ISSN 0078-1355

TRANSACTIONS
of the
Fifty-ninth North American
Wildlife and Natural Resources
Conference

Conference theme:
International Partnerships for Fish and Wildlife

March 18–23, 1994
William A. Egan Civic and Convention Center
Anchorage, Alaska

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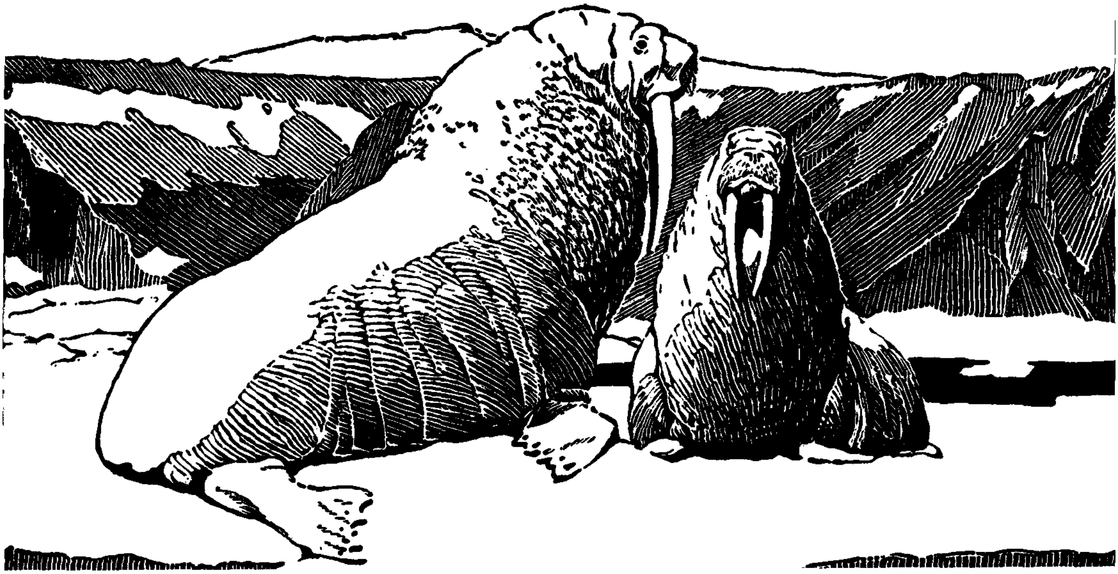

Published by the **Wildlife Management Institute**, Washington, D.C., 1994

Additional copies may be procured from the
WILDLIFE MANAGEMENT INSTITUTE
1101 14th Street, N.W., Suite 801
Washington, D.C. 20005

Composition by **AlphaTechnologies/mps**
Printing by **BookCrafters**

Transactions of the
59th
North American Wildlife and Natural Resources Conference
ISSN 0078–1355

Printed in U.S.A.

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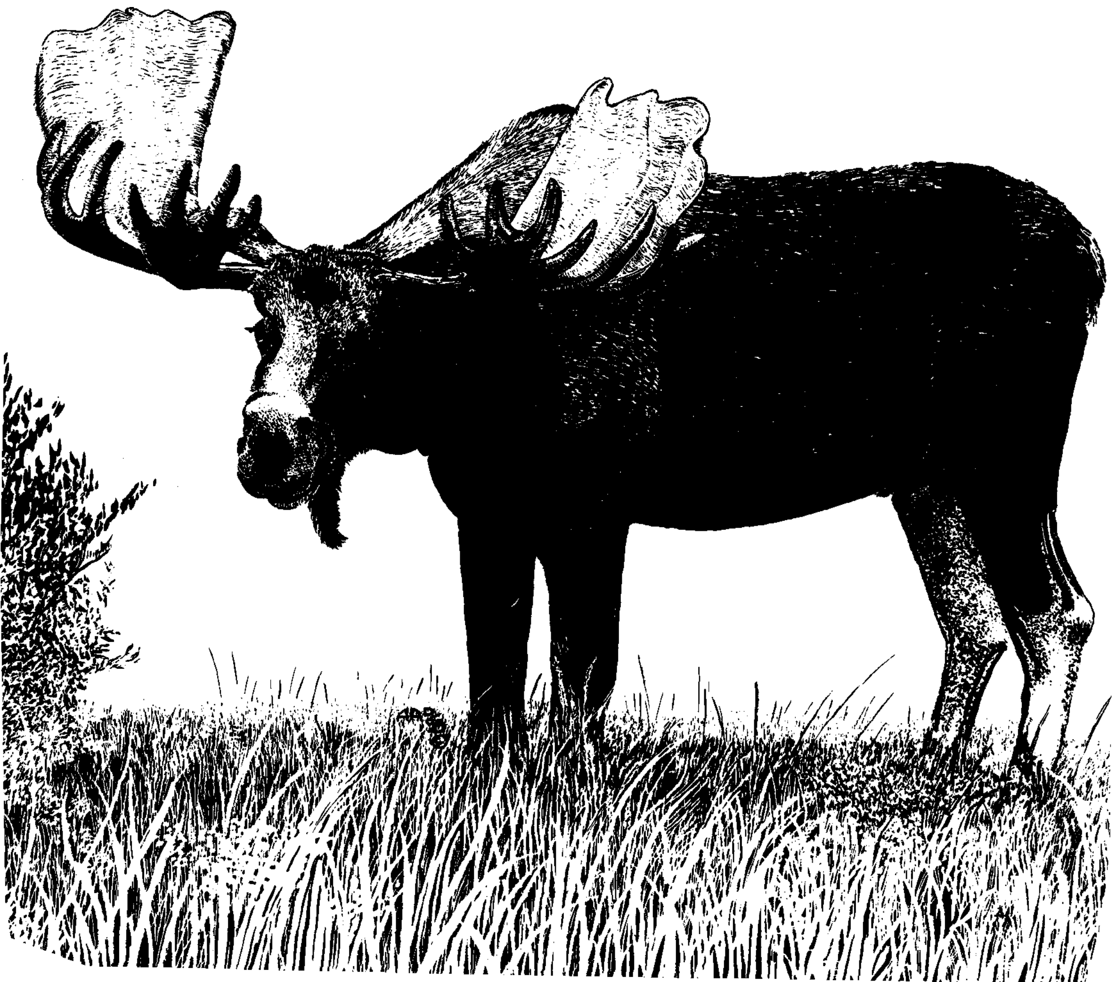

Opening Session. *Northern Lights and Northern Exposures*

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JEROME B. KOMISAR
University of Alaska-Fairbanks
Fairbanks, Alaska

Cochair
JERRY M. CONLEY
International Association of Fish and Wildlife Agencies and
Idaho Fish and Game Department
Boise, Idaho

Opening Statement

Rollin D. Sparrow
Wildlife Management Institute
Washington, D.C.

Welcome to the 59th North American Wildlife and Natural Resource Conference. The Conference theme, "International Partnerships for Fish and Wildlife," includes a focus on the North Pacific, and we are happy to see colleagues from Canada, Russia, Mexico, Japan and other countries here to participate. Many of you, as usual, have been here for several days and are deeply involved in committee meetings and other activities at which much of the real business of this Conference is conducted.

Special Sessions to follow will look at management concerns for internationally shared resources of migratory birds, mammals and fish. They will focus on genetic impacts of hatcheries, on wildlife population estimation, and on integrating traditional fish and wildlife management with the magical concepts of biological diversity and ecosystem management. Overall, sessions respond to the rapid changes occurring in society, the natural resources management agencies, and the attitudes, demands and ethical perspectives of resource users, managers themselves and the public.

The issue of partnerships resounds throughout government, domestically and internationally, in all our countries. Partnerships have been with us for decades and have achieved many things. Examples are the migratory bird treaties which include Canada, Japan, Mexico, Russia and the United States; various fishery management councils; Flyway Councils; an international caribou committee; the North American Waterfowl Management Plan and its joint ventures; an international shorebird reserve network; and many more. Existing partnerships focus on many species, habitats, and both private and public lands.

It is important to focus on what a partnership is and what it is not. In simple terms, "partnership" means coming together to share thinking, planning and resources to achieve common goals. Each participant often must sacrifice a little of its interests to achieve success and capitalize on the strengths of the overall effort. Some partners bring money; others contribute technical skills; some offer the land on which the work is done; and others provide political and other support. A successful partnership does not involve making decisions about new programs and *then* asking others for their input, support and understanding. Federal programs that originate from the top down without input by those whom they affect consistently are targets of discontent. Somehow this seems to haunt every new administration in Washington.

At last year's Conference, considerable attention focused on formation of the National Biological Survey. A main concern by long-standing partners with the Department of the Interior was that programs would be changed, essential functions could disappear, and the interests of the partners might not be taken into account in future development of dollars and programs. A National Research Council report recommended a broader concept of a National Biological Survey Partnership. To some extent, Interior seems to have adopted this model, but so far, it is mostly a limited partnership. Selective entities are involved in specific project planning, but the broad promises for outside coordination made subsequent to last year's Conference have not been kept.

Lack of focused leadership, bureaucracy, and most of all an apparent indifference to many long-standing partners and their needs still exist. The messages from within Interior are that senior employees in cooperative research units and research laboratories that make up the bulk of the staff that was transferred to the NBS largely are excluded from planning for the new agency, and from the designs for research and other programs to be done with its limited dollars.

Many among the Department of Interior's traditional constituency may find it hard to support NBS budget initiatives because so little is known about them. It is not clear how the research needs of agencies like the U.S. Fish and Wildlife Service will continue to be met. The undesirable separation of parts of the migratory bird management function from the Fish and Wildlife Service has disjointed long-standing international partnerships in processing data and managing resources. We had a useful dialogue opened before the Conference about the future of Cooperative Fish and Wildlife Units. More is needed. What seems to be holding things together is the personal relationships of employees now in separate agencies. We in the natural resources management community cannot afford to allow core capabilities in research to be lost by diffusion into as yet undefined programs. If the prevailing lack of communication, indecisiveness and failure to involve its own senior staff continue, the National Biological Survey surely will fail to meet its professed purpose.

Federal government agencies are being "reinvented" before our very eyes. Changes in approaches to budgeting, accountibilty, organizational structure and even in the basic parameters of agency goals are being expressed in new terms. While the ostensible purpose is to deliver better services to the public, realistic capability of such delivery remains unclear.

The USDA Forest Service has reached out to various partners to discuss how to approach "reinventing the Forest Service." Informal advice from such partners suggested key principles, such as focusing on the mission of the agency rather than administration and process, adopting performance measures for resource stewardship, emphasizing the agency's unique strengths, and diversifying its leadership. The Forest Service also was cautioned to avoid prompting change for the sake of change, not to lose sight of the "pieces" of the agency while trying to manage the whole of it, to communicate with and involve the public in new ways, and to build on partnership successes. Other agencies would do well to reach out to their partners as frequently as the Forest Service does. Perhaps the best advice throughout government, whether dealing with the general public or its various partners, is to involve people by asking them what they think and be responsive to what is said. Listening is a true art that can be expressed best by changing what is done on the basis of what is learned.

All has not looked positive at every stage in the reinvention of the Forest Service. Along with many agencies, the Service's budget categories are being homogenized into fewer and more general listings. It is unclear yet how accountability will be provided for expenditures on fishery or wildlife work, but there is a dialogue underway. Support by outside organizations to the Congress netted Forest Service fish and wildlife programs a more than 400-percent increase over the last decade. Ironically now, proposed reorganization under Ecosystem Management threatens visibility of supportable programs just as timber and road building categories did in the past. There is a clear lesson in this for the Departments of Agriculture and Interior. Accountability is important, whether it be the Forest Service or National Biological Survey or any other agency. If a constituency can't track what is done, budgets may not be supported and lack of funding may limit progress.

I mentioned earlier the magic concepts of biological diversity and ecosystem management. Because they are front and center with federal, provincial and state agencies, they deserve attention. This Conference has focused on the concepts and the need for practicality in their application. Biological diversity is important, but expecting the public to buy into such an esoteric and often vague topic as a goal for all the endeavors of life seems a bit naive. People still will view natural resources in terms of utility for the economy, for recreation and for personal uses. Another way to say this is that people will relate to what they understand, and they will demonstrate that understanding by paying for and supporting things they believe in. While we pursue globally stated goals for biological diversity, let's not forget how to relate to people.

Ecosystem management is expressed widely as the direction resource management is going from top to bottom. It is clear that our federal agencies are being directed from the Administration to shift to this poorly defined goal. Using common sense, ecosystem management generally means stepping back and looking at the whole forest, refuge, or park and its surroundings, and focusing on its functions as a basis for management. At a baseline level, this isn't magic, and it is supported by resource managers.

Ecosystem management in this context is not a new concept to fish or wildlife biologists in the federal, provincial or state fish and wildlife agencies. It is the concept many believe was a foundation for "multiple use"—even "wise use"—before those terms became negatively redefined through irresponsible application. Many experienced resource managers are concerned about the pace and fervor with which ecosystem management is being thrust upon agencies and the public. Like it or not, the concept often comes forward as if its advocates have a new idea, and they are going to sweep aside folks who have been doing things of lesser importance to do great new stuff. While that may not be the intention or the intended perception, it does not foster partnership, and projects an arrogance that will lose partners along the way.

There are many examples of practical advances toward ecosystem management. Here in Alaska, the large-scale lands assigned to national wildlife refuges, for example, satisfy one of the first premises of ecosystem management. Alaskan national parks and national forests are of the same magnitude. Enabling laws provide for development of baseline data on refuges, so that management can proceed with better knowledge of the resources present and how they function in an ecosystem. Hands-on management may be less needed in Alaska than elsewhere, but the point is that the building blocks are here for ecosystem management as a scale of application. Outside Alaska, the national wildlife refuges, national forests and national parks are beset by land-development pressures, watershed degradation, limits to the range of wildlife species and many human pressures.

In the Western United States, public lands managed by federal agencies are the model for thinking and policy. Managing single, public landownership on an ecosystem scale is easy to envision. On the Great Plains for grasslands, wheat or cornfields, or in eastern deciduous forests where landownerships shift more dramatically, private lands are the main resource base. Ecosystem management there must deal with altered systems and with private owners and different agendas that they may have for their landscape parcels. Much of the rhetoric about ecosystem management and biological diversity seems to center on so-called "natural systems," ignoring the essential role of the huge portion of North America that is farmed, grazed, logged and otherwise used by people. This is why agricultural policy is so vital to wildlife and fish in much of North America.

A cornerstone of land conservation in the United States for wildlife and fish has been the 1985 and 1990 Farm Acts. The Conservation Reserve Program and more recently the Wetland Reserve Program have combined the needs of the agricultural and conservation communities to produce wide-ranging soil erosion, water quality and wildlife benefits. Many at this Conference have worked to implement these programs and recognize their value. We invite those with "new" visions of the need for preserving ecosystems and biological diversity to work with conservationists to make the 1995 Farm Act even more valuable to achieving specified conservation goals.

National forests are moving in practical ways toward ecosystem management by considering watersheds as components for timber harvest, limiting cutting along streams, revising grazing programs, reducing open roads and managing for threatened and endangered species. Plans are being devised for timber harvest, burning, thinning and other practices with a specific desired future condition identified on visible maps where one can judge progress toward the goal. The public has a chance to have more say about this potential future condition as new forest plans are drawn up. While there are some who will advocate no management at all, the new approach looks like real progress to our Institute on several national forests.

The 12 joint ventures of the North America Waterfowl Management Plan all have management goals that far exceed designs to restore waterfowl. This is because the partners in these joint ventures have shared their mutual visions of what needs to be done to achieve each of their objectives. Those logically extend to restoration of wetlands, surroundings uplands, entire watersheds, and combinations of many different types of land protection or management through acquisition, easement and various cooperative agreements. Implementation of programs through the North American Wetlands Conservation Act has brought partners together in Canada, the United States and Mexico to secure and manage diverse wetlands and associated uplands.

National wildlife refuges and national parks both are looking outside their respective boundaries to identify needs on publicly and privately owned lands to achieve the original purpose for which parks and refuges were set aside. Migratory birds, anadromous fish and wide-ranging animals such as bears all need a greater landscape than was the vision when these conservation units first were formed. Since surrounding lands often are privately owned, expanded land management must be done with great sensitivity to private as well as public interests.

The Forest Service and Bureau of Land Management work with grazing and timber interests on adjacent private lands, just as the Fish and Wildlife Service has reached off its refuges to work with private landholders. From small initiatives adjacent to a refuge, to the Greater Yellowstone ecosystem concept, which includes national parks and all other categories of land ownership, there is a tremendous amount of energy already going in the direction that ecosystem management should take us. There may indeed be broader visions that are developing through time that should guide our future efforts. They must come about through building on what has been achieved to date, and recognizing on an equal basis the various motivations that bring people to the table to get things done. Otherwise, those who think they have a "new" cause and can ignore the views and needs of other players face the reality of looking around and finding little support.

All of this challenge requires some mutual vision. This is the greatest challenge to achieving ecosystem management. In general, the need for recognition of ongoing efforts and capitalizing on known partnerships is necessary because people will not support someone else's vision at the expense of their own. Yet, there are some who wish to force their ideas on others. So-called "reform" of the National Wildlife Refuge System is one example. The fact that the System was built with a multiplicity of objectives will not go away. Guiding principles for its management are needed, but they must recognize needs of a full range of supporters who are responsible for the establishment of the System as we know it. The ability of the various interests to focus on the real goal, namely improvement of the refuge system, is key to any reform legislation. Lack of operating funds, external threats and truly inappropriate uses that adversely affect wildlife can be a unifying focus. Most can agree that damaging the landscape for short-term gain or water skiing through bird production areas is inappropriate use. It is unfortunate that some have chosen to focus on hunting—an admittedly divisive issue—when it has not been highlighted as a primary problem by any of the recent refuge studies. This is the kind of thing that tears down consensus and strays from the main path of improving land management. It plays on the paranoia of embattled hunters, puts Congress in a difficult situation and just isn't the real problem that deserves our focus.

An ecosystem management challenge seems appropriate. Predator control continues to receive much adverse attention in North America. Public sensitivity to the issue, often fueled by specific interest groups, is highlighted by such things as the wolf issue here in Alaska and the wolf reintroduction issue in Yellowstone. A recent article in *Conservation Biology* focuses on writings in that journal that reflect an aversion to accepting the need for what is called the "nasty business" of controlling animals. Extensive data on North American waterfowl show that habitat modification for agriculture has dramatically modified waterfowl production habitat in the heartland of the United States and Canada. Combined with extended drought, the result is continentally depressed waterfowl populations. Likewise, recent data on population status of grassland-nesting songbirds indicate that these birds are the most rapidlly declining segment of the neotropical bird population. In fact, grasslands may be the most threatened ecosystem.

Simultaneously with these changes, the variety, range and abundance of more than a dozen predatory species, such as crows, gulls, skunks, foxes and ground squirrels, provide a powerful limiting influence on a truly magnificent, endemic and culturally important migratory component of North American fauna. Extensive data exist on the depressing impact on waterfowl of this imbalanced assemblage of predators throughout the prairies of North America. Nevertheless, public pressures based on emotion, and fanned by special interest groups that oppose killing of anything, have put resource agencies in a situation where they cannot include direct control measures to resolve these problems for fear of political reprisal. Reliance on changing habitat to resemble more closely earlier functional structure will help, but won't be fully successful without more direct attention to the predation problem. As we embrace ecosystem management, do we have the courage to face this with the biological facts and do something about it? The Nature Conservancy kills cowbirds to benefit black-capped vireos on an Oklahoma property—can similar measures be included as needed to restore prairie ecosystems in North America?

So why are we so confused about what ecosystem management means to existing programs? When one overlays the regional organizational structure of federal agencies—which all are different (except for Alaska) with state and provincial boundaries that control legal authorities, staff, money and lands—it becomes a complex picture. Another step in complexity includes county, city, community and individual landowners. How are ecosystems managed in this context?

Ecosystem management in the Forest Service certainly will expand beyond forest boundaries. BLM's huge landholdings are interspersed with a checkerboard of private holdings. The Forest Service and BLM recently announced that on the Eastside Forest initiative they will use "provinces." The Fish and Wildlife Service actively is considering watershed-based ecosystem management for the whole United States. The Nature Conservancy has had its "Last Great Places" identified nationally and internationally. The Partners in Flight program now is planning for neotropical birds down to physiographic regions. The North American Plan Joint Ventures are on a scale that crosses several of these jurisdictions. The Columbia Basin now is a major focus for ecosystem management, with stream quality and anadromous fish as an objective, but obvious larger implications to terrestrial systems. The Mississippi River, after last year's flooding, is being addressed similarly in an attempt to turn around flood-control policy directions. The "Wildlands Proposal," primarily for the United States, eventually would expand wilderness designation to half the country. Recently, the Sierra Club has identified 21 ecosystems in North America for its focus, as if it were a new idea.

If one were to look at all of this in context and be presented with the simple premise that "we are going to move toward ecosystem management," one might respond "enough already!" How do we get things done while work is going in so many directions? Who will lead and coordinate these efforts?

Hopefully, discussions at this Conference, the deliberations of agencies involved and wisdom in the application of science to management will lead us through this so that at future conferences we can simply talk about ecosystems, know exactly what we mean and have a full slate of truly involved partners at the same table.

Alaska's Unique Conservation Role

The Honorable Walter J. Hickel
Governor
State of Alaska
Anchorage

On behalf of all Alaskans, welcome to Alaska.

We appreciate that in a salute to our state you will focus today on "Northern Lights and Northern Exposures." Frankly, we have more fans in Alaska for northern lights than *Northern Exposure*. One is written in Hollywood and filmed in the State of Washington. The other is scripted in the heavens and performed right here across our fabulous northern sky.

The Need for Balance

I first spoke to this Conference in 1969 as U.S. Secretary of the Interior.

I have always liked the name of this Conference, now in its 59th year—"Wildlife and Natural Resources"—because it demonstrates balance. Without balance, we can win a battle now and then, but we will never win the war. We saw that in dramatic fashion at the Earth Summit in Rio de Janeiro.

I was asked to speak in Rio on the eve of the Summit alongside the Secretary General of that Conference, Maurice Strong. My theme was that the world's agenda cannot be addressed piecemeal. We must care about the total environment—people, people's needs and nature. I'm glad to report that that message got through. One U.S. environmentalist summed it up when she said, "Everyone came away from the Summit profoundly changed." In her words, delegates and observers alike realized that at the core of sustainable development lies "not just economics or pollution control, but equity and justice." Yes, it is finally hitting home.

We only will succeed in our efforts to care for nature, including international partnerships for fish and wildlife, when we care about people and their needs. The children of all nations need a chance to grow up healthy and free, and with an opportunity for a decent life. If that does not happen, we eventually will be overwhelmed.

But I believe it will happen. And I believe we will protect our wildlife and responsibly utilize our God-given natural resources. There is no other alternative.

To be realistic about the future, we must focus especially on the Pacific Rim. We must build partnerships here, the home of the great mass of the world's population, and the most powerful center of industrial strength. Like it or not, the developing nations are going to develop. They will insist on equity and justice. Let's help them do it right.

Alaska's Unique Conservation Role

How do we do that? And what is Alaska's role? Alaska's "unique conservation role" is not to wait, but to make that balance—stated in the name of your conference—a reality today. And that's exactly what we are all about. We are being successful because Alaskans are deeply committed. Fifty-three percent of Americans are involved in wildlife-related activities, from hunting and fishing to bird and wildlife watching. This is true of 93 percent of Alaskans. We have 130 million acres of National Parks and Wildlife Refuges in Alaska. That's 70 percent of all National Park acreage and 90 percent of all U.S. Wildlife Refuge lands. And we have another 11.5 million acres of state-owned parks and reserves. These reserves include Alaska's finest scenery and best wildlife habitat.

Alaska's Stewardship

Conservation requires more than preservation. It requires stewardship.

In 1959, after 50 years of struggle, and 16 failed bills in Congress, Alaska finally secured statehood. At that time, many of our wildlife resources were badly depleted. We have avoided a disaster through state management. There are almost twice as many caribou in Alaska—1 million—than there are people. And they provide a significant food resource to hundreds of villages. Our moose, mountain goats and deer are doing well. Alaska is unique in the United States in having healthy populations of large predators. We have more black and brown bears now than we did at Statehood. Our wolf population is larger than ever, and more widely distributed. In fact, it is flourishing. In some parts of the state, it is flourishing too well. We have 7,000 wolves. That's roughly 700 packs.

If you would like to help us with this over-population problem and take a wolf pack home with you please leave us your name and neighborhood.

Last year, in contrast to the other fishing grounds in the U.S., Alaska welcomed home a record run of fish, including 200 million salmon for starters. That's almost 10 times as many as the year we became a state (28 Million).

Our birds and waterfowl are doing well. The goose population declines on the Yukon-Kuskokwim Rivers have been reversed and now can support a sustainable harvest. The U.S. symbol, the American eagle, is abundant. And the Arctic peregrine falcon has made such a comeback it was removed last year from the endangered species list.

Some of today's environmental evangelists want to lock up Alaska. They don't trust local people. They want the federal government to have jurisdiction over fish and wildlife management.

Our experience is that it works much better the other way around. States should be the primary managers of fish and wildlife populations, especially in the Arctic and sub-Arctic. Local people know more about local species and local habitat, and care more. This traditional management role has worked for decades. It is respected in all other states, and Alaska should not be treated differently.

A Vision for Alaska

Alaska's role—a vision that most Alaskans share—is not just to take care of our own. We see ourselves as a model—a showcase—for other nations and ecosystems. The Arctic never will compete with the rest of the world for people. But the Arctic is rich with the resources people need.

Alaska produces 25 percent of our nation's oil, and many other products. Please don't think I'm boasting when I say that our oil development at the North Slope is the finest—not just in the Arctic world—but in the entire world. I urge you all to visit and to examine these pioneering marvels.

Learning from Our Mistakes

And we are learning from our mistakes.

After reassuming the governorship of Alaska in 1990, I set out to put the tragic 1989 *Exxon Valdez* oil spill behind us. Enlisting the support of the U.S. Justice Department, I negotiated a global legal settlement with Exxon Corporation for one billion dollars. I didn't want a repeat of the *AMOCO Cadiz* disaster of 1978 in which six times as much oil was spilled. The legal battles went on for 14 years. And the settlement was less than $300 million. Last report I've had, these funds were not yet released.

Since our settlement, however, which took us just 60 days to negotiate, we have begun to use that money to turn Prince William Sound into a living laboratory to study the long-term impacts of oil in our waters. And we have dedicated settlement funds to enhance the affected areas and purchase important habitat.

Other portions of that settlement will enhance our knowledge of the wildlife and ecosystems of the North. For instance, we will build a marine research and education center, on the scale of the Wood's Hole facility. It will be located in Seward, Alaska, on Resurrection Bay. Resurrection is the right word, and let me confirm that Prince William Sound is recovering rapidly, due to an all-out struggle by thousands of individuals and the remarkable capacity of Mother Nature to heal herself. This experience, technology and expertise are important for us.

The Northern Forum

But they also are important for our Arctic neighbors, such as the former Soviet Union, where environmental concern was ignored for decades. That nation is changing, and there is an open door we must not fail to enter. That's one of the reasons I have worked hard during the past three years to set up and make effective The Northern Forum. This organization, made up of the governors of 20 Arctic regions—including eight Russian regimes—is an ideal vehicle to help establish "International Partnerships for Fish and Wildlife." We currently have 13 established projects, including work on environmental research and monitoring, especially as it relates to nuclear waste dumping in the Arctic.

We are embarked on wildlife studies, human ecology, marine management and natural resource development in the North.

We do not subscribe to those who, out of fear, would lock up the Arctic and our people. Once again, we are stressing the total environment—people, people's needs and nature.

Our International Partnerships

Alaska has no national borders. All our borders are international.

Due to our size, geographic location and tremendous diversity of fish and wildlife habitat, we have to work internationally to protect our interests. Hundreds of our species of fish, birds and marine mammals spend part of each year in Alaska and then disperse to Europe, Asia, South America and the Antarctic. Conservation of these species requires national and international cooperation in scientific research and management.

Alaska is proud to be playing a leadership role in these efforts. A top priority for us is the resources in the North Pacific.

If the truth were known, the worst environmental disaster of the 1980s was not the Exxon Valdez. It was the rape of the North Pacific fisheries. Hundreds of millions of pounds of edible fish have been caught and discarded overboard—dead—every year. I raised this issue in Rio, and I believe I helped spark United Nations involvement. We worked hard on the abolition of drift nets.

Through Alaska's leadership, there is a new Salmon Convention that prohibits taking salmon on the High Seas beyond 200 miles. Our Department of Fish and Game drafted that Convention. And Russia, Canada, Japan and the U.S. all have signed on.

We also are working to protect the pollock in the so-called "Donut Hole"—a no man's land—in the Bering Sea. At Alaska's initiative, we finally have an agreement to protect those pollock. This Convention currently is in the capitals of six major fishing nations for ratification. It forbids any harvest of pollock in the Aleutian Basin, until the scientific community determines there is a biomass of at least 1.67 million metric tons. Beyond that, a method for establishing quotas has been designed.

In addition, Alaska has teamed with the Canadians in our common concern for the porcupine caribou herd. Our Eskimo Whaling Commission oversees a limited traditional harvest in conjunction with the International Whaling Commission. And Our migratory bird treaties are well known.

