivory's influences in forests. There is no wide consensus on the best model for integrating herbivory into forestry, and comparison of alternatives can be useful.

The Range Succession Model

Most research concerning ungulate herbivory has been conducted in rangeland and pasture settings. The dominant model in range management has been the range succession model, described by Westoby et al. (1989). This model describes vegetation in terms of a linear succession to a single persistent state in the absence of grazing, a monoclimax (Clements 1916). All possible states of vegetation occur along the continuum, and changes occur continuously and are reversible. Under the range succession model, managers seek herbivore stocking that establishes an equilibrium between the grazing pressure and the successional tendency of a given plant community, thereby stabilizing it in succession.

The range succession model has been criticized for lacking realism in several situations, chiefly where succession (i. e., trend) has not responded continuously to grazing modifications. The most extreme of such situations is represented by alternate, stable communities of apparently low seral status that persist despite modifications of grazing regime; these communities are referred to as alternate states (Westoby et al. 1989, Laycock 1991) or domains (Friedel 1991). Transitions to such states are often not linear, continuous or easily reversible. Often their inducement requires more than modification of the grazing regime. Forest succession is often dominated by overstory dynamics rather than by understory dynamics (where grazing is operative); thus, forest succession is problematic to this model, like the introduction of exotic species (Smith 1978).

State-and-transition Model

The state-and-transition model, described by Westoby et al. (1989) and Laycock (1991), is an alternative to the range succession model. This model does not require that vegetation succeed to monoclimax, and it does not require that vegetation responses to grazing modification be continuous. Instead, the state-and-transition model simply describes vegetation in terms of a catalogue of alternative states and a catalogue of transitions between states. States are considered stable within reasonable bounds, and they are typically identified by management significance rather than by theoretical significance. In state-and-transition modeling, it is not necessary to recognize climax vegetation as such; although, climax communities may indeed be approximated by one or more alternate stable states in a particular model. Transition between two stable states is induced by episodic events (e. g., drought, fire, chaining of juniper stands), by changes in herbivory regime or by combinations of episodic events and herbivory regime.

The most obvious benefit of the state-and-transition model to managers is its ability to categorize multiple states of stable vegetation and to conceptually link them via transitions that integrate episodic and chronic disturbance agents. However, where the occurrence and severity of episodic conditions necessary for transition cannot be predicted (e. g., precipitation, wildfire), implementation of the state-and-transition model may be difficult, and, in some settings, this constraint may render it less practical. Because the model conceptualizes switching, or inducing transitions, from one relatively stable state of vegetation to another and because these states are stable, it is insensitive to continuous changes in vegetation that occur within stable states.

Range Succession versus States-and-transitions in Northwestern Forests

The range succession model and the state-and-transition model of Westoby et al. (1989) are each derived largely from the labors of ecologists concerned with the management of arid rangeland. Because our focus is forests, comparing these models in forest settings might provide insight to their management utility there.

The range succession model has been widely applied in forests, principally by the U. S. Department of Agriculture, Forest Service (Forest Service). This application probably derives from both an adoption of early monoclimax ecological theory by range managers (see Westoby et al. 1989) and from a perceived need for operational consistency between forestry and range management programs. The logic of the state-and-transition model is recent but not new to northwestern forestry (Kessel 1979). State-and-transition logic has been employed explicitly to conceptualize forest overstory dynamics (Hemstrom et al. 2001), illustrated implicitly in diversity matrices for forest ecosystems (e. g., Haufler and Irwin 1993, Haufler et al. 1996), and used to model understory succession and herbivore carrying capacity (Jenkins and Starkey 1996). In none of these applications, however, has there been explicit consideration of quantitative herbivory, by either wild ungulates or livestock, as a transition-inducing agent.

The root criticism of the range succession model is its basis in the early Clementsian notion of a monoclimax preceded by linear succession in which grazing simply induces retrogression or retards succession. Later monoclimax theory, however, approached recognizing alternate states of vegetation, such as subclimax, proclimax, disclimax and neoclimax (Clements 1916, 1934; Weaver and Clements 1938; Oosting 1956), albeit lacking explicit recognition of the factors responsible for each alternative (see Tansley 1935).

The monoclimax theory's rudimentary treatment of alternate stable states and its simplified application in the range succession model might be mitigated to a substantial extent by integrating concepts of the polyclimax, stated by Tansley (1935), who drew attention to Godwin (1929) who insisted that subclimaxes might exist as stable, equilibrium communities because a factor could arrest or deflect a sere. The term "plagiosere" was applied to bent or twisted seres in which vegetation was not in equilibrium with the arresting factor, while the term "plagioclimax" was suggested for referring to vegetation that had actually, "come into equilibrium with the deflecting factor" (Tansley 1935:292). Tansley ultimately recognized three types of climax vegetation in equilibrium with primary factors (i. e., climatic climax, edaphic climax and physiographic climax), along with three others that could be derived from primary climaxes through equilibrium with disturbance agents (i. e., fire climax, biotic climax, mowing climax), among which the biotic climax was considered to develop where the, "incidence and maintenance of a decisive biotic factor, such as continuous grazing determines the species that persist" (1935). In the Northwest, these forms of disturbance climax are often referred to as disclimax vegetation, after Daubenmire (1968) who specifically applied the term to equilibrium vegetation that required chronic disturbance for maintenance. Nevertheless the term disclimax actually originated much earlier, as noted above.

Two salient points can be drawn from this review thus far. First, climax theory is not limited to the monoclimax construction but also includes the polyclimax alternative. The monoclimax and polyclimax are distinguished from one another largely by the latter's somewhat more overt accommodation of alternate climaxes (i. e., alternate equilibrium states), and by the latter's explicit recognition of the agents (some of which are disturbance agents) that produce alternate climaxes. Second, the notion of herbivore-induced, stable states and transitions was conceptually formed in classical ecology by 1935, albeit in terms of climax communities and successions rather than in terms of states and transitions.

As the monoclimax and polyclimax theories evolved through refinement of terminology, the distinction between the two theories came to be as much semantic as substantive (see Oosting 1956). Nevertheless it is important to understand the distinction to fully understand the application of climax theory in northwestern forest ecology. When applied in range management, the early monoclimax (after Clements 1916) is an equilibrium state of vegetation in which herbivory is not operative; herbivory is operative in the development of lower seral states only (Westoby et al. 1989). This application in range management, however, is not consistent with application of later climax theory in northwestern forest ecology. As applied in forest ecology, the climax is a polyclimax in which some forms are in equilibrium with herbivory (i. e., herbivory is operative in the climax vegetation). Northwestern forest classifications, exemplified by Daubenmire and Daubenmire (1968), Hall (1973), Pfister et al. (1977), Steele et al. (1981), Johnson and Simon 1987, and Johnson and Clausnitzer 1992 (all adapted after Daubenmire 1952) are all polyclimax classifications. Indeed, the regional practice of habitat typing northwestern forests, based on the composition of plant associations, amounts to taxonomic classification of forest sites (not vegetation), based on potential natural vegetation, after Tansley's (1935) polyclimax (see Daubenmire 1952, Pfister and Arno 1980). Habitat types (Daubenmire 1952), potential natural vegetation (Kückler 1964) and potential natural communities (Hall 1998) reflect polyclimax logic. In the Blue Mountains ecological province, only 7 of 31 climax forest associations have been successfully predicted based on their site characteristics alone (Kelly et al., in press). Documentation of these regimes, in context with those for seral vegetation (see Hall 1998) would clarify herbivory's role in determining stable states of forest vegetation in the region.

Nevertheless, the state-and-transition model is an attractive alternative to both the monoclimax and polyclimax models. It might be attractive to some because it does not require explicit distinction between climax and lower seral states. More importantly, the state-and-transition model clearly requires explicit definition of conditions that induce transitions (see Westoby et al. 1989), not merely identification of the agents that are involved in producing alternate stable states (see Tansley 1935) or knowledge that alternate stable states can exist (see Clements 1916, 1934; Weaver and Clements 1938). Thus, greatest management utility of state-and-transition modeling lies in the fact that it does not accommodate ambiguity to the extent of the monoclimax or polyclimax models.

Application of the state-and-transition model, nonetheless, can be challenging in the forestry context. Foremost is the basic problem of defining a stable state. Commercial foresters might classify forest stands based on apparent commercial value or position in a sequence of explicit management events (e. g., seedling, sapling, pole, small saw-log). Whereas, those not so concerned with commodity production might be attracted to classifications more reflective of gradual succession (e. g., stand initiation, stem exclusion, understory reinitiation, old growth [Oliver and Larson 1990]). In either case, the classification of stands amounts to a sequence of seral stages (some management-induced) that are explicitly linked by some combination of disturbance (i. e., management, herbivores) and succession. These stands also vary substantially in their composition while often sharing the majority of plant taxa. The initial and intermediate stages in these artificial sequences are virtually never stable but rather are dynamic, often highly and purposefully so. Switching can be induced between seral stages by invoking defined conditions, such as clear-cut logging to transform a saw-log stand into a herbaceous or shrub stage or controlled cattle grazing to facilitate development of a shrub understory. However, stability in the ecological sense is not achieved short of an equilibrium state in which the vegetation can be expected to persist indefinitely (i. e., some form of polyclimax). Thus, an explicit reconciliation of state-and-transition modeling with polyclimax theory seems unavoidable in northwestern forests.

Stromayer and Warren (1997) recognized that transition-inducing mechanisms in forests probably involve multiple disturbance agents, such as fire, logging and chronic herbivory, but they added that confirmation of new stable states might require monitoring over centuries. Kienast et al. (1999) concluded that extreme browsing by large herbivores would not dramatically alter successional pathways or long-term woody biomass dynamics. Such observations beg a critical question. How dramatic and persistent must an herbivore-induced change in forest vegetation be in order to represent a change in stable state? It is clear that disturbance-induced, seral forest vegetation does not typically represent stable states in the same sense, like stable, low-seral-state, rangeland communities do (Westoby et al. 1989). In absence of clear evidence to the contrary, seral forest stages should probably be modeled explicitly as transient rather than as stable states because these seral vegetations typically, do not, "persist indefinitely but rather [change] into one or other persisting states, depending on events while . . . in the transient state." (Westoby et al. 1989:268).

Productivity of arid land is usually constrained by moisture; thus, precipitation has played a pivotal role in state-and-transition models for rangeland applications (Westoby et al. 1989, Laycock 1991). Northwestern conifer forests, however, are strongly zonal in this respect (Daubenmire 1969, Franklin and Dyrness 1973), occurring along a two-dimensional gradient defined by drought stress and by number of optimum growing days. Five zones are arrayed typically (oak-juniper woodland, ponderosa pine, mixed conifer, coastal temperate and subalpine/boreal). The woodland, ponderosa pine and dry mixed-conifer forests are usually considered to be moisture-constrained and, in this sense, are similar to steppe and shrub-steppe vegetation. Subalpine and boreal forests often are considered to be temperature-constrained (Franklin and Dyrness 1973:50). Between these two extremes, most mixed-conifer and coastal-temperate forests are less likely to be constrained by moisture or temperature (Figure 4).

The nature and extent of herbivory's involvement in facilitating transitions between stable states should vary across these zones. In the dry, moisture-constrained forests, coupling of some herbivory regimes with relatively frequent drought, resultant moisture limitation and, in some cases, frequent, low-intensity fire should produce frequent opportunities for herbivory to induce transitions. In wetter forests, drought-induced moisture limitation is less frequent, and so is the fire frequency. But, fire intensity increases with the fire-return interval (Agee 1993). Thus opportunities for herbivores to interact with fire regimes to induce transitions between alternate stable states probably occur relatively infrequently in wetter, more productive forests, albeit more dramatically under normal fire-return intervals.

Anthropogenic disturbance regimes (e. g., logging, herbicides, fertilization, prescribed fire), however, should increase the frequency of such opportunities. For example, the typical fire-return interval on a cool-moist grand fir site might be 100 to 300 years (Agee 1993). Under intensive silvicultural management, however, the same site might have a disturbance-return interval (i. e., logging, herbicides, fertilization, fire) of only 10 to 15 years (Figure 3), thus increasing the potential frequency of interaction between episodic agents and

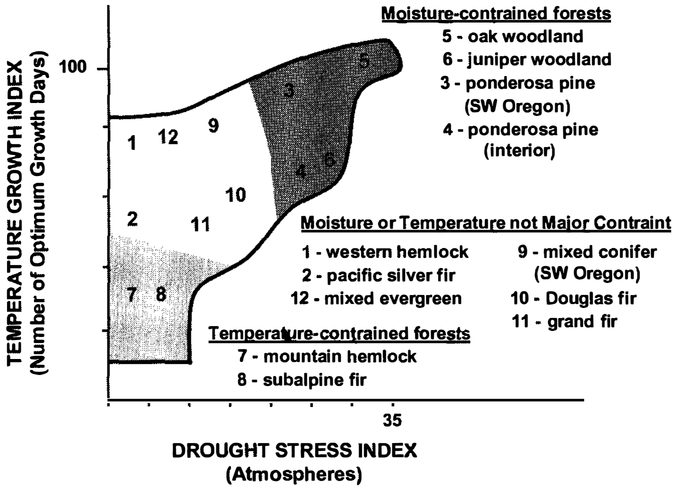


Figure 4. Forest zones in the northwest are distributed along a 2-dimensional gradient defined by drought stress and temperature. Oak woodlands, ponderosa pine, Douglas fir, and the drier mixed conifer types are those most likely to suffer frequent drought stress and periodic fire. Transition-inducing interactions that involve herbivores, drought, and fire are probably more frequent there than in coastal and more mesic mixed-conifer forests, but the frequency of such transitions can be increased through anthropogenic disturbances such as logging, prescribed fire, and persistently high herbivory regimes. Climate warming should increase the frequency of transition-inducing interactions along the warm-dry margin of each zone but would perhaps be most apparent in the temperature- and moisture-constrained forests. This figure is simplified from that of Franklin and Dymess (1973:50).

herbivores. Where episodic disturbances markedly alter nutrient pools in these forests and where subsequent herbivory alters nutrient accretion (accumulation, such as through selective suppression of nitrogen-fixing plants), alternate states of forest productivity may be inducible (Riggs et al. 2000).

The zonal distribution of northwestern forests might influence the effective magnitude of such interactions as well because the influences of herbivores depend on the rate of forest succession relative to the rate of herbivore population increase (Crawley 1983). In mesic coastal forests, wild ungulate populations may not be able to increase at sufficient rates to have much of an effect on the long-term stability of successions (i. e., from one forest rotation to the next) because sites in these forests are so rapidly colonized by plants and, ultimately, dominated by forest canopy. In drier forests of the Interior Northwest, however, secondary succession proceeds more slowly, providing a longer window of opportunity for wild ungulate populations to grow and to exert influences on secondary successions.