A Collective World

As the indigenous peoples learned long ago, in a cold, harsh environment, you have to care about others. You waste nothing. You share to survive. You care for the total. Every hunter's prize is a gift, not just to that hunter, but to his family and village. Sustainable living requires collective concern. Actually, this is true worldwide. Pollution knows no borders. All rivers eventually run into a common sea. All living things breathe the common air.

Yes, it is a collective world, but one in which we live so privately. Without concern for other people and peoples, for their needs and desires, activities for strictly private or local gain become destructive, not only to others but eventually to oneself. These truths were learned very early in the history of northern civilizations. Alaska's unique conservation role is to live by these truths and show others the way.

Interior's International Agenda

Mollie Beattie
Director
U.S. Fish and Wildlife Service
Washington, D.C.

Last week, I celebrated my six-month anniversary as Director of the U.S. Fish and Wildlife Service (Service). I want to thank you for your support during this transition period. Many of you have called to offer advice and assistance. I greatly appreciate this, and I hope to continue to hear from you.

This has been a busy time, and the Service is dealing with many issues in which you have a strong interest. These include the draft environmental impact statement for the Federal Aid program, the refuge system's budget shortfall, the "Refuges 2003" plan, the settlement of the compatibility lawsuit on secondary uses on the refuge system, and questions about non-indigenous aquatic species and the future of our fisheries programs.

Unfortunately, in the brief time I have to speak to you today, I can't adequately address all of these issues.

There is one issue I particularly want to deal with today. Although I am scheduled to speak about the Interior Department's international agenda, what we plan to do internationally is only a small piece of the Service's overall emphasis on an ecosystem approach to fish and wildlife management and conservation. I would like to talk about that today.

This ecosystem-based approach represents a change of direction for the Service. I am a Yankee from the mountains of Vermont, and Yankees generally don't cotton to changing the way things are done. We New Englanders still trap lobsters, tap maple syrup and bemoan the annual collapse of the Boston Red Sox the way we always have. Sometimes, it seems Paul Revere just rode through yesterday.

But I will tell you that, like a fellow Vermonter, the poet Robert Frost, I know when we've come to a fork in the road. And I firmly believe the Fish and Wildlife Service has come to such a place.

Looking in one direction, we see that the road has been well traveled. It represents our customary way of doing business—namely, management practices focused primarily on a single species with limited attention to the rest of the species and habitats in the surrounding ecosystem.

Looking in the other direction, the road has been less traveled. It represents a way of managing natural resources that takes into account the entire ecosystem and blends recreational use, economic development and conservation of wildlife so that *each* is definitely sustainable.

Many states and private organizations already have fine programs employing an ecosystem-based approach. And there also are many good examples of ecosystem conservation and restoration partnerships.

It is my intention to integrate this approach more fully at the federal level.

In short, the road less traveled is the road we *must* take, and it is the road the Fish and Wildlife Service *will* take. Fortunately, there is plenty of room on that road for everyone—for hunters, anglers and birdwatchers; for federal agencies and state wildlife managers; for Native American tribes and conservation organizations; for timber companies and farmers.

Some of you may ask, why go down this new road? What is wrong with what we are doing now?

I could offer up my own explanation, but I think I'd rather refer to the words of Aldo Leopold. Nearly 50 years ago, in *A Sand County Almanac*, Leopold wrote the following: "The disappearance of plants and animal species without visible cause, despite efforts to protect them and the irruption of others as pests despite efforts to control them, must, in absence of simpler explanations, be regarded as symptoms of sickness in the land organism."

Leopold went on to say: "The practices we now call conservation are, to a large extent, local alleviations of biotic pain. They are necessary, but they must not be confused with cures. The art of land doctoring is being practiced with vigor, but the science of land health is yet to be born."

I find it telling that after a half century and untold billions of dollars spent on conservation, Leopold's words still ring true. In fact, they are more true today than ever.

For all the doctoring we have done, we simply have not cured the basic ills that affect our land and water: the polluted and dying rivers and streams, the degraded wetlands, the growing numbers of imperiled species, the fragmentation and destruction of forest habitat—the list goes on.

Stated simply, it is time to change directions. To use Leopold's words, it is time to finally curtail the practice of land doctoring and give birth to the science of land health. And that is what the ecosystem approach to conservation is about.

Before I turn to how the Service is implementing this new ecosystem approach, let me digress for a moment and address those critics who warn that ecosystem-based conservation is a threat to hunting and fishing, as though the word "ecosystem" were some kind of code word for the animal rights movement.

In reality, there is absolutely no conflict between ecosystem-based conservation and fishing and hunting. In fact, I see ecosystem-based conservation as supportive of and even rooted in America's hunting and fishing legacy.

The tradition of hunting and fishing is steeped in a deep love and respect for the natural world. It is a tradition that cherishes the wholeness and wildness of the outdoor experience, regardless of whether a duck or fish is brought home. It is a tradition that led Izaak Walton to compare fishing to "the virtue of humility, which has a calmness of spirit and a world of other blessings attending upon it."

Certainly few of us would be content to hunt or fish in an environment stripped of its wholeness and diversity of life. The connection with nature in its primitive state is what is most alluring in these sports.

Leopold himself noted that there was a sharp division between one group of people that sees the land as commodity producing and another group that sees the land as a biota and its function as broader.

As Leopold did, I walk with those in the second group.

In moving to an ecosystem approach, we really are returning to the traditions of Leopold and Walton and the essence of what it means to be a hunter or angler.

Stated simply, the ecosystem approach supports, to use Leopold's words, a "land ethic" that "enlarges the boundaries of the community to include soils, waters, plants, animals, or collectively: the land."

Teddy Roosevelt, as avid a hunter as ever lived, summed it up when he said the nation is acting properly only when it "treats natural resources as assets which it must turn over to the next generation increased, and not impaired, in value."

Our goal, therefore, is the same as Leopold's and Roosevelt's and many other sportsmen who laid the groundwork for conservation in the country. This goal is to conserve and restore healthy ecosystems that will allow *sustainable* recreational use—including fishing and hunting—economic development and the well-being of all varieties of wildlife.

Obviously, there are a lot of questions about how the Service best can put an ecosystem approach into practice. I believe it's going to take at least three things.

First, we are going to have to look at the way we operate and make changes to allow our biologists from various program areas to work better together both with one another and with state wildlife managers and others outside the Service.

Second, we must engage in more, not fewer, partnerships, particularly with state wildlife agencies and Native American tribes, and also with conservation organizations, community groups, businesses, private landowners and other interests.

Third, we must educate the public about the importance of biodiversity and build support for an ecosystem-based approach to management. In short, we must change the way America thinks about its wildlife resources.

People need to grasp that often the obscure, unlovable species with the peculiar names, such as the unarmored threespine stickleback (a fish) or the Coffin Cave mold beetle, are as essential to the intricate network of life as the glamour species such as eagles and bears. They need to understand that, as insignificant as they may seem, molds, worms and insects are essential to the quality of life and the survival of humanity.

And they need to understand that often the decline of lesser-known species is a warning of serious problems in our environment that, if left unaddressed, eventually will harm humans. We can't allow the ecosystems upon which both wildlife and humans depend to continue to deteriorate or we will pay the price, not just in our quality of life but ultimately in our ability to survive.

With these three objectives in mind, therefore, we already are hard at work looking at how this might change the way the Service is structured and operates.

As you know, we are facing a number of challenges, and I am convinced the ecosystem approach will help us reduce duplication of others' efforts, use our resources more effectively, break through institutional barriers, and focus on getting ahead of the curve and dealing with environmental problems before they become crises.

For example, a multi-species, ecosystem approach to the conservation of declining and threatened species is far more likely to preclude the need for listings under the Endangered Species Act than dealing with one species at a time.

Why, for instance, deal with the decline of freshwater mussels independent of a decline in fish populations and riparian songbirds if they live in the same ecosystem and are affected by the same contaminants and degradation of habitat? Certainly it makes more sense, both biologically and economically, to take a broader approach to restoring the entire aquatic ecosystem.

And how can we tackle the problem of depleted fisheries in the Gulf of Mexico if we don't address the problem of midwestern farming practices that deposit sediment into the Mississippi, causing the degradation of the coastal marshes that are the nursing ground for many fish species?

The answer is—we cannot.

The Service's directorate met in Washington last month and approved a document entitled "An Ecosystem Approach to Fish and Wildlife Conservation," which I hope most of you have received by now.

The document defines the ecosystem approach as "protecting or restoring the function, structure, and species composition of an ecosystem, recognizing that all components are interrelated."

In a nutshell, this translates to three points. First, the Service and its partners cannot focus on any one piece of the ecosystem, we must look at all of it. Second, we have to go beyond managing for species and manage instead for habitat. Third, we need interagency cooperation at the federal level and, as I will discuss more fully in a moment, partnership with all of you.

A major question, quite obviously, is how we will define ecosystems on a map.

Each of our regional offices has worked on identifying ecosystem in their regions. The result uses watersheds as building blocks for our ecosystem units. We tentatively have identified 52 ecosystem units by grouping, or in some cases segmenting, watershed units. Vegetation cover types, physiography and optimum size are considered in these designations.

Our final ecosystem definition and boundaries must be worked out with all our partners. One of the next steps will be to work with our partners in the states, other federal agencies and conservation organizations to identify ecosystem units of highest emphasis.

Whichever way we proceed, however, the overall focus will be on action; planning and goal setting will be completed quickly, and cooperative work to bring about solutions will begin immediately.

So what does this mean for all of you?

I believe this is an unparalleled opportunity for state wildlife agencies, Native American tribes and villages, conservation organizations, local governments, community groups, businesses and others to influence and participate in the setting of this new course.

Nobody at the Fish and Wildlife Service is naive enough to think this approach to conservation possibly could be implemented through a sweeping mandate that comes down from the marble halls of Washington.

Success, if we are to achieve it, will come in small steps—acre by acre, streambed by streambed, wetland by wetland—in communities throughout American and, ultimately, the world. Clearly, it will not come without partnership.

One of the great achievements of the conservation movement in America is the foundation of partnership it has created.

Programs ranging from Federal Aid to the North American Waterfowl Management Plan to Partners for Wildlife and Partners in Flight are typical of the dozens of partnerships, both large and small, that have accomplished far more than any agency working unilaterally.

The ecosystem approach to conservation will build on, and *not* replace, this foundation.

To those who say it can't be done, I would say it's already being done. Ecosystem-based management may be a road less traveled but it has been traveled.

For example, we already have an excellent example of how an ecosystem-based partnership program works with the Washington State Ecosystems Conservation Plan.

The Service and Washington State government have teamed with Native American tribes, conservation groups and private landowners to use an ecosystem approach to restoring degraded habitat throughout the state. Already, there are 481 partners participating in the program improving habitat for all species on 300,000 acres through restoration projects.

But it also is ecosystem-based management because the projects involved, from stream restorations to wetlands enhancement to revegetation of upland areas, benefit the entire network of plant and animal life in Washington.

The North American Waterfowl Management Plan itself stands as a wonderful example of a successful partnership that benefits ecosystems. Under the auspices of the plan's 12 joint ventures, more than 2 million acres of wetland habitat have been conserved, restored or enhanced through hundreds of partnerships involving the Service, state agencies, corporations, conservation groups and landowners.

The plan has done as much to promote biodiversity as it has to conserve waterfowl. The Service continues to be fully committed to meeting all the plan's long-term objectives and goals. In fact, the joint ventures of the North American Plan may provide excellent models of the scope and type of partnerships we envision for ecosystem-based conservation and management.

There are many other joint projects with an ecosystem focus, from the Chesapeake Bay to the Sandhills ecosystem between the Platte and Niobrara rivers in Nebraska to the Trinity and Klamath rivers in California. In each case, working in partnership with state, local, Native American and private interests has allowed the Service to accomplish far more than it could by itself—the whole is indeed greater than the sum of its parts.

At this point, let me reiterate that we still are in the preliminary stages of developing this new ecosystem approach. We are eager for your suggestions, criticisms and participation. Nothing is cast in concrete. Now is the time to get involved.

The Service has sent out the draft document and a formal request for comments to state wildlife agencies and private conservation groups. We hope to hear from many, if not all of you.

Before I conclude, let me return to where I was supposed to start—and that is the Service's efforts to expand our ecosystem approach beyond our national borders.

Periodically, events and circumstances remind us just how closely we are linked to other countries.

We see pollution from U.S. smokestacks falling as acid rain in Canada. Radiation from a nuclear accident in the former Soviet Union drifts over Europe. A hole appears in the ozone over Antarctica and the best scientific evidence points to CFCs released into the atmosphere thousands of miles away.

We see the decline of migratory birds that fly from one hemisphere to the other and back each year. In South America, there is the destruction of the rain forests so aptly described as the earth's lungs. And we remember the industrialized countries of the northern hemisphere already have destroyed most of their forests and wetlands.

All this reminds us how slow the countries of the world have been in recognizing basic ecological problems—let alone addressing how to solve them—and how these problems threaten to explode into environmental crises that transcend national boundaries.

I believe these international problems call for the same ecosystem approach that we are seeking to employ domestically.

Certainly we can point with some pride to historic international conservation agreements, starting with the Migratory Bird Treaty of 1916 between the United States and Great Britain on behalf of Canada, and followed by other significant treaties and conventions, including a migratory bird treaty with Mexico in 1936 and the Western Hemisphere Convention in 1940.

More recently, the Convention on International Trade in Endangered Species has been effective in controlling and managing trade in many imperiled species around the globe. And we have worked closely with many countries to establish wildlife management training and graduate programs to help them better manage their resources.

To improve the Service's ability to carry out its international responsibilities, we are in the process of establishing a new position for an Assistant Director for International Affairs, which will become the focal point for programs that formerly were scattered throughout the Service's management structure. I think this will enable us to more effectively focus our considerable in-house international expertise on the global challenges we face.

In a sense, this reorganization simply reflects our belief that to make real progress in tackling international problems, such as the decline of migratory birds and the unprecedented loss of both plant and animal species, we cannot deal with them in isolation from each other.

As I mentioned earlier, I believe this makes the success of an ecosystem approach here in the United States that much more vital. We are, in a sense, a laboratory for the world. Where we succeed, other countries will follow.

Building a Better Moosetrap? Investigating Biodiversity in Alaska

Thomas R. McCabe
National Biological Survey
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National Biological Survey
Anchorage, Alaska

My coauthor, Larry Pank and I are especially gratified and mildly intimidated to be able to make a presentation at this opening session, which normally and otherwise is reserved for people who have prominent positions and access to speechwriters.

We were asked to address biodiversity in Alaska. Being wildlife researchers and Alaska residents, we may have a unique perspective on this topic and on the matter of biodiversity in general. We had all winter to mull and refine this perspective and, in Alaska, a winter's worth is a lot of mulling. It suffices to say that our remarks should be taken either as the professional insights of two biologists with collectively two decades of experience in dealing with the Alaskan landscape and its renewable resources, or they can be taken as the rantings of two cabin-fevered Alaskans who would say anything for a chance to see new faces.

For this presentation, our first order of business was to define some terms. "Alaska" was easy. It is this huge chunk of land at the top of the map that represents one-sixth of the total area of the United States and fully one-third of the U.S. Coastline, yet is home to only one-fifth of 1 percent of the total U.S. population. Relative to the rest of the United States, Alaska's per-capita rankings include first in birth rate, first in aircraft pilot license, first in ice cream consumption, second in household income, second in unemployment, first in high school education attainment or higher, and first in energy expenditure. With regards to resources, Alaska ranks forty-seventh in manufacturing, first in fisheries catch and value, first in energy extraction, first in waterfowl production, and first, by far, in the amount of pristine land and water reserved and conserved. And so on and so on. Alaska is not a stereotyped admixture of grizzly bears, blackflies, barflies, free-floating oil tankers, Iditarod groupies and igloos. It isn't a stereotype. It is much more. It is the "Great Land," which is in fact, the translation of the word Alaska as corrupted from the language of the Aleut.

Alaska is too big, too diverse and too unique to be defined or characterized adequately by standards that apply elsewhere or by ephemeral measures of comparison. Such circumscription is what we humans do to grasp the dimension of a place, and all too often this is done at the sacrifice of grasping the essence of the place itself. Nevertheless, Alaska is a huge and diverse cornucopia, and for the most part, not yet fully sectioned, siphoned and sacrificed to last-gasp manifest destiny. It is a place of self-reliance, of pioneering spirit, of enterprise. In many ways, it is an attitude with a geographic boundary.

To avoid lapsing further into the prosaic, let me switch to our efforts to examine "biodiversity"—a term for which there most certainly is not a prosaic definition. Larry and I had no trouble finding a plethora of literature on biodiversity, but we did have trouble locating a consensus as to what it is. At best, it seems to refer to holism—that a thing or place is replete with what it is or was capable of supporting and/or sustaining. At worst, it is a prematurely hackneyed neologism promising new abstraction, enthusiasm, funding and rallying point for carrying on the extraordinary territorial business of natural resource management or conservation or stewardship, assuming—and cautiously so—that those strategies are synonymous or at least compatible.

Larry and I read many learned papers on biodiversity, and we conferred with a number of learned people about the subject. It seems that everyone intuitively intellectualizes the concept, but very few share a common definition. Almost all see the concept of biodiversity as the universal underpinnings to maintaining a healthy biosphere, but the term is vastly overexposed, giving the appearance that it is not so much an ecological condition as it is a biopolitical agenda of one sort or another.

We found, in the course of investigating our topic, that virtually everyone who has written on the subject of biological diversity has attempted to define its meaning and scope. Among the most comprehensive, in our opinion, was that proposed by the U.S. Fish and Wildlife Service in 1992, that read: "The variety of life in an area, including genetic composition, richness of species, distribution and abundance of ecosystems and communities, and the processes by which all living things interact with one another and with their environment."

Notwithstanding that this definition entertains nearly every biological concept short of "What is the meaning of life?" we agree. To what grander, more noble objective could we, as a biological scientists, dedicate ourselves than to the preservation of all living things and their processes of life and interaction? Hopefully, that is what we have been doing all along. But somehow it seems that, with emergence of the term "biological diversity" as a professed paradigm and moral imperative, the proverbial waters of practical management have been seriously muddied. For the concept of biodiversity to become functional for land-based agencies, it must be supported by viable management alternatives. Proof of the validity of a concept is dependent upon whether it can be tested and if it can be implemented to produce tangible results. The loss of 10,000 more hectares of tropical rainforest or a few more holes found in the ozone layer will be of interest to a land manager in Alaska when he or she reads the latest issue of *Wilderness* magazine. However, those matters will not address the immediate management protocol and wherewithal necessary to administer a refuge, park or wilderness area.

During our tenure in Alaska with the US Fish and Wildlife Service, and now the National Biological Survey, Larry and I have worked alongside researchers, managers and technicians who tangle with polar bears, wolves and grizzlies, who wrestle muskox and caribou, who brave subzero temperatures, blizzards and horizontal winds, who contend with frostbite in the winter and mosquitos the size of small Cessnas in the summer, and who survive weeks in the field without dry socks or television. They are a hardy lot. Yet, these same folks can develop facial tics and shingles when confronted by exhortations to place greater emphasis on biodiversity. They—we—are versed in how to conduct research and manage wildlife. But, our Achilles' heel is that we do not always know how best to manage public, bureaucratic or political expectations—expectations that can be short-lived, short-sighted and budget-insensitive.

"When the human mind deals with any concept too large to be easily visualized, it substitutes some familiar object which seems to have similar properties." Aldo Leopold wrote these words in 1939. He wrote them in reference to the popular "balance of nature" theme. He went on to explain that the concept of "balance of nature" was convenient for describing the biota to laypersons, but had both merits and defects. Its merits lay in the conception of a collective system and the recognition of some order and utility of all species. Its defects were that it inferred only one point of balance and that the balanced state was static. We think that Leopold's reservations about large biological conceptualizations are analogous and applicable to our present notion of biodiversity. In other words, biodiversity explains something everybody can understand in terms nobody can understand.

A more recent admonition was prescribed by an individual writing on the theme of "motherhood, apple pie and biodiversity". The skeptic asked: "Before this salad-bar term shows up in many more statutes, shouldn't there be consensus on what it means . . . and . . . who is going to pay for whatever 'new' research and management and realignment of the cosmos it requires?" We could not help but agree.

Here in Alaska, we are, as they say, covered in biodiversity. Since we have not yet paved over, cut down, or turned under but a minuscule portion of this state, human impact often is only as an added predator to systems with pristine biodiversity. To adjust for our predatory nature or consequences, the managing agencies set limits and seasons on wildlife harvest and, when necessary, attempt to realign the total "predator" impact. To our knowledge, the natural resource agencies in this state do not manage for biological diversity. They manage for the natural ecological integrity that prevails despite the incursion of man for what he claims as his welfare.

Alaska has reputation as "The Last Frontier." And so it is in many respects. This state, from an ecological standpoint, is blessed with 9 to 20 ecoregions-depending on which agency perspective you endorse. It also is blessed by a climate more nurturing to fair-weather tourism than to year-round homesteading, by a manifestly independent citizenry that has not been weaned of respect for the landscape, and by a political leadership tenacious in applying its public trust mission to Alaska's lands, waters and wildlife as well as to its people. Alaska is America's last frontier because it is relatively pristine. And it is pristine because it retains its biological diversity.

To be sure, there are problems of natural resource management here. Witness controversies over petroleum development on the Arctic National Wildlife Refuge, subsidized timber harvest on the Tongass National Forest and the recent wolf-reduction issue. But these are matters that can and should be addressed in the scientific context of biodiversity, not just in the political context of it.

In our opinion, biodiversity should not be viewed simply as a crusade to save endangered species. The melodramas involving snail darters, California condors, red-cockaded woodpeckers and spotted owls, to name a few, were pretty convincing testimony that human society is not yet willing to accede to the limits of resource tolerances. The Judeo-Christian ethic is alive and well, even in Alaska, but until the human animal is willing to share planet Earth with all other life forms, embarking on a quest for biodiversity may be quixotic. Certainly, rededicating ourselves to the objective of biological diversity is the grail worthy of our continued professional enterprise. But first we must understand that biodiversity is an ecological condition, not a strategy. Strategies are transitory processes subject to constant revision and perpetually influenced by economic and political pressures. Whereas, biodiversity must be thought of as the maximum potential level of natural ecological integrity attainable. Recognition of these facts should provide the basis and impetus for establishing management goals to accommodate and accomplish the condition of biological diversity.

The question relative to biodiversity, in our opinion, also does not imply a divine mission to save all of the world's genotypes. Rather, it should be addressing the pandemic alteration of habitats and moderation of ecosystems due to overexploitation, overpopulation, and toxification of the land, water and air. The fundamental unit of biodiversity is habitat. In 1939, Leopold, in his classic paper "A Biotic View of Land," presented the "biotic pyramid" to show the interrelationship, however simplistically, among all of the trophic levels in a community. At that time he disparaged the "unprecedented violence, rapidity, and scope" of man's ability to change what evolution created. In the 55 years since publication of that paper, humankind has preempted every nook and cranny of the Earth and much of the solar system as its habitat, and revelled in its technological capability to reshape science and to commit its resource base to short-term demands. One cannot help but be awestruck, for example, by the engineering feat of the Prudhoe Bay complex and the TransAlaska Pipeline, and equally appalled at the destruction wrought by the Exxon Valdez. Although we now know more of the specifics of the interrelationships within communities that Leopold eluded to, we are far from understanding all of the ramifications of our impacts on the environment, however awesome or unintentional. Our lack of understanding of the interactions among the myriad trophic levels within a community dictate that we must conserve the whole, so as not to eliminate, unwittingly, any essential element. This is the basis for the concept of landscape management.

We have heard it said in jest, that Alaska's state motto should be "*Carpe Resourceum*," but in truth it is "North to the Future," and relative to landscape management, Alaska has lived up to its motto. Nearly 200 million acres of this state have been set aside to conserve essential aspects of its unique ecosystems. That these systems are virtually untrammeled by modern man makes their value that much greater, for it provides an opportunity to view, study and maintain natural systems afflicted only by the vagaries of nature.

None of the foregoing is meant to imply that we believe there is anything wrong with espousal of biodiversity—in Alaska or anywhere else. To be sure, and again, it is what our profession long has been dedicated to. But the perspective on biodiversity here is different than most other places on the North American continent. It is and has been viewed in Alaska as a management objective— *the* management objective—not as a management strategy. Perhaps and probably because of the prevailing luxury of a diverse resource base, management and research personnel in Alaska are not readily drawn to the new banners of idealism without the surety of pragmatic gain. And certainly, we Alaskans are not given to banners that portend further diffusion of already stretched budgets and workloads. In other words, until the dominant resource agencies firmly and convincingly define the common boundaries of biodiversity and commit to developing uniform management objectives within that definition, the Alaska perspective will not wholly embrace a reinvention of the wheel or the building of a supposedly better moose trap. But that is not to say that heightened idealism and philosophical adaption are absent from the resource management psyche here in the north. To the contrary, we readily if not eagerly embrace practical new ideas and practicable new objectives . . . weather permitting.

The National Biological Survey is manifestation of a practical idea that Alaskans, for the most part, are willing to adopt and endorse if it truly fulfills the promise to enhance research efforts for the resource management agencies and becomes, as Interior Secretary Babbit has stated, "a useful tool for sound resource management decisions." As the new research arm of the Department of the Interior, NBS will be charged with quantifying biological resources and examining the questions of ecosystem biodiversity. In Alaska, the Interior Department is responsible for management of over one half of the state's land. Through partnerships between NBS, the state and native interests, the ecosystems of Alaska can serve as a cornerstone for the national database on biological resources. Because Alaska is yet essentially untrammeled, the data accumulated for these ecosystems will reflect a natural order of the biota. If it is to be demonstrated that resource management can be accomplished within the concept of biodiversity, it must be achieved first within an ecosystem that is fairly devoid of man's influences, has a simple complex of species and is easily susceptible to disturbance. The Arctic tundra is just such an ecosystem. And if we may suggest, modestly, we know the perfect place to start.

Earlier, we mentioned that this state is an attitude with a geographic boundary. It is one of self-reliance, self-determination, survival and resilience. It also is an attitude of pride in the immenseness, wildness and beauty of this state. The vast array of fauna that inhabit this land and the flora that adorns it are as nature intended. Some people call it biodiversity; we call it Alaska.

Acknowledgements

Although they should not be held accountable for the opinions expressed here, we would like to thank R. A. McCabe, J. T. Ratti, B. Griffith, N. E. Walsh, D. D. Young, and especially R. E. McCabe, for their helpful comments and suggestions on earlier drafts of this manuscript.

4-H Wildlife and Fisheries Recognition Awards, 1993

Opening Remarks

Mollie Beattie, Director
U.S. Fish and Wildlife Service
Washington, D.C.

A number of you here this morning had the opportunity, as I did, to meet the six National 4-H Wildlife and Fisheries Adult Volunteer Leader winners for 1993 at the reception held for them last evening. They truly are fine people, giving generously of their time and energies, leading some of our nation's most promising young people. I am delighted to publicly recognize and thank these people—winners who represent thousands of other 4-H adult volunteer leaders—for their essential contribution, inspiring 4-Hers to become life-long stewards of our Nation's fish and wildlife resources.

I am pleased to continue this U.S. Fish and Wildlife Service tradition: it is the 14th consecutive year we have worked in partnership with USDA's Cooperative Extension Service, to recognize outstanding volunteer leaders for their significant contributions to our young people.

James E. Miller, Acting Administrator for Extension
U.S. Department of Agriculture
Washington, D.C.

I, too, am pleased to participate in this program to honor these six 4-H Wildlife and Fisheries Volunteer Leaders, winners for 1993.

Once again, on behalf of the Cooperative Extension System and U.S. Department of Agriculture, thanks to the U.S. Fish and Wildlife Service for their continuing support of this annual program, and to these outstanding volunteers for their personal commitment to wildlife and fisheries 4-H youth education programs.

Ellen DeBacker, *Boulder, Colorado*

Ellen Debacker is a science teacher in secondary schools and has been a 4-H volunteer leader for four years. Ellen realized that the traditional classroom did not offer all aspects of learning about natural resources that could be accomplished by exposing youngsters to wildlife through a 4-H program. Her group of 4-H kids were enthusiastic and knowledgeable, and tours of the Rocky Mountain National Park, local wildlife refuges and the Rocky Mountain Arsenal gave them the opportunity to see a variety of habitats and to add to their list of wildlife sightings. She has assisted many other young people, for example, classes of elementary children often have been taken on tours of nearby wildlife refuges. Ellen also serves as a counselor and teacher on wildlife projects and guides the development of junior wildlife leaders at a number of conservation camps. Ellen says that ". . . nature and the sharing of nature has changed my life and I hope through my work the lives of others will be changed."

Elizabeth Jordan, *Shreveport, Louisiana*

Elizabeth Jordan has been a 4-H leader for 12 years. As a volunteer 4-H wildlife and fisheries leader, Elizabeth has guided 4-Hers in preparing forestry science fair exhibits for the parish and state fair competition, set up wildlife exhibits made by parish 4-Hers at Earth Day activities, guided youngsters preparing wildlife tabletop demonstrations and arranged for speakers on Project Wild activities. Elizabeth can be depended on to lend a hand wherever needed. She has assumed responsibility for registering contestants and tallying results at parish fishing derbies, helping members make squirrel boxes and developing the wildlife exhibit for state 4-H leaders conference. She plans to become more active as leader for the parish Wildlife and Forestry Club. She says, based on her experience, wildlife projects and programs for urban young people instill in them a greater awareness of our renewable natural resources and how they contribute to our lives and well-being.

Kay Stewart, *Pontotoc, Mississippi*

Kay Stewart has been a 4-H volunteer leader for nine years. The list of activities summarizing her experiences and accomplishments as 4-H volunteer leader is extraordinary! Kay is a Shooting Sports club leader, Field and Stream County Coordinator, Wildlife Judging team coach, Project Learning Tree facilitator, Soil Conservation Earth Team member and State Park volunteer, Hunter Education County Coordinator and instructor, and much more! She coordinated the planning and construction of a three-mile self-guiding nature trail that teaches about forest management, wildlife, wildflowers, reptiles, butterflies and soil conservation practices. Kay says she has enjoyed every minute of her so-called "work," leading and teaching 4-H youngsters and emphasizing the conservation aspects of wildlife and fisheries management, while hunting, fishing, canoeing, studying and coaching the wildlife habitat judging team, and working with kids, other leaders and natural resource professionals.

Twila Buffington, *Mebane, North Carolina*

Twila Buffington has been a volunteer wildlife and fisheries leader in North Carolina since 1977, when she started a 4-H club in her community. She also served as a leader in Iowa for five years prior to moving to North Carolina. She is employed as a secretary in a pre-school center for children with developmental disabilities. Twila is a 4-H wildlife and fisheries adult leader on the community, county and district levels. Project areas have included wildlife, forestry, entomology, photography, archery fishing and natural resource conservation. Twila serves as a county wildlife club leader, developing wildlife food plot planting activities, an archery instructor and served on the district level for 11 years, where over 600 4-H wildlife project books were completed under her direction. She currently is recognized as a North Carolina Master Volunteer for environmental education programs. She indicated that she intends to continue investing her volunteer time in the 4-H program to help promote a better understanding and appreciation for our natural resources.

Philip Genova, *Ithaca, New York*

Philip Genova is a self-employed contractor and has been a volunteer 4-H leader for the past four years. He lives on a farm with his wife and young son. He also is a devoted fly fisherman. A Trout Unlimited newsletter suggested there was a need for volunteer instructors for its 4-H Sportsfishing and Aquatic Resource Education Program. There began a warm relationship between Phil and hundreds of youngsters who share an interest in sport fishing. His 4-Hers were taught to tie knots, cast, sample water quality, and they learn about food webs. They also spent plenty of time fishing, and fly fishing has become the method of choice. Over 500 young beginners and nearly 100 instructors/leaders have been involved in his events, raising the visibility of 4-H, fishing and conservation in his community. In 1992, he and his 4-Hers worked in the "Spawn Room" of the Altman Hatchery, extracting eggs and milt from spawning salmon and learning more about the salmon restoration program. Phil says his goal is to make the fly fishing apprentice program a full-time endeavor and introduce more young people to an old and established tradition.

Nancy Tucker, *Knoxville, Tennessee*

Nancy Tucker has been a 4-H volunteer wildlife and fisheries leader for the past 11 years. She describes herself as a 4-H promoter, teacher, wife and mother. Her interests in wildlife and environmental stewardship began with a 4-H Environmental Leaders Forum. She returned home with a variety of new ideas and enthusiasm—and a county without a Project Group in wildlife. She organized all the 4-Hers interested in wildlife, conservation, forestry and shooting sports and set goals for that first year. Her 4-H group developed an environmental education area, including a natural wetland area that the students could use to observe wildlife, identify plants and trees, study the wetlands and water quality—yet an area that still could be enjoyed by everyone in the community on a daily basis. Her club applied for grants and obtained $8,000 in funds and services, enabling them also to develop a boardwalk in the area to make it accessible to the handicapped. They planted mast trees, native grasses and wildflowers, and taught primary and intermediate school teachers how to utilize the area with suggested lesson plans and ideas for students. Nancy's four children—her homegrown 4-Hers—are growing up and are off to college.

Concluding Remarks

Mollie Beattie

We extend our sincere appreciation to these six winners and to the thousands of other volunteer 4-H wildlife and fisheries leaders they represent. It has been an honor and privilege to present these awards. Best wishes and continued great success in the years ahead for your continued exemplary leadership for our most important resources—the young people of this nation.

The 1994 Guy Bradley Award

Mollie Beattie, Director
U.S. Fish and Wildlife Service
Washington, D.C.

The Guy Bradley Award recognizes excellence in wildlife law enforcement. Together with biologists, habitat managers, and hosts of other state and federal land management professionals, law enforcement agents are an integral part of this nation's effort to conserve our fish, wildlife and plant resources for future generations.

The Award is given annually to that person, or persons, whose dedication and service to the protection of the country's natural resources demonstrate outstanding leadership, extended excellence and lifetime commitment to the field of wildlife law enforcement, and whose actions advance the cause of wildlife conservation. The Award is given in the spirit of Guy Bradley, an Audubon game warden killed in the line of duty in July 1905, while preserving a Florida rookery from plume hunters. Guy Bradley is believed to have been the first warden to give his life in the line of wildlife law enforcement.

Established in 1988 by the National Fish and Wildlife Foundation, the Award has recognized both state and federal wildlife law enforcement officials. Last year, the Foundation presented the award to two individuals: Tom Moore, a Forensic Scientist for the Wyoming Game and Fish Department, and Richard Moulton, a Special Agent for the U.S. Fish and Wildlife Service. This year, the Foundation is pleased to recognize Ken Goddard, Director of the U.S. Fish and Wildlife Service's National Forensics Laboratory in Ashland, Oregon.

Ken was selected from an outstanding group of nominees by a volunteer panel of judges comprised of representatives from federal and state wildlife agencies and conservation organizations.

Ken Goddard, Director, National Forensics Laboratory

Ken Goddard serves as the Director of the U.S. Fish and Wildlife Service's National Forensics Laboratory in Ashland, Oregon. Ken is a nationally recognized expert in wildlife forensics who, for years, has been at the forefront of wildlife forensics, advancing its important role in fish and wildlife conservation.

Ken was a driving force behind the development of the first national and international wildlife forensics laboratory in the world. In this role, he engaged in numerous political, administrative and economic battles before the lab even was built and staffed. The lab opened for operation on October 1, 1988. Since that time, Ken has developed strategic long-range plans for the lab geared to the wildlife forensics needs of not only the U.S. Fish and Wildlife Service, but of the 50 state fish and game agencies and 119 signatory countries of the CITES Treaty.

To handle these monumental tasks, Ken has staffed the laboratory with a team of highly qualified experts. Together, their efforts have had dramatic impacts on the overall effectiveness of wildlife forensics. For example, the lab's work on DNA testing and analysis has led to the resolution of cases that had been impossible to solve. One case which recently garnered national attention was a deer poaching problem on Clint Eastwood's northern California ranch. Through the work of the lab, and one of the first implementations of DNA testing in a wildlife case, a successful prosecution was obtained.

Without question, the excellent work performed by the lab could not be accomplished without the dedication of many volunteers and employees. However, Ken remains the driving force behind the scenes with his leadership, innovation and dedication. Largely because of his professional tutelage, the lab has affectionately become known as the "Wildlife Scotland Yard" among members of the media. Somehow, Ken also finds time to write and he is the author of four nationally recognized works of fiction including his most recent novel, *Prey*.

The Award

In recognition of Ken's efforts on behalf of wildlife conservation, the National Fish and Wildlife Foundation is pleased to present him with a commemorative plaque featuring the Foundation's 1994 Conservation Print, together with a check for $1,000.

In presenting this award, it is with the recognition that Ken is one of the hundreds of dedicated individuals in the larger law enforcement community who deserve similar recognition. The Foundation would like to thank John Doggett, Terry Crawforth, Jim Timmerman, Terry Grosz, Rollie Sparrow and Max Peterson for their willingness to serve as Guy Bradley Award judges. Finally, our thanks go to the Wildlife Management Institute for its help in this presentation.

Special Session 1. *Conserving International Resources of the North Pacific Rim*

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The Status of Sea Ducks in the North Pacific Rim: Toward Their Conservation and Management

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Background

Species and Status

Sea ducks (tribe *Mergini* after Johnsgard 1960) are the most northerly distributed ducks, and species diversity is greatest in the North Pacific. They exploit a diversity of inshore and offshore marine habitats during the non-breeding season, and their use of habitat during breeding varies from coastal through freshwater wetlands of the tundra and taiga (Figure 1, Appendix 1). Non-breeding cohorts frequent marine habitats most of the year. Sea ducks thus are important indicators of the quality of freshwater and marine ecosystems of northern biomes.

Of the 17 species discussed in this manuscript, at least 13 are reported to be declining (Appendix 2). However, the basis for many of those assessments is equivocal because there has been little effort to monitor populations. The efforts to more

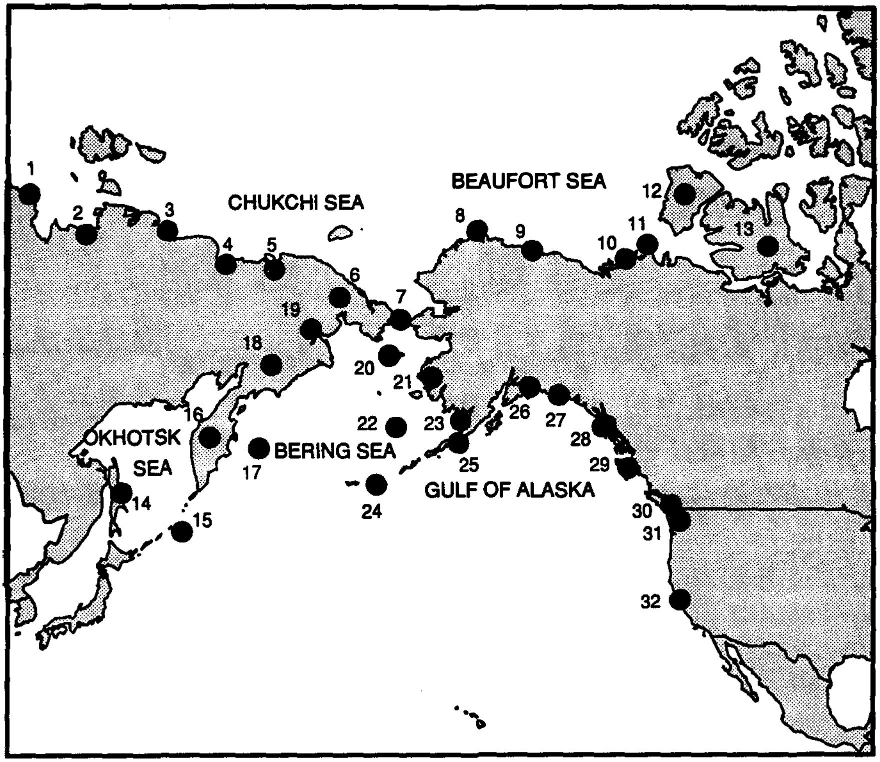


Figure 1. Important sea duck areas of the North Pacific Rim.

25 Alaska Peninsula
24 Aleutian Islands
28 Alexander Archipelago
19 Anadyr River
12 Banks Island
23 Bristol Bay
11 Cape Bathurst
27 Cape Yakataga
5 Chaun Bay
6 Chukotst Peninsula
17 Commander Islands
7 Diomede Islands
29 Hecate Strait
3 Indigirka Delta
16 Kamchatka Peninsula
4 Kolyma Delta
15 Kuril Islands
1 Lena Delta
8 Point Barrow
22 Pribilof Islands
26 Prince William Sound
9 Prudhoe Bay
31 Puget Sound
20 Saint Lawrence Island
14 Sakhalin Island
32 San Francisco Bay
30 Strait of Georgia
10 Tuktoyuktuk Peninsula
13 Victoria Island
18 Yakutio Koryak
2 Yana Delta
21 Yukon-Kuskokwim Delta

precisely assess their status point to catastrophic declines (Kertell 1991, Stehn et al. 1993). Conservation problems related to sea ducks have a long history throughout the holarctic. For example, the Labrador duck (*Camptorynchos labradorius*) became extinct in 1875 (Phillips 1925); common eiders (*Somateria mollissima*) declined seriously throughout the northern hemisphere (Townsend 1914, Phillips 1925, Doughty 1979); harlequin ducks (*Histrionicus histrionicus*) experienced declines in Iceland and Greenland (Gudmundsson 1971, Salomonson 1950), and more recently have been designated *endangered* in eastern Canada (Committee On the Status of Endangered Wildlife in Canada 1990). In Russia, all species of eider and harlequin ducks have been closed to sport hunting since 1981, and Chinese mergansers (*Mergus squamatus*) presently are extremely rare and fully protected, i.e., category one of the red book (Solomonov 1987).

Current issues. Bartonek (1993) noted an increased concern for the status of sea ducks in the Pacific Flyway due to (1) the listing of the spectacled eider (*S. fischeri*) as a *Threatened* species throughout its range in the United States (U.S. Fish and Wildlife Service 1993a); (2) the finding that the Alaskan nesting population of the Steller's eider (*Polysticta stelleri*) warranted listing as a *Threatened* species; (3) losses of harlequin ducks stemming from the *Exxon Valdez* oil spill; and (4) inexplicable mortality of scoters (*Melanitta* spp.) summering in the Gulf of Alaska. This concern, however, has not changed management approaches to most sea duck populations. Sea ducks are subjected to extremely liberal hunting regulations enhanced by a perception of little hunting interest and insignificant harvest rates of this group (Bartonek 1993, Gillelan 1988, Reiger 1987, 1989, U.S. Fish and Wildlife Service 1993b), and management of hunting kill may be lacking (e.g., Seller's eiders in Russia prior to 1981), or seriously compromised by conflicting interests in subsistence and aboriginal use (Kondratyev 1988, Nichols et al. 1988, Wentworth 1993, Wolfe et al. 1990).

Conservation of wildlife species requires a fundamental understanding of population status, mortality and natality in order to make informed decisions. This knowledge is lacking for sea duck population. Here we review aspects of life histories and simulate demography of sea ducks. By developing matrix models to integrate life history parameters we analyze the effects of varied mortality rates on population dynamics. We present recommendations that redirect our approach to the management and protection of sea ducks.

An Ecological Basis For Conservation

Mortality Theories

Compensation. Patterson (1979) highlighted the need for management based on ecological principles, and integrated theories of compensatory mortality with life history patterns. Empirical evidence suggested that hunting and non-hunting mortality may largely be compensatory for the mallard (*Anas platyrhynchos*), up to some threshold (after Anderson and Burnham 1976); however, that hypothesis has been largely repudiated (*see* Johnson et al. 1988). Patterson (1979) expressed concerns that the mallard would be used as a "yardstick" with which numerical kill of other species is evaluated. Also, Mortalbano et al. (1987) were concerned that compensatory mortality had become a philosophical cornerstone of regulatory programs for waterfowl. This philosophy condones an approach to management which can result in over-exploitation of species (Bartonek et al. 1984.).

Additivity. Anderson and Burnham (1976) noted that above a certain level, hunting mortality in the mallard must be additive and this "threshold" must be less than the natural mortality rate. Therefore, species with low natural mortality rates are less capable of "compensating" for hunting mortality than species with high mortality rates. Patterson (1979) noted that mallards and canvasbacks (*Aythya valisneria*) are at opposite ends of the threshold spectrum, i.e., 0.40 and 0.10 harvest rates, respectively. He therefore emphasized the need for conservative approaches in the management of hunting kill in the diving ducks (also Pirot and Fox 1990, Hochbaum and Caswell 1978).

We expand this suggestion to include sea ducks. Our analyses indicate that sustainable harvest rates may not exceed about 0.03 of the adult population in some sea duck species. Therefore, our perception of the significance of losses to hunter kill will change based on life history patterns for each species.

The r-K Continuum

Life history. Waterfowl span the entire r-K continuum, and sea ducks exhibit extreme K-selection relative to other species of ducks (Eadie et al. 1988). Like seabirds, sea ducks have deferred sexual maturity, low annual recruitment rates to breeding age, variable annual rates of non-breeding by adults and high annual adult survival rates (*see* Ricklefs 1990). The highly variable environment of the northern marine ecosystem favors a life history strategy of minimized annual investment in reproduction and extended longevity.

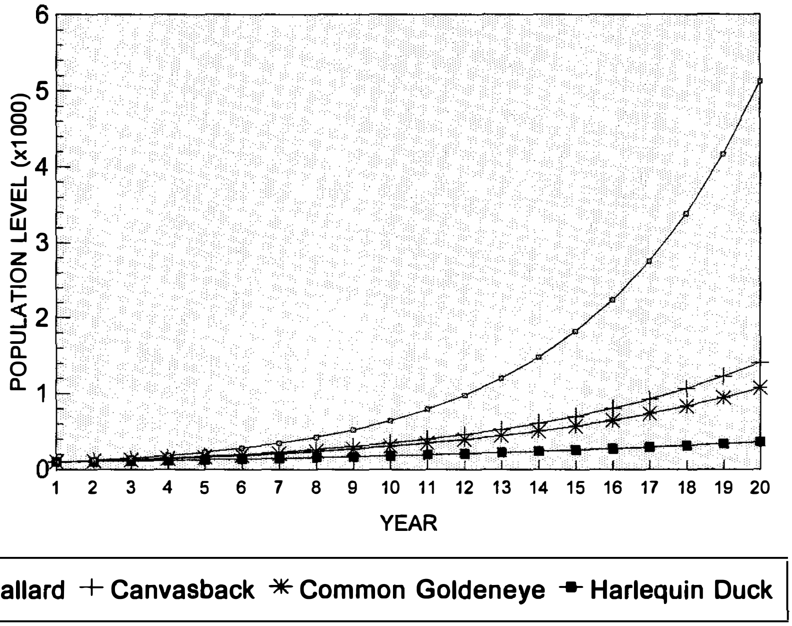
Ecological time. Population stability of sea ducks is dependent on high adult survival and a few successful years of reproduction (e.g., Milne 1974, Swennen 1991). This results in population growth that is stepped, and average annual rates of increase can reach 5 to 10 percent. Considerations of ecological time become important because infrequent Arctic ice event can cause mass mortality for some species (Barry 1968), and/or might affect body condition and fitness of birds (*see* Goudie and Ankney 1986). Hence, gains in populations during a few decades of favorable environmental conditions likely are important to buffer against extirpation during harsh conditions.

Species sensitivity. Species which maintain population stability through high adult survival are sensitive to increased mortality (Shaw, 1985). In sea ducks, sensitivity is exacerbated by the relatively high proportionate losses of adults in events such as hunting and oil contamination.

Population Modeling

Intrinsic differences. We generated theoretical populations of various species of ducks over a 20-year period (figure 2, Appendix 3). In this exercise, the mallard population increased to over 5,000 females, whereas the harlequin duck population increased to 400 females. It is clear that the ability of these populations to sustain mortality and/or recover from population declines are dramatically different. Johnson et al. (1988) pointed out that modelling is no panacea for waterfowl management, but it helps to consolidate our understanding of population dynamics. Here modelling supports the need for a different approach to the management of sea duck populations.

Demography. We modelled theoretical populations of harlequin ducks using a Leslie matrix approach (Caswell 1989). We incorporated data on harlequin ducks from Iceland (*see* Bengtson 1972, Bengtson and Ulfstrand 1971, Gardarsson and Einarsson 1991). Our analysis suggests that population stability occurs when adult survival rates are about 0.85 (Figure 3), a level somewhat less than unhunted popu-



Calculations using a transition matrix approach of data in Appendix 3.

Figure 2. Hypothetical population growth of four species of ducks.

lations of common eiders in Scotland (*see* Coulson 1984). An increasing population of harlequin ducks, i.e., 9.3 percent per year at Lake Myvatn, Iceland from 1975 to 1989) (*see* Gardarsson and Einarsson 1991), was simulated when adult survival rates approximated 0.95.

Adult survival appears to be the main factor influencing population stability for sea ducks (Appendix 4), suggesting that little can be achieved through management of other biological parameters, such as survival and production of young.

Defining Sustainable Mortality

Simulating mortality. Simulated annual kills of harlequin ducks suggest that losses exceeding 3 to 5 percent of the initial adult population are not sustainable (Figure 3). This is similar to our earlier estimates of harvest rate thresholds. This finding highlights the need to reduce mortality on some species of sea ducks in areas where harvest rates are high, such as in Alaska, Newfoundland and the eastern United States (*see* Wentworth 1993, Reed and Erskine 1986, Goudie 1989, Krohn et al. 1992, Wendt and Silieff 1986) and where chronic oil pollution is severe (Piatt et al. 1990a, Chadwick 1993).

Estimating mortality. Because minor increments of mortality can negatively affect populations of sea ducks, the estimation of mortality is fundamental for wise management decisions. However, precision in these estimates is lacking. For example, estimates of hunter kill of sea ducks vary by orders of magnitude depending on the

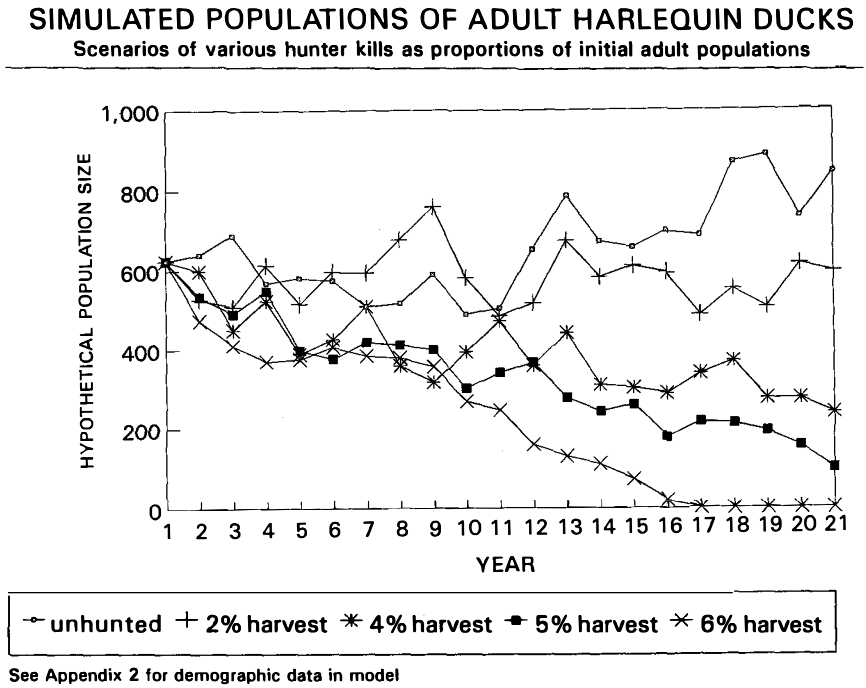


Figure 3. Simulated population growth model for harlequin ducks with and without hunting mortality. Individual lines represent a mean of 10 simulations with random annual productivity having a mean of 1.95 fledged young per experienced female.

approach to sampling hunters (Goudie 1989, Wendt 1989, Wendt and Silieff 1986, Wentworth 1993, Wolfe et al. 1990). Also, actual losses due to oil spill events are thought to be 5 to 10 times the number of observed corpses (*see* Piatt et al. 1990b, Patten and Crawley 1993). Furthermore, mortality of sea ducks in the North Pacific may be exacerbated through sublethal contamination of food chains (Henny et al. 1991, 1994).

Estimating trends. Managers are reluctant to take action until declining trends can be demonstrated, yet most sea ducks lack sufficient survey coverage for trend analyses (Appendix 2). Trends are difficult to generate for sea ducks because of inherent stochasticity in the populations and high standard errors in aerial survey techniques. It is unlikely that we have the luxury of awaiting such tenuous results. Our simulation corroborate the long recovery time necessary to rehabilitate some stocks (>50 years). Therefore, managers should expect very little change in trend statistics over 5 to 10-year periods.

Conclusions

We suggest that a fundamental realignment of our management of sea ducks is needed. The recent listing under endangered species programs of three species of sea ducks in the northern hemisphere suggests that current management practices are inadequate. The poor effectiveness of past management practices stems from a lack of knowledge of the ecology of sea ducks relative to populations of other waterfowl. Because of high sensitivity of sea ducks to very slight changes in adult mortality, we conclude that managers should adopt conservative measures in the management of mortality. In most cases there is insufficient information on which to base wise management decisions, and therefore managers should take a conservative approach because of the slow recovery rate of sea duck populations.

Recommendations

Management

We stress the need for fundamental changes to the current approaches to sea duck management. These include:

(1) Apply sea duck management at a population level which recognizes the existence of high philopatry and discrete geographic sub-populations.
(2) Hunting regulations to reduce or curtail unsustainable annual mortality to adult sea ducks.
(3) Integrate government and subsistence interests to manage spring and summer kills of sea ducks at sustainable levels.
(4) Integrate "sport" and "subsistence" kills into collective management actions.
(5) Control chronic oil disposal and catastrophic oil spills in coastal waters.
(6) Identify and protect key habitat areas, and manage them to limit hunting and disturbance, buffer against intertidal and benthic habitat alteration, minimize contamination and pollution.
(7) Integrate data on sea duck distribution with coastal zone management to ensure development activities, such as aquaculture, mariculture, commercial fisheries and oil exploration, are sustainable.
(8) Improve enforcement of existing and future regulations aimed to conserve sea ducks.
(9) Identify and implement monitoring programs of "indicator" species in suitable geographic areas. These should serve to indicate the status of the guild of sea ducks.

Research

Very little is known of the ecology of sea ducks. Some approaches to improve our understanding include:

(1) Review existing literature, and identify information gaps and establish priorities.
(2) Refine data on basic demographics of sea duck populations in order to improve our ability to model population dynamics.
(3) Improve our understanding of the trophic web of the marine ecosystems through further studies of the ecology of sea ducks during molt and winter.
(4) Initiate long-term studies of sea ducks that aim to identify ecological factors controlling the "boom and bust" phenomenon of productivity of young, and the influence of natural mortality that periodically can be catastrophic (e.g., delayed ice break-up and starvation).

(5) Analyze foraging activity and habitat use of sea ducks to better understand the species-specific requirements for habitat structure.

Acknowledgments

Special thanks to A. Breault, S. Boyd, H. Hogan and G. Kaiser for reviewing various drafts of this manuscript. We are grateful to G. E. John Smith for developing the deterministic model of population growth used to compare relative population growth over time. We are very grateful to Shelagh Bucknell for her patience in processing the document especially the tedious tables, and to P. Whitehead for assistance in preparation of some figures.

Appendix 1. Population sizes, ranges and trends for sea ducks in the North Pacific Rim.

Species	Range		Breeding population[a]	Current 10-year trend	Comments
	Breeding	Wintering			
Someteria mollissima v-nigra	From Victoria Is., NWT west along Beaufort Sea & Bering Sea Coasts of AK, Aleutian Islands & Siberia east from Chaun Bay & along Bering Sea Coast	Bering Sea esp. in Bering Strait near Diomede Is. & St. Lawrence Is. Chukchi Sea & east coast of Kamchatka Peninsula	Unknown *Guestimates:* 81,500 — W. Can. Arctic 25,500 — AK 20,000+ — Russia	Declining in Russia by three- to four-fold since early 1970s. Thought to be declining in Alaska	Hunted species in autumn and winter and especially for subsistence in spring
S. spectabilis	Northern Russia, eastern Siberia, Alaska and Arctic Canada west of Victoria Is.	Bering Sea, notably near Chukotka coasts, St. Lawrence Is., Pribilof Is., Alaskan Pen. & Aleutian Is.	Unknown *Guestimates:* >100,000+ — Russia ~1 million in central & W. Arctic and Alaska	Thought to be declining in Alaska & western Arctic. Stable in Russia	Most common marine duck. Heavily hunted in certain portions of its range, especially for spring subsistence
S. fischeri	Narrow coastal strip from Yana R. to Chaun Bay in Siberia, Y-K Delta & Pt. Barrow to Prudhoe Bay, Alaska	Unknown but probably Bering Sea toward the Siberian Coast	<50,000	Declining in Alaska & western Arctic at 14 percent per year	*Threatened*—listing in U.S. Hunting in Siberia & Alaska for spring subsistence
Polysticta stelleri	In Russia, narrow coastal strip from western Siberian coast to N. of Chukotski Pen., esp. in Lena & Yana deltas, and in AK from Norton Sound to Pt. Barrow	Southern Bering Sea, notably along Aleutian Islands & Alaskan Pen. Occurs along Kamchatka Pen., Commander Is., & Pribilof Islands	<100,000	Declining	*Threatened*—U.S. listing proposed. Hunting in Siberia & AK esp. in spring. Formerly an important component of sport outfitting hunts in AK due to accessibility

Appendix 1. *Continued.*

Species	Range		Breeding population[a]	Current 10-year trend	Comments
	Breeding	Wintering			
Histrionicus histrionicus	In Russia NW coast of Baikal east to Kamchatka & on Commander Is. and S.E. Siberia. Alaska south to British Columbia, Alberta, Washington, Idaho, Montana & Oregon	Commander Islands & Aleutian Islands. Alaskan coast south to California, SE portions of Russia esp. Kamchatka, Kuril Is., Sakhalin Is. to Sea of Japan	USA & Canada—165,000 Russia—*Guestimate:* 50,000–100,000	Declining rapidly in eastern Siberia. Slight decreases in B.C. Retraction from former breeding range in southwestern U.S.	Hunted species especially in Alaska. Breeding population of W. Prince William Sound, AK decimated by *Exxon Valdez* oil spill
Clangula hyemalis	Circumpolar & ubiquitous to Arctic	At edge of ice in Bering Sea, esp. Aleutian Islands, Commander Islands. South to Korea and California	Unknown *Guestimates*: Russia—500,000 USA & Canada—-2.5 million	Declining in AK & Yukon Stable in Russia	Hunted species throughout range esp. in Russia where of considerable commercial importance
Melanitta nigra	Northeast Siberia and NW North America	Kamchatka and Commander Islands. South to Sea of Japan, Korea and occasionally China. Aleutian Islands & E. Alaskan Pen. south to Mexican border	Unknown *Guestimates:* NW N.A. = 282,000 Russia—200,000	Declining in NW North America; possibly stable in Russia	Hunted species. Very important component of market hunt in spring in northern Siberia & sport hunt in eastern U.S.
M. perspicillata	Western Canada to Yukon and northeast Alaska	Aleutian Islands & S. coast of AK to California. Only stragglers to the Pacific northwest	Unknown *Guestimate:* NW N.A. = 536,200	Declining in northwest North America	Hunted species in North America, especially eastern U.S.