Growth and Yield Models

We have discussed that northwestern forests can be characterized by complicated interagent disturbance regimes. Many of these regimes involve intensive forestry practices, such as preprogrammed thinning, herbicide use and fertilizer applications. In these settings, forest growth simulators play a crucial role in building deterministic projections of timber growth and yield, based on model strategies that dictate both type and sequence of management intervention. The dominant simulator in the Northwest is probably the forest vegetation simulator (FVS; Wykoff 1986, Wykoff et al. 1982), of which many regional variants exist in public and private domains (see U. S. Department of Agriculture, Forest Service 2001). These simulators project conifer growth and yield based on input for site productivity, conifer in-growth, mortality and applied silviculture. Output is in the form of periodic projections (e. g., 3-, 5-, 10- or 20-year periods) of stand attributes (e. g., trees per acre, diameter distribution, volume) that can be interpreted to classes or stages of forest vegetation. In some of these regional variants, tree growth is calibrated according to polyclimax associations, and postprocessor programs are also available for linking understory dynamics with overstory dynamics (after Moeur 1985; see U. S. Department of Agriculture, Forest Service 2004).

The FVS and its postprocessors can be calibrated to herbivory regimes as well, thereby integrating herbivory and silviculture in an operational context. However, none of the regional variants have been calibrated to integrate herbivory effects explicitly. Nevertheless, growth simulators are the primary tools used by foresters to simulate forest dynamics in forest management and, if properly calibrated, can provide powerful projections of wildlife habitat quality under varying silviculture regimes (e. g., Roloff et al. 1999, McGrath et al. 2003). Thus, they are a logical contact point for integrating herbivores into the context of operational forestry.

Landscape Models

Ecosystem management implies a landscape focus and a planning horizon measured in tens to hundreds of years (Tester et al. 1997). The range succession model, the state-and-transition model (as described by Westoby et al. 1989) and the vegetation simulators that are represented by the FVS focus on vegetation trajectories, or on states and transitions within a patch of vegetation over relatively short time-periods (e. g., a few decades). It has been argued that these models cannot mechanistically model either the processes or responses involved in the ecosystem (Starfield et al. 1993, Tester et al. 1997), and they cannot discern how the ecosystem will respond to specific changes in the variables or sporadic environmental events (Starfield et al. 1993). In these models, herbivores modify successions or induce transitions in the patch, but the range of possibilities is defined by expectations considered realistic for the site, based on historical ranges of variation at similar sites. Events or trends outside of that range—which might be expected due to climate change, for example—cannot be explicitly considered. Also, because these models focus on sites or patches, they are not well suited to consider how events or conditions in nearby patches may influence events or processes in the focal patch (e. g., seed production and dispersal, fire intensity and spread, herbivore density and the predictable spatial pattern of grazing intensity). Thus, neither the range-succession or state-and-transition models are suited to modeling herbivory effects across landscapes or in context with transient climate. There are better alternatives.

Frame-based models are a type of top-down, state-and-transition model emerging from landscape ecology (Starfield et al. 1993, Tester et al. 1997). In their simplest form, each frame represents an ecosystem (e. g., grassland, deciduous forest, conifer forest). Each ecosystem frame runs as an independent subroutine—one at a time—and transitions occur from one ecosystem frame to another when the conditions necessary for maintenance of a frame are no longer met (Figure 5). Frame-based models have been used to explore how herbivores could interact with disturbance to alter vegetation in a variety of ecosystems (Starfield et al. 1993, Starfield and Chapin 1996, Tester et al. 1997). Using a frame-based model, Starfield and Chapin (1996) concluded that management policies related to logging and moose populations could affect vegetation stability as much or more than could climate change. Like the other models we have considered, frame-based modeling was initially patch-focused, but it now incorporates spatially explicit relationships in landscape contexts (see Rupp et al. 2000a, b).

Like the more traditional state-and-transition models used in rangeland, frame-based models are relatively insensitive to linear trajectories of vegetation within their frames. Frame-based models are probably best used to explore relatively long-term and large-scale vegetation dynamics. Frame-based models should be explored in northwestern forests, given the apparent likelihood of global

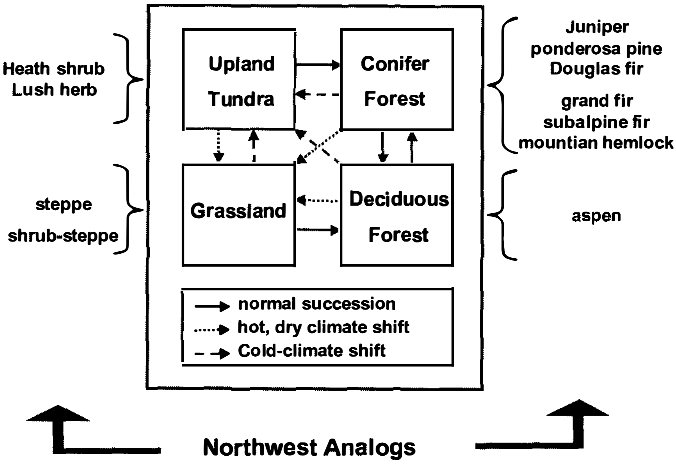


Figure 5. Frame-based model structure (large box, after Starfield and Chapin 1996) shows normal succession among arctic and boreal ecosystem types (frames) and transitions under transient climatic conditions. In this model, herbivory interacts with fire and logging to influence the probability of each transition occurring (see Starfield and Chapin 1996:856). Bracketed vegetation (outside the large box) represent Northwest analogs to the more northern vegetation.

warming, the region's zonal limitations to forest growth and distribution, and the likelihood that transition-inducing interactions occur between climate, fire, forest management and herbivory. Calibration of frame-based models to northwestern forests will probably be challenging because of zonal factors, complicated interagent disturbance regimes and complex herbivore assemblages.

Multiherbivore systems often present the challenge of evaluating influences of modifying herbivore regimes. An example of this is comparing the influence of modifying livestock numbers to that of modifying wild herbivore populations (Weisberg et al. 2002). This problem involves interacting submodels for weather, carbon, nitrogen, water, light, fire, predators, vegetation production and herbivore population dynamics, simulating vegetation dynamics in terms of plant composition and responses to climate, fire and herbivory and simulating animal responses in terms of distribution, production and population growth (Coughenour 1993).

Summary and Conclusions

Influences of large herbivores on northwestern forests may range from benign to profound, and wild herbivores are just as capable of affecting forest ecosystem properties and processes as are domestic livestock. Wild herbivores and livestock are agents of chronic disturbance that interact with episodic agents in sequence, modifying transient states that are catalyzed by episodic disturbance. Enlightened management of herbivory effects in northwestern forest ecosystems must be based on knowledge of how herbivores interact sequentially with episodic disturbance agents. We offer the following suggestions for integrating herbivory science in the regional forestry model.

First, continuing management with only the range succession model would be as problematic for forest managers as it has been for range managers. The range succession model remains useful for managing short-term responses of vegetation, where herbivory is the primary agent responsible for change. However, it does not reconcile easily with the polyclimax theory of forest succession, with regional classifications of stable forest vegetation or with disturbance-agent interactions.

Second, in seeking alternatives to the range succession model, we should seek models that can distinguish herbivory regimes that merely alter successions in the short-term from those that are likely to produce alternate stable states over the long-term. Long-term changes in vegetation states will probably result from interactions involving herbivores and other controlling agents, and they will probably be more likely in forests that are characterized by high-frequency disturbances and by frequent water or nutrient stress. The state-and-transition model represents an attractive alternative for conceptualizing patch-level disturbance regimes and their consequences. However, state-and-transition models are not necessarily sensitive to gradual changes in vegetation. They require unambiguous articulation and calibration and, as proposed in the range management model, are not capable of addressing landscape-level dynamics.

Third, the selection of a model to integrate herbivory into forest management should also depend on the nature and the scale of the manager's information need. Growth simulators, like the FVS, provide a convenient tool for integrating herbivory influences into the operational context with overstory management, and their calibration for herbivory-induced effects could provide unambiguous management utility. State-and-transition models provide a convenient tool for organizing herbivory's influences in context with those of episodic disturbances, provided that stable-states (i. e., polyclimaxes) and their transitions are not confused with transient-state dynamics (i. e., of seral stages). Landscape models offer ecosystem frameworks in which managers can consider spatially-explicit interactions between herbivores and other disturbance agents, and managers can consider the long-term effects on forest ecosystem stability. Still, these models remain to be calibrated in the Northwest. Forest managers interested in integrating herbivory into their management programs are more constrained by inadequate data for calibration than by availability of models, and applied research should focus on calibration.

Fourth, some experimental research will be required to understand how herbivores interact with other disturbance agents to influence forest vegetation. Ultimately, the management utility of these experiments will depend upon their blending of ungulate biology, range science, forestry science, landscape ecology and even climatology to an extent not often seen. Agency and university administrators would do well to force that integration into their research programs as rapidly as possible. Because the nature and magnitude of herbivore effects can vary across the region's environmental gradient, long-term funding for interdisciplinary research should be undertaken at several experiment stations.

Finally, integrating large herbivores into the organizational model for northwestern forestry will not be a mere exercise in modeling effects. Ultimately, it must involve managing herbivore populations in context with specific landscape disturbance regimes. Managing herbivore populations will be as important to that integration as managing the pattern of forestry practices. Thus, to the extent that landscape management defines herbivore carrying capacity, game departments and their constituents have a vested interest in understanding the biological processes and forest dynamics involved.

Reference List

Agee, J. K. 1993. *Fire Ecology of Pacific Northwest Forests.* Washington, DC: Island Press.

Alldredge, M. W., J. M. Peek, and W. A. Wall. 2001. Alterations of shrub communities in relation to herbivory in northern Idaho. *Northwest Science*. 75:137–44.

Alverson, W. S., and D. M. Waller. 1997. Deer populations and the widespread failure of hemlock regeneration in northern forests: In search of a smoking muzzle. In *The Science of Overabundance: Deer ecology and population management*, eds. W. J. McShea, H. B. Underwood, and J. H. Rappole, 280–97. Washington, DC: Smithsonian Institute Press.

Augustine, D. J., L. E. Frelich, and P. A. Jordan. 1998. Evidence for two alternate stable states in an ungulate grazing system. *Ecological Applications*. 8:1,260–69.

Bartos, D. L., J. K. Brown, and G. D. Booth. 1994. Twelve years biomass response in aspen communities following fire. *Journal of Range Management*. 47:79–83.

Bryant, J. P., F. S. Chapin, III, and D. R. Klein. 1983. Carbon/nutrient balance of boreal plants in relation to vertebrate herbivory. *Oikos*. 40:357–68.

Clements, F. E. 1916. Plant succession: An analysis of the development of vegetation. *Carnegie Institute of Washington Publication*. 242:1–512.

Clements, F. E. 1934. The relict method in dynamic ecology. *Journal of Ecology*. 22:39–68.

Coughenour, M. B. 1993. *The SAVANNA landscape model-documentation and user's guide*. Fort Collins, Colorado: Natural Resources Ecology Laboratory, Colorado State University.

Crawley, M. J. 1983. *Herbivory: The dynamics of animal-plant interactions*. Berkely, California: University of California Press.

Daubenmire, R. F. 1952. Forest vegetation of northern Idaho and adjacent Washington, and its bearing on concepts of vegetation classification. *Ecological Monographs*. 22:301–30.

Daubenmire, R. F. 1968. *Plant communities: A textbook of plant synecology*. New York, New York: Harper and Row Publishers.

Daubenmire, R. F. 1969. Ecologic plant geography of the Pacific Northwest. *Madroño*. 20(3):111–28.

Daubenmire, R., and J. B. Daubenmire. 1968. *Forest vegetation of eastern Washington and northern Idaho, technical bulletin No. 60.* Pullman, Washington: Washington State University, Agricultural Experiment Station.

DeCalesta, D. S. 1994. Effect of white-tailed deer on songbirds within managed forests in Pennsylvania. *Journal of Wildlife Management*. 58:711–18.

DeToit, J. T., J. P. Bryant, F. D. Provenza, J. Pastor, P. B. Reichardt, and T. P. Clausen. 1991. Interactions between woody plants and browsing mammals mediated by secondary metabolites. *Annual Review of Ecology and Systematics*. 22:431–46.

Edgerton, P. J. 1987. Influence of ungulates on the development of the shrub understory of an upper slope mixed conifer forest. In *Proceedings of the symposium on plant-herbivore interactions, general technical report INT-GTR-222*, comps. F. D. Provenza, J. T. Flinders, and E. D. McArthur, 162–7. Ogden, Utah: U. S. Department of Agriculture, Forest Service.

Erickson, L. R. 1974. Livestock Utilization of a Clear-cut Burn in Northeastern Oregon. M.S. thesis, Oregon State University, Corvallis.

Franklin, J. F., and C. T. Dyrness. 1973. *Natural vegetation of Oregon and Washington, general technical report PNW-GTR-8*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A viewpoint. *Journal of Range Management*. 44:422–6.

Godwin, H. 1929. The subclimax and deflected succession. *Journal of Ecology*. 17:144–7.

Hall, F. C. 1973. *Plant communities of the Blue Mountains in eastern Oregon and southeastern Washington, area guide 3–1*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region, Region 6.

Hall, F. C. 1998. *Pacific Northwest ecoclass codes for seral and potential natural communities, general technical report PNW-GTR-418*. Portland, Oregon. U.S. Department of Agriculture, Forest Service.

Haufler, J. B., C. A. Mehl, and G. J. Roloff. 1996. Using a coarse-filter approach with species assessment for ecosystem management. *Wildlife Society Bulletin*. 24:200–8.

Haufler, J. B., and L. L. Irwin. 1993. An ecological basis for forest planning for biodiversity and resource use. *Proceedings of the International Union of Game Biologists*. 21:73–81.

Hedrick, D. W., J. A. Young, J. A. B. McArthur, and R. F. Keniston. 1968. *Effects of forest and grazing practices on mixed coniferous forests of northeastern Oregon, technical bulletin no. 103*. Corvallis, Oregon: Oregon State University, Agricultural Experiment Station.

Heinselman, M. L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In *Fire regimes and ecosystem properties, general technical report WO-26*, eds. H. A. Mooney, T. M. Bonnicksen,, N. L. Christensen, J. E. Lotan, and W. A. Reiners, 7–57. Washington, DC: U. S. Department of Agriculture, Forest Service.

Hemstrom, M. A., J. J. Korol, and W. J. Hann. 2001. Trends in terrestrial plant communities and landscape health indicate the effects of alternative management strategies in the interior Columbia River Basin. *Forest Ecology and Management.* 153:105–26.

Hett, J., R. Taber, J. Long, and J. Schoen. 1978. Forest management policies and elk summer carrying capacity in the *Abies amabilis* forest, western Washington. *Environmental Management.* 2:561–6.

Hobbs, N. T. 1996. Modification of ecosystems by ungulates. *Journal of Wildlife Management.* 60:695–713.

Hofmann, R. R. 1973. *The ruminant stomach: Stomach structure and feeding habits of East African game ruminants.* Nairobi, Kenya: East African Literature Bureau.