Appendix 1. *Continued.*

Species	Range: Breeding	Range: Wintering	Breeding population[a]	Current 10-year trend	Comments
M. deglandi	E. Siberia, notably middle Anadyr, Kolyma, Yakutio Koryak and other high mountain plateaus. Arctic Ocean to Prairie plateau, west to B.C. & east to Manitoba	From Kamchatka Pen. to Japan, Korea & China noteably, Sakhalin & Kurile Islands. From Alaska to California	Unknown *Guestimates:* W.N.A. = 592,600 Russia — <200,000	Possibly unchanged in North America. Declining rapidly in east Siberia where delines of 10-fold are reported since the 1970s	Hunted species especially in eastern U.S. Some market & spring hunting in Russia. Large kills of molters localized
Bucephala clangula	Most of Boreal Forest Zone of Europe, Asia and North America	In Asia, in Issykkal & in Iran, some in India, Baluchistan & Mongolia. Commander Islands & Kamchatka, south in large numbers to Korea, Japan, China & Taiwan. Aleutian Islands to California	Unknown *Guestimates;* W.N.A. ~570,000 Russia ~400,000	Declining in NW N.A. Perhaps stable in Russia	Hunted species. Not of market significance in Asia. Reported to have declined considerably in Europe attributed to deforestation
B. islandica	Southern B.C. to Western Alaska	Alexander Arcipelago, AK to California, including interior lakes & rivers. Accidental straggler in Russia	Unknown *Guestimates:* W.N.A. ~150,000	Declining in B.C.	Hunted species in autumn & winter range
B. albeola	Central and western North America to western Alaska	Aleutian islands to southern California. Only a straggler to Russia	Unknown *Guestimates:* NW N.A. — 887,000	Declining in NW N.A.	Hunted species in autumn & winter range
Lophodytes cucullatus	Areas of boreal western North America	Southeastern Alaska south to California. Coastal & interior lakes & rivers	Unknown *Guestimates:* W.N.A.—15,000	Unknown	Hunted species in autumn & winter range

Appendix 1. *Continued.*

	Range				
Species	Breeding	Wintering	Breeding population[a]	Current 10-year trend	Comments
Mergellus albellus	Forested zone in Russia, Siberia and Far East	Japan, Korea & China. Coastal & large lakes & rivers	Unknown *Guestimates:* Russia = 100,000 Dispersed in low densities	Perhaps stable in Russia	Hunted species, incidental to other diving ducks
M. serrator	Throughout southern tundra lakes of boreal Russia escept for northern coastal zone. Throughout boreal N.A.	From Kamchatka and Commander Islands to Kuril Is., Japan, Korea, China & Taiwan. Aleutian Islands and S.E. AK to Washington	Unknown *Guestimates:* W.N.A. = 237,000 Russia = 100,000	Increasing in W.N.A. Possibly stable in Russia	Hunted species of low interest
M. merganser	Closed boreal forests of Eurasia & N.A.	Aleutian Islands to Mexico and from Kamchatka and Kuril Islands south to Japan, Korea, China & Taiwan	Unknown *Guestimates:* W.N.A. = 641,000 Russia = 140,000	Increasing in W.N.A. Possibly stable in Russia	Hunted species of low interest
M. squamatus	Mid & S. portions of Sikhote—Alin Range & hilly portions of N.E. Manchuria	Korea, China, Tonkin & Burma moreso on river habitats	Unknown but very small 500 to 1,400	Declining rapidly	Very rare with an extremely restricted range Red Book Category 1

[a]Virtually no estimates of wintering populations exist. Because sea ducks do not breed until two to three years of age, juvenile and subadult cohorts can comprise significant components of non-breeding flocks. Few independent measures of juvenile and subadult cohorts exist, and those reported generally are low, i.e., 5 to 10 percent (*see* Joensen 1972, Bourget et al. 1986). Larger components assumed to be immatures and subadults may, in part, comprise adults that have deferred breeding (*see* Bengtson and Ulstrand 1974, Coulson 1984) which can be considerable in some years. Inappropriate assumption of juvenile recruitment/composition can result in gross overestimate of sustainable harvest, for example, *see* Reed and Erskine (1986) on S.m. borealis.

Sources for Appendix 1 and Appendix 2: Alison 1975, Barry 1986, Bellrose 1976, Bocharnokov 1990, Breault and Savard 1991, Brown and Brown 1981, Cassirer et al. 1993, Dau 1977, Dementiev and Gladkov 1952, Degtyarev and Larionov 1982, Dzinbal and Jarvis 1984, Erskine 1972, Flint and Krivenko 1990, Gabrielson and Lincoln 1959, Gerasimov 1990, Gusakov 1988, Hodges et al. 1994, Johnson and Herter 1989, Kertell 1991, Kistchinski 1973, 1980, Kistchinski and Flint 1974, Koehl et al. 1984, Kondratyev 1988, 1989, 1990, Kondratyev and Zadorina 1992, Labutin and Revin 1985, Lobkov 1986, Palmer 1976, Portenko 1952, Savard 1988, Stehn et al. 1993, Vermeer 1981, 1982, 1983, Vermeer and Bourne 1984, Vermeer and Ydenberg 1989.

Appendix 2. Aspects of ecology of sea ducks of the North Pacific Rim

Species	Habitat				Special Notes
	Breeding	Wintering	Molting	Migration	
Somateria mollissima v-nigra	Colonial nester on coastal islets & islands. Many hens raise broods on freshwater lakes and lagoons adjacent to the coast	Shallow coastal waters <20m depth. Extensive use of polynas & leeward open water leads during winter	Poorly documented but occur along Chukchi Sea & Bering Sea coast. Some (<10,000) reported in the Beaufort Sea	May cross land during spring migration e.g., Pt. Barrow, AK and NE. Chukotski Pen, Siberia. Does not migrate very far south, e.g., vagrant in B.C.	Most closely tied to marine habitats than any other sea duck
S. spectabilis	Arctic tundra meltwater ponds & lakes in proximity to the coast. Highest densities reported in Lower Lena & Kolyma lowlands of 1 to 2 pr/km^2 and Prudhoe Bay, AK of 2.3 pr/km^2. Breeding population of 60,000 reported for Banks Is.	Somewhat pelagic & occurs at margin of pack ice, polynas & open waters in ice floes up to 60m depth	Poorly documented but overlaps with areas of winter range	Spectacular migration that often cross close to land in spring & late summer, from Aleutians, AK to W. Can. Arctic. Occur closer to shore during spring and stages at certain locations e.g., Bristol Bay, AK. Uncommon south of Alaska & Siberia	Rarely encountered in winter. Inshore individuals are often juveniles & subadults
S. fischeri	Associated with deltas and coastal plains of large river systems emptying into the Arctic Ocean and Bering Sea. May sometimes form colonies	Unknown. Thought to be offshore at ice edges & polynas on Russian side of Bering Sea	Unknown, perhaps Bering, Chukchi, and Beaufort Sea	Unknown. Arrive from the north to the Yukon-Kuskokwim Delta breeding area	Utilize rich planktonic crustaceans during brood rearing

Appendix 2. *Continued.*

	Habitat				
Species	Breeding	Wintering	Molting	Migration	Special Notes
Polysticta stelleri	Wetlands associated with river deltas. Dispersed breeder with highest densities of 4 pr/km^2 in Russia	Winters in inshore haunts preferring rocky shoals <10m depth	Extensive concentration at lagoons along the n. side of the AK Peninsula	Forms large concentrations in spring in Bristol Bay. Vagrant south of AK and Siberia	It's visibility in winter has resulted in a misconception of abundance
Histrionicus histrionicus	Along rivers and streams of mountainous terrain often associated with limestone bedrock. Densities rarely exceed 1 pr/km of river	Outer marine archipelagoes & headlands to protected rocky shorelines with large tidal amplitudes. Occur in small flocks, general 10s	Remote marine islands & rocky shorelines with an abundance of crustaceans and gastropods. Molts in groups of 10s to 100s	Can congregate in large groupings in spring especially in association with herring spawn along B.C. coast (up to 2,000 to 3,000 individuals)	Mostly feeds on insect larvae on freshwater but may experience enhanced nutrition from Salmonid roe in some areas
Clangula hyemalis	Extensive breeding range encompassing a wide spectrum of ecological land types. Densities can reach 5 pr/km^2 but average about 1 pm/km^2 across the tundra. Somewhat colonial in certain areas	Inshore to offshore marine zones especially headlands & archipelagoes where they occur in small groups, 10s to 100s. Some noteable areas, such as Bristol Bay, AK with 10–15,000 in winter	Poorly known. Some molt along coastal zone of N. slope of AK and NWT, e.g., Tuktoyuktuk Pen.	Poorly known. Cross continental migration likely in North America	Very active & proficient diver. Reported to depths of 50m.

Appendix 2. *Continued.*

	Habitat				
Species	Breeding	Wintering	Molting	Migration	Special Notes
Melanitta nigra	Rocky-shored lakes & ponds of the boreal forest/ tundra zone where densities can reach 0.7pr/km^2	Shallow marine coastal waters <10m usually over substrates of cobbles & boulders	Poorly known & to some extent coincide with portions of winter range. Molting concentration found in coastal AK & NWT, e.g., Tuktoyuktuk Pen.	Poorly known. Some large spring & autumn assemblages observed in the coastal regions of the Queen Charlotte Islands, B.C. Cross continental migration likely in North America. Aggregations over herring spawn noted in spring in B.C.	Very poorly studied species. Die-offs due to possible food-chain contamination at Cape Yakataga, AK
M. perspicillata	Rocky-shored lakes & ponds of the boreal forest/tundra zone esp. with calcareous bedrock influence	Shallow marine coastal waters <10m usually over substrates of pebbles & sand	Poorly known & to some extent may coincide wth portions of winter range. Molting concentration found in coastal AK & NWT, e.g., Tuktoyuktuk Pen. and B.C.	Poorly known. Some large spring & autumn assemblages observed in the coastal regions of the Queen Charlotte Islands, B.C. Cross continental migration likely in North America. Aggregations over herring spawn noted in Spring in B.C.	Virtually unstudied species especially during breeding. Endemic to Nearctic. Die-offs due to possible food-chain contamination in Cape Yakataga, AK

Appendix 2. *Continued.*

Species	Habitat: Breeding	Habitat: Wintering	Habitat: Molting	Habitat: Migration	Special Notes
M. deglandi	Deep lakes rich in crustaceans in the boreal parklands of N.A. & mountain plateaus of Asia. Dispersed densities of 0.45/km^2 reported. Some colonial nesting.	Shallow marine coastal waters <20m over a variety of rocky, pebble & sand substrates	Poorly known & to some extent may coincide with portions of winter range. Molting concentration found in coastal AK & NWT, e.g., Tuktoyuktuk Pen., and B.C.	Poorly known. Some large spring & autumn assemblages observed in the coastal regions of the Queen Charlotte Islands, B.C. Cross continental migration likely in North America. Aggregations over herring spawn noted in spring in B.C. Some spectacular spring migrations of 10s of 1000s along Kamchatka Pen.	Die-offs due to possible food-chain contamination in Cape Yakataga, AK
Bucephala clangula	Obligate (tree) cavity nester of the boreal forest zone	Shallow protected coastal waters <5m as well as interior lakes & rivers. Widespread	Poorly known. Males probably molt on marine coasts whereas females & subadults molt on interior lakes	Poorly known	In winter, roost at night far offshore
B. islandica	Obligate (tree) cavity nester of the boreal forest zone of western North America favoring eutrophic lakes & ponds	Shallow protected coastal water usually of estuarine influence	Totally unknown for adult males. Females & subadults molt on interior eutrophic lakes	Poorly known	Strong philopatry to natural areas
B. albeola	Obligate (tree) cavity nester of the boreal forest zone of North America favoring eutrophic lakes & ponds	Shallow protected coastal waters <5m. Usually feed over cobble, rock & boulder substrates	Totally unknown for adult males. Females & subadults molt on interior eutrophic lakes	Poorly known	Endemic to North America

Appendix 2. *Continued.*

	Habitat				
Species	Breeding	Wintering	Molting	Migration	Special Notes
Lophodytes cucullatus	Obligate (tree) cavity nester of western N.A. favoring wetlands of fluvial systems	Shallow protected temperate coastal waters & interior lakes & rivers	Unknown	Unknown	Endemic to North America
Mergellus albellus	Obligate (tree) cavity nester of forested zones of Russia, Siberia & Far East	Shallow protected temperate coastal waters & interior lakes & rivers	Unknown	Unknown	Endemic to Asia
M. serrator	Widespread on lakes & rivers of n. boreal zone. Frequently nests on islands & can be somewhat colonial. May nest in coastal marine situations	Widespread on shallow coastal marine waters <10m. May remain far north in winter often to the limit of pack ice	Poorly known	Unknown	
M. merganser	Obligate (tree) cavity nester of closed boreal forest zones	Widespread on shallow coastal marine waters to inland lakes & rivers	Unknown	Unknown	
M. squamatus	Obligate (tree) cavity nester of southern boreal zones of east Asia	Primarily rivers	Unknown	Unknown	Very rare & virtually unstudied. Endemic to Asia

Sources for Appendix 1 and Appendix 2: Alison 1975, Barry 1986, Bellrose 1976, Bocharnikov 1990, Breault and Savard 1991, Brown and Brown 1981, Cassirer et al. 1993, Dau 1977, Dementiev and Gladkov 1952, Degtyarev and Larionov 1982, Dzinbal and Jarvis 1984, Erskine 1972, Flint and Krivenko 1990, Gabrielson and Lincolon 1959, Gerasimov 1990, Gusakov 1988, Hodges et al. 1994, Johnson and Herter 1989, Kertell 1991, Kistchinski 1973, 1980, Kistchinski and Flint 1974, Koehl et al. 1984, Kondratyev 1988, 1989, 1990, Kondratyev and Zadorina 1992, Labutin and Revin 1985, Lobkov 1986, Palmer 1976, Portenko 1952, Savard 1988, Stehn et al. 1993, Vermeer 1981, 1982, 1983, Vermeer and Bourne 1984, Vermeer and Ydenberg 1989.

Appendix 3. Life History parameters of four duck species in North America (from Bellrose 1976, Bengtson 1972, Bengtson and Ulfstrand 1971, Cassirer et al. 1993).

Life History Parameter	Mallard	Canvasback	Common Goldeneye	Harlequin
Population size (x 1000)	10667	642	1469	165
Life span (years) (y)	7	10	12	18
Adult survival (S_a)	0.65	0.75	0.75	0.85
Juvenile survival (S_y)	0.35	0.40	0.50	0.50
Average age at first breeding (A)	1	1 to 2	2	3
Clutch size	9	7.9	8.8	5.6
Renesting capacity (additional nests per pair) (R)	1.15	0.5	0.3	0
Fledged young per female (F)	4	3	3	2
Nest success (K)	0.6	0.6	0.8	0.9
Philopatry (probability of return)	0.1	0.75	1	1
Rate of non-breeding	0	0	0	0.44

Where N_{ij} = the size of the population of age *j* in year *i*
(age 0 indicates a juvenile)
and $N_{i+1,1} = N_{i,0} * S_y$
$N_{i+1,j+1} = N_{ij} * S_a$ if $j>0$

$$N_{i+1,0} = \frac{KF}{2}(1+R)\,S_a \sum_{j=A}^{y} N_{ij}$$

Then, the total population in year i (T_i) is

$$T_i = \sum_{j \geq 0} N_{ij}$$

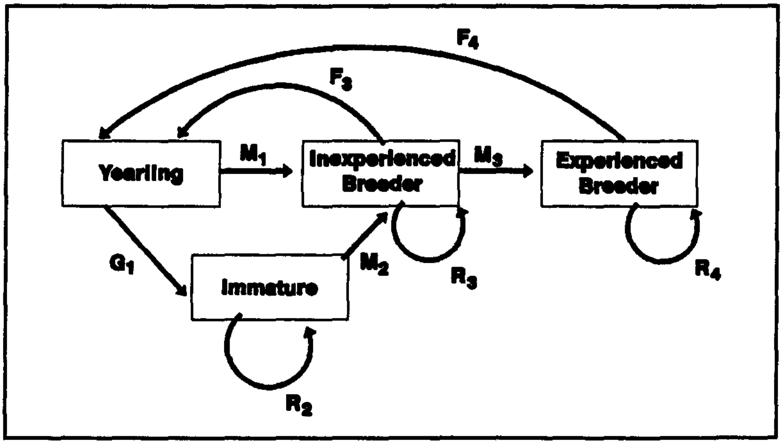
Populations are projected over 20 years once the stable age distribution has been established (Figure 2).

Appendix 4. Life history parameters used in the estimation of the population rate of growth, and diagramatic representation of stage-classified matrix model for the harlequin duck.

Life History Parameter		Value	Elasticity[a]
Yearling survival	(S_1)	0.5	0.138
Immature survival	(S_2)	0.75	0.2482
Survival of inexperienced breeders	(S_3)	0.85	0.1434
Survival of experienced breeders	(S_4)	0.85	0.4706
Probability of yearling maturation	(P_1)	0.05	0.0074
Probability of immature maturation	(P_2)	0.32	0.0657
Proportion of females breeding	(E)	0.66	0.1627
Mean fecundity of inexperienced	(M_3)	0.55	0.0134
Mean fecundity of experienced	(M_4)	1.95	0.1247

[a]The proportional sensitivity of population growth to changes in respective life history parameters.

LIFE CYCLE REPRESENTATION



$G_1 = S_1 \cdot (1-P_1)$
$R_2 = S_2 \cdot (1-P_2)$
$M_2 = S_2 \cdot P_2$
$R_3 = S_3 \cdot (1-P_3)$
$M_3 = S_3 \cdot E_3$
$R_4 = S_4$
$F_3 = S_3 \cdot E_3 \cdot M_3$
$F_4 = S_4 \cdot E_4 \cdot M_4$

$M_1 = S_1 \cdot P_1$ where:
S_i = survival probability at stage i
P_i = maturation probability at stage i
M_i = mean fecundity at stage i
E_i = proportion of stage i birds breeding

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Management of Pacific Brant: Population Structure and Conservation Issues

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Introduction

Pacific brant (*Branta bernicla*) nest from the Yukon-Kuskokwim (Y-K) Delta in southwestern Alaska along the coast of North America to the central Canadian arctic (Pacific Flyway Subcommittee on Pacific Brant 1992) (Figure 1). Birds from this population also nest in the Canadian arctic islands south of Prince Patrick Island and on the coast of the Chukotka Peninsula. Brant nest principally in colonies associated with productive river deltas but isolated nests and small aggregations are common, especially in the arctic. Most Pacific brant breed on the Y-K Delta (Sedinger et al. 1993).

The Pacific brant population is comprised of two distinct genetic stocks (Shields 1990): (1) the gray-bellied form that breeds nearly exclusively on Melville, Prince Patrick and adjacent islands in the Canadian high arctic (Boyd and Maltby 1979); and (2) black brant (*B. b. nigricans*), with a broader breeding range. Band recoveries show that the gray-bellied brant winter exclusively in Padilla Bay in northern Puget Sound, Washington (Boyd and Maltby 1979, Reed et al. 1989a). Since 1980, 80 percent of the counted Pacific population, or 91 percent of black brant have wintered in Mexico (Pacific Flyway Subcommittee on Pacific Brant 1992), a small percentage of which stop in Puget Sound during autumn migration (Reed et al. 1989a). Most of the remainder of the population winters along the Pacific coast of North America from Izembek Lagoon through California. A small number of brant from this population (<5,000) has been described wintering in Japan and Korea (Owen 1980). Brant begin moving north by early February, with a large proportion of brant wintering in Mexico stopping in San Quintin Bay in northern Baja California (Ward et al. 1993a). About 18 percent of the population uses the Strait of Georgia in southern British Columbia during March and April (Nygren 1991). Some brant remain in California and Oregon until early June.

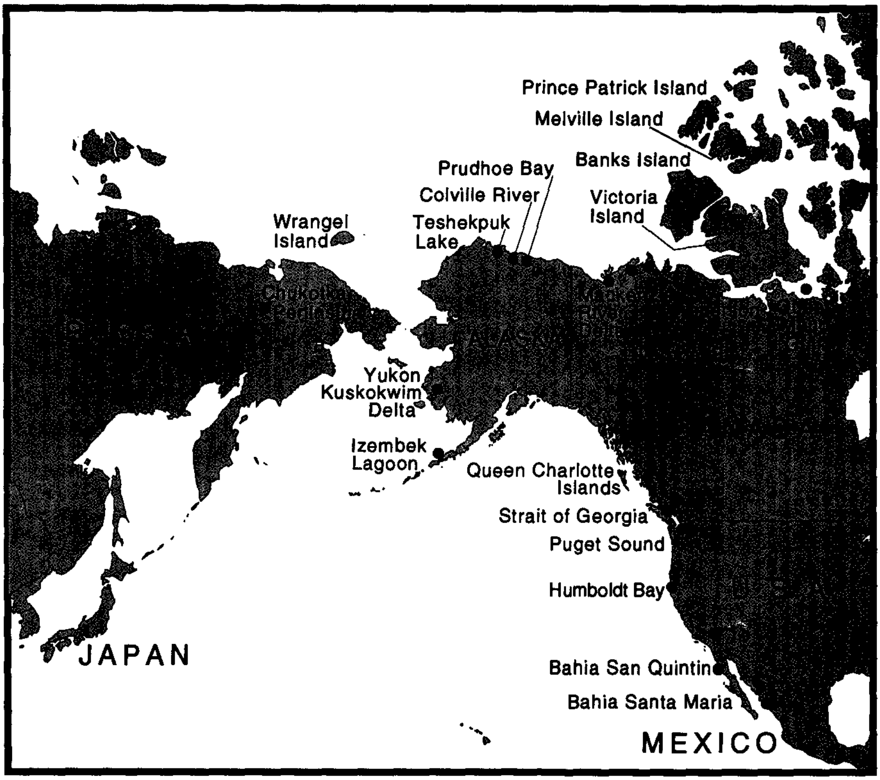

Figure 1. Locations of important Pacific brant breeding, molting, migration and wintering areas.

The distribution and size of the Pacific brant population has been dynamic over the last three decades. The most dramatic change is the reduction in the number of brant using bays along the California coast during February and March, from an average of 26,000 brant in the 1950s to 8,000 in the 1980s (Pacific Flyway Subcommittee on Pacific Brant 1992). It is unclear whether this trend represents a population decline or a change in spring migration behavior of Pacific brant, because winter surveys were inadequate to detect temporal and spatial shifts in migration areas. Sport harvest in California was changed to autumn (from February) and substantially reduced in an attempt to allow reestablishment of "traditional" patterns. Washington state closed its sport harvest of brant during the years of 1983–1986 in response to declining numbers of brant wintering in Padilla Bay (Pacific Flyway Subcommittee on Pacific Brant 1992).

Numbers of brant nesting in two of three major colonies (>1,000 pairs) monitored on the Y-K Delta declined substantially in the 1980s (Sedinger et al. 1993). Numbers of dispersed-nesting brant also are thought to have declined during this period, continuing a trend, likely extending back to the 1970s. Declines in brant nesting on the Y-K Delta occurred coincidentally with those of three other species, emperor geese (*Anser canagicus*), greater white-fronted geese (*A. albifrons frontalis*) and cackling

Canada geese (*B. canadensis minima*) (Raveling 1984, King and Derksen 1986). As a result of concern about the Pacific brant population, brant were included in the Y-K Delta Goose Management Plan, established to foster cooperation among Pacific Flyway states, the federal government and subsistence users in the recovery and management of geese nesting on the Y-K Delta (Pamplin 1986). The plan establishes a population objective of 180,000 Pacific brant, based on the midwinter survey and calls for the cessation of sport and subsistence harvest when the three-year moving average of the midwinter survey falls below 120,000, which nearly occurred in 1984, and in 1993 was only prevented by inclusion for the first time of brant wintering in Alaska (Pacific Flyway Subcommittee on Pacific Brant 1992).

Recent analysis of autumn age ratios at Izembek Lagoon and reproductive parameters (nest success, clutch size, gosling survival and number of nesting pairs) on the Y-K Delta indicate that >75 percent of total production of young occurs on the Y-K Delta, which is consistent with historic knowledge of the breeding distribution (Spencer et al. 1951, Sedinger and Derksen 1992, Derksen and Ward 1993, Sedinger et al. 1993). These analyses estimated >80,000 nonbreeders and failed breeders in the population during July and August 1990. Only about 28,000 nonbreeders could be accounted for in 1990 at the best studied molting areas, Teshekpuk Lake and Wrangel Island, combined.

An extensive color-marking and observation program has been conducted during the last decade throughout the range of Pacific brant. In addition, new technologies for monitoring brant recently have been developed. New data and enhanced monitoring capabilities, combined with concern about management of brant, make an examination of management practices and objectives timely. Our goal in this paper is to review recent data on Pacific brant, evaluate current and past surveys, and recommend a new basis for management of these populations.

Surveys

Midwinter Survey

The midwinter survey is the principal survey used for management of the Pacific brant population. This survey is flown in January, along the Pacific coast of Baja California and the mainland coast of Mexico north of, and including, Bahia Santa Maria, combined with surveys of bays and estuaries used by wintering brant in California, Oregon, Washington and, since 1993, Alaska. These areas (except Alaska) have been completely surveyed annually since 1961 (Pacific Flyway Subcommittee on Pacific Brant 1992). Brant wintering at Izembek Lagoon have been surveyed since 1986. Brant are known to winter in the Queen Charlotte Islands and the Strait of Georgia of British Columbia (Hansen and Nelson 1957), but regular surveys of these areas are not conducted. Traditionally, numbers of brant wintering in Mexico, California, Oregon and Washington have been used to calculate the midwinter index used to make decisions about harvest management. In 1993, brant wintering in Alaska were added to the midwinter index for making management decisions (Bartonek 1993).

The midwinter Pacific brant survey may be among the most accurate midwinter goose surveys because brant winter in well-defined locations in reasonable numbers for estimating flock size using standard methods (Conant et al. 1993). Nevertheless, there are several problems with the midwinter Pacific brant survey that reduce its effectiveness as a management tool. First, as now constituted, the survey combines brant from two distinct populations, gray-bellied and black brant. The midwinter population index has been somewhat erratic over its history; variation (s_{yx}) around the long-term trend is 16 percent of current population size. Two years, 1964 and 1981, deviated substantially from preceding and following years. The 1964 estimate, 185,282, was 32 percent higher than that from the previous year and 11 percent higher than the succeeding year, while the 1981 estimate was 33 and 60 percent higher than estimates from adjacent years. Such variation is not atypical for midwinter surveys of geese, but these fluctuations exceed the estimated change in the Pacific brant population over the period 1961–1993.

Between 1981–82 and 1986 the number of nests on the Y-K Delta declined by about 12,000, or 24,000 birds (Sedinger et al. 1993). We regressed the Mexican portion of the midwinter index (thus excluding Melville-Prince Patrick Island brant) against year from 1980 through 1987, which is the last year of expected decline if production was low through 1985. We used the mean indices from 1978–80 as an estimate of the number of birds in 1980. We excluded the 1981 index which was 33 percent larger than the 1980 index and 60 percent larger than the 1982 index. Even with this selective use of data, the relationship between the midwinter index and year was not significant ($r^2 = 0.3$, $P > 0.05$). It is, however, interesting that the predicted decline from the regression between 1980 and 1987 was 25,823, within 7 percent of that expected, based on the decline in nesting brant on the Y-K Delta. The failure of such dramatic declines to produce a significant trend in the midwinter survey indicates the difficulty of management of black brant, based only on the current midwinter survey.

An additional assessment of the midwinter survey is provided by comparing changes in the midwinter survey with production of young each year. We calculated an index of number of young in autumn by multiplying the previous year's midwinter index by the current year's autumn age ratio at Izembek Lagoon (Pacific Flyway Subcommittee on Pacific Brant 1992). This index of production was only weakly correlated with change in the midwinter index from one year to the next ($r = 0.17$, $P > 0.05$). We would expect that changes in the number of brant estimated in the population in January should be correlated more closely with production of young the previous summer; the population should increase following summers of good production.

Autumn Survey

Number of brant staging at Izembek Lagoon before autumn migration have been counted nearly annually since 1975. Multiple counts (≥4) have been conducted each year since 1984, except 1985 and 1986, when two and three counts, respectively, were conducted (Hodges and Conant 1992, C. P. Dau, U.S. Fish and Wildlife Service, personal communication). Multiple counts offer the advantage that the precision of the estimates can be calculated. Since 1984, SE's of estimates have averaged 13 percent of the estimated number of brant at Izembek Lagoon. The accuracy of these counts has not been assessed because of the general difficulty in estimating the number of brant in flocks on the water other than by aerial survey. Estimating numbers of geese in large flocks has been shown, however, to have the potential for large errors (McLandress 1979). Experimental video surveys of brant at Izembek Lagoon estimated 119,077 ±26,429 were present (Anthony 1992), while ocular based estimates averaged 112,115 ±22,480 (Hodges and Conant 1992). The video-based survey has technical problems, such as missing or double-counting flushing birds, and the effects of changing tide levels on distribution of brant during sampling. This technique, however, shows promise for estimating the number of brant in the population, as it has for colonially nesting brant (Anthony et al. 1994).

Autumn Age Ratios

Proportion of young in the population has been estimated each autumn since 1963 based on the number of individuals with juvenal plumage (Pacific Flyway Subcommittee on Pacific Brant 1992). These estimates have high precision (SE < 2 percent of the estimate) based on examination of >5,000 individuals each year. Age ratios at Izembek Lagoon do not reflect production by gray-bellied brant, however, because brant from this population occupy a segment of Izembek Lagoon that is relatively inaccessible to observers (Reed et al. 1989b) and are, therefore, not adequately sampled. Age ratios for black brant also may be biased in some years because timing of estimation of age ratios has been inconsistent, relative to the migration of the breeding and nonbreeding segments of the population. Nonbreeders and failed breeders radio marked on arctic molting areas arrive later in autumn at Izembek Lagoon than breeding brant from the Y-K Delta (D. H. Ward unpublished data). Also, arctic breeding brant arrive later than those from the Y-K Delta (Reed et al. 1989b). Therefore, unless age ratios are determined at the same time each year, relative to the arrival of these various populations segments, considerable annual variation unassociated with actual production, will be introduced into age ratio estimates.

Molting Areas

Two important molting areas currently are recognized, the large oriented lakes northeast of Teshekpuk Lake on Alaska's north slope (Derksen et al. 1979) and Wrangel Island (Ward et al. 1993a). Numbers of molting brant using the Teshekpuk Lake area have been counted using aerial surveys annually since 1982 (R. J. King, U.S. Fish and Wildlife Service, unpublished data) and numbers of brant molting on Wrangel Island were counted in 1990 (Ward et al. 1993a). These two areas account for only about 36 percent of the nonbreeders and failed breeders in most years (Sedinger et al. 1993). A significant proportion (45 percent) of variation in numbers of brant molting in the Teshekpuk Lake area between 1982 and 1989 was explained by nest success in the Tutakoke and Kokechik Bay colonies on the Y-K Delta; numbers of brant increased at Teshekpuk Lake in years when nest success on the Y-K Delta was lower (Sedinger and Derksen 1992). Past surveys of these areas have contributed substantially to our understanding of the Pacific brant population but annual surveys of these molting areas are unlikely to play an important role in brant management in the future. Resources currently used to survey the Teshekpuk Lake area could be better directed toward locating and surveying other important molting areas, especially the Y-K Delta.

Surveys in Canada

Regular surveys are not flown in Canada so our understanding of the distribution of black brant in Canada is less precise than that for Alaska. Recent surveys of Queen Maud Gulf indicate from 3,000 to 6,000 black brant present in late summer; few of these are breeders (R. Alisauskas, Canadian Wildlife Service, unpublished data). An estimated 12,000 brant occurred on Banks Island in 1993; as many as 50 percent of these could be breeders but this remains to be verified (J. Hines, Canadian Wildlife Service, personal communication). Bromley estimated about 2,000 breeding and molting brant on Victoria Island in the late 1980s (J. Hines, Canadian Wildlife Service, personal communication) and Hines (personal communication) estimated an additional 5,800 brant, of which 50 percent were breeding, in the Liverpool Bay-Mackenzie River Delta area during 1991–93.

Harvest

Historically, total harvest of Pacific brant was approximately 20,000 annually (Sedinger et al. 1993), which represented 12 percent of the average midwinter index in the 1960s. This level of harvest is well below sustainable harvest levels for other North American goose populations, which frequently exceed 20 percent of the population annually (Grieb 1970, Hanson and Eberhardt 1971, Boyd et al. 1982, Brownie et al. 1985). The harvest of Pacific brant differs from that of other goose populations in several ways, however. First, adults generally represent >60 percent of the Pacific brant harvest (Pacific Flyway Subcommittee on Pacific Brant 1992, D. Ward unpublished data), whereas adults usually comprise a smaller proportion of the bag in other geese, except Canada geese (Padding et al. 1992). Therefore, brant harvest may have a greater impact on the breeding population than is true for other geese. Second, the subsistence harvest on the Y-K Delta is concentrated during the migration period for breeding pairs from the Y-K Delta, and before migration of nonbreeders and breeders from arctic nesting areas. Spring subsistence harvest on the Y-K Delta may be comprised nearly entirely of breeding brant from this area (Sedinger et al. 1993), and bird for bird, will have a substantially greater impact on the breeding population than sport harvest. Finally, harvest in Washington state is concentrated on gray-bellied brant (Reed et al. 1989a) and must be evaluated based on its effect on this population alone. For example, brant harvest in Washington averaged 5,200 per year between 1969 and 1976, which represented 78 percent of the average midwinter index for Washington in those years. These harvest levels clearly were not sustainable; the brant population wintering in Washington declined 67 percent between the early 1960s and the early 1980s, precipitating a complete closure of the harvest in Washington in 1983.

Harvest levels have been dramatically lower during the 1980s and '90s. Total sport harvest of black brant in the states of Alaska, Oregon and California has been less than 1,000 brant annually in the 1990s from a population (excluding brant wintering in Washington state) exceeding 100,000 brant. Annual harvest in British Columbia and Mexico totaled about 1,800–2,600 brant in 1990–92 (D. Ward unpublished data, I. Goudie, Canadian Wildlife Service, unpublished data). Complete closure of sport harvest throughout the Pacific Flyway, including Canada and Mexico, would improve annual adult survival by only about 3 percent and we conclude that reduction of sport harvest below known current levels is unlikely to significantly benefit the Pacific black brant population. We acknowledge, however, that there are limitations in the harvest data and better information on age composition, origin and total numbers of brant in each component of the harvest is needed.

Harvest of brant in Washington state has been between 800 and 900 birds annually in the 1990s, which represents about 6 percent of the brant wintering in Washington (Washington Department of Game unpublished data). These should represent sustainable harvest levels unless several years of poor production on the arctic breeding grounds occur sequentially or gray-bellied brant are subject to substantial subsistence harvest in Canada. Because of the greater relative harvest levels on this population, however, greater impacts on population dynamics are likely to be achieved by managing sport harvest of gray-bellied brant than is the case for black brant.

Estimated subsistence harvest by Yupik Eskimos on the Y-K Delta in Alaska apparently has declined from ca. 8,000 in the 1960s (Klein 1966) to an average of 2,200 since 1985 (Wentworth 1993). Subsistence harvest on the Y-K Delta represents about 5 percent of the breeding adults on the Y-K Delta. Because the Y-K Delta segment of the breeding population experiences both subsistence and sport harvest, a complete harvest closure likely would improve adult survival for these brant.

Dynamics of the Pacific Brant Population

Numbers of Pacific brant in the midwinter index have declined significantly ($r^2 = 0.34$, $P < 0.01$) since 1961 (Figure 2) and the three-year moving average has approached 120,000, the lower threshold that triggers a complete harvest closure, twice since 1980. Net reduction in the midwinter index between 1961 and 1993 is 39,138, based on the regression of the midwinter index on year. Of this decline, 8,351 fewer brant wintered in Washington in the 1990s than in the first half of the 1960s (Pacific Flyway Subcommittee on Pacific Brant 1992). Based on current wintering distributions, most of these brant were from the population breeding on Melville and Prince Patrick islands. Numbers of brant wintering in Mexico declined by 25,480 between the early 1960s and the 1990s (Pacific Flyway Subcommittee on Pacific Brant 1992). We note that this decline is partially a result of a large shift (7,000 brant) into California coincident with heavy rains in Baja California during the winter of 1992–93 (Ward et al. 1993b).

Relatively high midwinter counts in California during the 1950s do not reflect the California wintering population because these counts generally were conducted in February and March (California Department of Fish and Game unpublished data), well after brant began migrating north from Mexico (Ward et al. 1993b). The midwinter survey for California in 1962 was conducted in February and was more than 15,000 greater than either the preceding or following years, when surveys were done in January. We believe that problems in timing and methodology in California surveys conducted before the 1960s argue against use of these surveys to estimate the potential number of black brant wintering in California.

Numbers of brant using California bays during spring migration apparently have declined since the 1950s, which could represent changes in migration behavior, which are common in North American geese (Bellrose 1968, Owen 1980), rather than reductions in the size of the population. We suggest that the decline in the number of brant wintering in Mexico between the 1960s and the 1990s, 25,480 brant, represents the most reasonable estimate of the potential for the black brant population to increase. Based on the current proportion of breeders in the black brant population, such an increase would add approximately 3,700 nests to the population. We note

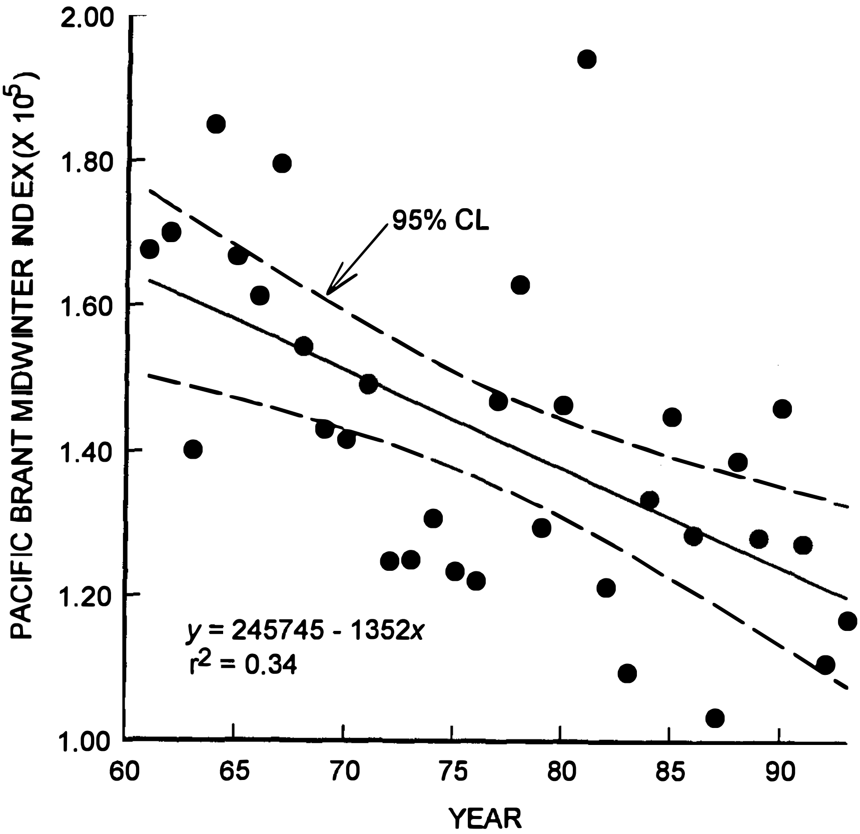

Figure 2. Pacific brant midwinter index, 1961–93. Index generated from the annual midwinter survey conducted by Migratory Bird Management, USFWS and the Pacific Flyway states (Pacific Flyway Subcommittee on Pacific Brant 1992).

that three of the four major colonies on the Y-K Delta have been relatively stable in numbers during the 1990s (R. M. Anthony unpublished data), although the Kigigak Island and Tutakoke River colonies could increase by a combined total of 5,800 nests to return to their sizes of the early 1980s (Sedinger et al., 1993, R. M. Anthony unpublished data). Total brant counted on breeding pair surveys for geese on the Y-K Delta have increased steadily since the mid-1980s (W. L. Butler, unpublished data), which, combined with the patterns observed within the major colonies, suggests a steady increase in the numbers of brant nesting as single pairs and small aggregations outside the major colonies. We currently are unsure how many dispersed-nesting brant the Y-K Delta can support and analysis of current surveys and dynamics will be necessary to estimate this potential.

Dynamics of arctic colonies are more poorly understood but sufficient data exist to document changes in three nesting areas during the last two decades. Historically, the largest known colony in the arctic, at the mouth of the Anderson River (Barry 1967) contained ca. 1,000 nests. Since 1990, this colony has contained only about

300 nests (Sedinger and Derksen 1992, Sedinger et al. 1993). In contrast, the Colville River Delta supported fewer than 100 brant nests in the early 1960s (Pacific Flyway Subcommittee for Pacific Brant 1992), but currently about 400 pairs of brant nest there (Derksen and Ward 1993). Annual monitoring of numerous small colonies in the Prudhoe Bay area during the 1980s indicates a generally increasing trend in nesting brant over this period (Ritchie et al. 1990). Nevertheless, numbers of nests and nest success have fluctuated dramatically in response to weather conditions during nesting and the local presence of foxes. These fluctuations in production are consistent with those observed by Barry (1967) and indicate that maintenance of arctic colonies is dependent on relatively infrequent successful breeding. Increases in numbers of brant in the Colville River Delta are partially associated with deterrence of predators (P. Martin personal communication) and recent increases in numbers of brant nests in the Prudhoe Bay area also could be associated with changes in the predator community or predator behavior associated with human presence in the oil fields. The decline in numbers of brant nesting in the Anderson River Delta may have resulted from substantial deterioration of salt marsh foraging habitat (M. S. Lindberg personal communication) similar to that on the west coast of Hudson Bay (Kerbes et al. 1990). A small number of black brant previously nested at several locations on the Seward Peninsula (Pacific Flyway Subcommittee on Pacific Brant 1992), but brant were absent from one of the largest of these areas, at the Nugnugaluktuk River in 1992 (E. Peltola personal communication).

Addition of 25,480 black brant to the three-year moving average would produce an index of 129,333, excluding gray-bellied brant. We note that 9,100 fewer brant wintered in Washington during the 1990s than the early 1960s. Restoration of brant numbers in Washington to historic levels would produce a total midwinter index of 153,000. This value is comparable to midwinter indices of the 1960s, excluding 1962 when double counting occurred, and 1964, which was an outlier. Also, the three-year average Pacific brant index has exceeded this level only twice since 1970.

Management of Pacific Brant

Population Considerations

We believe that recent studies throughout the range of Pacific black brant and the development of new survey methods have created the opportunity to redesign the basis for management of the black brant population. Because analyses are not yet sufficiently complete to recommend specific management formulas, our purpose here is to propose a new approach to the management of Pacific brant for consideration when the Pacific Flyway brant management plan is revised.

Paramount to revision of our thinking about the Pacific brant population is an understanding of (1) the basic structure of the brant population in the Pacific Flyway and (2) the capabilities and limitations of past and current surveys. The most fundamental component of structure in the Pacific brant population is the genetic and distributional differentiation between gray-bellied brant that nest primarily on Prince Patrick and Melville islands and winter in Padilla Bay, Washington, and black brant that nest primarily on the Y-K Delta (but throughout an extensive range) and winter principally in Mexico. These two stocks should be explicitly recognized and managed separately.

The black brant population can be subdivided into three components for the purpose of interpreting surveys and assessing management action: successful breeding adults, nonbreeders and failed breeders, and young-of-the-year. Presently, breeding adults comprise about 30 percent of the autumn population (Sedinger et al. 1993), while young-of-the-year have represented 16 to 28 percent of the autumn population in the 1990s (Pacific Flyway Subcommittee on Pacific Brant 1992). Nonbreeders comprise the remaining 40–50 percent. It is currently not possible to more precisely partition the autumn population because we lack understanding of geographical and temporal variation in population parameters, such as nest success and postfledging survival. Nevertheless, there is little doubt that nonbreeders have comprised a substantial proportion of the population in recent years. The presence of this large pool of nonbreeders influences population indices based on autumn or midwinter surveys, but because they do not contribute directly to production, should be of less concern to managers than breeders. The fact that the nonbreeding component of the population is substantially larger than the number of yearlings, produced the previous year, indicates that some factors are limiting the number of breeding brant in the population, similar to the situation in European populations of geese (Ebbinge 1985). Furthermore, repeated recaptures of adult-plumage individuals on molting areas indicate that a substantial fraction of the nonbreeding segment of the population remain in this status for up to three consecutive years (Ward et al. 1993a, K. S. Bollinger unpublished data).

Despite the persistence of a consistently nonbreeding component of the population, we believe an understanding of the movement between the nonbreeding and breeding population is important to our understanding of the potential size of the black brant population. For example, our best assessment of the current potential of the population to increase would not completely replace all of the nests thought to have disappeared from the Y-K Delta if the current proportion of breeders and nonbreeders in the population persisted. Dynamic interchange between the breeding and nonbreeding components of the population is further suggested by the increase in the number of breeding pairs on the Y-K Delta since the mid 1980s while the midwinter index generally has declined over the same period.

In contrast to nonbreeders, declines in the breeding population will reduce production of young in the short term. Therefore, changes in the breeding population should trigger management decisions more rapidly than similar declines in the nonbreeding population.

Fluctuations in the nonbreeding segment of the population should not be of concern to managers in the short term because this segment of the population can quickly be replaced by a few years of successful breeding. For example, the entire nonbreeding segment of the population could be replaced by five years of successful reproduction, based on current estimates of annual survival (Ward and Sedinger unpublished data). The nonbreeding segment of the population provides a buffer that potentially can replace individuals lost from the breeding population, although the dynamics of this process currently are unknown. This process may partially explain the rapid increase in numbers of nesting brant on the Y-K Delta following several years of nesting failure in the early 1980s (Sedinger et al. 1993), although the presence of such a large number of non-breeders in a population which still is increasing requires explanation.

Data Required for Management and Management Objectives

We believe that the Pacific black brant population can be best managed using a combination of breeding pair surveys on the Y-K Delta, surveys of the population in autumn at Izembek Lagoon and age-ratios in the population at Izembek Lagoon in autumn. Surveys on the Y-K Delta should combine videographic surveys of major colonies (Anthony et al. 1994), which have SEs of about 10 percent of estimated colony sizes, and estimates of the number of brant nesting in small aggregations and singly. These latter estimates can rely on the breeding pair surveys already conducted to estimate nesting pairs of dispersed geese on the Y-K Delta by Migratory Bird Management, U.S. Fish and Wildlife Service. Data from breeding pair surveys must be adjusted to eliminate double counting of brant in the major colonies, which also are estimated using videography. This combination of surveys will allow managers to closely monitor the predominate segment of the breeding population, annual production and the size of the entire population. We recommend that population objectives and population thresholds for management of harvest be based on this group of surveys. Intensive analyses of demographic parameters and modeling of population processes currently is in progress, based on extensive data generated from intensive color marking and resightings over the last seven years. Biologically realistic population objectives can be developed as a result of these analyses.

We recognize that our recommendations deviate from current management practices. Nevertheless, we believe our proposal provides for management based on population processes most responsible for achieving population goals: breeding pairs and production of young. Our proposal seemingly ignores the arctic segment of the breeding population. The current system of monitoring, with its substantial potential for error, fails to adequately track small arctic breeding colonies. Complete loss of the entire arctic component of the breeding population would not likely be detected by the current midwinter survey. If there is concern about these brant they must be monitored on their breeding areas.

Our proposal also would replace the black brant midwinter survey as the principal management tool for the population. Abandonment of this survey would result in the loss of substantial data on winter distribution, especially in Mexico. Impending development in key wintering locations in Mexico may require regular surveys of these areas to monitor the effects of development on brant distribution, although it may not be necessary to conduct these surveys annually. Individual states within the Pacific Flyway likely will continue to monitor winter distribution within their states to address local management concerns. Management of gray-bellied brant can be accomplished best by continuing to survey these brant during winter in Padilla Bay, Washington. Additional data on the composition of the harvest in Washington would enhance the management of this population. While harvest levels generally are low, the existence of an extensive marking program has created a unique opportunity to characterize the age and sex composition, and geographical derivation of the harvest throughout the Pacific Flyway. We encourage increased efforts to recover data from harvested birds. Better characterization of the harvest, combined with enhanced surveys and population modeling will allow for more precise management of Pacific brant in the future.

We have focused primarily on issues of population assessment and management. It is important to recognize that, ultimately, the health of Pacific brant populations will depend on preservation of habitats throughout their range. Substantial development already has occurred in migration areas within the U.S. and Canada, and there are threats to critical wintering areas in Mexico. We strongly encourage monitoring of development activities in estuaries used by brant and protection of these areas where appropriate.

Acknowledgments

R. N. Smith, Deputy Director, U.S. Fish and Wildlife Service (USFWS), established funding for brant research and challenged us to consider management questions. R. Pospahala and R. West, USFWS, provided support for aspects of this work and helped us coordinate these international studies with Pacific Flyway states. S. Kohl, P. Ward and C. Hayes (USFWS) were especially helpful in coordinating our cooperative studies with colleagues in Russia, Canada and Mexico. In Russia, S. and I. Kharitonov (Bird Ringing Center, Moscow) and staff of Wrangel Island Reserve provided logistical support and assistance in banding and censuses. In Mexico, Secretaria de Desarrollo Social (SEDESOL) staff E. Ezcura, J. Aguilar and M. Martinez assisted with permitting and collection of data. We also thank the staffs of Humboldt Bay, Izembek, Tijuana Slough and Yukon Delta National Wildlife Refuges for logistic and financial support, and cooperation in collecting data. Y. Miyabayashi (Japanese Association of Wild Geese Protection), P. Martin (USFWS), R. Ritchie and B. Anderson (Alaska Biological Research), S. Johnson (LGL Ltd.), J. Hines (Canadian Wildlife Service [CWS]) and R. Bromley (NWT Department of Renewable Resources Management) cooperated in brant marking and resightings. R. Alisauskas (CWS), W. Butler and C. Dau (USFWS), M. Lindberg (University of Alaska), P. Martin and D. Yparraguirre (California Department of Fish and Game) provided unpublished data. We benefited from discussions with W. Butler, R. King (USFWS), J. Hines and R. Stehn (National Biological Survey), although conclusions and recommendations presented here are our own.

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Conservation of North Pacific Shorebirds

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Introduction

In his introduction to the 1979 Symposium proceedings entitled "Shorebirds in Marine Environments," Frank Pitelka stressed the need for studies and conservation programs that spanned the western hemisphere (Pitelka 1979). In the 15 years since Pitelka's call to arms, the locations of many important migratory and wintering sites for shorebirds have been identified in the Americas (Senner and Howe 1984, Morrison and Ross 1989, Morrison and Butler 1994) and in the East Asian-Australasian flyway (Lane and Parish 1991, Mundkur 1993, Watkins 1993). However, assessments for Central America, the Russian Far East and most of Oceania remain incomplete or lacking.

The recognition that shorebird conservation required the protection of habitats throughout the birds' range (e.g., Morrison 1984, Davidson and Evans 1989 *in* Ens et al. 1990) prompted the establishment of the Western Hemisphere Shorebird Reserve Network (WHSRN) in the Americas in 1985 (Joyce 1986). This program complemented the 1971 Convention on Wetlands of International Importance Especially for Waterbirds (Ramsar Convention, Smart 1987), recognized by more that 50 countries world-wide.

Our purpose for writing this paper is to: (1) describe the distribution of North Pacific shorebirds throughout their annual cycle; (2) review the locations of and threats to important sites used by North Pacific shorebirds during the breeding, migration and wintering periods; and (3) outline a program for international conservation of Pacific shorebirds.

Distribution in the North Pacific

The North Pacific region is the area bounded by British Columbia, Alaska and the Russian Far East. The status, distribution and scientific names of the 93 species and subspecies of shorebirds that occur in this region are shown in Table 1.

Breeding

The North Pacific region represents a relatively small portion of the Holarctic landmass, but it is one of the world's most important breeding areas for shorebirds. The region not only supports a disproportionately large assemblage of species with a high degree of endemism, but also hosts the majority of the global populations for many other more widespread taxa. Compared to the world's shorebird fauna, the portion breeding in the North Pacific is represented by 4 of 12 families, 22 of 55 genera and 75 of 212 species (Table 1). This region, more so than anywhere else in the world, is characterized by the Scolopacidae, the largest and most diverse of the shorebird families. Within the North Pacific, the Scolopacidae are represented by 17 of 22 genera (77 percent) and 65 of 87 species (75 percent). The polytypic genera within this family are especially well represented within the region. All species of godwits, shanks, phalaropes, dowitchers and turnstones (genera *Limosa, Tringa, Phalaropus, Limnodromus* and *Arenaria*, respectively), 7 of 9 species of curlews (Tribe Numeniini), and 17 of 19 species of typical sandpipers (genus *Calidris*) breed in the North Pacific. Lastly, several of the genera and many of the species in this family largely are endemic to the region or the majority of their populations occur there. These include the monotypic genera *Eurynorhynchus* (spoon-billed sandpiper) and *Aphriza* (surfbird), both species of tattlers (*Heteroscelus incanus* and *H. brevipes*), black turnstone (*Arenaria melanocephala*), bristle-thighed curlew (*Numenius tahitiensis*), western sandpiper (*Calidris mauri*), all five races of rock sandpiper (*C. ptilocnemis*), great knot (*C. tenuirostris*), American black oystercatcher (*Haematopus bachmani*), and the endangered spotted or Nordmann's greenshank (*Totanus guttifer*).

The biogeographic distribution of shorebirds breeding within the North Pacific is depicted in Figure 1. Fifty-eight species or races nest within the Russian Far East, including 37 that occur only within the Palearctic (*see* Table 1). Compared to the Russian Far East, Alaska has slightly fewer overall breeding taxa (48) and only a third as many taxa restricted to its region (13). The 21 taxa that breed both in the Russian Far East and in Alaska are dominated by no single group, but include a mixture of plovers, godwits, curlews, phalaropes and sandpipers. Seventeen species breed in British Columbia, 16 of which also breed in Alaska. Only one species, the red-necked phalarope (*Phalaropus lobatus*), breeds commonly throughout the entire region.

Migration

Shorebirds breeding in the region migrate over a vast area of the globe, including at least 40 different countries throughout North, Central and South America, Oceania, Asia, Australasia, and Africa (Figure 2). Although the migration corridors along

Table 1. Status of shorebirds within the North Pacific Region.