Irwin, L. L., J. G. Cook, R. A. Riggs, and J. M. Skovlin. 1994. *Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystems, general technical report PNW-GTR-325.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Jenkins, K., and E. Starkey. 1996. Simulating secondary succession of elk forage values in a managed forest landscape, western Washington. *Environmental Management.* 20:715–24.

Johnson, C. G., and R. R. Clausnitzer. 1992. *Plant associations of the Blue and Ochoco Mountains, R6-EDW-TP-036-92.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Johnson, C. G, and S. A. Simon. 1987. *Plant associations of the Wallowa-Snake province, R6-ECOL-TP-255A-86.* Portland, Oregon: U. S Department of Agriculture, Forest Service.

Kelly, A., D. Powell, and R. Riggs. In press. Predicting potential natural vegetation in an interior northwest landscape using classification tree modeling and a GIS. *Western Journal of Applied Forestry.*

Kessell, S. R. 1979. *Gradient Modeling: Resource and fire management.* New York, New York: Springer-Verlag, Inc.

Kienast, F., J. Fritschi, M. Bissegger, and W. Abderhalden. 1999. Modeling successional patterns of high-elevation forests under changing herbivore pressure—Responses at the landscape level. *Forest Ecology and Management.* 120:35–46.

Kilgore, B. M. 1981. Fire in ecosystem distribution and structure: Western forests and shrublands. In *Fire regimes and ecosystem properties, Forest*

Service general technical report WO-26, eds. H. A. Mooney, T. M. Bonnicksen, N. L. Christensen, J. E. Lotan, and W. A. Reiners, 58–89. Washington, DC: U. S. Department of Agriculture.

Kückler, A. W. 1964. *Potential natural vegetation of the conterminous United States, special publication no. 36.* New York, New York: American Geographical Society.

Krueger, W. C. and A. H. Winward. 1974. Influence of cattle and big game grazing on understory structure of a Douglas fir-ponderosa pine/ Kentucky bluegrass community. *Journal of Range Management.* 27:450–3.

Laycock, W. A. 1991. Stable states and thresholds of range condition on North American rangelands: A viewpoint. *Journal of Range Management.* 44:427–33.

McGrath, M. T., S. DeStefano, R. A. Riggs, L. L. Irwin, and G. J. Roloff. 2003. Spatially explicit influences on northern goshawk nesting habitat in the Interior Pacific Northwest. *Wildlife Monograph.* 154:1–63.

McNaughton, S. J. 1984. Grazing lawns: Animals in herds, plant form, and coevolution. *American Naturalist.* 124:863–86.

McNaughton, S. J. 1992. The propogation of disturbance in savannas through food webs. *Journal of Vegetation Science.* 3:301–14.

McShea, W. J. 1997. Herbivores and the ecology of forest understory birds. In *The science of overabundance: Deer ecology and population management*, eds. W. J. McShea, H. B. Underwood, and J. H. Rappole, 280–97. Washington, DC: Smithsonian Institute Press.

McShea, W. J., and J. H. Rappole. 1992. White-tailed deer as keystone species within forested habitats of Virginia. *Virginia Journal of Science.* 43:177–86.

Moeur, M. 1985. *COVER: A user's guide to the CANOPY and SHRUBS extension of the Stand Prognosis Model, general technical report INT-190E.* Ogden, Utah: U. S Department of Agriculture, Forest Service.

Molvar, E. M., R. T. Boyer, and V. Van Ballenberghe. 1993. Moose herbivory, browse quality, and nutrient cycling in and Alaskan treeline community. *Oecologia.* 94: 472–70.

Monfore, J. D. 1983. Livestock: A useful tool for vegetation control in ponderosa pine and lodgepole pine plantations. In *Proceedings of the forest land* *grazing symposium*, eds. B. F. Roche, and D. M. Baumgartner, 105–7. Pullman, Washington: Washington State University, Cooperative Extension Service.

Oliver, C. D., and B. C. Larson. 1990. *Forest stand dynamics*. New York, New York: McGraw-Hill, Inc.

Oosting, H. J. 1956. *The study of plant communities*. San Francisco, California: W. H. Freeman and Co.

Paine, R. T. 1969. A note on trophic complexity and community stability. *American Naturalist*. 103:91–3.

Pastor, J., B. Dewey, R. J. Naiman, P. F. MiInnes, and Y. Cohen. 1993. Moose browsing and soil fertility in the boreal forests of Isle Royale National Park. *Ecology*. 74:467–80.

Pastor, J., and R. J. Naiman. 1992. Selective foraging and ecosystem processes in boreal forests. *American Naturalist*. 139:691–705.

Pastor, J., and Y. Cohen. 1997. Herbivores, the functional diversity of plant species, and the cycling of nutrients in ecosystems. *Theoretical Population Biology*. 51:165–79.

Peek, J. M. 1989. Another look at burning shrubs in northern Idaho. In *Proceedings of the symposium on prescribed fire in the intermountain region*, eds. D. M. Baumgartner, D. W. Breuer, B. A. Zamora, L. F. Neuenschwander, and R. H. Wakimoto, 157–9. Pullman, Washington: Washington State University.

Peek, J. M., F. D. Johnson, and N. N. Pence. 1978. Successional trends in a ponderosa pine/bitterbrush community related to grazing by livestock, wildlife, and to fire. *Journal of Range Management*. 31:49–53.

Pfister, R. D., B. L. Kovalchik, S. F. Arno, and R. C. Presby. 1977. *Forest habitat types of Montana, general technical report INT-GTR-34*. Ogden, Utah: U.S. Department of Agriculture, Forest Service.

Pfister, R. D., and S. F. Arno. 1980. Classifying forest habitat types based on potential climax vegetation. *Forest Science*. 26:52–70.

Riggs, R. A., A. R. Tiedemann, J. G. Cook, T. M. Ballard, P. J. Edgerton, M. Vavra, W. C. Krueger, F. C. Hall, L. D. Bryant, L. L. Irwin, and T. Delcurto. 2000. *Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains Ecological Province, research paper, PNW-RP-527*. Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Riggs, R. A., S. C. Bunting, and S. E. Daniels. 1996. Prescribed fire. In *Rangeland wildlife*, ed. P. R. Krausman, 295–319. Denver, Colorado: Society for Range Management.

Roloff, G. J., B. Carroll, and S. Scharosch. 1999. A decision support system for incorporating wildlife habitat quality into forest planning. *Western Journal of Applied Forestry*. 14:91–9.

Rupp, T. S., A. M. Starfield, and F. S. Chapin, III. 2000a. A frame-based spatially explicit model of sub-arctic vegetation response to climatic change: A comparison with point model. *Landscape Ecology*. 15:383–400.

Rupp, T. S., F. S. Chapin III, and A. M. Starfield. 2000b. Response of sub-arctic vegetation to transient climate change on the Seward Peninsula in north-west Alaska. *Global Change Biology*. 6:541–55.

Schreiner, E. G., K. A. Krueger, P. J. Happe, and D. B. Houston. 1996. Understory patch dynamics and ungulate herbivory in old-growth forests of the Olympic National Park, Washington. *Canadian Journal of Forest Research*. 26:255–65.

Shipley, L. A. 1999. Grazers and browsers: How digestive morphology affects diet selection. In *Grazing behaviour of livestock and wildlife, bulletin no. 70*, eds. K. L. Launchbaugh, K. D. Sanders, and J. C. Mosley, 20–7. Moscow, Idaho: University of Idaho, Forest Wildlife and Range Experiment Station.

Smith, E. L. 1978. A critical evaluation of the range condition concept. In *Proceedings of the 1st international rangeland congress*, 266–7. Denver, Colorado: Society for Range Management.

Starfield, A. M., D. H. M. Cummings, R. D. Taylor, and M. S. Quadling. 1993. A frame-based paradigm for dynamic ecosystem models. *Artificial Intelligence Applications*. 7(2):1–13.

Starfield, A. M., and F. S. Chapin. 1996. Model of transient changes in arctic and boreal vegetation in response to climate and land use change. *Ecological Applications*. 6:842–64.

Steele, R. W., R. D. Pfister, R. A. Ryker, and J. A. Kittams. 1981. *Forest habitat types of central Idaho, general technical report INT-GTR-114*. Ogden, Utah: U. S. Department of Agriculture, Forest Service.

Stromayer, K. A. K., and R. J. Warren. 1997. Are overabundant deer herds in the eastern United States creating alternate stable states in forest plant communities? *Wildlife Society Bulletin*. 25:227–34.

Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. *Ecology*. 16:284-307.

Tester, J. R., A. M. Starfield, and L. E. Frelich. 1997. Modeling for ecosystem management in Minnesota pine forests. *Biological Conservation*. 80:313–24.

U. S. Department of Agriculture, Forest Service. 2001. FVS Variants. U. S. Department of Agriculture, Forest Service, Forest Management Service Center. http://www.fs.fed.us/fmsc/fvs/variants/index.php.

———. 2004. FVS Models. http://www.fs.fed.us/fmsc/fvs/description/model.php.

Vallentine, J. F. 1990. *Grazing management*. San Diego, California: Academic Press, Inc.

Waller, D. M., and W. S. Alverson. 1997. The white-tailed deer: A keystone herbivore. *Wildlife Society Bulletin*. 25:217–26.

Weaver, J. E., and F. E. Clements. 1938. *Plant ecology*. New York, New York: McGraw Hill.

Weigand, J. F., R. W. Hayes, A. R. Tiedemann, R. A. Riggs, and T. M. Quigley. 1993. Economic assessment of ungulate herbivory in commercial forests of eastern Oregon and Washington, USA. *Forest Ecology and Management*. 61:137–55.

Weisberg, P. J., N. T. Hobbs, J. E. Ellis, and M. B. Coughenour. 2002. An ecosystem approach to population management of ungulates. *Journal of Environmental Management*. 65:181–97.

Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*. 42: 266–74.

Woodmansee, R. G., and L. S. Wallach. 1981. Effects of fire regimes on biogeochemical cycles. In *Fire regimes and ecosystem properties, general technical report WO-26*, eds. H. A. Mooney, T. M. Bonnicksen, N. L. Christensen, J. E. Lotan, and W. A. Reiners, 379–400. Washington, DC: U. S. Department of Agriculture, Forest Service.

Wykoff, W. R. 1986. *Supplement to the user's guide for the stand prognosis model: Version 5.0, general technical report GTR-INT-208*. Ogden, Utah: U.S. Department of Agriculture, Forest Service.

Wykoff, W. R., N. L. Crookston, and A. R. Stage. 1982. *User's guide to the stand prognosis model, general technical report INT-GTR-133*. Ogden, Utah: U.S. Department of Agriculture, Forest Service.

The Role of Ungulate Herbivory and Management on Ecosystem Patterns and Processes: Future Direction of the Starkey Project

Martin Vavra
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

John G. Kie
U. S. Department of Agriculture, Forest Service,
Pacific Northwest Research Station
La Grande, Oregon

John G. Cook
National Council for Air and Stream Improvement
La Grande, Oregon

Robert A. Riggs
Boise Cascade Corporation
La Grande, Oregon

Introduction

In the national forests of the Blue Mountains, a high percentage of commercial tree species, such as Douglas fir (*Pseudotsuga menziessi*) and true firs (*Abies* spp.), have died as a result of overcrowding on drier sites, drought and insects (Johnson et al. 1995, Quigley and Arbelbide 1997). These conditions are typical of forested land throughout the West (Covington and Moore 1994, Quigley and Arbelbide 1997). Traditional forest management practices (fire exclusion, harvest practices) (Johnson et al. 1995) and livestock grazing (Belsky and

Blumenthal 1997) have contributed substantially to the current situation (Hann et al. 1997). Because of these influences, trees in many stands exist at higher basal areas and higher densities (live and dead) than occurred historically, creating ladder fuels that have dramatically increased the risk of catastrophic wildfires covering very large acreages. In the coming years, it can be expected that fires will continue on forests where excessive fuel build-ups have occurred. And, extensive fuel reduction projects will be initiated to prevent them per the Healthy Forest Restoration Act of 2003.

Secondary succession of both understory and conifer components is initiated following either large fires or fuel reduction treatments. These areas often become focal points of ungulate herbivory for two reasons: (1) vegetation developing after disturbance is often more palatable to ungulates than that available on undisturbed sites (Asherin 1976), and (2) forage production in recently disturbed areas is often greater than in surrounding forest communities with dense canopies (McConnell and Smith 1970, Klinka et al. 1996). Large herbivores are attracted to areas that are characterized by relatively high biomass of palatable food resources; thus, they can be expected to focus foraging activity in recently disturbed areas.

Only rudimentary data exist (Riggs et al. 2000), but these data implicate ungulate herbivory as a significant agent in altering successional trajectories following disturbance (fire, logging, fuel reduction) in the Blue Mountains. Raedeke (1988) stated that selective feeding by forest animals can result in complete changes in the structure, composition and productivity of the forest. In general, plant communities within grazing exclosures are more diverse than the surrounding forest community subjected to continual herbivory (Raedeke 1988). Recent literature reviews (Hobbs 1996, Augustine and McNaughton 1998) clearly indicate the important role of herbivory not only in modifying the composition of plant communities but of ecosystems.

In the Blue Mountains, herbivory has long been recognized to be a competitive factor in ungulate relationships (Cliff 1939, Pickford and Reid 1943) and in suppressing the understory shrub component (Mitchell 1951). However, the role of herbivory as a major disturbance agent is not well recognized in the predominant management paradigms, both in the Blue Mountains and in all forest ecosystems of the western United States. Moreover, knowledge of herbivory effects is more anecdotal than predictive (Riggs et al. 2000).

In this paper, we briefly describe current knowledge regarding the effects of ungulate herbivory on ecosystem patterns and on processes in forests of western North America. We then describe new research that is being initiated at the Starkey Experimental Forest and Range (Starkey) to understand ungulate herbivory effects. Finally, we identify some of the key policy implications associated with management of ungulate herbivory on national forests in the western United States.

Why Study and Manage Ungulate Herbivory?

In the western United States, state and federal land management plans outside of national parks have not recognized the ecological effects of foraging by wild ungulates, as evidenced by the lack of its mention in land management plans. Aber et al. (2000) made no reference to ungulate herbivory, either wild or domestic, in their publication, *Applying Ecological Principles to Management of the U. S. National Forests*. In Europe, the effects of ungulate herbivory are better recognized but primarily for impacts on regeneration of conifers (Kräuchi et al. 2000). At the same time, livestock grazing has long been identified as a potent agent of change in the composition, structure and production of plant communities (Fleischner 1994), and management and research have attempted to address the potentially deleterious effects of this agent. Historically, however, native herbivores, such as deer and elk, usually were perceived as benign constituents of the environment. In this context, native herbivores are considered a product of management rather than a disturbance factor that might influence other resources and various ecological processes. More recently, empirical evidence increasingly indicates that ungulate herbivory, wild or domestic, can have dramatic effects on ecosystem structure and function (Hobbs 1996, Augustine and McNaughton 1998).