Species[a]	Breeding			Migration			Wintering		
	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia
Haematopodidae									
Eurasian oystercatcher (*Haematopus ostralegus osculans*)	xE[b]			xE					
American black oystercatcher (*Haematopus bachmani*)		x	x		x	x		x	x
Recurvirostridae									
Black-winged (black-necked) stilt (*Himantopus himantopus*)	+			+					
Charadriidae									
Pacific golden plover (*Pluvialis fulva*)	x	x		x	x	x		+	x
American golden plover (*Pluvialis dominica*)	?	x	+	+	x				
Grey (black-bellied) plover (*Pluvialis squatarola*)	x	x		x	x	x		+	x
Ringed plover (*Charadrius hiaticula tundrae*)	x	+		+					
Semipalmated plover (*Charadrius semipalmatus*)	+	x	+		x	x		+	x
Long-billed plover (*Charadrius placidus*)	+T			+T					
Little ringed plover (*Charadrius dubius curonicus*)	x			x					
Killdeer (*Charadrius vociferus*)		x	x		+	x		+	x
Kentish (snowy) plover (*Charadrius alexandrinus*)	+			+		x			x
Lesser sandplover (*Charadrius mongolus stegmanni*)	x	+		x	+				
Eurasian dotterel (*Charadrius morinellus*)	+	+		+	+				
Northern lapwing (*Vanellus vanellus*)	x			x					
Scolopacidae									
Black-tailed godwit (*Limosa limosa melanuroides*)	x			x	+				
Hudsonian godwit (*Limosa haemastica*)		x	+		x	+			
Bar-tailed godwit (*Limosa lapponica baueri*)	x	x		x	x				
(*L. l. menzbieri*)				x					
Marbled godwit (*Limosa fedoa*)		x			x	x			x
Little curlew (*Numenius minutus*)				+					
Eskimo curlew (*Numenius borealis*)		+E[c]			+E[c]				
Whimbrel (*Numenius phaeopus variegatus*)	x			x					

Table 1. *Continued.*

	Breeding			Migration			Wintering		
Species[a]	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia
(*Numenius p. hudsonicus*)		x			x	x			x
Bristle-thighed curlew (*Numenius tahitiensis*)		x			x				
Eurasian curlew (*Numenius aquarta*)				+					
Far eastern curlew (*Numenius madagascariensis*)	x			x					
Long-billed curlew (*Numenius americanus*)			x			x			+
Upland sandpiper (*Bartramia longicauda*)		x	x		+	+			
Spotted redshank (*Tringa erythropus*)	x			x					
Redshank (*Tringa totanus ussuriensis*)	x			+					
Greenshank (*Tringa nebularia*)	x			x					
Marsh sandpiper (*Tringa stagnatilis*)	+			+					
Spotted (Nordmann's) greenshank (*Tringa guttifer*)	xE			xE					
Greater yellowlegs (*Tringa melanoleuca*)		x	x		x	x			x
Lesser yellowlegs (*Tringa flavipes*)		x	x		x	x			x
Green sandpiper (*Tringa ochropus*)	x			x					
Solitary sandpiper (*Tringa solitaria*)		x	x		x	x			
Wood sandpiper (*Tringa glareola*)	x	+		x	+				
Willet (*Catoptrophorus semipalmatus*)						x			+
Terek sandpiper (*Xenus cinereus*)	x			x					
Common sandpiper (*Actitis hypoleucos*)	x			x					
Spotted sandpiper (*Actitis macularia*)		x	x		x	x		+	x
Grey-tailed tattler (*Heteroscelus brevipes*)	x			x	+				
Wandering tattler (*Heteroscelus incanus*)		x	x	+	x	x			+
Ruddy turnstone (*Arenaria interpres*)	x	x		x	x	x			x
Black turnstone (*Arenaria melanocephala*)		x			x	x		x	x
Wilson's phalarope (*Phalaropus tricolor*)		+	x		+	x			
Red-necked phalarope (*Phalaropus lobatus*)	x	x	x	x	x	x			x
Grey (red) phalarope (*Phalaropus fulicarius*)	x	x		x	x	x			x

Table 1. *Continued.*

Species[a]	Breeding			Migration			Wintering		
	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia
Eurasian woodcock (*Scolopax rusticola*)	x			+					
Solitary snipe (*Gallinago solitaria japonica*)	x			+			x		
Japanese snipe (*Gallinago hardwickii*)	x			+					
Pintail snipe (*Gallinago stenura*)	+			x					
Swinhoe's snipe (*Gallinago megala*)	x			x					
Common snipe (*Gallinago g. gallinago*)	x			x					
(*Gallinago g. delicata*)		x	x		x	x		x	x
Short-billed dowitcher (*Limnodromus griseus caurinus*)		x	x		x	x		x	x
Long-billed dowitcher (*Limnodromus scolopaceus*)	x	x		x	x	x		+	x
Asiatic dowitcher (*Limnodromus semipalmatus*)	+			+					
Surfbird (*Aphriza virgata*)		x			x	x		x	x
Red knot (*Calidris c. canutus*)				x					
(*Calidris c. roselaari*)	x	x		+	x	x			x
(*Calidris c. rogersi*)	x			x					
Great knot (*Calidris tenuirostris*)	x			x					
Sanderling (*Calidris alba*)		x		x	x	x		x	x
Semipalmated sandpiper (*Calidris pusilla*)	+	x			x	x			
Western sandpiper (*Calidris mauri*)	x	x		+	x	x			x
Red-necked (rufous-necked) stint (*Calidris ruficollis*)	x			x	+				
Little stint (*Calidris minuta*)	+			+	+				
Temminck's stint (*Calidris temminckii*)	x			+	+				
Long-toed stint (*Calidris subminuta*)	x			x	+				
Least sandpiper (*Calidris minutilla*)		x	x		x	x			+
White-rumped sandpiper (*Calidris fuscicollis*)		x			+	+			
Baird's sandpiper (*Calidris bairdii*)	x	x		+	x	x			
Pectoral sandpiper (*Calidris melanotos*)	x	x		x	x	x			
Sharp-tailed sandpiper (*Calidris acuminata*)				x	x	+			

Table 1. *Continued.*

	Breeding			Migration			Wintering		
Species[a]	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia	Russian Far East	Alaska	British Columbia
Rock sandpiper (*Calidris ptilocnemis couesi*)		x			x			x	
(*Calidris p. tschuktschorum*)	x	x		+	x	x		x	x
(*Calidris p. ptilocnemis*)		x			x			x	
(*Calidris p. quarta*)	x			x			x		
(*Calidris p. kurilensis*)	xT			xT			xT		
Dunlin (*Calidris alpina pacifica*)		x		+	x	x		x	x
(*Calidris a. articola*)		x		x	x				
(*Calidris a. sakhalina*)	x			x	?				
(*Calidris a. kistchinski*)	x			x					
(*Calidris a. actites*)	xT			xT					
Curlew sandpiper (*Calidris ferruginea*)	+	+		+					
Stilt sandpiper (*Calidris himantopus*)		x			+	+			
Broad-billed sandpiper (*Limicola falcinellus sibirica*)				x					
Spoon-billed sandpiper (*Eurynorhynchus pygmaeus*)	x			x					
Buff-breasted sandpiper (*Tryngites subruficollis*)	+	x		+	+	+			
Ruff (*Philomachus pugnax*)	x	+		+	+	+			

[a]Taxonomic and vernacular names from Hayman et al. (1986), except we do not recognize *Calidris paramelanotus* as a species, and we include stilt sandpiper within *Calidris*.

[b]Breeding (May–June): (x) = significant portion of a population of a species or subspecies breeds within this region; (+) = breeds in low numbers within a region. Migration (July–October and March–May): (x) = occurs in significant numbers within the region, primarily on coastal or intertidal habitats; (+) = occurs regularly but in small numbers within the region; (?) = status uncertain. Wintering (November–March): (x) = relatively large numbers occur within the region, primarily on coastal or intertidal habitats; (+) = occurs regularly but in small numbers within the region. E = endangered, T = Threatened. Source: Brazil (1991), Campbell et al. (1990), Flint et al. (1984), Gabrielson and Lincoln (1959), R. Gill (unpublished data), Gochfield et al. (1984), Hayman et al. (1986), Kessel and Gibson (1978), Lane (1987), Paulson (1993), Stepanyan (1990), Stishov et al. (1991), Tomkovich (1986, 1992a, 1992b, 1992c, unpublished data), Vaurie (1965), Watkins (1993).

[c]Inclusion for region based on historical accounts. There has been no substantiated record for the curlew in Alaska since 1899 and the species now may be extinct (Gollop et al. 1986).

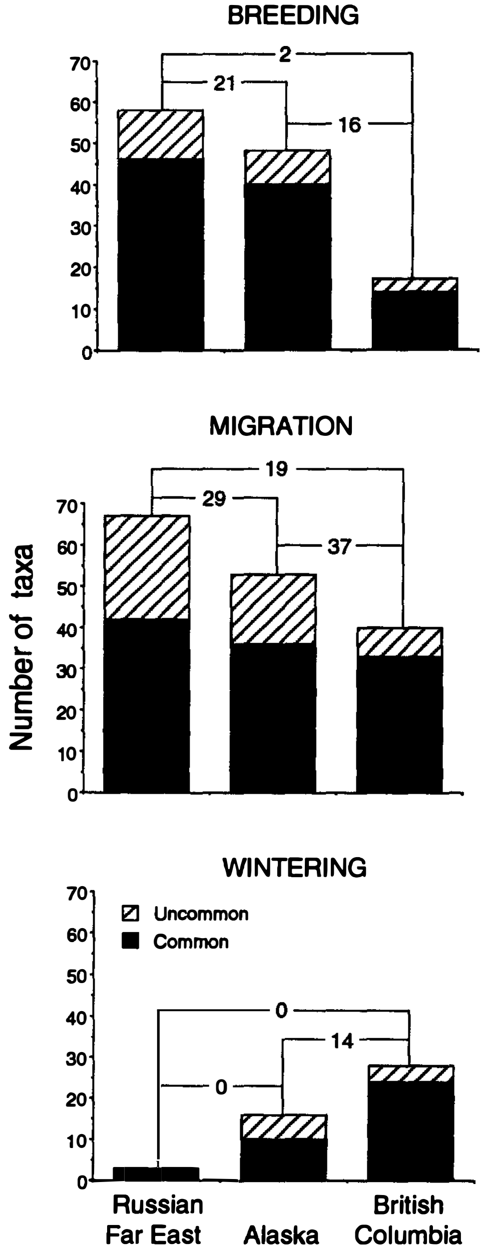

Figure 1. Biogeographic distribution of shorebirds within three areas of the North Pacific region during the breeding, migration and wintering periods. Solid portion of bars indicates the number of taxa (species and subspecies) occurring in significant numbers within each area; cross-hatching shows those occurring regularly but in small numbers (*see* Table 1). Connections between bars show the number of taxa shared between areas.

which North Pacific shorebirds travel are fairly well known, specific links between different breeding and wintering populations within broad-ranging species are virtually unknown. The routes taken are as varied as the species and the migration strategies they employ. Migrations entail distances ranging from only a few hundred kilometers (*e.g.*, rock sandpiper) to several thousand kilometers in a single flight (*e.g.*, bristle-thighed curlew).

Shorebirds traveling to and from the region use a number of migration corridors which sometimes differ between spring and autumn. Corridors used during spring or autumn within the western hemisphere have been summarized by Morrison and Myers (1987). Those used during autumn throughout Oceania and during autumn and spring in east Asia also are generally well known (Baker 1951, Parish et al. 1987, Weishu and Purchase 1987, Parish 1989). Most birds migrating to the region in spring from western hemisphere wintering grounds follow routes along the east coast of the Pacific Ocean or pass through the interior of North America (Morrison and Myers 1987). Shorebirds migrating to the Russian Far East from eastern hemisphere wintering areas primarily follow the west coast of the Pacific Ocean (Parish 1989), but also use several interior routes. The termini of both the Pacific and Central flyways of the western hemisphere and the East Asian flyway overlap in Beringia (Hopkins 1982) and result in considerable interchange of species between Asia and North America (Figure 2). The third major migration corridor to the region is a transoceanic route from over-winter sites in Australia, New Zealand, and the myriad atolls and islands of southern Oceania (Baker 1951, Parish et al. 1987, Parish 1989).

In general, the major southward migration routes of shorebirds from the North Pacific are the reverse of those used in spring. The autumn migration period, however, is much more protracted (June–October) than in spring (March–May) and birds use more stopover sites, many that differ from those used in spring (Page and Gill 1994). These differences are mainly attributable to age- and sex-related differences in the timing of postbreeding movements (*e.g.*, Gill and Handel 1981, 1990, Butler et al. 1987).

The continental routes in North America are used mainly by birds that nest at high latitudes and winter in the Neotropics (Pitelka 1979, Boland 1991). The continental flyways in Asia are used primarily by birds migrating from central Siberia to the East Asian coast and from the Russian Far East to the Indian Ocean and Africa (Parish et al. 1987, P. Tomkovich unpublished data). One particular feature of autumn migration, however, is the greater number of species with long, transoceanic migrations. From the North Pacific, these transoceanic migrants include populations of Pacific golden plovers (*Pluvialis fulva*), dunlin (*Calidris alpina*), long-billed dowitchers (*Limnodromus scolopaceus*), bar-tailed godwits (*Limosa lapponica*), whimbrels (*Numenius phaeopus*), bristle-thighed curlews, ruddy turnstones (*Arenaria interpres*) and sanderlings (*Calidris alba*). After breeding, red-necked and grey (red) phalaropes (*Phalaropus fulicarius*) migrate exclusively at sea, the former along the continental shelf and the latter mostly across pelagic waters.

Wintering

The distribution of shorebirds within the North Pacific region during winter is very different from that during breeding. Only three species winter in the Russian Far East, while 16 occur in Alaska and 28 occur in British Columbia during winter (Table

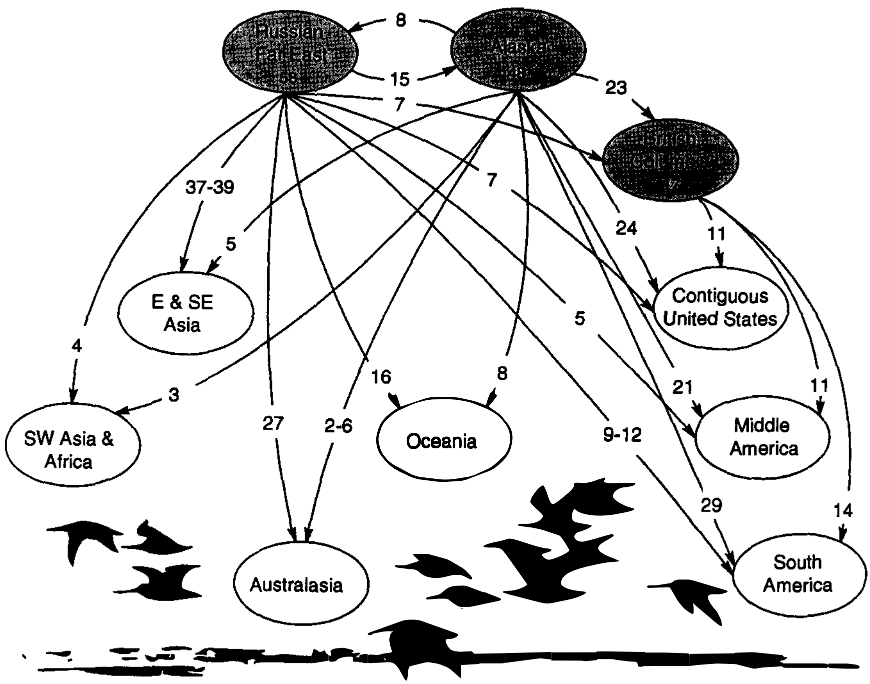

Figure 2. Post-breeding dispersion of shorebirds from the North Pacific region. Number of taxa breeding within each of the three areas is shown inside shaded ovals. Connections between areas within the North Pacific show the number of these taxa exchanging during autumn migration. Connections to other regions of the world (clear ovals) show the number of taxa dispersing to winter in those regions. Many species winter in more than one region, and exact connections between specific breeding and wintering populations are poorly known for most species.

1, Figure 1). Only species associated with rocky intertidal habitats or sandy beaches (*e.g.,* American black oystercatcher, sanderling, rock sandpiper, surfbird and black turnstone) are common in Alaska during winter. Most species breeding in the Russian Far East and about half of those breeding in Alaska and British Columbia spend the boreal winter in tropical or subtropical latitudes encompassing both hemispheres of the globe. The patterns of post-breeding dispersion shown in Figure 2 underscore the need for a truly international perspective for the conservation and management of North Pacific shorebirds.

Important wintering sites in the Pacific region for populations of shorebirds breeding in the North Pacific occur in the Americas from southern Canada to Chile (Morrison and Ross 1989, Morrison et al. 1992, 1993, Page and Gill 1994). These include numerous estuaries along the coast of Washington and California, especially San Francisco Bay (Page et al. 1992), estuaries along the coast of Baja and west coast of mainland Mexico (Morrison et al. 1992, G. Page unpublished data), and the Bay of Panama (Morrison and Butler 1994). In Oceania and Eastern Asia, most North Pacific species winter south of about 30 degrees N (Weishu and Purchase 1987), although large numbers of dunlin and a few other species winter along the coasts of Korea, Japan and China (Long et al. 1988, Brazil 1991). The bristle-thighed curlew is the only migratory species whose entire population is confined to Oceania during the nonbreeding period (Gill and Redmond 1992).

Conservation of Shorebirds

The high degrees of endemism and species diversity make the North Pacific one of the world's most important regions for shorebirds. The responsibility for their conservation rests on the will for international cooperation. One of the most effective mechanisms for the conservation of shorebirds is the protection of critical breeding, staging and nonbreeding areas along entire flyways, which transcend international boundaries.

Along the Pacific coast of the Americas, there are 26 areas known to qualify as sites of hemispheric or international importance to North Pacific shorebirds under the WHSRN program (Table 2, Figure 3). To date, an additional eight sites along the western rim of the Pacific Ocean have been identified as important to North Pacific shorebirds under these criteria. Identification of critical sites is incomplete, however, especially in the Russian Far East, Central America, East Asia and Oceania. Within the North Pacific region, 5 areas potentially qualify as international sites and 11 areas qualify as hemispheric sites (Table 2). Among these, only three have been officially designated under the Ramsar or WHSRN programs. Izembek Lagoon in Alaska and the Alaksen National Wildlife Area on the Fraser River Delta in British Columbia are official Ramsar sites, and the Copper River Delta, Alaska, is a WHSRN hemispheric site. Elsewhere in the Pacific, 12 areas qualify as international sites and 6 areas qualify as hemispheric sites according to WHSRN criteria (Table 2). Among these, only San Francisco Bay and Grays Harbor have been officially designated as WHSRN sites. In addition to the 26 Pacific Rim sites identified here, numerous other sites are important to North Pacific shorebirds, especially to species with mid-continent or Atlantic migration routes or those wintering along the Atlantic coast of Central and South America. Such sites include Cheyenne Bottoms in Kansas, Laguna Madre along the east coast of Mexico, and Bahia Lomos, Chile (Senner and Howe 1984, Morrison and Ross 1989, Morrison et al. 1992, 1993).

Most sites in Alaska currently are afforded some level of official protection under various land conservation measures (*e.g.*, as National Wildlife Refuges, National Monuments or State Critical Habitat Areas). Boundary Bay in the Fraser River delta, British Columbia, likely will receive official protection as a Provincial Wildlife Management Area in 1994. Conservation efforts in Alaska and British Columbia should be directed primarily at preventing habitat deterioration, especially from oil spills. In the Russian Far East major efforts should be directed at identifying the many important sites that are likely to exist. The effects of hunting that occur locally along the coast also should be assessed, particularly the impacts on populations of Eurasian woodcock (*Scolopax rusticola*), whimbrel, Eurasian oystercatcher (*Haematopus ostralegus*) and the endangered spotted greenshank.

The major threats to North Pacific shorebirds in Central America, South America and the East Asian-Australasian flyway are from destruction of mangrove habitats, hunting, and pollution from oil, mining and pesticides (Delgado 1986, Mundkur 1993, I. Davidson personal communication: 1994). Most shorebird populations are judged to have rebounded from the market hunting that occurred during the past century in North America (Morrison and Harrington 1979, Senner and Howe 1984). The long

Table 2. Coastal wetlands thoughout the Pacific basin that qualify as important sites for North Pacific shorebirds under criteria of the Western Hemisphere Shorebird Reserve Network (WHSRN).[a] Sites referenced by number on Figure 3.

Site	WHSRN designation	Source
United States—Alaska		
1. St. Lawrence Island	H[b]	Gill and Tibbitts unpublished data
2. St. Matthew Island	I	Gill and Tibbitts unpublished data
3. Pribilof Islands	H[b]	Gill and Tibbitts unpublished data
4. Nunivak Island	I[b]	Gill and Tibbitts unpublished data
5. Central Yukon-Kuskokwim River delta	H	Gill and Handel (1990)
6. Kuskokwim River delta	H	Gill and Tibbitts unpublished data
7. Cinder River lagoon	I	Gill and Tibbitts unpublished data
8. Nelson Lagoon	I–H[c]	Gill and Jorgensen (1979), Gill et al. (1981), Gill and Tibbitts unpublished data
9. Mud Bay	I–H[c]	Gill and Tibbitts unpublished data
10. Redoubt Bay	I	Gill and Tibbitts unpublished data
11. Fox River delta	I	Gill and Tibbitts unpublished data, G. West unpublished data
12. N. Montague Island	H[d]	Gill and Tibbitts unpublished data
13. Copper River delta	H	Senner and Howe (1984)
14. Stikine River delta	H	C. Iverson unpublished data
Canada		
15. Fraser River delta, B.C.	H	Morrison et al. (1992)
United States—contiguous states		
16. Grays Harbor, Washington	H	Senner and Howe (1984), Wilson (1993)
17. Humboldt Bay, California	I	Senner and Howe (1984)
18. San Francisco Bay, California	H	Senner and Howe (1984), Page et al. (1992)
Mexico		
19. Rio Colorado	I	Morrison et al. (1993)
20. Laguna Ojo de Liebre	I	Morrison et al. (1993), G. Page unpublished data
21. Esteros Tobari and Lobos	I	Morrison et al. (1993)
22. Culiacan-Los Mochis	I	Morrison et al. (1993)
Panama		
23. Panama Bay	I	Morrison and Butler (1994)
Peru		
24. Virrila estuary	H[e]	Morrison and Ross (1989)
25. Chiclayo region	H	Morrison and Ross (1989)
Chile		
26. Chiloe region	H[f]	Morrison and Ross (1989)
Russian Far East		
27. Moroshechnaya River delta	H	P. Tomkovich unpublished data
Sumatra		
28. Banyuasin Musi River delta	I	Mundkur (1993)
Australia		
29. Lake McLeod	I	Watkins (1993)
30. Port Hedland Saltworks	I	Watkins (1993)
31. Eighty Mile Beach	H	Watkins (1993)

Table 2. *Continued.*

Site	WHSRN designation	Source
32. Roebuck Bay and Plains	I	Watkins (1993)
33. S. E. Gulf of Carpentaria	I	Watkins (1993)
34. The Coorong	I	Watkins (1993)

[a]Under WHSRN criteria, an international site (I) must annually support at least 100,000 shorebirds or 15 percent of a flyway population; a hemispheric site (H) must support at least 500,000 shorebirds or 30 percent of a flyway population.
[b]Based on percentage of rock sandpiper population using this site.
[c]Site qualifies as (I) based on numbers and as (H) based on percent of flyway population (dunlin and bar-tailed godwit). Additional studies also likely to support (H) designation based on total numbers.
[d]Based on percentage of surfbird population using this site.
[e]Based on percentage of sanderling population using this site.
[f]Based on percentage of Hudsonian godwit and whimbrel populations using this area.

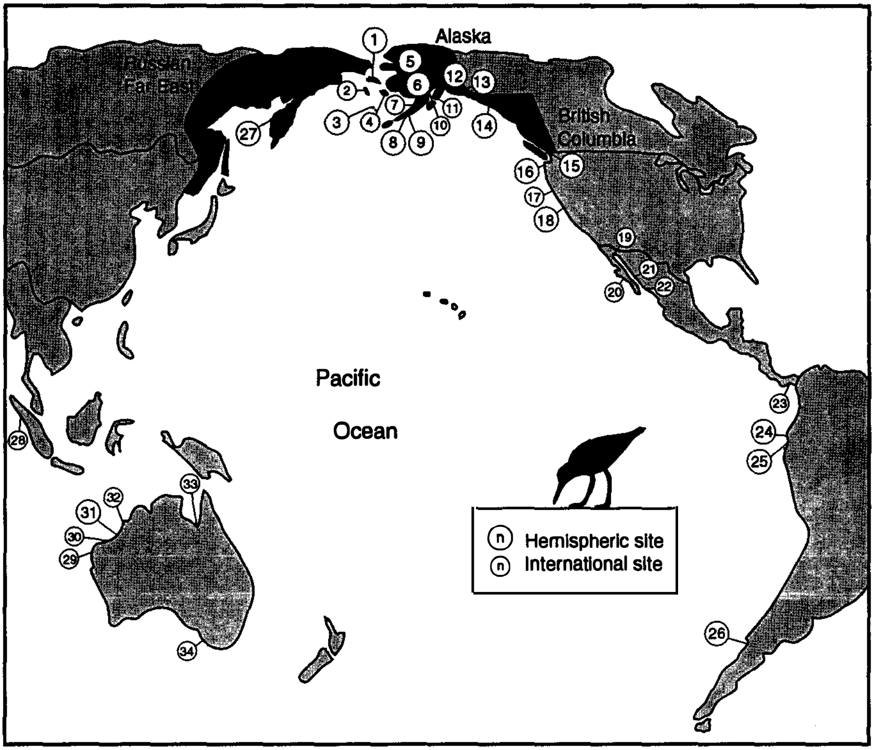

Figure 3. Locations of coastal wetlands throughout the Pacific basin that meet Western Hemisphere Shorebird Reserve Network criteria for sites of international or hemispheric importance (*see* Table 2 for criteria, names and designations).

period required for recovery, however, highlights the need for effective protection from severe impacts throughout their range. Humans have devastated the avifauna of Oceania, which is one of the fastest growing human population centers on earth (Holyoak 1973, Moors 1985, Loope et al. 1988, IUCN 1991). There is a particular need for information on the bristle-thighed curlew because of its restricted range on small islands and atolls, where it may be vulnerable to human disturbance and exotic animals, especially during its flightless molt (Marks et al. 1990, Gill and Redmond 1992). Red-necked phalaropes, which winter throughout southern Oceania, may be threatened by ingestion of plastic particles (Connors and Smith 1982) and oil spills. Only international cooperation will ensure that oceanic and coastal habitats remain free of such pollution.

Coordinated International Research and Conservation

Many countries are involved in migratory bird conservation throughout the Pacific. However, conservation information is dispersed, resources are limited and data necessary for conservation actions are not always available. The global scale of shorebird conservation problems requires coordinated efforts to direct results to appropriate decisionmakers. We see this happening at two levels, one involving the hands-on biologists, the other wildlife administrators, but both working jointly through all phases of the program.

In the past two decades numerous organizations have formed to promote the study and conservation of shorebirds, including the Western Hemisphere Section of the Wader Study Group of Europe, the Australasian Wader Studies Group, the Asian Wetland Bureau, Wetlands for the Americas and the Russian Working Group on Waders, to name a few. These groups have been very active in their areas of geographic interest and readily have made information available to others. Recently, they have recognized the need to form partnerships and expand their focus throughout a flyway. For example, the Wader Study Group developed a formal protocol for international cooperation in research efforts in the eastern hemisphere, including the East Asian-Australasian flyway (Wader Study Group 1992). They also developed a formal agreement to provide advice on shorebird research and conservation issues to the International Wetlands Research Bureau (N. Davidson personal communication: 1994). The protocol and agreement are being used as models to establish arrangements between the western hemisphere section of the Wader Study Group and Wetlands for the Americas (Canavari 1993). The Australasian Wader Studies Group, in conjunction with Russian shorebird biologists, recently has supported work on Palearctic nesting species using the East Asian flyway. All of these partnerships are aligned around north-south shorebird migration corridors. We have shown in this paper that shorebirds throughout the Pacific, but especially the North Pacific, involve east-west associations as much as they do those north-south. It is time for the various shorebird groups and national conservation agencies throughout the Pacific Rim nations to recognize this east-west link and begin to work toward new partnerships. Further, these arrangements should extend to include Pacific island nations that individually support many small populations of shorebirds but collectively account for substantial numbers of birds.

What specifically can be done? First, on a regional basis, but through international programs, we need to identify important sites using objective criteria. The Russian

Far East, Central America and Oceania need particular attention. By the nature of habitats and preliminary studies, we know that critical sites exist in these areas, but there is no funding available or programs established to identify them. It is in the interest of all Pacific Rim nations to identify and evaluate the relative importance of critical sites used by North Pacific shorebirds during their annual cycle.

As a second step, we need to establish programs to link each of these sites to the specific populations that use them during various stages of the annual cycle. It is hollow conservation to have identified a critical staging site in Alaska, for example, if sites used by these same birds the other 10 months of the year are not known and if potential threats to the areas are not assessed. These links can be established through large-scale marking and censusing programs that are organized along flyways by core staff in each nation, and that function with mostly volunteer help. New advances in genetics and systematics show much promise as another tool that can be used by research biologists to link populations to specific breeding, staging and wintering sites. If these links can be established, it will be much more cost-effective to initiate international monitoring programs at appropriate sites throughout the annual cycle than to have a single country try to cover all aspects by itself. Such programs, however, will require a strong, long-term commitment by the participating governments to support their portion of such an international monitoring program. It may be in the best interests of some of the nations to assist others, particularly the developing countries, in organizing such programs and developing their own expertise.

Last, once sites have been identified, linked and their threats assessed, they need to be recognized as critical components of an *international* shorebird reserve network. This will require the continued financial and political support of existing programs such as WHSRN, Ramsar, Wetlands for the Americas and the Asian Wetland Bureau. Mostly, it will require a strong commitment from the three North Pacific countries—the United States, Russia and Canada—to expand the scope of such programs and forge partnerships that encompass the entire Pacific basin.

Acknowledgments

We thank Ian Davidson for information from Central and South America, Gary Page for information from Mexico, and Chris Iverson and George West for information from Alaska. Bob Elner critiqued an earlier draft of this paper.

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Population Models as Tools for Research Cooperation and Management: The Wrangel Island Snow Geese

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Introduction

Restoring the Wrangel Island lesser snow goose (*Anser caerulescens caerulescens*) population to its historical level is a major objective of the Pacific Flyway Study Committee (Kraege 1992). Given that the geese nest on Wrangel Island, Russia, and winter on the Pacific coast of Canada and the U.S., many research and management questions need to be addressed at the international level. Recognizing this fact, scientists in the three countries have started collaborating more closely on the research and monitoring of the Wrangel Island (WI) population. However, developing management prescriptions for a population requires finding a way to obtain a quantitative assessment of status of that population and of factors influencing its size.

The aim of this paper is to present the framework that we have initiated to obtain such a quantitative assessment (Brault 1994). In this paper, we present basic information on the WI population, the analysis methods being used and some preliminary results.