In the Blue Mountains, recent findings indicate that wild ungulates dramatically alter the successional pathways of forest understories following disturbance (Riggs et al. 2000) and impact grasslands (Johnson and Vavra 2001, but see Coughenour 1991, Singer 1995). This evidence largely refutes the notion that successional pathways can be predicted without regard to the herbivory regimes that may be expected and, thus, underscores the need to understand how interactions between herbivores and episodic disturbance function, in the context of both wild and domestic herbivory. We define ungulate herbivory as referring to both wild and domestic herbivores unless otherwise specified. Identifying how ungulate herbivory influences composition and structure of forest understories following disturbance is critical to the success of forest planning over the next several years for several reasons.

Forest conditions in the western United States are such that major changes in management focus probably will occur in the next several years. Years of fire suppression, resulting from forest ingrowth, and tree mortality, caused by insect and disease outbreaks, have all contributed to the development of forests that exceed the natural range of variability and are susceptible to conflagrations (Quigley and Arbelbide 1997). For example, 17 percent of the Wallowa-Whitman National Forest has burned in the last 10 years. As a result, management actions are being planned to reduce tree density and fuels and to increase prescribed burning. Nevertheless, vast areas will probably remain untreated well into the future, so the risk of conflagrations will remain high in many areas. Management to restore more natural conditions and continued wildfire conflagrations both set in motion secondary plant successions that will be influenced by ungulate herbivory. The nature and extent of these herbivory effects are currently unknown, but sufficient evidence indicates the effects may be relevant and perhaps deleterious to other wildlife species, biodiversity and ecosystems processes in general. Additionally, herbivory-induced changes in understory may affect productivity of native ungulate herds via negative feedback (Irwin et al. 1994) and may increase the degree of interspecific competition among ungulates (Vavra and Riggs 2004).

New Research at Starkey: Effects of Ungulates on the Forest Ecosystem

Past research at Starkey focused on deer and elk responses to management (Wisdom et al. 2004). New research, however, also will focus on the effects of deer, elk and cattle as disturbance agents in the ecosystem. This new research will specifically measure effects of ungulate herbivory on plant succession and associated changes in nutrient cycling and biodiversity. Results are expected to provide insights to how herbivore regimes can be sustainably integrated with other disturbances in conifer-dominated ecosystems in the Blue Mountains.

The new herbivory research at Starkey specifically is designed to evaluate the effects of ungulate grazing, the interaction with episodic disturbances, the vegetation development and the development of a variety of other vegetative resources (such as grand fir [*Abies grandis*]) and Douglas-fir forests) that are predominant across much of the interior western United States. Within this context, we will assess effects of herbivory by elk and by cattle, the two dominant ungulate species in these ecosystems, using a variety of response variables, including (a) understory forest development, (b) elk and cattle forage selection and digestibility, (c) nitrogen availability and availability of other key nutrients that affect forest productivity, (d) insect herbivory and biodiversity, and (e) small mammal herbivory and biodiversity.

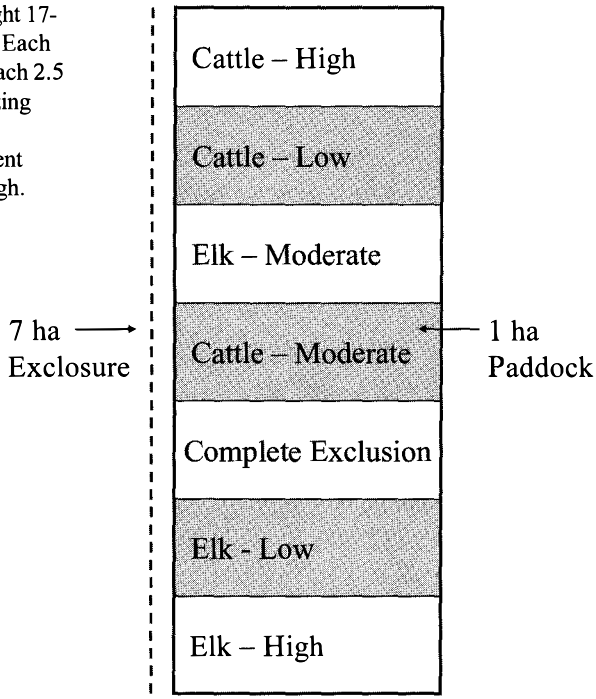
The research is designed to answer the following questions.

1. How does herbivory by elk and by cattle affect alien plant invasions and the composition, life forms, species richness, cover and structure of forest understories of grand fir and Douglas-fir forests?
2. How do effects of herbivory by elk and by cattle on vegetation development vary with low, moderate and high densities of each ungulate, in contrast to no herbivory by either ungulate?
3. How does nitrogen availability and that of other key nutrients change in relation to low, moderate and high densities of elk and cattle, and how might this affect vegetation development?
4. How does diet selection by elk and by cattle interact with herbivory effects on vegetation development under low, moderate and high ungulate densities?
5. How does diet selection by elk and by cattle and the resultant digestibility of forage consumed change over time in relation to low, moderate and high ungulate densities, as changes in forage biomass and quality occur?
6. What are the implications of question 5 on the long-term performance of elk and cattle populations?
7. How do herbivory effects by elk and by cattle change in relation to episodic disturbances of intensive timber harvest and prescribed fire, in contrast to effects with no harvest or fire?
8. How does herbivory by insects and by small mammals affect vegetation development as such forms of herbivory interact with ungulate herbivory?
9. Does biodiversity of insects and small mammals change in relation to ungulate herbivory?
10. How might conceptual models of ungulate herbivory-episodic disturbance interactions be refined and parameterized for stand-level

management of grand fir and Douglas-fir forests of the Interior Northwest?

These questions are being addressed with the use of 8 ungulate exclosures, each 17 acres (7 ha) that have been completed or are under construction at Starkey. Exclosure sites were selected within the strata of timber harvest and fire versus no timber harvest or fire. Each exclosure has been subdivided into 7 enclosed paddocks, each 2.5 acres (1 ha) for stocking density manipulation of ungulates (Figure 1).

Figure 1. Schematic of one of eight 17-acre (7-ha) exclosures at Starkey. Each exclosure contains 7 paddocks, each 2.5 acre (1 ha), within which the grazing trials for cattle and elk will be conducted. Each of seven treatment levels will be low, moderate or high.



Each of the 7 individual paddocks that compose a 17-acre (7-ha) exclosure (Figure 1) are randomly assigned 1 of 7 different treatment levels: 3 levels of grazing by tame elk, 3 levels of grazing by cattle (low, moderate or high density) and 1 level of total ungulate exclusion.

Construction of the eight exclosures began in summer of 2002 and will continue through 2004. Summer grazing trials will take place each July, using tame elk and cattle at each stocking level in the paddocks. Over 40 response variables related to effects on plants, nutrients, ungulate diets, ungulate nutrition and insect and small mammal diversity will be measured over the life of the study, which is intended to last at least 10 years.

This new herbivory study will address effects of multiple densities of ungulates, of wild versus domestic ungulates, and of the interactions of herbivory with episodic disturbances of reduction in wood fuels and prescribed fire. These treatment effects have rarely been evaluated in past herbivory research, which typically compared complete exclusion versus extant herbivore use. There was little measurement of stocking levels and identification of herbivore species, and there was little consideration of interactions with other disturbances (Riggs et al. 2000, Wisdom et al. in review).

To complement this research, models will be developed to predict effects of varying densities of ungulates on plant succession and fire risk (Wisdom et al. in review). The new research at Starkey will be used to parameterize the models for integrated management of ungulate herbivory with episodic disturbances of timber harvest, fuels reduction and fire.

New Research at Starkey: Ungulate Response to Forest Management Practices

As a result of the recently passed Healthy Forests Restoration Act of 2003, efforts to reduce fuel loadings throughout eastern Oregon, eastern Washington and western Idaho will be intensified. These efforts will likely include mechanical treatments, prescribed burning and other techniques used in the past. One such project was recently completed on Starkey. In 1996 the Pacific Northwest Research Station and the Wallowa-Whitman National Forest began planning an Adaptive Resource Management (ARM) fuel reduction program. ARM projects offer unique opportunities for collaboration between research and management. Rarely do research stations have the funds, personnel and expertise to conduct land management activities at such a broad scale. Conversely, national forests lack personnel trained in experimental design and the infrastructure to conduct sound, scientific experiments. By combining the talents and resources of both the Pacific Northwest Research Station and the U. S. National Forest System in a collaborative ARM project, benefits accrue that would be impossible for either branch if operating independently. Many professional, scientific organizations have identified the value of such adaptive, collaborative resource management efforts (Lancia et al. 1996).

The fuel reduction treatments began in spring 2001 and were completed in fall 2003. A total of 46 stands of true fir and Douglas-fir that had suffered substantial mortality during the spruce budworm outbreak in the late 1980s were chosen for treatment. A similar number of stands were left untreated as experimental controls. The treated stands totaled 1,999 acres (809 ha) (mean 34.6 acres [14 ha], range 2.8 to 114.0 acres [1.1–46 ha]). In many of those stands, fuel loadings exceeded 60 tons per acre. The objective of the treatments was to reduce fuel loadings to no more than about 15 to 20 tons per acre. All stands were mechanically thinned using a feller-buncher. About 75 percent of the treated stands were then either broadcast burned or piled and burned to achieve final fuel objectives.

On Starkey, we are using an automated radio telemetry system to track elk, mule deer and cattle to determine how they respond to fuel reduction treatments. Our goal is to determine how they respond to the fuel treatments at the landscape level. Some specific hypotheses we are testing include: (1) animals will be attracted to the treatment areas because of increased production of preferred forages, (2) elk and deer use of treatment areas will be influenced by the presence of roads and traffic levels, (3) spatial memory (remembering where the treatment areas are located and the amount of forage in each) and exploratory trips (to find newly treated patches) will influence the process by which animals use fuel treatment areas, and (4) the shape, size and spacing of treatment patches will affect their use by elk, mule deer and cattle. The automated telemetry system is critical to successfully testing these hypotheses.

This study will yield critically needed information about how overstory and understory vegetation responds once the fuel reduction treatments have been completed. The success of future fuel treatments, even the ability to complete such treatments, will be greatly enhanced if we understand the vegetation response to treatments. In addition, identification of cattle, deer and elk responses will be useful for future environmental assessments and planning efforts.

Similar future research is in the planning stages that will incorporate prescribed fire and, if needed, prior fuel reduction treatments in ponderosa pine (*Pinus ponderosa*) communities. The design of the experiment will be similar to that just described in that experimental protocols will be utilized, but the size of the treatments applied will be on a management scale.

Policy Implications

Herbivory exists as an unrecognized disturbance on landscapes in western North America. Extant plant communities may be influenced by herbivory and may be termed grazing disclimaxes (Riggs et al. 2000). However, the influence of herbivory to alter successional trajectories may be most critical following disturbance, such as fire or fuels reduction treatments. After disturbance, plants often reestablish either from rootstocks or seeds and are more susceptible to foraging by herbivores. Research results relating herbivory and disturbance from Starkey and from elsewhere in eastern Oregon are critically important as the new Healthy Forest Initiative is implemented.

- Important management and policy implications are relevant to herbivory.
- Ungulates act as keystone species that control ecosystem function and properties. These effects have substantial ecological, economic and social consequences that deserve increased focus in forest research and management. Improved understanding of the effects of wild and domestic ungulates, under varying population densities, is a critical need for improved management of ungulate herbivory.
- Structure and composition of vegetation can change dramatically in relation to the intensity of ungulate herbivory, and it can have significant impacts on biodiversity. Biodiversity responses to varying levels of ungulate herbivory are not well documented and deserve high priority in research.
- Marked changes in structure and composition of vegetation due to herbivory may increase error associated with identification of vegetative communities (Riggs et al. 2000) in the context of potential natural vegetation, reducing the accuracy and the value of the predominant vegetative and land classification systems currently used by federal management agencies throughout the Northwest (e. g., see Henderson et al. 1992).
- Ungulate herbivory can negatively affect ungulate productivity through negative feedback mechanisms mediated through suppression of food resources. Declining biomass of forage is likely to increase the intensity of ungulate herbivory. In turn, the increased herbivory may not allow preferred forage plants to establish, recover and persist unless disturbances, such as fire or timber harvest, occur across spatial extents

sufficient to providing a forage biomass far in excess of what can be utilized by ungulates. On the other hand, where disturbance is not sufficient, ungulate populations will have to be reduced to avoid degradation. In any event, managers and policy makers might consider how the balance between landscape disturbance and herbivore population management influences long-term carrying capacity and ecosystem stability. These relations deserve attention in future ungulate-landscape research.

- Ungulate herbivory may contribute to conifer ingrowth in forest understories, particularly when high levels of herbivory are combined with fire suppression, resulting in substantially higher risk of stand-replacement fires than existed historically.
- When considering forest management practices, such as timber harvest, fuels reduction and prescribed burning, the size and number of treated areas will affect the recovery of those areas, with smaller treatments likely to attract intensive herbivory and associated changes in vegetation development and other ecosystem processes.

Reference List

Aber, J., N. Christensen, I. Fernandez, J. Franklin, L. Hidinger, M. Hunter, J. MacMahon, D. Mladenoff, J. Pastor, D. Perry, R. Slangen, H. van Miegroet. 2000. Applying ecological principles to management of the U. S. National Forests. *Issues in Ecology*. 6:1–20.

Asherin, D. A. 1976. Changes in elk use and available browse production on north Idaho winter ranges following prescribed burning. In *Proceedings of the elk-logging-roads symposium*, ed. S. R. Hieb, 122–34. Moscow, Idaho: Forest, Wildlife, and Range Experiment Station, University of Idaho.

Augustine, D. J., and S. J. McNaughton. 1998. Ungulate effects on the functional species composition of plant communities: Herbivore selectivity and plant tolerance. *Journal of Wildlife Management*. 62:1,165–83.

Belsky, A. J., and D. M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior West. *Conservation Biology*. 11:315–27.

Cliff, E. P. 1939. Relationship between elk and mule deer in the Blue Mountains of Oregon. In *4th North American wildlife conference.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Cook, J. G., L. Irwin, L. Bender, B. Johnson, and D. Boyce. 1999. *Declining elk herd productivity: Assessing habitat's role in the Pacific and inland Northwest—A Proposal.* unpublished.

Coughenour, M. B. 1991. Biomass and nitrogen responses to grazing of upland steppe on Yellowstone's northern winter range. *Journal of Applied Ecology.* 28:71–82.

Covington, W. W., and M. Moore. 1994. Southwestern Ponderosa forest structure, changes since Euro-American settlement. *Journal of Forestry.* 92:39–47.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. *Conservation Biology.* 8:629–44.

Hann, W. J., J. L. Jones, M. G. Karl, P. F. Hessburg, R. E. Keane, D. G. Long, J. P. Menakis, C. H. McNicoll, S. G. Leonard, R. A. Gravenmier, and B. G. Smith. 1997. Landscape dynamics of the basin. In *An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins, general technical report PNW-GTR-405*, eds. T. M. Quigley, and S. J. Arbelbide. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Henderson, J. A., R. D. Lesher, D. H. Peter, and D. C. Shaw. 1992. *Field guide to the forested plant associations of the Mt. Baker-Snoqualmie National Forest, technical paper R6 ECOL TP 028–91.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Hobbs, N. T. 1996. Modification of ecosystems by ungulates. *Journal of Wildlife Management.* 60:695–713.