The Wrangel Island Snow Geese

Lesser snow geese breed on Wrangel Island (Figure 1) from May to August. Weather conditions there are harsh and extremely variable; in some years the entire hatchling or fledgling cohort have died due to weather-related phenomena (V. Baranyuk personal communication). The birds migrate to their wintering grounds in family units, along two migration routes. The northern population (Group 1 in Figure 1) travels to Alaska and then along the west coast of Canada to winter on the Fraser (BC) and the Skagit (WA) river deltas. A small proportion of the southern population (Group 2) migrates down the coast and stops at the Fraser/Skagit (F/S) deltas before moving on to California, but most are thought to travel inland from Alaska and join up with the Banks Island population (Group 3) in its migration through inland western

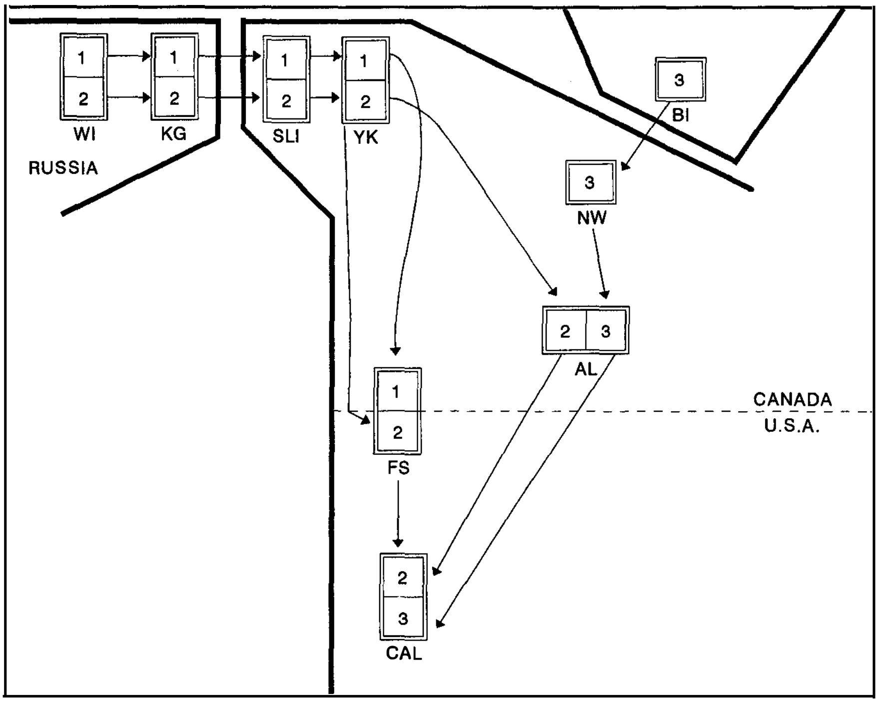


Figure 1. Migration patterns of the Wrangel Island and Banks Island populations. Numbers in boxes represent the three lesser snow geese populations discussed in the text: (1) northern population; (2) southern population; and (3) Banks Island population. WI = Wrangel Island; KG = Chukotka; SLI = St. Lawrence Island; YK = Yukon Kuskokwim deltas; BI = Banks Island; NW = Northwest Territories; AL = Alberta; FS = Fraser Skagit deltas; and CAL = California.

Canada to California (J. Takekawa unpublished data). Spring migration operates along similar routes; however most WI geese wintering in California travel back north through the prairies. While the northern group is homogeneous, the southern group and the Banks Island birds mix on the wintering grounds in California. WI geese represent only about 5 percent of all white geese (including Ross' geese, *Anser rossii*) in California, which are estimated at around 500,000 birds.

The snow goose population on Wrangel Island numbered about 150,000 individuals in 1969. During the early to mid-1970s, however, a precipitous decline occurred (Figure 2A), bringing the population down to 57,000 in 1975. The population appeared to recover thereafter, but its numbers dropped again in recent years. There are at least two possible reasons for the original decline: survival of immature (i.e., less than 1 year-old) birds was very low in 1971–1974 (less than 1 percent survived in each of these years; Figure 2B) and survival in the California wintering population (Group 2) was low. As seen in Figure 2A, the Northern Group 1 did not decline to the same extent as the total population, so that most of the decrease must have occurred in the southern Group. The northern Group now accounts for about 60 percent of the WI population, whereas it was only 30 percent in 1970 (S. Boyd, unpublished data).

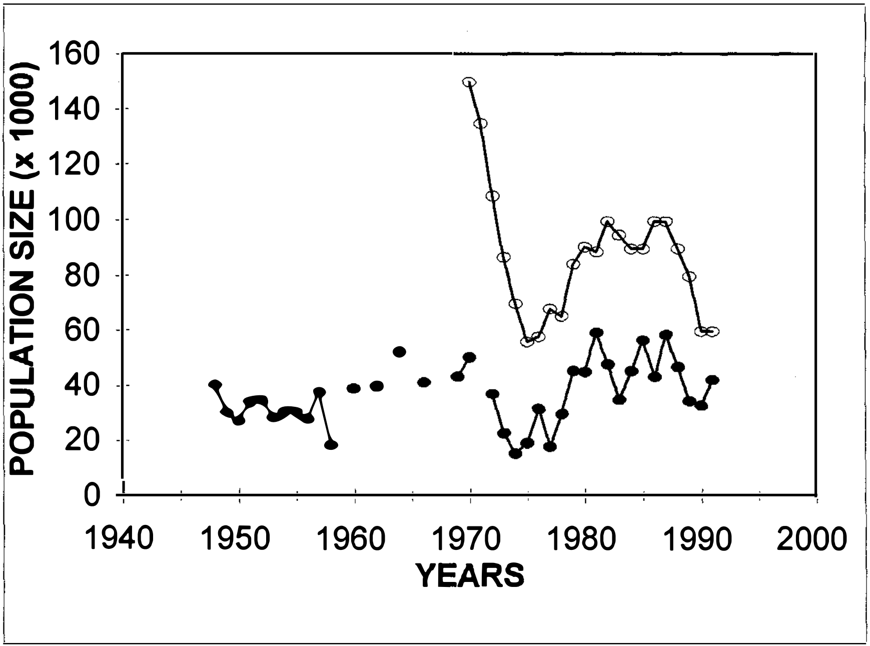


Figure 2A. Changes in population size of the WI snow goose. Open circles; population at Wrangel Island, from Russian sampling program; closed circles: population estimates at the Fraser and Skagit deltas from visual estimates and photo counts.

The WI snow goose population has been protected in Russia since 1976, but it still is hunted in Canada and the U.S. Harvest data for the Fraser and Skagit deltas since the mid-1940s are presented in Figure 2B. There is a significant decreasing trend in the proportion of birds harvested (for 1962–92, regression slope $= 0.5$, $p < 0.001$, $R^2 = 0.419$) although the variance in this proportion is very large. Proportion harvested is correlated with the proportion of immatures in the deltas ($r = 0.45$), and the year-to-year changes in these two proportions are more highly correlated ($r = 0.62$). Hunting activity (or success) thus appears to be influenced by the proportion of immatures in the wintering population.

A Population Model Framework

A useful framework for the assessment of the population status, and of the need for further research, should (1) include realistic estimates of biological characteristics of the species; (2) be capable of taking into account the population structure and movements; (3) allow for data from different sources to be combined; (4) allow for the calculation of population trends, to be compared with the observed trends; and (5) allow for the analysis of the effects of disturbance on the fate of the population. The last point is connected to questions of management and research. Disturbance can be caused by environmental variations or by human action, and in both cases needs to be evaluated in a management plan. This analysis also can be used to address

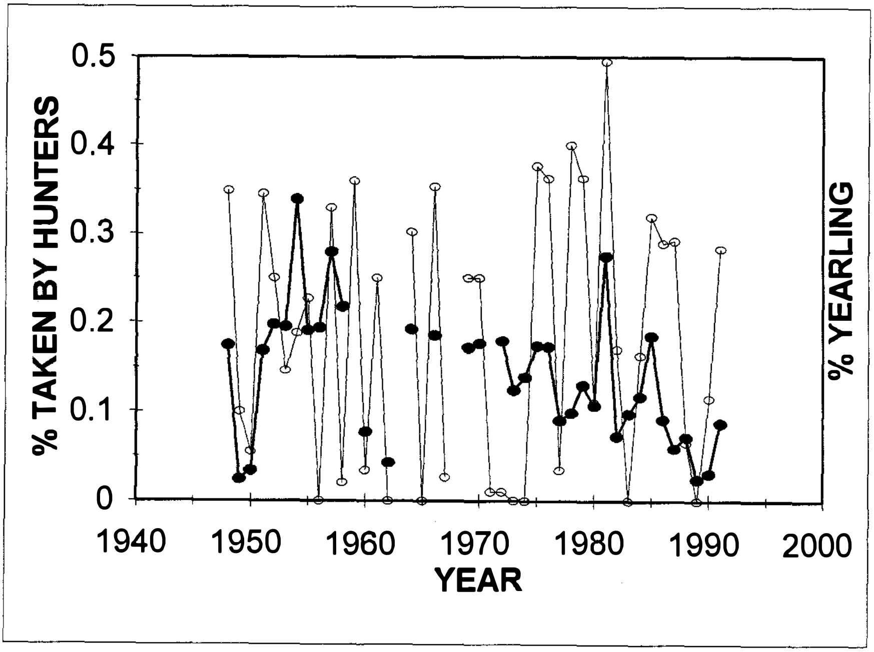


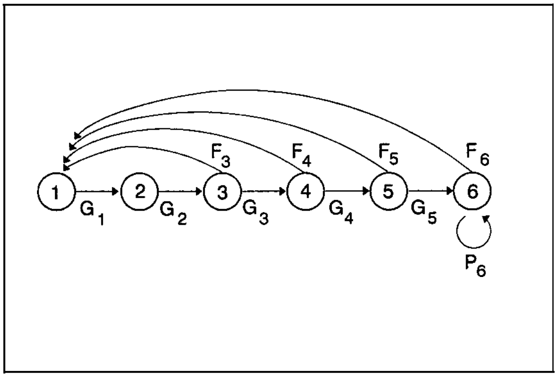
Figure 2B. Closed circles: proportion of the Fraser/Skagit wintering population harvested each year; fine line: proportion of juveniles in the population each year.

the most appropriate research questions, by asking how useful it would be to fill certain data gaps. The examples below will clarify these points.

The Basic Model

Method. The stage-structured formulation we have chosen stems from the Leslie matrix (Leslie 1945). The model is a mathematical representation of the life cycle of the geese, but unlike the Leslie model, it does not require knowledge of the ages of individuals. Instead, these can be grouped in stages which can represent any common characteristic of these groups: size, reproductive status and age. This makes it a useful tool to study bird population dynamics, because aging birds is difficult (McDonald and Caswell 1993). The formulation also requires fewer parameters than an equivalent Leslie-matrix model, making it more "economical." Stage-structured models have been used to study the population dynamics of such disparate forms of life as perennial plants, turtles, corals and whales (*e.g.*, Caswell 1989, Brault and Caswell 1993). For a complete discussion see Caswell 1989, for a simpler presentation oriented to birds see MacDonald and Caswell 1993.

Figure 3A illustrates a stage-structured model based on data collected for lesser snow geese at La Perouse Bay, Canada (Cooke et al. 1994). Stages 1 to 5 correspond to the first five years of life. Individuals remain in the last stage (i.e., mature adult stage) as long as they survive; in matrix form (transition matrix A of Figure 3B), this translates into a non-zero element on the diagonal. The parameters of the model are the survival probability at each stage, the probability of moving on to the next stage



$$A = \begin{pmatrix} 0 & F2 & F3 & F4 & F5 & F6 \\ G1 & 0 & 0 & 0 & 0 & 0 \\ 0 & G2 & 0 & 0 & 0 & 0 \\ 0 & 0 & G3 & 0 & 0 & 0 \\ 0 & 0 & 0 & G4 & 0 & 0 \\ 0 & 0 & 0 & 0 & G5 & P6 \end{pmatrix}$$

$$A = \begin{pmatrix} 0 & 0 & 0.46 & 0.79 & 0.84 & 0.98 \\ 0.13 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.76 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.76 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.81 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.81 & 0.81 \end{pmatrix}$$

$$W = \begin{pmatrix} 0.4611 \\ 0.0667 \\ 0.0559 \\ 0.0469 \\ 0.0393 \\ 0.3301 \end{pmatrix}$$

$$E = \begin{pmatrix} 0 & 0 & 0.0051 & 0.0073 & 0.0066 & 0.0641 \\ 0.0831 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.0831 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.078 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.0706 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.0641 & 0.5381 \end{pmatrix}$$

Figure 3. A = Life-cycle graph for the lesser snow geese. Symbols (P, G and F) are as described in Table 1. Stage 1 is the immature stage; stages 2 through 5 are the eartly maturing years of adult stage; state 6 is the fully mature adult stage. B = Matrix form of the life=cycle graph. C = Values for this matrix using means of parameters (*see* Table 1 and text) from 1970–1987. D = Stable stage distribution obtained using matrix **A**. E = Elasticity (= proportional sensitivity) matrix calculated from matrix **A**.

(in an age-structured model, this probability is one; in a stage-structured model, it can be less than one), and parameters affecting female fertility at each stage. We used a post-breeding birth pulse formulation (Caswell 1989).

Transition probabilities for the fertility elements of matrix A were estimated from data collected at Wrangel Island on the proportion of birds nesting, brood size and nesting success. Surveys on the Fraser/Skagit deltas provided census data and proportion of juveniles during autumn migration; these data, combined with WI data on hatchling survival and survival to end of the first year (birds returning to WI) were used to calculate transition element G_1. Survival probabilities for stages other than stage 1 are not well defined for this population; data for lesser snow geese at La Perouse Bay provide a rough estimate of 0.76 for subadults (stages 2 through 5) and of 0.8 for mature adults (stage 6) (Cooke and Rockwell 1988, Francis et al. 1992), and these values were used in the basic model. The effect of varying these values was checked and discussed below. Parameters used in the model are detailed in Table 1.

The model was used to obtain the population rate of increase and to perform a sensitivity analysis to examine how a change in any of the vital rates affects the rate of increase (*see* Caswell 1989 or McDonald and Caswell 1993). A series of simulations also was performed using data collected at Wrangel Island and on the F/S deltas from 1970 to 1987. These data allow the estimation of year-to-year variation in the following parameters: female nesting probability, nest success, survival from egg to hatch, survival from hatch to juvenile (at F/S), and survival from juvenile to one year old returning to Wrangel Island the following year. In the simulation runs, the annual values for these parameters are used in the calculation of the transition matrix elements G_1 and Fs. The simulation analysis thus asks whether these variations in fertility and first-year survival parameters can explain the observed trends in population size during the 1970–87 period. Simulations also are used to study the effect of a hypothetical change in hunting intensity on the F/S deltas.

Results. Using means for all years (1970–87) for all parameters, we obtain the transition matrix A shown in Figure 3C. Two features of this matrix are worth noting.

Table 1. Parameters used to calculate the entries of the transition matrix of the Wrangel Island snow goose stage-structured model. The P, G, and F are the vital rates presented in Figure 2.

$G_1 = sjuv * syrl * hunt_1 * \sigma_1$
$G_i = \sigma_i * hunt_i$
$F_i = \sigma_i * hunt_i * breed_i * clutch_i * segg$
$P_6 = \sigma_6 * hunt_6$

Where:
i = age or stage (i.e., n_i is the nesting success of females at stage i);
segg = survival probability from egg to hatch;
sjuv = survival probability between hatchling and fall census in the Fraser/Skagit;
syrl = survival probability from fall census to return to WI the next spring
σ = yearly probability of survival (from natural causes);
hunt = probability of surviving human exploitation;
clutch = mean number of female egges per female;
breed = proportion of females breeding.

Fertility, expressed as the mean number of female hatchlings per female, is very low; even for the fully mature stage 6 it is less than 1. First-year survival also is very low; on average only 13 percent of hatchlings survive to return to the nesting grounds the following year. The intrinsic rate of increase λ (= exp (r)) calculated from matrix A is 0.91 (r = -0.094), which means that if these were constant conditions, the population would decline. The vector ***W*** of Figure 3D is the stable stage distribution, that is, the proportion of individuals in each stage under the assumption of stable conditions. This vector shows that most individuals in the population are either stage 1 (immature) or stage 6 (i.e., fully mature) birds. The low survival probability in the first year of life results in low proportions for stages 2 through 5, while individuals over five years old accumulate in stage 6 (lesser snow geese can live form more than 15 years in the wild—the oldest band recovery at La Perouse Bay was 27 years old; Francis et al. 1992).

The main results from the sensitivity analysis are shown in Figure 3E. The values in this matrix are the proportional change in rate of increase (λ) due to proportional changes in each element (the G, P and F) of matrix A. The larger the value of an element in matrix E, the stronger the effect of a change in the corresponding element of matrix A on λ. (The values in matrix E are all on the same scale.) The rate of increase is most sensitive to proportional changes in survival of fully mature birds (P_6). Parameters contributing to other elements such as fertility ($F_{2\text{-}6}$) have to vary much more to affect the rate of increase to the same extent as P_6. A small change in adult survival thus will have a very strong impact on the population trajectory.

Simulation results are compared with observed changes in population size from 1970–87 in Figure 4. The simulations closely track the observed decline in population until the mid-'70s; the lower curve (σ_{2-6} = 0.75) follows this decline closely, but later fails to match the observed recovery. In contrast, the highest curve (σ_{2-6} = 0.9) parallel the observed trend during this population increase. The results suggest that (1) adult survival has changed from a low value during the early '70s decline to a high value thereafter, and (2) variations in first-year survival and, to a lesser extent, fertility in the early '70s, are partially responsible for the decline.

Results from simulations where hunting intensity on the F/S deltas was varied (Figure 5) illustrate the sensitivity analysis results. Each line is the mean of 100 simulation runs, where sets of fecundity and immature survival parameters are picked randomly from the 17 years of data. Harvesting 5 to 20 percent of the immatures (curves B and C) does not affect the population trajectory substantially, compared to a no hunt scenario (curve A). However, harvesting 5 to 10 percent of the adult stages (curves D and E) results in a strong depression of the population; most of this effect is due to harvesting stage 6 individuals (curve F).

A Metapopulation Model

Although the basic model brings the major elements of the life cycle of the snow geese, it does not provide the structure necessary to study the winter segregation of two WI groups, or the interaction of WI birds with other population (Figure 1). Such a structure is desirable if we want to understand the change in relative size of the two groups that has occurred in the last two decades, or possible influences of other goose populations on the California wintering grounds through common habitat use of disease transmission (Wobesor 1981). This necessitates a metapopulation approach

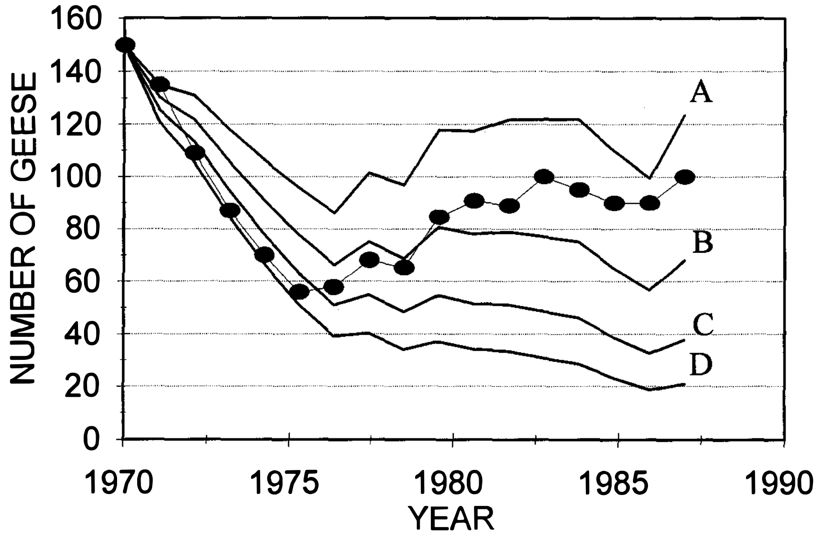


Figure 4. Comparison of observed changes in population size at Wrangel Island (closed circles) with simulations using yearly estimates of fertility parameters and first year survival, but where adult survival probability (σ_{2-6}) is controlled. Line A: $\sigma_{2-6} = 0.9$; Line B: $\sigma_{2-6} = 0.85$; Line C: $\sigma_{2-6}0.8$; Line D: σ_{2-6} 0.75.

i.e., a model where local (sub) populations are interacting portions of a larger entity (the metapopulation) (Gilpin and Hanski 1987, Hastings 1991). Because this work requires input from many sources in all three countries, our aim in this section is to present the modelling procedure rather than preliminary results (Kraege 1992, Brault 1994).

The life-cycle graph and matrix representations of the approach are shown in Figure 6. To clarify the figure, we use a simplified version of the basic model form presented above for each of the three sub-populations, where Y are the young or immatures, SA are the sub-adults and A are the mature adults. The sub-populations then are linked either through exchange of individuals or sharing of common environmental conditions (weather, hunting pressure, epidemics, etc.) We use two time steps in a year (rather than a single one in the basic model) to take into account the very different interactions and conditions occuring on the nesting and wintering grounds. For example, the two WI sub-populations are submitted to the same weather conditions during nesting and the early period of yearling stage, but experience separate sets of conditions on their respective wintering grounds; the F/S area has more severe winters than California, but possibly lower hunting pressure, less competition for food and minimal risk of disease transmission from other species. In matrix form, this translates into two matrices for each sub-population that are multiplied at each yearly step.

These sub-population matrices are linked by possible exchange of individuals at the appropriate stages and time of the year. For instance, some maturing individuals

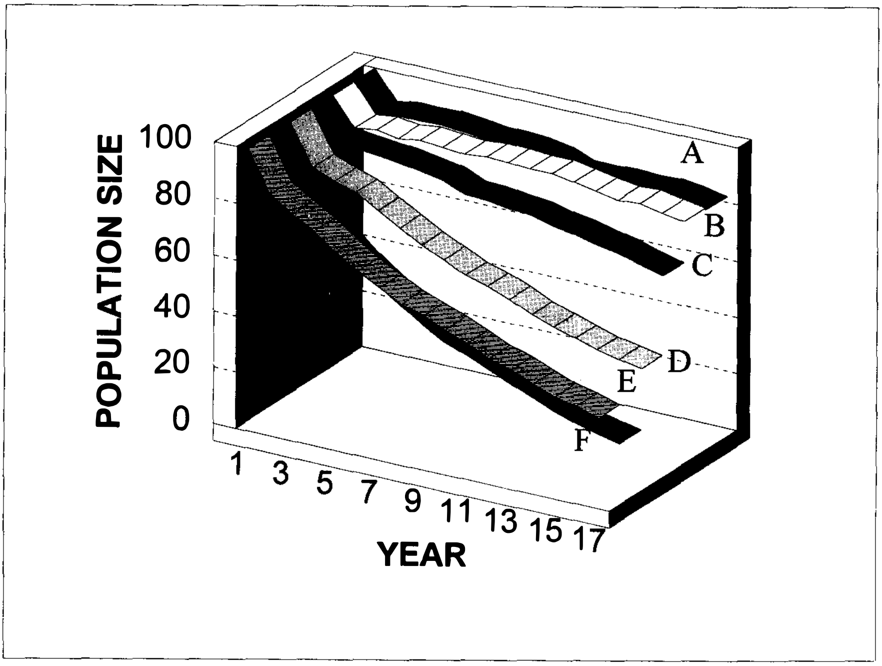


Figure 5. Results from simulations where yearly sets of estimates of fertility parameters and immature survival probability are picked randomly, and harvesting is an extra mortality factor. Each curve is the mean of 100 runs. A = No hunting, $\sigma_{2\text{-}6}$ = 0.9 (baseline simulation); B = 5 percent of stage 6 (mature adults) only is harvested; C = 10 percent of stage 6 only is harvested; D = 5 percent of stagfe 1 (yearling) only is harvested; E = 10 percent of Stage 1 only is harvested; F = 5 percent of stages 2 to 6 is harvested..

are likely to change group through pairing with a member of another sub-population. This linkage results in a large matrix composed of the sub-population elements and the elements quantifying the amounts of exchange. Although the structure has become much more complex, some of the analytical tools used to examine the basic model, such as the sensitivity analysis, still are applicable. This type of analysis will be combined with strategic use of simulation analysis to study the effects of interchange between Wrangel Island and Banks Island populations, of hunting in California and of potential change in habitat conditions on both wintering grounds.

Discussion

Nesting conditions at Wrangel Island are harsh and variable so that it is natural to hypothesize that they will have the strongest impact on population growth. They caused some of the observed downward trends because of new cohort failure several years in a row, but they fail to explain the subsequent pronounced increase in the late 1970s. The sensitivity analysis suggested a possible reason for this; a small change in adult survival can have more of an effect on the population dynamics than a large change in fertility or immature survival. Adult survival is notoriously difficult to estimate, and requires concerted research efforts. However, the results from our

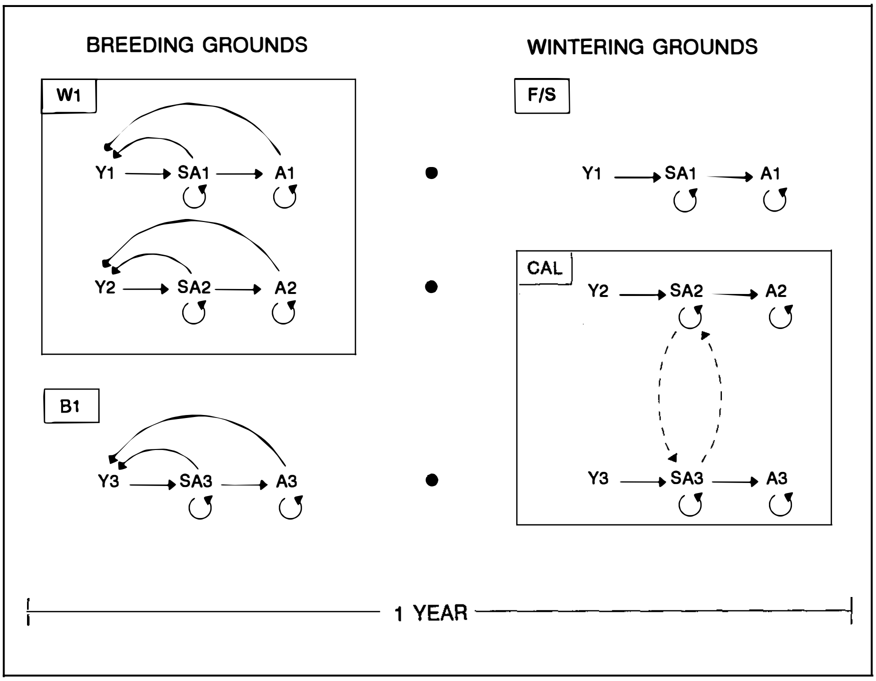


Figure 6. Structure of the metapopulation model. The life-cycle graphs are simplified to Young (= Immature), Subadult and Adult stages; Subscript numbers are Group numbers as identified in Figure 1, and abbreviations of breeding and wintering grounds names are also from Figure 1. Frames around pairs of Groups represent shared environmental conditions. Arrows with dotted lines represent some of the possible interchanges of individuals between Groups. In matrix form, one multiplies event probabilities for the two periods of the year to obtain yearly transition elements; this is symbolized by the dots between life-cycle graphs of each Group.

analysis clearly point toward the need for such information. It also shows that hunting mortality (possibly a major cause of mortality of adult geese) can affect the trajectory of this population, and thus is not a negligible factor in the dynamics of the WI snow geese. A similar analysis of the endangered loggerhead sea turtle in the Caribbean was instrumental in changing the conservation emphasis from protecting nesting beaches to reducing adult mortality. In the shrimp fishery, where turtles are a by-catch, all fishing nets now are required to have devices allowing the turtles to escape if caught (Crouse et al. 1987, Crowder et al. 1994.)

The sensitivity and simulation are results specific to the WI population, and are not directly applicable to other snow goose populations. For example, sensitivity analysis of a model with identical structure of the La Perouse Bay population shows that the influence of stage 6 (adult) survival on the rate of increase is less pronounced than in the WI population; survival probabilities in previous stages also have a strong effect (Brault 1994). The low average survival probability of immatures in the WI population—due in large part to the specific climate conditions at Wrangel Island—apparently is responsible for this difference.

Such a model framework and analysis can be used in exploring causes for population variations and pointing to potential management prescriptions. The same structure can be the basis for a risk analysis, where the probabilities of some outcomes are estimated for different management decisions (such as reducing hunting mortality, or increasing protected wintering habitat) (Burgman et al. 1993). Population viability analysis is an example, but other less drastic outcomes than population extinction can be addressed (Restrepo et al.1992, Rosenberg and Brault 1994).

This modelling approach should be interactive with field research. For example, the metapopulation structure suggested here was constructed to address questions generated through field research; how important is it to understand the interaction between Banks Island and Wrangel Island birds? Can we explain the decrease in the proportion of California birds at WI? How can the WI population be rehabilitated? In turn, results from the model analysis can provide further research questions (and even some answers!). They also can help determine the optimal use of research funds by comparing return from different experiments in terms of better understanding of population dynamics. Models are not aimed at replacing field work. Like any hypothesis, they are based on assumptions which have to be kept in mind when interpreting model results and, if possible, should be verified. Also, like any hypothesis, they are constructions of the mind that allow us to think more clearly and more in depth about a problem; it is possible that a different model would refute the current one and be a better tool. This critical assessment is maintained by interaction with field research.