Irwin, L. L., J. G. Cook, R. A. Riggs, and J. M. Skovlin. 1994. *Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystem, general technical report PNW-GTR-325.* Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Johnson, K. N., J. Agee, R. Beschta, J. Beuter, S. Gregory, L. Kellogg, W. McComb, J. Sedell, T. Schowalter, and S. Tesch. 1995. *Forest health* *and timber harvest on national forest in the Blue Mountains of Oregon*. Corvallis, Oregon: Oregon State University.

Johnson, C. G., and M. Vavra. 2001. Permanent monitoring points: An example of value and utility in the study of elk impacts on bunchgrass communities in Hells Canyon. *Abstracts Society for Range Management Annual Meeting*. February 2001.

Klinka, K., H. Y. H. Chen, Q. Wang, and L. de Montigny. 1996. Forest canopies and their influence on understory vegetation in early-seral stands on West Vancouver Island. *Northwest Science*. 70:193–200.

Kräuchi, N., P. Brang, and W. Shönenberger. 2000. Forests of mountainous regions: Gaps in knowledge and research needs. *Forest Ecology and Management*. 132:78–82.

Lancia, R. A., C. E. Braun, M. W. Collopy, R. D. Dueser, J. G. Kie, C. J. Martinka, J. D. Nichols, T. D. Nudds, W. R. Porath, and N. C. Tilghman. 1996. ARM For the future: Adaptive resource management in the wildlife profession. *Wildlife Society Bulletin*. 24:436–42.

McConnell, B. R., and J. G. Smith. 1970. Response of understory vegetation to ponderosa pine thinning in eastern Washington. *Journal of Range Management*. 23:208–12.

Mitchell, G. E. 1951. Status of browse on ranges of eastern Oregon and eastern Washington. *Journal of Range Management*. 4:249–53.

Pickford, G., and E. Reid. 1943. Competition of elk and domestic livestock for summer range forage. *Journal of Wildlife Management*. 7:328–32.

Quigley, T. M., and S. J. Arbelbide. 1997. *An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins, general technical report PNW-GTR-405*. Portland, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Region and Bureau of Land Management.

Raedeke, K. J. 1988. Forest management and wildlife in the Pacific Northwest. *Northwest Environmental Journal*. 4:263–78.

Riggs, R. A., A. R. Tiedemann, J. G. Cook, T. M. Ballard, P. J. Edgerton, M. Vavra, W. C. Krueger, F. C. Hall, L. D. Bryant, L. L. Irwin, and T. DelCurto. 2000. *Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains Ecological Province, research paper PNW-RP-527*. La Grande, Oregon: U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Singer, F. J. 1995. Effects of grazing by ungulates on upland bunchgrass communities of the northern winter range of Yellowstone National Park. *Northwest Science*. 69:191.

U. S. Department of Agriculture and U. S. Forest Service. 2003. National Fire Plan. Wildland Fire Leadership Council. http://www.fireplan.gov.

Vavra, M., and R. Riggs. 2004. Managing multiple herbivores across heterogeneous landscapes. In *Abstracts of 60th annual meeting of the Society of Range Management*. Salt Lake City, Utah: Society of Range Management.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:455–74.

Wisdom, M. J., M. Vavra, J. M. Boyd, M. A. Hemstrom, A. A. Ager, and B. K. Johnson. In review. Understanding ungulate herbivory-episodic disturbance effects: knowledge gaps and research needs. *Wildlife Society Bulletin.*

Has the Starkey Project Delivered on Its Commitments?

Jack Ward Thomas
Boone and Crockett Club and College of Forestry and Conservation, University of Montana
Missoula

Michael J. Wisdom
U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station
La Grande, Oregon

Introduction

The Starkey Project was conceived from intense debate about how best to manage habitats and populations of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) in western North America (Rowland et al. 1997). Founders of the research effort promised to provide definitive knowledge about effects of the dominant public land uses on mule deer and elk and to transfer that knowledge in relevant and synthesized forms for easy management application.

In our paper, we review the commitments of the Starkey Project and summarize its major achievements. We also identify key research opportunities that, in our view, are optimally addressed by the project's unique research facility and compelling partnerships. Our review is focused on the historical performance of the Project in generating knowledge of high management utility, and the efficient transfer of that knowledge to land and wildlife managers in western North America.

Commitments and Accomplishments

The Starkey Project was designed to address the most contentious of resource uses of public land and to provide cause-effect responses of mule deer and elk to those uses. When the project was proposed in the early 1980s (Wisdom et al. 2004a), four resource issues were the focus of debate in relation to deer and elk: (1) road management, (2) intensive timber production and thermal cover, (3) competition with cattle, and (4) breeding efficiency of male elk. These four issues became the foundation of the Starkey Project's studies that began in 1989 and ended during the mid- to late 1990s at the Starkey Experimental Forest and Range (Starkey) and the Kamela research sites in northeastern Oregon (Rowland et al. 1997, Cook et al. 1998, Wisdom et al. 2004a).

Road Management

Project scientists undertook new research to estimate the long-term spatial relations of mule deer and elk to the type of road management (open, closed and administrative use) and to the rate of motorized traffic (number of vehicle trips per unit of time) on the roads. Scientists designed the research to estimate these relations, with sufficient sample sizes to clearly portray spatial patterns and at landscape scales commensurate with ungulate use and management decisions.

Investigators met this commitment by generating the largest data set ever amassed on locations of deer and elk in relation to distance from roads of different types and of different rates of traffic; scientists characterized these relations with spatially explicit models that could be directly applied in management (Rowland et al. 2000, 2004; Wisdom 1998; Wisdom et al. 2004b). Results are now used by state wildlife agencies and federal land management agencies throughout western North America. Findings constituted part of the foundation for the national roads policy established by the U. S. Department of Agriculture, Forest Service (FS), thus affecting road management on all national forests.

Intensive Timber Production and Thermal Cover

Thermal cover is defined as, "cover used by deer or elk to assist in maintaining homiothermy," and as, "any stand of coniferous forest trees 40 feet (12 m) or more tall with an average canopy closure exceeding 70 percent" (Thomas et al. 1979:113-4). Whether thermal cover was a requirement of elk (that is, whether animal performance suffered without thermal cover) constituted a major question that the Starkey Project promised to examine. A corollary question that the project pledged to answer was whether intensive timber harvest had any measurable effects on habitats and populations of mule deer and elk. Prior investigations had shown that elk avoided or substantially reduced their use of areas subject to timber harvest and associated road use. But, what if the entire landscape was so treated and deer and elk had to react to the resultant conditions?

The question of whether elk require or benefit from thermal cover was addressed in an experimental study at Kamela (Cook et al. 1998). Here, the nutritional condition of tractable elk maintained in pens was monitored in relation to varying amounts of thermal cover and no cover. Results, which detected no positive physiological benefits to elk from the presence of thermal cover, have changed managers' thinking about elk-cover relations (Cook et al. 2004a). As a result, many land managers have modified their thermal cover direction for elk. (Elk use of dense forest stands, such as for hiding cover and escapement, remain important considerations in management [Lyon and Christensen 2002]). Notably, contentious litigation related to meeting thermal cover standards on national forests was resolved, saving millions of dollars to the FS planning process.

Effects of timber management (harvest and roading) were addressed by conducting a landscape experiment to evaluate cattle and elk responses to intensive harvest and associated human activities at Starkey from 1989 to 1996 (Rinehart 2001, Wisdom et al. 2004c). Thousands of ungulate locations were collected before, during and after completion of a timber sale that substantially reduced canopy closure on over half the forested study site, and more than doubled the road density. Measures of animal response provided little evidence of lasting negative effects with the major exception of the substantial increase in elk vulnerability to harvest by hunters (Wisdom et al. 2004c). Results (Rinehart 2001, Wisdom et al. 2004c) are now available for timber sale planning on national forests.

Competition with Cattle

The degree to which mule deer, elk and cattle compete for food and space was another key research problem. By evaluating the spatial distributions, resource selection patterns and behavioral interactions of the three ungulates as cattle were rotated through the livestock pastures each summer during the 1990s (Coe et al. 2001, 2004), scientists estimated a realistic forage allocation among cattle, deer and elk by month and season (Johnson et al. 1996). Combined with a subsequent study of diet overlap among the three species (Findholt et al. 2004), scientists used the results to build new models capable of assessing the trade-offs and benefits of different grazing management scenarios on summer ranges shared by cattle, mule deer and elk (Ager et al. 2004).

These models of forage allocation (Johnson et al. 1996) and grazing management (Ager et al. 2004) allow rangeland managers to evaluate trade-offs of changing stocking rates among the three ungulates under different assumptions that reflect ecological differences in ungulate use of summer ranges. Application of these tools is expected to facilitate timely completion of new allotment plans now under development in national forests in the western United States.

Breeding Efficiency of Bull Elk

Determining the degree to which elk productivity is affected by age of breeding males was a major objective of the Starkey Project in the 1990s (Noyes et al. 2004). During two separate, five-year studies, conception dates in female elk were shown to occur earlier and to be more synchronous with increasing breeding age of male elk (Noyes et al. 1996, 2002). That is, breeding by older bulls resulted in calves being born earlier and over a shorter time period each spring, conferring a number of survival benefits (Noyes et al. 2004).

As a result, states and provinces throughout western North America modified their hunting regulations to protect older male elk from being taken by hunters. Protection of older males from hunting is now recognized and implemented as an important management strategy, owing to the benefits of older males as breeders, in combination with the recognized social and aesthetic benefits of viewing these animals (Bunnell et al. 2002).

Additional Accomplishments

As project investigators completed their initial studies during the 1990s, several new resource issues in public land management emerged that could be addressed with the use of the project's novel research technologies (Wisdom et al. 2004a). Those issues became the focus of subsequent ungulate research at Starkey. New research completed or underway includes:

- effects of woody fuels reduction on distributions and on forage conditions for mule deer, elk and cattle (Vavra et al. 2004)
- deer and elk responses to off-road recreation, including travel by all-terrain vehicles, horseback, mountain bike and foot (Wisdom et al. 2004d)
- development and testing of new road models for elk management (Rowland et al. 2004)
- synthesis and modeling of factors that affect elk vulnerability to harvest by hunters (Vales 1996, Johnson et al. 2004)

- energetic costs to deer and elk exposed to, but not harvested under, varyious levels of hunting pressure and season designs (Johnson et al. 2004)
- hourly, daily and seasonal changes in movements and habitat use by mule deer and elk, measured at fine resolution with one of the largest data sets on ungulate locations ever amassed (Ager et al. 2003)
- effects of sampling design on resource selection and home range estimators for wildlife research (Garton et al. 2001, Leban et al. 2001)
- exploration and use of diffusion theory to model animal movements (Brillinger et al. 2004)
- consideration of nutrition demands and animal condition to enhance elk productivity (Cook et al. 2003, 2004b)
- effects of ungulate herbivory on vegetation development and ecosystem processes (Riggs et al. 2004, Vavra et al. 2004)
- validation of resource selection patterns for elk to strengthen and expand inference space for management (Coe 2003).

Results from these follow-on studies have yielded compelling benefits to managers. For example, defensible options for managing off-road vehicles and other off-road recreation on public land are now being developed. Findings are expected to provide information about effects of off-road recreation, particularly motorized recreation, the most-rapidly growing use of public land in the United States (Havlick 2002).

Another example is the current research on management of deer, elk and other wildlife in relation to fuel treatments to reduce fire risk (Vavra et al. 2004). The FS identified Starkey as a national research site to evaluate success of fuel treatments to reduce fire risk in national forests and to assess effects on wildlife.

Two other studies of keen interest and utility to managers are (1) energetic and movement responses of deer and elk to hunting, specifically when animals are exposed to hunting activities but not harvested (Johnson et al. 2004), and (2) effects of nutrition on elk productivity (Cook et al. 2003, 2004b). State, federal, private and provincial resource managers are using findings from this new research to meet increasing demands for hunting and viewing of elk, which generate hundreds of millions of dollars annually to local and regional economies in western North America (Bunnell et al. 2002).

Effective Transfer of Knowledge to Management

The Starkey Project has been one of the few research programs in the FS with a full-time focus on the science of technology transfer (Rowland et al. 1997). The transfer program has served as an effective liaison between management and research. As a result, the project has shared research technologies and results directly with more than 200,000 recipients, encompassing local, regional, national and international organizations, groups and agencies. These communication mediums beyond scientific publications included field tours, presentations, workshops, symposia, news releases, newspaper features, magazine articles, radio interviews and television coverage. More than 600 field tours and presentations have been given, and scientists have organized over 15 field tours and more than 20 presentations per year during the past decade.

The technology transfer program helped garner widespread public acceptance and support of the project's research and results (Rowland et al. 1997). Transfer efforts have been viewed as one of the primary reasons that many land and wildlife managers have adopted Starkey research findings.

Starkey's Future: Best Uses of a Unique Facility

The unique research facility at Starkey (see Rowland et al. 1997, 1998; Wisdom et al. 2004a) is an optimal environment in which to conduct manipulative experiments and to elucidate cause-effect responses of ungulates at landscape scales. We know of no other research facility that has the combination of experimental controls and animal tracking technologies needed to conduct such cause-effect research at landscape scales. There are emerging, significant research questions ideally suited for study within the project's unique research facility.

Effects of Elk Removal on Mule Deer

Scientists have noted the anecdotal evidence suggesting that as elk invade mule deer ranges and become plentiful, mule deer numbers decline (e. g., see Wallmo 1981, Wisdom and Thomas 1996). In support of this hypothesis, one of the more interesting findings at Starkey has been the consistently strong avoidance shown by mule deer for elk (Johnson et al. 2000; Coe et al. 2001, 2004; Stewart et al. 2002; Wisdom et al. 2004b). Interference competition may be operating, causing mule deer avoidance of elk and reducing population performance of deer.

The only definitive manner in which to document whether mule deer avoidance of elk results in reduced population performance of deer is to conduct a manipulative experiment, where elk are removed from an area and the population response of deer is measured. Then, elk could be reintroduced to the study area, first at low density, as the mule deer population response was measured. Finally, elk could be allowed to reach high density, and mule deer response again could be measured. The cycle could be repeated to validate the initial set of mule deer responses to elk removal and reintroduction.

Starkey is one of the few sites in which such removal experiments could be done in a definitive manner to document whether interference competition is operating between mule deer and elk. Given that mule deer populations are declining throughout much of western North America (Kie and Czech 2000), this research should be viewed as one of the highest priorities of the Starkey Project.

Effects of Hunting Season Designs on Mule Deer and Elk

Research at Starkey has documented substantial energetic costs to mule deer and elk under different types and lengths of hunting seasons (Johnson et al. 2004). What remains to be evaluated, however, are the effects of the full spectrum of different hunt types, season lengths, hunter densities and associated options for road and off-road access on deer and elk. In particular, how all-terrain vehicles (ATVs) with full access to the hunting landscape affect harvest rates of targeted animals and energetic costs of nontargeted animals has not been studied. This issue is a major concern of state wildlife agencies. Starkey, with its capability to design and administer a variety of hunts as part of landscape experiments and to accurately measure the population responses of ungulates, is well-suited for such research.

Off-road Vehicle Effects on Wildlife and Other Resources

Use of ATVs during nonhunting periods can substantially increase movement rates of elk and, consequently, is likely to increase the daily energetic expenditures of animals (Wisdom et al. 2004d). What remains unclear, however, is the effect of ATVs on elk during hunting seasons. Moreover, the effects of ATVs and of other forms of off-road recreation on additional species of wildlife, on exotic plant invasions and on other resources, such as soil productivity and water quality, have not been studied experimentally.

These topics are well suited for new research at Starkey, to complement and to validate current research on effects of off-road recreation (Wisdom et al. 2004d). ATV riding is the most rapidly-growing recreation on public land (Havlick 2002), and other forms of off-road recreation, such as mountain biking, horseback riding and hiking, also are increasing. New research on the comparative effects of these different forms of off-road recreation is urgently needed. Starkey is ideally suited for conducting such research with appropriate experimental controls to elucidate cause-effect relations.

Effects of Ungulate Herbivory on Forest Development and Productivity

Ungulate herbivory has profound effects on vegetation development and productivity in forest ecosystems (Hobbs 1996; Riggs et al. 2000, 2004). Livestock grazing has long been recognized as an agent of change in composition, structure and development of plant communities (Fleischner 1994), but herbivory by wild ungulates has not always been recognized as an ecological force in western North America (Augustine and McNaughton 1998).

New research is being planned at Starkey that would evaluate effects of varying levels of grazing by cattle and elk on plant succession, soil nutrients, animal biodiversity and ungulate nutrition, as measured over long (10 or more years) time periods (Vavra et al. 2004). This research will use a series of 18-acre (7.3-ha) exclosures, each subdivided into 7 pastures with different levels of elk and cattle grazing during summer. The research, however, requires substantial funding (at least $500,000 per year) that currently is unavailable. Moreover, herbivory by cattle versus elk is a highly contentious issue, fraught with uncertainty. And, it is clouded by strong political interests. For example, stringent utilization standards are often imposed on cattle grazing in riparian habitats to meet objectives for management of salmonids. Yet, effects of herbivory by wild ungulates may also affect habitat conditions in ways neither acknowledged nor understood. The new herbivory research at Starkey is needed to understand the effects of wild versus domestic ungulates in relation to management goals for wildlife, vegetation, silviculture, nutrient availability and other measures of forest productivity (Vavra et al. 2004).

Habitat Management Relations with Ungulate Nutrition

The individual, stand-level effects of timber management, fuels reduction, prescribed burning, wildfires, insect defoliation and other episodic disturbances on the nutritional condition of ungulates is understood at a coarse level. However, the nutritional effects of the combination of these disturbance events in time and space on the landscape are complex and poorly documented. For example, episodic disturbances create open-canopy forests, resulting in substantial increases in forage biomass for ungulates (Peek et al. 2001). Yet, the degree to which these episodic disturbances meet the nutritional demands of wild and domestic ungulates is not well understood or well studied (Cook et al. 2003).

New studies of ungulate foraging dynamics and of nutritional intake are needed with the use of tame elk, deer and cattle. The Starkey Project and its many partners maintain tractable mule deer, elk and cattle for such grazing studies (Cook et al. 2003). These animals and these scientists experienced in the use of these animals could be quickly put to work at Starkey to discern the nutritional benefits of fuels reduction treatments, of timber harvest and of other habitat manipulations at the mosaic of stands across the landscape.

Moreover, project scientists have developed a compelling set of maps depicting the probabilities of habitat use by elk, mule deer and cattle across the landscape, estimated by month and season from years of research at Starkey (Ager et al. 2004). These probabilities could be linked to the underlying nutritional conditions of each habitat type, using the tractable cattle, deer and elk available at Starkey. Such research would provide insight about the nutritional consequences of ungulate habitat choices. Given current concerns about declining productivity in elk and mule deer populations (Kie and Czech 2000, Johnson et al. 2004), a basic management question still needs to be addressed. That is, how well are current habitat conditions and conditions planned under future landscapes providing for the nutritional needs of wild and domestic ungulates? New studies on ungulate nutrition in relation to habitat conditions at Starkey could fill this important gap in current knowledge.

Conclusions

The Starkey Project has delivered on its promises. In fact, the project has paid off far in excess of those promises. To drive home that point, consider the following: over 140 publications are complete or are in press; over 40 partners are involved in the research; over 50 studies are complete or are underway; more than 600 tours and presentations to local, regional, national and international audiences are provided. Results are now being used by state, federal, tribal and private land and wildlife managers across western North America. Keys to the project's remarkable success have been many: (1) unique technologies that have facilitated the completion of needed studies, previously considered difficult or impossible to conduct under appropriately controlled conditions, (2) long-term commitments by diverse federal, state, private and tribal interests, (3) continued focus on ungulate issues of highest relevance to wildlife management and (4) constant and open dialogue and sharing of the research with all interests and resource users (see Kie et al. 2004, Quigley and Wisdom 2004, Wisdom et al. 2004a).

Perhaps most impressive is the economic return from the research, which is conservatively estimated to substantially exceed the 20 million dollars invested by partners during the project's 22-year history. This economic return will increase in the future as use of resulting information by land and wildlife managers continues to expand. This unique and long-lasting research program is a shining example of applied research conducted under controlled and rigorous conditions and made available through publications, tours, presentations and media outlets. We urge partners in the Starkey Project to continue supporting this long-term research for the benefit of land and wildlife managers and for the benefit of those who care about forests, rangelands and the wildlife these habitats support.

Reference List

Ager, A. A., B. K. Johnson, J. W. Kern and J. G. Kie. 2003. Daily and seasonal movements and habitat use by female Rocky Mountain elk and mule deer. *Journal of Mammalogy*. 84:1,076–88.

Ager, A. A., B. K. Johnson, P. K. Coe, and M. J. Wisdom. 2004. Landscape simulation of foraging by elk, mule deer and cattle on summer range. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:687–707.

Augustine, D. J., and S. J. McNaughton. 1998. Ungulate effects on the functional species composition of plant communities: Herbivore selectivity and plant tolerance. *Journal of Wildlife Management*. 62:1,165–83.

Brillinger, D. R., Preisler, H. K., Ager, A. A., and Kie, J. G. 2004. An exploratory data analysis (EDA) of the paths of moving animals. *Journal of Statistical Planning and Inference*. 122:43–63.

Bunnell, S. D., M. L. Wolfe, M. W. Brunson, and D. E. Potter. 2002. Recreational use of elk. In *North American elk: Ecology and management*, eds. D. E. Toweill, and J. W. Thomas, 701–47. Washington, DC: Smithsonian Institution Press.

Coe, P. K. 2003. V*alidating resource selection functions for elk in the Blue Mountains of northeast Oregon*. M. S. thesis proposal, Oregon State University, Corvallis.

Coe, P. K., B. K. Johnson, J. W. Kern, S. L. Findholt, J. G. Kie, and M. J. Wisdom. 2001. Responses of elk and mule deer to cattle in summer. *Journal of Range Management*. 54:205, A51–A76.

Coe, P. K., B. K. Johnson, K. M. Stewart, and J. G. Kie. 2004. Spatial and temporal interactions of elk, mule deer and cattle. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:656–69.

Cook, J. G., L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 1998. Relations of forest cover and condition of elk: A test of the thermal cover hypothesis in summer and winter. *Wildlife Monographs*. 141:1–61.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2003. Effects of summer-autumn nutrition and parturition date on reproduction and survival of elk. *Wildlife Monographs*. 155:1–61.

Cook, J. G., L. L. Irwin, L. D. Bryant, R. A. Riggs, and J. W. Thomas. 2004. Thermal cover needs of large ungulates: A review of hypothesis tests. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:708–26.

Cook, J. G., B. K. Johnson, R. C. Cook, R. A. Riggs, T. DelCurto, L. D. Bryant, and L. L. Irwin. 2004. Nutrition and parturition date effects on elk: Potential implications for research and management. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:604–24.

Findholt, S. L., B. K. Johnson, D. Damiran, T. DelCurto, and J. G. Kie. 2004. Diet composition, dry matter intake and diet overlap of mule deer, elk and cattle. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:670–86.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. *Conservation Biology*. 8:629–44.

Garton, E. O., M. J. Wisdom, F. A. Leban, and B. K. Johnson. 2001. Experimental design for radiotelemetry studies. In *Radio tracking and animal populations*, eds. J. Millspaugh, and J. Marzluff, 1–42. San Diego, California: Academic Press.

Havlick, D. G. 2002. *No place distant: Roads and motorized recreation on America's public lands*. Washington, DC: Island Press.

Hobbs, N. T. 1996. Modification of ecosystems by ungulates. *Journal of Wildlife Management*. 60:695–713.

Johnson, B. K., A. A. Ager, S. A. Crim, M. J. Wisdom, S. L. Findholt, and D. Sheehy. 1996. Allocating forage among wild and domestic ungulates—A new approach. In *Proceedings symposium on sustaining rangeland ecosystems, special report 953*, eds. W. D. Edge, and S. L. Olson-Edge, 166–69. Corvallis, Oregon: Oregon State University Extension Service.

Johnson, B. K., A. A. Ager, J. H. Noyes, and N. Cimon. 2004. Elk and mule deer responses to variation in hunting pressure. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:625–40.

Johnson, B. K, J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk in spring. *Journal of Wildlife Management*. 64:685–97.

Kie, J. G., and B. Czech. 2000. Mule and black-tailed deer. In *Ecology and management of large mammals in North America*, eds. S. Demarais, and P. R. Krausman, 629–57. Upper Saddle River, New Jersey: Prentice Hall, Inc.

Kie, J. G., A. A. Ager, N. J. Cimon, M. J. Wisdom, M. M. Rowland, P. K. Coe, S. L. Findholt, B. K. Johnson, and M. Vavra. 2004. The Starkey database: Spatial-environmental relations of North American elk, mule deer and cattle at the Starkey Experimental Forest and Range in northeastern Oregon. *Transactions of the North American Wildlife and Natural Resources Conference*. 69:475–90.

Leban, F. A., M. J. Wisdom, E. O. Garton, B. K. Johnson, and J. G. Kie. 2001. Effect of sample size on the performance of resource selection analysis. In *Radio tracking and animal populations*, eds. J. Millspaugh, and J. Marzluff, 291–307. San Diego, California: Academic Press.

Noyes, J. H., B. K. Johnson, L. D. Bryant, S. L. Findholt, and J. W. Thomas. 1996. Effects of bull age on conception dates and pregnancy rates of cow elk. *Journal of Wildlife Management.* 60:508–17.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2002. Effects of male age and female nutritional condition on elk reproduction. *Journal of Wildlife Management.* 66:1,301–07.

Noyes, J. H., B. K. Johnson, B. L. Dick, and J. G. Kie. 2004. Influences of age of males and nutritional condition on short- and long-term reproductive success of elk. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:572–85.

Peek, J. M., J. J. Korol, D. Gay, and T. Hershey. 2001. Overstory-Understory biomass changes over a 35-year period in southcentral Oregon. *Forest Ecology and Management.* 150:267–77.

Quigley, T. M., and M. J. Wisdom. 2004. The Starkey project: Long-term research for long-term management solutions. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:442–54.

Riggs, R. A., J. G. Cook, and L. L. Irwin. 2004. Management implications of ungulate herbivory in northwest forest ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:759–84.

Riggs, R. A., A. R. Tiedemann, J. G. Cook, T. M. Ballard, P. J. Edgerton, M. Vavra, W. C. Krueger, F. C. Hall, L. D., Bryant, L. L. Irwin, and T. DelCurto. 2000. *Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains Ecological Province, research paper PNW-RP-527.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rinehart, J. M. 2001. *Effects of intensive salvage logging on Rocky Mountain elk at the Starkey Experimental Forest and Range.* M. S. thesis, University of Montana, Missoula.

Rowland, M. M., L. D. Bryant, B. K. Johnson, J. H. Noyes, M. J. Wisdom, and J. W. Thomas. 1997. *The Starkey project: History, facilities, and data collection methods for ungulate research, general technical report PNW-GTR-396.* Portland, Oregon: U. S. Department of Agriculture, Forest Service.

Rowland, M. M., P. K. Coe, R. J. Stussy, A. A. Ager, N. J. Cimon, B. K. Johnson, and M. J. Wisdom. 1998. *The Starkey habitat database for ungulate* *research: Construction, documentation and use, general technical report PNW-GTR-430.* Portland, Oregon: U. S Department of Agriculture, Forest Service.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. *Journal of Wildlife Management*. 64:672–84.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: Implications for management in forested ecosystems. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:491–508.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: Resource partitioning and competitive displacement. *Journal of Mammalogy*. 83:229–44.

Thomas, J. W., H. Black, Jr., R. J. Scherzinger, and R. J. Pedersen. 1979. Deer and elk. In *Wildlife habitats in managed forests—The Blue Mountains of Oregon and Washington, handbook number 553,* 124–37. Washington, DC: U.S Department of Agriculture Forest Service.

Vales, D.J., 1996. *User's manual for ELKVUN, an elk vulnerability, hunter, and population projection program, version 1.0.* Moscow, Idaho: University of Idaho, Department of Fish and Wildlife Resources.

Vavra, M., M. J. Wisdom, J. G. Kie, J. G. Cook, and R. A. Riggs. 2004. The role of ungulate herbivory and management ecosystem patterns and processes: Future direction of the Starkey project. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:785–97.

Wallmo, O. C., editor. 1981. *Mule and black-tailed deer of North America.* Lincoln, Nebraska: University of Nebraska Press.

Wisdom, M. J. 1998. *Assessing life-stage importance and resource selection for conservation of selected vertebrates.* Ph.D. dissertation, University of Idaho, Moscow.

Wisdom, M. J., and J. W. Thomas. 1996. Elk. In *Rangeland wildlife*, ed. P. R. Krausman, 157–81. Denver, Colorado: Society for Range Management.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2004d. Effects of off-road recreation on mule deer and elk.

Transactions of the North American Wildlife and Natural Resources Conference. 69:531–50.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004a. Overview of the Starkey project: Mule deer and elk research for management benefits. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:455–74.

Wisdom, M. J., N. J. Cimon, B. K. Johnson, E. O. Garton, and J. W. Thomas. 2004b. Spatial partitioning by mule deer and elk in relation to traffic. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:509–30.Wisdom, M. J., B. K. Johnson, M. Vavra, J. M. Boyd, P. K. Coe, J. G. Kie, A. A. Ager, and N. J. Cimon. 2004c. Cattle and elk responses to intensive timber harvest. *Transactions of the North American Wildlife and Natural Resources Conference.* 69:727–58.

Registered Attendance

Alabama
James Barry Grand, Michael J. Hardy, David C. Hayden

Alaska
Doug Alcorn, Gene V. Augustine, Ellen Campbell, Ellen M. Clark, Cleveland J. Cowles, Maureen S. Cowles, Tina Cunning, Christopher Estes, Christopher D. Garner, Rowan W. Gould, Jeff Hughes, Winifred Kessler, Gary Larsen, Tom Liebischer, Kellie Peirce, Lou Regelin, William Regelin, Steve Reidsma, Matthew H. Robus, Mark Sledge

Alberta
Bill D. Gummer

Arizona
Jay R. Adkins, Kerry Baldwin, Bob Broscheid, Jim de Vos, Randy English, Luke A. Fedewa, Dennis Fenn, Jim Hessil, Terry B. Johnson, Carol L. Krausman, Paul R. Krausman, Sam R. Lawry, Bryan L. Morrill, Sergio Obregon, Mike J. Rabe, Lawrence M. Riley, Duane L. Shroufe, Linda Shroufe, Bruce Taubert, Bill Van Pelt

Arkansas
Marilyn K. Bentz, Michael D. Gibson, David Goad, Donny Harris, Robert J. Luce, Donald F. McKenzie, Gregg S. Patterson, Amanda J. Riggs

British Columbia
Rick Taylor

California
Bill Berry, Jack A. Blackwell, Lauren Bonin, Kirsten Christopherson, Liz Clark, Tammy Conkle, Diana L. Craig, Rhys M. Evans, James Fenwood, Bill Ferrier, Scott Fletcher, Danielle Flynn, Nancy Francine, Bob Frost, Chad E. Garber, Dave Gibbons, Don Guidoux, Paul Growald, Jake Hartwick, Wally Haussamen, Steve Holl, Bob Holmes, M. Brent Husung, Manuel Joia, Jr., Marti J. Kie, Robert N. Knight, Mary K. Lamb, Steve Loe, Wayne Long, Neil Lynn, Ryan Mathis,

Deborah C. Maxwell, Michael R. Miller, Rodney J. Mouton, Laura Muhs, Steve Pennix, Carl Pope, Bruce Reinhardt, Rudolph Rosen, Richard Rugen, Bob Schallmann, James R. Sedell, Bobbie A. Stephenson, Steve Thompson, Robert M. Timm, Peter Torrey, Phyllis Trabold, Thomas S. Yager

Colorado
Arthur W. Allen, Carol Beidleman, Gary L. Belew, John Blankenship, Richard L. Bruggers, Janice Carpenter, Len H. Carpenter, Richard D. Curnow, Matt Dunfee, Michael Dunning, Thomas J. DeLiberto, Peter Dratch, Michael W. Fall, Jennifer Gaines, Paul E. Gertler, Dennis Haddow, Aran Johnson, Mead Klavetter, David Klute, James A. Lawrence, Carol A. Lively, Kenneth A. Logan, Bruce McCloskey, Robert G. McLean, Pat Mehlhop, Loyal Mehrhoff, Brian Mihlbachler, Ralph Morgenweck, Pete Morton, John W. Mumma, Larry Nelson, Sandy Nelson, Dan Nolan, Eric Nuse, Eileen Regan, Randy Robinette, Stan Rogers, Dave Sharp, Greg Speights, Richard Storick, Gene Stout, Paul J. Sweeney, Jeff Trousil, Jeffrey M. Ver Steeg, Melanie Woolever, Michael V. Worthen

Connecticut
Cyndi Dalena, Edward C. Parker, R. Richard Patterson, Sharon Rushton

Delaware
Patrick J. Emory, Eugene Greg Moore, Bill Whitman

District of Columbia
Kevin Adams, Dan Ashe, John Baughman, Sally L. Benjamin, John Berry, Brad Bortner, Marc Bosch, Dale N. Bosworth, Gordon Brown, Jack C. Capp, David Chadwick, Kathleen Clarke, William H. Clay, Vaughn T. Collins, Sean Cosgrove, Jeff Crane, Dave Cross, Drue DeBerry, Naomi Edelson, Jacob Faibisch, Nina Fascione, Dwight Fielder, Jeffrey M. Fleming, Luke Forrest, Gary D. Frazer, James Gladen, Debbie Hahn, Bill Hartwig, Polly E. Hays, Tami Heilemann, Evan M. Hirsche, Matt Hogan, Lorraine Howerton, Bengt "Skip" Hyberg, Chris Iverson, Donna M. Janisch, Marshall Jones, Jim Kenna, Daniel E. Kugler, Jeff Lerner, Fran Mainella, Jina Mariani, Tom Melius, Martin Mendoza, Jr., Phil Million, Jen Mock, Lanny Moore, Robert Naney, Angela Nelson, Kelly Niland, Maribeth Oakes, Peggy Olwell, Ira F. Palmer, Mamie A. Parker, Clint Riley, Paul Schmidt, Eric Schwaab, Ken Schwartz, Anna Seidman, Bart Semcer, Edward H.

Shepard, Melissa Simpson, Len Singel, Liz Skipper, Allan T. Smith, David P. Smith, Mike Soukup, Fred Stabler, Casey Stemler, Scott A. Sutherland, Gary Taylor, Jim Terry, Tom L. Thompson, Samara Trusso, Leonard Ugarenko, Fred Wahl, David L. Walker, William A. Wall, Greg Watson, Rebecca W. Watson, Steve Williams, Tad Williams, James Williamson III, Paul Wilson, Mike Yost

Florida

Edward Barham, Jimmie Bartee, John W. Bridges, Martha M. Carroll, Chris Chaffin, Angy L. Chambers, Kenneth Conley, Nat B. Frazer, Julie L. Jones, Marian J. Lichtler, Steve Lowrimore, Elizabeth Martin, Jack E. Mobley, Frank Montalbano, Daniel Nichols, Kevin Robertson, Randall D. Rowland, Rob Southwick, Jeff Sprinkmann, Dennis Teague

Georgia

Albert E. Bivings, Tom Darden, Robert Drumm, John Fischer, Ernie Garcia, Sam D. Hamilton, Noel Holcomb, Robert T. Jacobs, Gregory W. Lee, Michael Riegert, John Saccomanno, Ron Smith, James M. Sweeney, Peter K. Swiderek, Charles E. White

Hawaii

Paul J. Conry, Randy Miyashiro, Timothy Sutterfield

Idaho

Jon Bart, Steve Barton, K Lynn Bennett, Susan Bernatas, Michele Beucler, Frances Cassirer, Jim Clark, Vicki Clark, Dan Davis, Russell L. Davis, Mark Elsbree, Nicky Elsbree, Kevin Frailey, Scott Gamo, Susan P. Giannettino, Jeff Gould, Joseph C. Greenley, James Haggengruber, Jerome Hansen, Tom Hemker, Ray Hennekey, Mark Hilliard, Steve Huffaker, Bill Hutchinson, Corey Kallstrom, Brian J. Kernohan, Joe Kraayenbrink, Gary Machlis, Terry Mansfield, Allen May, Cal McCluskey, Marjorie McHenry, George Pauley, James Peek, Paul J. Pence, Jon Rachael, Karen Rice, Terry Rich, Scott R. Robinson, Jeff Rohlman, Mike Scott, Gregg Servheen, Jill Silvey, Kathrine Strickler, Mark Taylor, Mike C. Todd, Al Van Vooren, Jim White, Michael Whitfield, Pete Zager

Illinois

Jay Alexander, Joel Brunsvold, Robert A. Clevenstine, David Delaney, Michael L. Denight, Dick Gebhart, Janet Gebhart, Steve Hodapp, Joe Jordan, Steve

Mowbray, Larry Pater, Don Pitts, Gary E. Potts, Andreas Rodriguez, Eric Schenck, Jay Truty, Christopher Whalen

Indiana
Gary Armstrong, David Case, Todd Eubank, Monica Hardy, Art Howard, Ronald E. Moore, John Olson, Mark Reiter, Glen Salmon, Brad Schneck, Phil Seng, Gwen White

Iowa
Louis Best, Judy Bishop, Richard A. Bishop, Marion Conover, Dale L. Garner, Joseph J. Haffner, J. Michael Kelly, Terry W. Little, David Otis, Jeffrey Vonk

Kansas
Ken Brunson, Kevin Jones, Joe Kramer, Ron Little, Mike Mitchener, Doug Nygren, Anice M. Robel, Robert J. Robel, Keith Sexson, Gibran M. Suleiman, Amy Thornton, Roger Wells

Kentucky
Tom Bennett, Richard A. Fischer, Jonathan Gassett, David S. Maehr

Louisiana
Philip E. Bowman, Mark Gates, Gerald Grau, John J. Jackson III, Edmond C. Mouton, Delia Nunez

Maine
Charles D. Duncan, Gino Giumarro, Col. Tim Peabody

Manitoba
Michael G. Anderson, Rick Baydack, Bob Carmichael, Dale Caswell, Jonathan Scarth, Merlin W. Shoesmith

Maryland
Paul J. Baicich, Jim Bailey, L. Peter Boice, Betty Boyland, Caitlin Burke, Julia K. Burzon, Helene Cleveland, Tom Franklin, David Goad, Paul W. Hansen, Joe Hautzenroder, Ronald R. Helinski, Marshall Howe, Judd A. Howell, Eric Lawton, Andrew Manale, Richard E. McCabe, Bette S. McKown, Jim Mosher,

Toni M. Patton-Williams, Tim Richardson, James D. Ripley, Jay Rubinoff, Celeste Ruth, Tom Sadler, Jane J. Sandt, Joshua L. Sandt, Bryan Seipp, Steve Sekscienski, Jerry Serie, Jacqueline C. Smith, Melissa Smith, William A. Spicer, Elizabeth L. Stallman, Kay Stratman, James Swift, George Teachman, Robert Wardwell, Todd C. Wills, Carolyn Woods, Thomas Wray II

Massachusetts
Wayne F. MacCallum, Sherry Morgan, Marvin Moriarty, John Organ, Tom Poole

Michigan
Mark Hirvonen, Robert D. Hoffman, Rebecca A. Humphries, Tina Yerkes

Minnesota
Tim Bremicker, John Christian, Steve Cordts, Joe Duggan, Doug Grann, John Guenther, Detta Jack, James Jack, Harvey K. Nelson, Dave Nomsen, Barbara J. Pardo, Jim Perry, Brett Richardson, Bill Stevens, Robyn Thorson, Howard Vincent, Stephen D. Wilds, Rick Young

Mississippi
Ken Babcock, Pamela Bailey, Scott Baker, Jim Copeland, Bob Karr, Bruce D. Leopold, Chester O. Martin, Ross Melinchuk, Doris J. Miller, James E. Miller, Michael F. Passmore, Dave Tazik

Missouri
Lorna H. Domke, Dave Erickson, Carole Evans, Ray Evans, Ken Gamble, Denise L. Garnier, Thomas F. Glueck, Dale D. Humburg, Carol Kragskow, Bill McGuire, John W. Smith, Liz Smith, Bill Renken, John H. Schulz, Dennis Steward, Ollie Torgerson, Bryant White, Daniel Zekor

Montana
Keith Ayne, Ed Bangs, Naomi Baucom, Dale M. Becker, Barney Benkelman, Timothy M. Bertram, George A. Bettas, Chuck Bowey, Stan Bradshaw, Don Childress, Allen Christophersen, Angela Daenzer, Peter J. Dart, Rick DiFiore, Doug Epperly, Dennis L. Flath, Lisa Flowers, Jeff Hagener, Jon Haufler, Mark Helweg, Jeff Herbert, Erin Inkley, Larry L. Irwin, Richard L. Jachowski, Pride Johnson, Eric Johnston, Abigail R. Kimbell, Skip Kowalski, Andrew J. Kroll, Lee

C. Lamb, David Ledford, Ron Marcoux, Michael McGrath, Mike McMahon, Don Merritt, Sterling Miller, Brendan Moynahan, Mike Mueller, Sunni Nabozney, Larry Peterman, Dan Pletscher, Stan Rauch, Jack Reneau, Susan Reneau, Laird Robinson, Ralph Rogers, William C. Ruediger, Gretchen Rupp, Rebecca Sharp, Alex Sienkiewicz, Gayle Sitter, Christian A. Smith, Ty D. Smucker, Mark Steinbach, Bob L. Summerfield, Jack Ward Thomas, Tom Toman, Dave Torell, Kate P. Walker, Ken E. Wall, John Waller, John P. Weigand, Gary J. Wolfe

Nebraska

Rex Amack, James Douglas, Kirk Nelson, Steve Riley, John Sidle

Nevada

Robert W. Buonamici, Sandy Canning, Kelly Clark, Terry Wayne Cloutier, Terry R. Crawforth, Robert J. Turner

New Brunswick

Keith McAloney

New Hampshire

Stephen John Najjar, Lee E. Perry, Judy Stokes, Steven J. Weber, Scot J. Williamson

New Jersey

Richard L. De Feo, Lewis E. Gorman III, John Joyce, Martin J. McHugh

New Mexico

Chuck Bartlebaugh, Don G. DeLorenzo, Lisa Evans, Bill Ferranti, Nancy Glowman, Dale Hall, Joyce Johnson, Junior D. Kerns, Gary A. Littauer, H. Stevan Logsdon, David Mehlman, Ruth Musgrave, Jim Ramakka, Luke Shelby, John W. Sigler, John P. Taylor, Daisan E. Taylor-Glass, Bruce C. Thompson, Gail Tunberg

New York

Gerald A. Barnhart, James A. Beemer, Tommy L. Brown, Daniel J. Decker, Donald P. Finamore, Joan Finamore, Christopher C. Pray, Raymond Rainbolt, Mark L. Shaffer, Leonard Vallender

North Dakota

Steve Adair, Jay Hestbeck, Nann Hestbeck, Dean Hildebrand, Michael A. Johnson, Randy Kreil, Greg Link, Jeff Nelson, Jim Ringelman, Kenneth Sambor, Ron Stromstad, Genevieve Thompson, Keith Trego

North Carolina

Scott Bebb, Charles S. Brown, Janet Bryant, Michael R. Bryant, David Cobb, Elizabeth J. Evans, Bryan R. Henderson, Francine Long, Matthew E. Long, John Rogers, John R. Townson

Nova Scotia

Mike O'Brien

Ohio

Carolyn Caldwell, Tony Celebrezze, Steve Gray, Gary Isbell, Roy Kroll, Julie McQuade, Thelma Peterle, Tony J. Peterle, Dave Risley, Pat Ruble, David P. Scott, Rob Sexton, Kendra Wecker, Jim Wentz

Oklahoma

James E. Bellon, Richard Hatcher, Toni M. Hodgkins, Jeff Howard, Raymond W. Moody, Alan D. Peoples, Glen Wampler

Ontario

Rob Cahill, Brigitte Collins, James Gibb, Trevor Swerdfager, John Williamson

Oregon

Alan Ager, John Alexander, David B. Allen, Robert Alvarado, Robert Anthony, Brad Bales, Jennifer Boyd, Larry R. Bright, Elaine Brong, George R. Buckner, Thomas J. Carlson, Norm Cimon, Cilla Coe, Rachel Cook, John Cook, Nancy Curry, Robert P. Davison, Larry De Bates, William Daniel Edge, David Filippi, Scott Findholt, Erik K. Fritzell, Michael Haske, Keith Hay, David Heller, Susan Holtzman, Bruce Johnson, Cal Joyner, John Kie, Gail McEwen, Estyn R. Mead, Steve Mealey, Holly Michael, Jim Noyes, Paul Phillips, Tom Quigley, Henry M. Reeves, Merilyn B. Reeves, Terry Robot, Mary Rowland, Hal Salwasser, Susan Sawyer, Joan Seevers, Robert Trost, Martin Vavra, Sara E. Vickerman, David Wesley, Michael Wisdom

Pennsylvania

Bob Boyd, Gary J. Crossley, Michael A. Dubaich, Calvin W. DuBrock, John Dunn, J. W. Gibson, Mary Jo Gibson, Joseph Hovis, Scott R. Klinger, Dave B. Messics

Quebec

Luc Belanber, Danielle Bridgett, Christiane Hudon, Barbara Robinson, Raymond Sarrazin, Steve Wendt

South Carolina

Buddy Baker, Breck Carmichael, Emily Cope, Dennis Daniel, Drenia Frampton, John E. Frampton, Robert G. Hotchkiss, James Earl Kennamer, Ron Kinlaw, Mark Mohr, Laurel Moore-Barnhill, Betty Jean Osgood, Tammy Sapp, John R. Sweeney, Kirk Thomas, Ben Wigley, James Alvin Wright

South Dakota

John Cooper, Doug Hansen, John Morgenstern, Charles G. Scalet, Art Smith, George Vandel

Tennessee

Bruce Batt, Brooks Garland, Amy Henry, Larry C. Marcum, Gary T. Myers, Greg Wathen, Alan Wentz, Scott C. Yaich, Don Young

Texas

Mary Anderson, Mike Berger, Vernon Bevill, Clint Boal, Bob Brown, Kirby Brown, Linda Campbell, Rafael Corral, D. Lynn Drawe, Kay Drawe, Russel T. Farringer, Daniel D. Friese, Ronnie R. George, Selma Glasscock, Lee A. Graham, Dennis M. Herbert, John S. Herron, Jack Hill, Lynne Lange, Raquel I. Leyva, Richard E. Moore, Dave Morrison, Charles E. Pekins, Kevin G. Porteck, Nova J. Silvy, Bob Spain, Michael Tewes, Anna Toness, Kim Vaughn

Utah

Marcus W. Blood, Dwight Bunnell, Alan Clark, Jim Cole, Mike Conover, Rick Danvir, Brian Ferebee, Barrie K. Gilbert, Richard Griffiths, Frank Howe, Bill LeVere, Julie Moretti, Miles Moretti, Don S. Paul, Jack M. Payne, Steve Plunkett, Jack Troyer

Vermont
Stephen Hill, Ronald J. Regan

Virginia
Terry L. Bashore, Pamela M. Behm, Bob Blohm, Hannibal Bolton, Robert L. Byrne, Thomas Busiahn, SA David Carey, Gabriela Chavarria, Mark Duda, Chris Eberly, Robert W. Ellis, Jim Fleming, Dorothy Gibb, Anne Glick, Jim Greer, Charles G. Groat, Susan D. Haseltine, Tim Hess, Heidi Hirsh, Brian Hostetter, Stephanie Hussey, Mark Indseth, Doug Inkley, Gary Kania, Rick Kearney, Mary Klein, SA Rick Klink, James W. Kurth, Johanna Laderman, Genevieve Pullis LaRouche, Pam Matthes, Bruce Matthews, Janet McAninch, Jay McAninch, Steven J. McCormick, Steve L. McMullin, Bruce Menzel, Mark E. Mobius, Vivian Nolan, Jim Omans, Donald J. Orth, David Pashley, Carol J. Peddicord, Bill Poole, Jim Preacher, Jennifer Rahm, Susan Recce, Tom Reed, Kathryn B. Reis, Gordon Robertson, I. Teiko Saito, Robin Schrock, David A. Smith, Gregory J. Smith, Billy Templeton, Beatrice Van Horne, Meegan Wallace, Ken Williams, Dave Woodson

Washington
Judith-Ann H. Agan, Carolyn Alfano, John Andrews, Len Barson, Rocky Beach, Ken Berg, Mike Bireley, Rance R. Block, Jenevieve Borin, Nathan Boyer, Patrick Brauer, Rick Brazell, Ben Brigham, Dave Brittell, Melissa L. Brown, Joseph K. Buesing, John D. Buffington, Pam Camp, George R. Carlson, Kye Carpenter, Rod Clausnitzer, Will Clegern, Tina Coley, Jeremiah Collins, Mick Cope, Hilary Cruickshank, Dennis D. Dauble, Chase Davis, David Douglas, Sarah Ekerholm, Valerie Elliott, Tyrell Erlebach, Bob Everitt, Scott Fisher, William Frymire, Cara Gardner, Ron Gayman, Brian Gilbert, Barbara Graham, Brom Granger, Mark Halupczok, Kellie Halverson, Shalen Hamar, Lacie Hearst, Hans Helmstetler, Sara Hornor, Scott Horton, Joe Jertberg, Gerald T. Johnson, Robert L. Johnson, Lacey E. Jones, Lacey K. Jones, Jennifer Kindred, Anne Kinsinger, Jeff Koenings, Matthew W. Klope, Megan Kranenburg, Rich Landers, Matthew J. Landt, Don Larsen, Joe La Tourrette, Amy P. Lawrence, Cora Lininger, Tamar Loeffler, Carolyn Lovano, Nicholas Lovrich, Stuart N. Luttich, William MacDonald, John Mankowski, Susan Martin, Bob McCready, Ryan McDonald, Meghan McGarry, Jim McGowan, John A. Miller, Mark G. Miller, Stan Moberly, Crystal Montoya, Amie Morrison, Sandra L. Mosconi, Lisa

Moss, Bob Nelson, Jerry Nelson, Jim O'Donnell, John R. Phillips, Mark Plummer, Steve Pozzanghera, Cavin Richie, Marshall C. Richmond, Stephanie Ridgway, Steve Robinson, James "Rusty" Ruchert, Mike Rule, Sandra Rule, Erin Rose Rundquist, Ronnie Sanchez, Mike Sardinia, Glenn E. Schmitt, Michael A. Schroeder, Sarah Shogren, Carey Smith, Rachel Sparks, Rick Spaulding, Sandra Staples-Bortner, Dale Swedberg, Greg Sweeney, Todd D. Thompson, Adam Vanter, Matthew Westman, Richad Whitney, Karen Zirkle

West Virginia

Paul R. Johansen, Suzette Kimball, John R. Lemon, Randy Rutan, Curtis I. Taylor

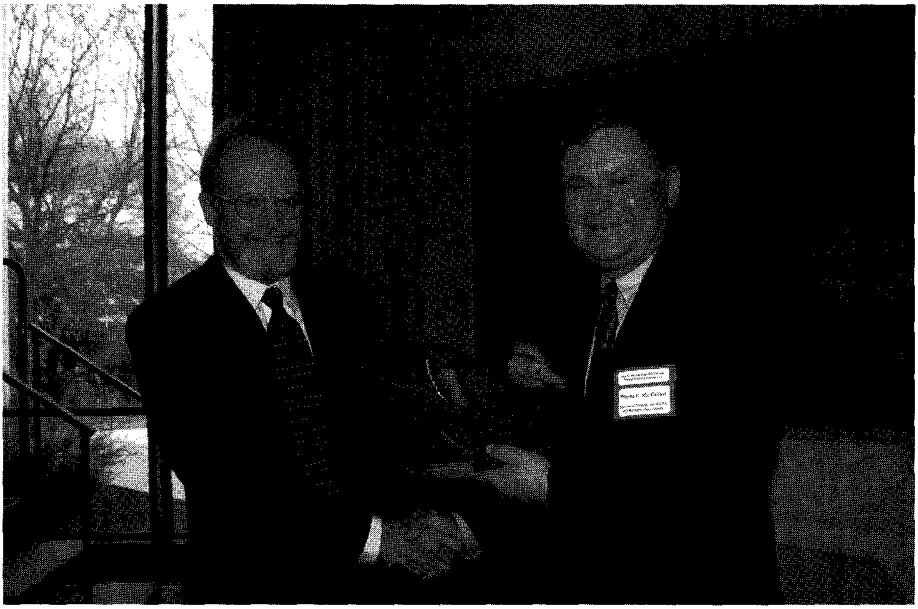
Wisconsin

Kimberly Anderson, Jimmy S. Christenson, Dan Dessecker, Leslie A. Dierauf, Don Gonnering, Tom Hauge, Robert H. Holsman, Diane Lueck, Butch Marita, J. Kim Mello, Thomas Niebauer, Christopher D. Risbrudt, Paul I. V. Strong, Darrel VanderZee, Leonard H. Wurman

Wyoming

Joseph Bohne, Larry D. Hayden-Wing, Ken Jones, John Kennedy, Larry L. Kruckenberg, Jay Lawson, Ray Lee, Fred Lindzey, Robert Model, Hall Sawyer, Mandy Scott, Bettina Sparrowe, Rollin D. Sparrowe, Bill Wichers

Wayne F. McCallum Receives Distinguished Service Award



Rollin D. Sparrowe, president McCallum, director of the Massachusetts Distinguished Service Award

Wayne F. McCallum, Director of the Massachusetts Division of Fisheries and Wildlife, received the Wildlife Management Institute.s 2004 Distinguished Service Award at a special ceremony on March 17, during the 69th North American Wildlife and Natural Resources Conference in Spokane, Washington. This international award recognizes individuals who have made extraordinary and enduring, but largely unsung contributions to natural resources conservation.

For more than 30 years, MacCallum has dedicated himself to natural resources conservation in both the public and private sectors. As director of the Massachusetts agency since 1988, his professionalism, integrity, and intellect have dramatically improved leadership and resource management within the agency, evolving to improvements on other state, regional and national levels.

"This tradition of willing leadership fulfills our most highly regarded criteria for distinguished service," said Wildlife Management Institute president Rollin D. Sparrowe. "Wayne has a long history of never saying no to tackling state or national challenges in conservation. He has met resource demands and stakeholder expectations in Massachusetts, one of the most contentious forums for wildlife management in the country. He has dealt successfully with strident antihunting, antitrapping and antimanagement interest groups. He has stood strong in one of the most bare-knuckled political climates of any state. Through it all, Wayne MacCallum has ensured that traditional hunting and fishing programs are founded on a solid base of cutting edge science and public support.

"As well as being Division of Fisheries and Wildlife director, MacCallum served as president of both the Northeast Fish and Wildlife Directors Association and the International Association of Fish and Wildlife Agencies (IAFWA). He has been a member of the North American Wetlands Council, Sea Duck Joint Venture and the IAFWA task force on waterfowl regulations. He served as chair of the Atlantic Flyway Council, Atlantic Coast Joint Venture and Woodcock Task Force.

"We celebrate Wayne for his high professionalism over the years, but also for his ability to work warmly with people, no matter their stance on issues he holds dear to his heart,. said Dr. Sparrowe. .Wayne has a wonderful ability, at the end of the day or at a critical juncture, to make the impersonal personal and the tedious enjoyable through his marvelous humor and personal interest in those he meets. It is that quality that has made him a great and effective leader, and most deserving of the Wildlife Management Institute.s Distinguished Service Award."

Illinois River Valley Habitat Initiative Receives Touchstone Award

The Illinois River Valley Habitat Initiative Receives the Touchstone Award (from left: Eric Schenck, Steve Mowbry, Rollin D. Sparrowe and Joel Brunsvold).

The Illinois River Valley Habitat Initiative received the Wildlife Management Institute.s (WMI) coveted Touchstone Award for 2004. This award recognizes persons, groups or agencies involved in professional natural resources management, whose ingenuity and initiative result in a program or product that notably advances sound resource management and conservation in North America. The award was conferred on March 19, during the 69th North American Wildlife and Natural Resources Conference in Spokane, Washington.

The Illinois River Valley Habitat Initiative~~-a~~-cooperative venture uniting Caterpillar, Inc., Ducks Unlimited, Inc. (DU), Illinois Department of Natural Resources and a variety of businesses and citizens-~~was~~ recognized for its diligence and creativity in developing more than 170 wetlands, as well as improving and maintaining thousands of acres of habitat in the Illinois River Valley since 2001.

Mike Hitchcock and Jay Alexander conceived the notion of using available resources to improve and enhance waterfowl habitat in the IllinoisRiver Valley, a vitally important segment of the Mississippi Flyway migration corridor. Mike, owner of Hitchcock Scrap Yard, and Jay, then the manager of building equipment for Caterpillar, Inc., from Canton and Washington, Illinois, respectively, recognized that habitat improvement work was constrained by lack of heavy equipment available for the necessary work. Caterpillar, Inc., already with a strong record of support of conservation, was an answer to that constraint.

Caterpillar administrators Doug Oberhelman and Kurt Tisdale gave the thumbs up, and Steve Mowbray of Altorfer Rents in East Peoria, provided the machinery. Next, Mike and Jay went to Eric Schenck, a DU biologist in Peoria, to involve DU as advisor. Eric, in turn, contacted Jim Modglin and Rick Messinger, regional land managers for the Illinois Department of Natural Resources (DNR), to identify sites from the backlog of wetlands needing attention.

It was proposed that the DNR would do the needed work they could not otherwise afford, and Caterpillar equipment would be used. Significant expenses were involved, so DU organized a matching grant to cover costs. Caterpillar and Mike Hitchcock donated much of the cash that DU matched, exceeding $70,000.

In late 2001, seven DNR areas were beneficiaries of the cooperative venture. That year, the equipment and funds enabled creation of 20 new wetland acres, improvement of 200 acres of degraded habitat, and needed maintenance of about 5,000 acres. Since then, approximately 150 new wetland acres have been created or reclaimed, 400 acres have been improved and maintenance has been done on nearly 10,000 acres. Joel Brunsvold of the DNR has coordinated the work.

"The Wildlife Management Institute salutes all the players of the Illinois River Valley Habitat Initiative, not because of the magnitude of wetlands enhanced, but for parlaying a simple, good idea into reality," said Rollin D. Sparrowe, WMI president. "Simple, good ideas often are most difficult to achieve and most often shelved. This idea succeeded because it was championed by individuals, businesses, a conservation organization and a receptive, progressive natural resource management agency. We thank all members of the Illinois River Valley Habitat Initiative—those we have recognized here, as well as other heroes behind the scenes. All have been dedicated to cooperative accomplishment for the sake of conservation."