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Genetic Diversity in Arctic-nesting Geese: Implications for Management and Conservation

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Introduction

The North Pacific Rim harbors breeding populations of many unique wildlife resources, of which waterfowl are among the most abundant and taxonomically diverse. Arctic nesting geese in particular are wide-spread in distribution (Figure 1), and though only seasonal residents, they have evolved many unique adaptations for breeding in northern latitudes. This diversity has been recognized and managed at many taxonomic and geographic levels (Figure 2). Populations are spatially structured on macro- and micro-geographic scales reflecting taxon-specific migratory tendencies, and breeding and winter site fidelity.

The preservation of this diversity is a major goal of many state and federal management programs. However, there may be little time; although nesting habitats are largely unaltered since the last glaciation, many goose populations have been increasingly impacted on wintering areas in terms of population numbers and distribution (O'Neill 1979, Raveling 1984). Concomitant with these changes, many species and populations have experienced declines in levels of genetic diversity.

Effective conservation of any species must be based on a solid understanding of demographic and life history parameters (Lande 1988). Unfortunately, for migratory species, it often is difficult to obtain estimates of population parameters needed to assess the effects of factors regulating populations and to make predictions of species or population status due to complexities posed by high dispersal ability and use of numerous regions and habitats throughout the year. Populations may be affected by numerous factors intrinsic to both breeding and wintering areas. However, assessment of potential underlying factors may be monitored best in northern breeding areas where populations are spatially segregated. While direct techniques (Slatkin 1985, 1987) such as survey data, banding and telemetry have revealed much of the current information regarding the status of arctic goose populations, these techniques often are inadequate or prohibitively expensive to employ for assessing conservation-related questions.

One augmentative approach to resolving relationships at taxonomic and population levels involves the collection of genetics data (Smith et al. 1976). Genetics data offer several perspectives not available for direct observations or from morphological, behavioral or ecological data. Molecular techniques provide unambiguous information about specific gene regions or gene products with a known heritable basis, as analyses are based on homologous regions of the genome. Researchers can quantitatively compare differences across a wide array of taxonomically diverse species (Avise 1983), thus affording a historical component to investigate the timing and causes of events which underlie contemporary patterns and levels of diversity. Just as movements of banded or radio-collared individuals have been used to infer migratory affinities and dispersal, data of population differences in gene frequencies can be

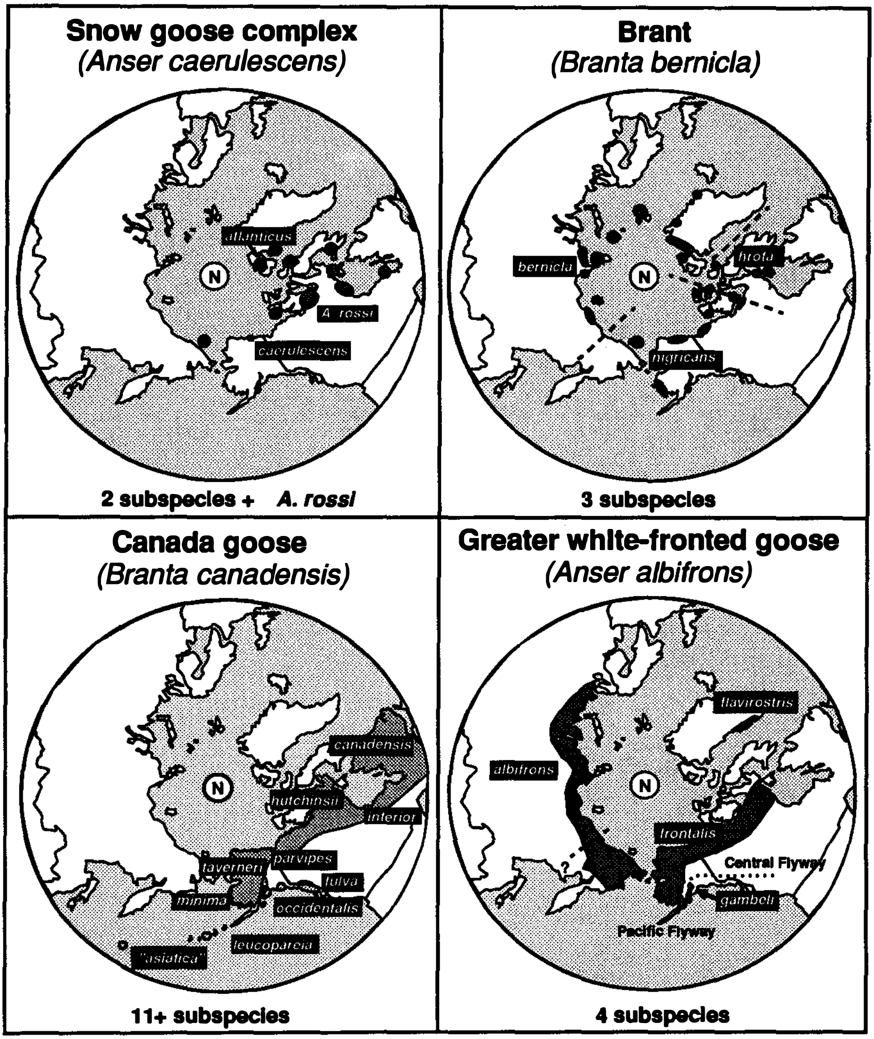

Figure 1. Breeding distribution of four species of arctic-nesting geese whose range encompasses all or part of the North Pacific Rim (after Delacour 1954, Owen 1980).

used for similar purposes. Genetic methodologies and underlying theory have proven fundamental in resolving questions in ecological genetics and evolutionary biology (*see* Burke et al. 1992 for review). There is growing appreciation of the application of population genetics and molecular systematics to management-related issues (Avise and Nelson 1989, Smith and Rhodes 1992).

Here we present a general overview of genetic and ecological data for several species of arctic nesting geese, and also provide preliminary findings from our studies of greater white-fronted geese (*Anser albifrons*) as examples of the broad application of different types of genetics data in species conservation. We also review many of

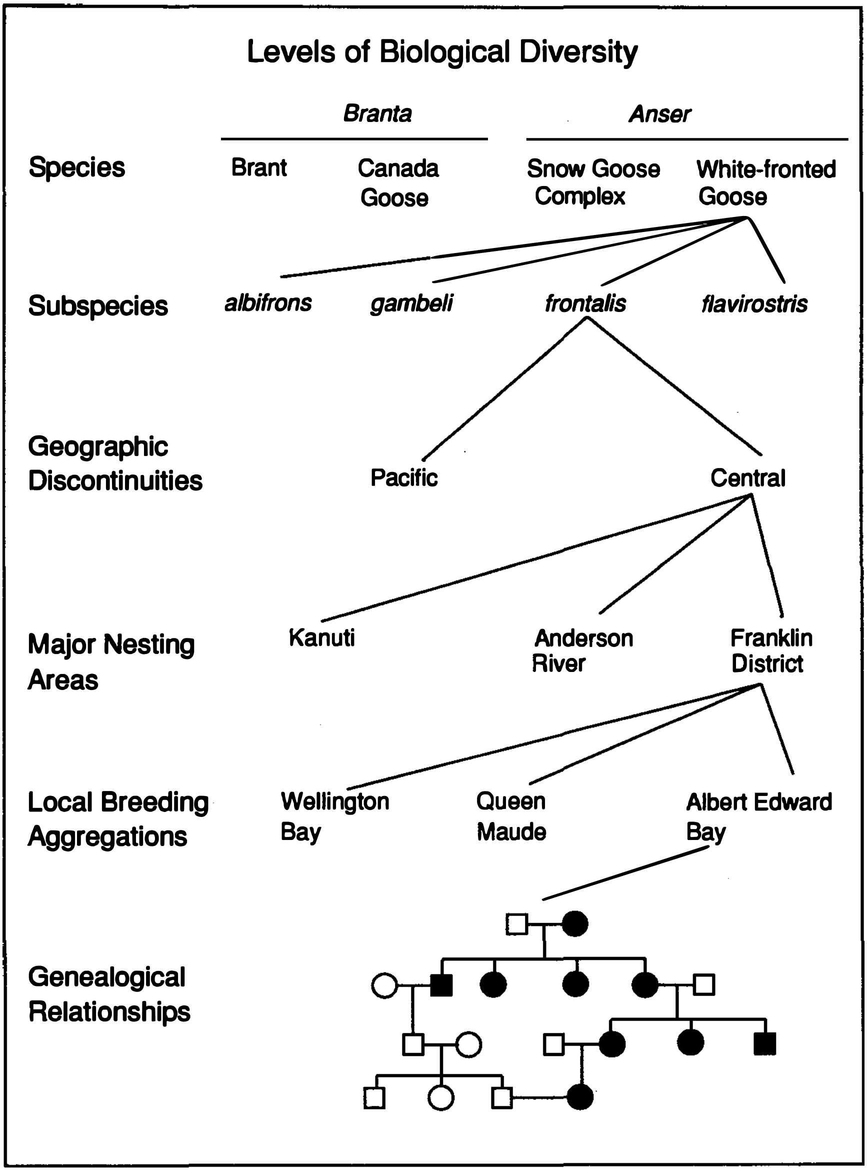

Figure 2. Schematic representation of the genetic diversity of arctic geese at taxonomic, macrogeographic, microgeographic and individual levels.

the molecular techniques which have been used to investigate relationships at the species, population and individual levels, and interpret observed patterns of genetic structuring with respect to life history attributes.

Direct Evidence of Population Structuring

Systematic status of various populations of arctic geese (Figure 1) predominantly has been based on degree of phenotypic variation relative to geographic distribution (Delacour and Mayr 1945, Delacour 1954, Owen 1980). In cases where phenotypic variation is extreme (e.g., Canada geese), subspecific status often has been conferred even if conspecific populations are sympatric. In the absence of unequivocal phenotypic characters, allopatry generally has been a requirement for subspecific designation (e.g., black and Atlantic brant). In instances of moderate phenotypic variation among sympatric or nearly sympatric populations, the taxonomic status is less clear (e.g., Tule white-fronted goose, Melville Island brant and many Canada goose populations) and inferences of spatial structuring may be made from studies documenting the degree of movement of birds between populations.

Phenotypic Variation and Taxonomic Subdivisions

Different species of arctic nesting geese exhibit varying degrees of phenotypic variation. The extent of morphological variation in Canada geese is legendary (Bellrose 1976). Divergence of body size between several subspecies is so extreme (e.g., the largest subspecies of Canada geese are three to four times larger than the cackling Canada goose) that some subspecies are effectively reproductively isolated. In contrast, lesser snow geese exhibit little morphological and, hence, taxonomically-recognized variation, although the sympatric-nesting Ross' goose (Figure 1) could be considered a smaller form of snow goose (Anderson et al. 1992). Greater white-fronted geese (Krogman 1979, Owen 1980, Timm et al. 1982) and brant (Boyd and Maltby 1979, Owen 1980) exhibit an intermediate degree of morphological variation. Emperor geese (*Anser canagicus*) apparently are mono-typic (Delacour 1954). Some caution should be used when taxonomic relationships are based solely on morphological characteristics, as phenotypic expression may be strongly influenced by environmental effects (James 1983, Cooch et al. 1991).

Studies of Movements and Distribution

Analyses of movements and distribution typically are based on recoveries or recaptures of birds fitted with metal leg bands, resightings of geese fitted with colored and coded markers (generally neck bands or leg bands), or with radio or satellite transmitters. The earliest and still most common of these studies rely on recoveries and recaptures of leg-banded birds. Much of our current flyway management of arctic goose populations is based on distributions inferred from recoveries of leg-banded birds (e.g., Miller et al. 1968, Lensink 1969, Bellrose 1976). Unfortunately, most of these studies simply document the presence or absence of inter-population movement of individuals without actually determining if dispersing birds contribute reproductively to another population (Erlich et al. 1975, Rockwell and Cooke 1977).

An analysis of the distribution of 11,500 recoveries from leg-banded greater white-fronted geese banded throughout North America indicates it is unlikely there is current gene flow between North American and Siberian populations (no cross continental recoveries), and little opportunity for gene flow between the Pacific and Central flyways (0.5 percent of recoveries of northern-banded birds were recovered outside flyway boundaries, C. Ely personal files: 1993). In contrast, a similar analysis of lesser snow geese reveals a much greater extent of movement between areas as exemplified by wintering distributions of geese banded on Banks Island, Northwest Territories. Over 85 percent of the 2,500 wintering-ground recoveries of birds banded on Banks Island have been in Pacific Flyway states (predominantly California) where they winter sympatrically with birds from Wrangel Island, Russia. The other nearly 15 percent of Banks Island geese have been recovered in the Central Flyway where they winter with birds breeding as far east as Hudson Bay. However, longitudinal affinities between breeding and wintering areas indicate that even continental populations of lesser snow geese are not completely panmictic (Dzubin 1979, Cooke et al. 1988).

Actual inferences of gene flow have best been documented by recaptures of birds banded on breeding areas. In studies based on recaptures and resightings of lesser snow geese, Cooke and his colleagues (Geramita and Cook 1982, Cooke 1987, Cooke et al. 1988) have reported a high degree of natal- and breeding-site fidelity in females relative to males (Rockwell and Cooke). However, even 5–10 percent effective dispersal rate of females (Cooke et al. 1975) may be sufficient to homogenize gene frequencies among breeding populations, particularly over long periods of time (Avise et al. 1992). Recaptures and recoveries of leg-banded black brant have revealed a moderate degree of movement of individuals among nesting and molting areas in eastern Siberia, Alaska and Western Canada (King and Hodges 1979, Ward et al. 1993, C. Ely personal files: 1994). Similar data indicate that populations of brant from the central and eastern Canadian arctic each have unique wintering areas (Boyd and Maltby 1979, Owen 1980, Reed et al. 1989a).

The advent of small, long-lived conventional (Tacha et al. 1989) and satellite transmitters (Ely et al. 1993) has allowed individual birds to be followed among breeding, molting, staging and wintering areas. Such studies are costly, but in remote areas, as is characteristic of much of the north Pacific Rim, the use of such devices may be the only safe and cost-effective way to document movements directly. Radio transmitters have been used to detect significant differences in the chronology of use of staging areas among different breeding populations of black brant (Reed et al. 1989b) and greater white-fronted geese in the Pacific Flyway (C. Ely and J. Takekawa personal files: 1993).

Marking programs currently are in place for nearly every species of arctic-nesting goose in North America, with additional programs underway in Siberia on lesser snow geese, greater white-fronted geese and bean geese (*Anser fabilis*). Results from these studies will provide important information on distribution, habitat use, affinities between breeding and wintering populations, and inferences of population structuring.

Genetic Techniques

The simplicity of the genetic code and linear arrangement of just four nucleotides of which all DNA molecules are derived belies the tremendous variation in DNA complexity, rates of change and selective pressures. Most organisms possess several distinct genomes (e.g., mitochondrial and nuclear) which differ in mode of inheritance (i.e., maternal versus bi-parental, respectively) and in the rate of evolution (quantified in terms of the number of mutations per segment of DNA per unit of time). Different molecular markers offer differing levels of resolution. Their respective properties make some more amenable for certain applications than others. For example, certain markers are employed best at higher taxonomic levels as variation among closely related species and within species are negligible. Choice of appropriate markers also depends on the question being addressed. For certain analyses it may be sufficient to have one or few diagnostic markers (e.g., species-specific alleles are useful for forensics purposes, Oates et al. 1983; unique mitochondrial DNA haplotypes are useful for the identification of Canada goose subspecies, Shields and Wilson 1987a, Van Wagner and Baker 1986, 1990). Additional techniques are available for other purposes such as establishing phylogenetic or biogeographical relationships, or for determinations of the spatial pattern and extent of variation among groups or individuals.

Sources of Material

DNA may be extracted from nearly all tissues. Of particular interest are materials which may be obtained via non-destructive methods and which may be collected and preserved for long periods of time using a minimum of effort, cost and with limited supporting facilities. Avian red blood cells are nucleated and are particularly attractive sources of DNA. Large quantities of high molecular weight DNA may be obtained from fractions of a milliliter. Feathers (Taberlet and Bouvet 1991), epithelial scales from the legs and egg shell membranes all have been successfully used in our laboratory (S. Miller et al. personal files: 1993). Collections of samples for DNA analysis can be taken directly from birds on breeding, molting, staging and wintering areas coincident with trapping and banding efforts, from hunter bag checks, or federal parts surveys. Samples may be indirectly obtained from feather or egg shells in nests. With the advent of polymerase chain reaction (PCR) techniques samples also may be obtained from museum specimens which may be hundreds of years old (Ellegren 1991). This latter source of material opens exciting research opportunities by facilitating comparisons between historical (i.e., pre-exploitation) and contemporary populations.

The predominant method of sample storage has been freezing. Recently, additional chemical preservation protocols involving ethanol (Smith et al. 1987) and high salt buffers (Bruford et al. 1992, Longmire et al. 1988, Seutin et al. 1991) have proven useful for preserving DNA in blood and tissue samples in undegraded condition suitable for most laboratory analyses for periods of several weeks to months at ambient temperatures. Alternatively, blood may be dried onto filter paper or glass slides. Dried egg shell membranes and feathers also may be kept at ambient temperatures for long periods of time.

The quantity and quality of DNA required will vary depending on the method employed. Methods such as single and multilocus DNA fingerprinting (Figure 3c) require large quantities ($\geq 5 \mu g$) of undegraded DNA. Methods employing PCR require far less quantity (several *ng*) (e.g., sequencing, mtDNA and nuclear DNA RFLPs, and microsatellites—Figure 3). Amplification of specific segments of DNA theoretically can be obtained from a single copy.

Applications

DNA sequences. DNA sequence variation ideally is assessed by direct determination of nucleotide sequences from homologous segments of specific gene regions, assayed from a series of individuals (*see* Hoelzel and Green 1992, Hillis et al. 1990,

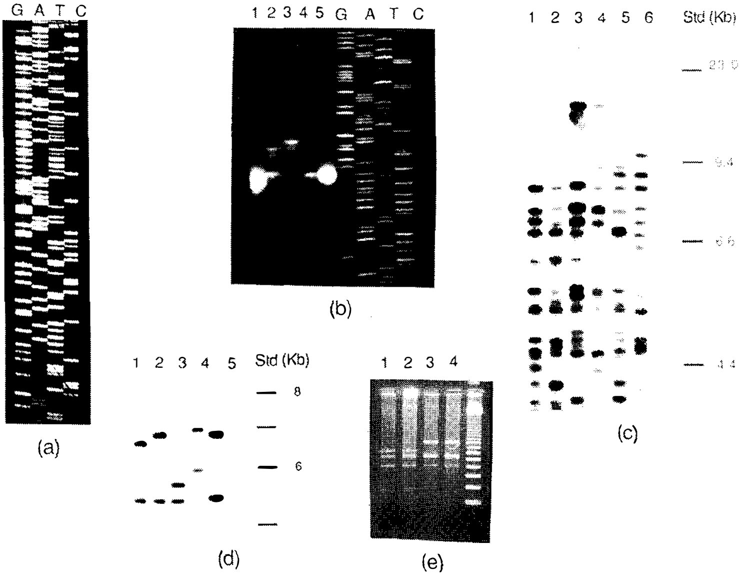

Figure 3. Genetic variation in greater white-fronted geese quantified using five molecular genetic techniques: (a) DNA sequence data from a portion of the cytochrome b region of the mitochondrial genome; (b) microsatellite allelic variation, an adjoining DNA sequence allows determination of size differences among alleles; (c) multilocus minisatellite profiles (DNA fingerprints); (d) allelic variation revealed using a single locus minisatellite probe; and (e) mitochondrial DNA restriction fragment length (RFLP) polymorphisms.

Simon 1991, for excellent technical descriptions). This technique is suited for taxonomic studies, although recently, applications of sequence analysis to population studies have been facilitated by the ease with which specific regions of DNA can be amplified using the polymerase chain reaction (PCR) technique (Kocher et al. 1989). Nucleotide sequence data (e.g., Figure 3a) are particularly attractive because characters (nucleotides) are the basic units of information encoded by organisms and the size of most genomes, and thus the potential size of data sets is quite large.

Restriction fragment length polymorphisms (RFLPs). An alternative method for obtaining information on sequence variation involves comparing the number and size of fragments produced by digesting DNA with restriction endonucleases. Resulting RFLP variation has been used extensively in inter- and intra-specific analyses (*see* Wilson et al. 1985, Avise et al. 1987, Moritz et al. 1987, for reviews of mitochondrial DNA literature). Excellent references detailing methods for isolation and characterization of mtDNA RFLP variation can be found in Lansman et al. (1981), Chapman and Powers (1984), Shields and Helm-Bychowski (1988), Dowling et al. (1990) and Solignac (1991). Figure 3e shows restriction site polymorphisms in PCR amplified mtDNA among four greater white-fronted geese. Specific PCR primers also can be used to amplify regions within the nuclear genome (Quinn and White 1987, Karl and Avise 1992, Aquadro et al. 1992).

Nuclear variable number of tandem repeat (VNTR) loci. One class of nuclear markers which are receiving considerable attention are the variable number of tandem repeat (VNTR) loci (Tautz et al. 1986, Burke 1989, Tautz 1989, Burke et al. 1991). These loci are tandemly repeated segments of DNA which can show extensive allelic differences in length due to variation in repeat copy number. Microsatellites (Figure 3b) are tandem repeats composed of short (1–5 bp) motifs (*see* Rassman et al. 1991 for general techniques). This method utilizes PCR and thus is appropriate for population-level analysis, as well as analysis of individual-specific variation. In contrast, mini satellites are composed of tandem repeats of larger size (15–100 bp). Alleles may differ in size by several thousand base pairs. Probes containing a specific cloned mini satellite sequence allow the characterization of specific alleles at a single locus (single locus fingerprinting, Bruford et al. 1992, Figure 2d). Alternatively, multiple loci may be resolved simultaneously using a probe containing the core repeat sequence (multilocus fingerprinting, Figure 3c). Single and multilocus VNTR techniques have been utilized primarily to establish identity and relatedness among individuals. However, these loci also may prove to be a powerful tool for addressing ecological questions at the population level.

Indirect Estimates of Population Structuring

Contemporary distributions of arctic nesting geese do not reflect the complex series of historical events which have led to the establishment of nesting, brood rearing and molting sites, or of migratory routes, as arctic nesting geese are recent residents of high arctic habitats (Ploeger 1968). Thus, direct observations alone are insufficient to address questions of species evolutionary history or ecology. Direct markers provide information concerning the movements of individuals but not the genetic consequences of migration (i.e., whether individuals successfully breed in a new location). Further, current measurements of straying, or of extirpation or recolonization events provide no direct information about the magnitude, duration or consistency of these events over time.

In general, birds at all taxonomic levels exhibit less genetic diversity than has been documented for other vertebrate groups (Avise and Aquadro 1982, Barrowclough et al. 1985, Kessler and Avise 1984, 1985, Patton and Avise 1985). Low levels of inter-specific variation are suggestive either of relatively recent speciation or a decelerated rate of evolution within the specific genetic regions assayed. The lack of appreciable differentiation among geographic populations has been attributed to high levels of gene flow and moderately large effective population sizes (Barrowclough 1980), or to recency of population separation.

Genetic studies of Pacific Rim geese have not been conducted to address management issues directly. Rather, genetic markers have been used to resolve questions pertaining to rates of gene flow, phylogeny, historical biogeography and behavioral ecology. The collective literature are by no means extensive and sample sizes from which conclusions were drawn are, in many cases, quite small. However, the existing studies do address a large number of issues, and collectively greatly enhance our knowledge of evolutionary relationships and contemporary features of these species' biology. The distribution of genetic variation within and among species suggests a number of generalizable scenarios.

Systematic relationships. Understanding phylogenetic relationships is a fundamental prerequisite for understanding the adaptive significance of phenotypic variation among taxa (Harvey and Pagel 1991) or of historical biogeographic events which have contributed to present species distributions and movement patterns. In a phylogenetic sense, macroevolution (i.e., speciation) is an extrapolation of contemporary processes (i.e., movements, breeding structure, stochastic events, selection). Organisms have parents, who in turn have parents, and so forth back through evolutionary time. Thus, branches in phylogenetic trees have a substructure that consists of smaller branches, ultimately resolved as generation-to-generation pedigrees (Figure 2, Avise et al. 1987).

Levels of sequence divergence estimated from DNA RFLP or sequence analysis suggest that the divergence of present-day species and subspecies predates the Pleistocene glaciations. Mitochondrial DNA's from three species of *Anser* (Ross, snow and white-fronted geese) form a closely related group and are highly divergent from two species of *Branta* (the Canada goose and brant), which are themselves quite distinct genetically (Shields and Wilson 1987a, 1987b). Additional phylogenetic analyses have revealed large differences between the Emperor goose and *Anser* and *Branta* species (Quinn et al. 1991). Estimates of mtDNA sequence divergence among *Anser, Chen* and *Branta* (supported by fossil data) suggest a divergence time of between 4–5 million years BP.

Analysis of mtDNA variation among *Anser* species suggests a lack of appreciable genetic differentiation. Estimates of percentage sequence divergence between snow geese and Ross' geese (0.80, Shields and Wilson 1987b) is less than that described among many subspecies of Canada geese (range 0.11–2.54, Van Wagner and Baker 1990). Avise et al.(1992) found Ross', and greater and lesser snow geese all share the same mtDNA genotypes.

Subspecific relationships. Taxonomic affinities at the subspecies level often are assigned based on morphology (e.g., body size) and color. Studies which have investigated the degree of subspecific variation in genetic characteristics and the degree of concordance between morphological and genetic divergence have met with mixed success. Several studies have documented unique mtDNA genotypes in each of several subspecies of Canada geese (Shields and Wilson 1987a, Van Wagner and Banker 1990). Assuming a clocklike accumulation of genetic divergence in mtDNA sequence (Wilson 1988), Van Wagner and Baker (1990) estimated divergence times from a common ancestor between large- and small-bodied forms at approximately 700,000 years. Using these same calibrations these authors estimate divergence times of 100,000 years among subspecies within large- and small- bodied forms.

Several researchers have attempted to relate taxonomic differences assigned based on plumage coloration to estimates of genetic divergence. Shields (1990) found brant from Melville Island which differed in color phase to be genetically diverged from other arctic nesting black brant populations. Avise et al. (1992), using mtDNA restriction fragment polymorphisms, and Quinn (1992), using mtDNA sequence analysis, found little evidence of genetic differentiation among lesser snow geese of different color phases (dark versus light). Cooke et al. (1988) did observe slight but significant differences in allozyme allele frequency among dark and light color phases.

Macrogeographic structure. Estimates of the degree of divergence among genotypes suggest that populations of many arctic nesting geese existed in allopatry, presumably within regional glacial refugia. Co-occurrence of genetically divergent genotypes within the same breeding population suggests that for some species (e.g., the snow goose complex) considerable mixing of birds from formerly allopatric regions has occurred (Avise et al. 1992). This is in contrast to Canada geese which exhibit morphological and genetic variation among breeding populations (Shields and Wilson 1987a, Figure 4a), but may winter sympatrically. Our findings suggest that geographic variation in greater white-fronted geese may be similar to that found in Canada geese (Figure 4b).

Comparisons across species suggest a relationship between the distribution of nesting areas and degree of population genetic structuring. For colonial nesting species (snow geese and brant), relatively little geographic differentiation has been observed. In contrast, for species such as greater white-fronted geese and Canada geese, which nest in a more dispersed and continuous manner, considerable evidence has been found for spatial genetic differentiation. These results could reflect differences in effective population sizes and the possibility of stochastic drift in gene frequencies. Founder effects and some degree of population bottlenecking has been inferred based on findings of low levels of mtDNA genotype diversity within geographic locations. Alternatively, differences simply could reflect species-specific propensities for dispersal.

The disparity of direct and indirect measures of gene flow are greatest for species such as arctic nesting geese which live in different areas during different times of the year. For species whose dispersal capabilities are constrained by natural barriers (e.g., freshwater fishes), molecular data consistently reveals a high degree of concordance between genealogical relationships and geographic proximity. Many freshwater fish species in the United States and arctic regions of Canada have retained the evolutionary signature of past vicariant events (i.e., glacial refugia, Bernatchez and Dodson 1991, Avise 1992) as inferred from the distribution of mtDNA genotypes.

Microgeographic Variation

Few studies have addressed issues related to the degree of genetic structuring within local areas. Van Wagner and Baker (1986), using allozymes, found no significant differences in allele frequency between Canada goose nesting from Rankin Inlet and Eskimo Point, Northwest Territories. Rockwell and Cooke (1977), citing evidence from mark/recapture studies, concluded that gene flow among nesting colonies within western light color phases and eastern dark color phases would preclude local differentiation. Scribner and Ely (personal files: 1993), using hypervariable mini satellite and micro satellite VNTR loci, have found significant differences in allele frequency among populations of greater white-fronted geese in Alaska and around Victoria Island, Northwest Territories. For certain species, observations of localized nesting aggregations in otherwise uniform nesting habitat, high natal-site fidelity and maintenance of family groups during migration (Table 1), suggest some degree of genealogical structuring within localized nesting populations. Non-zero additive genetic variance (heritability) for several important life history traits (clutch size and hatching

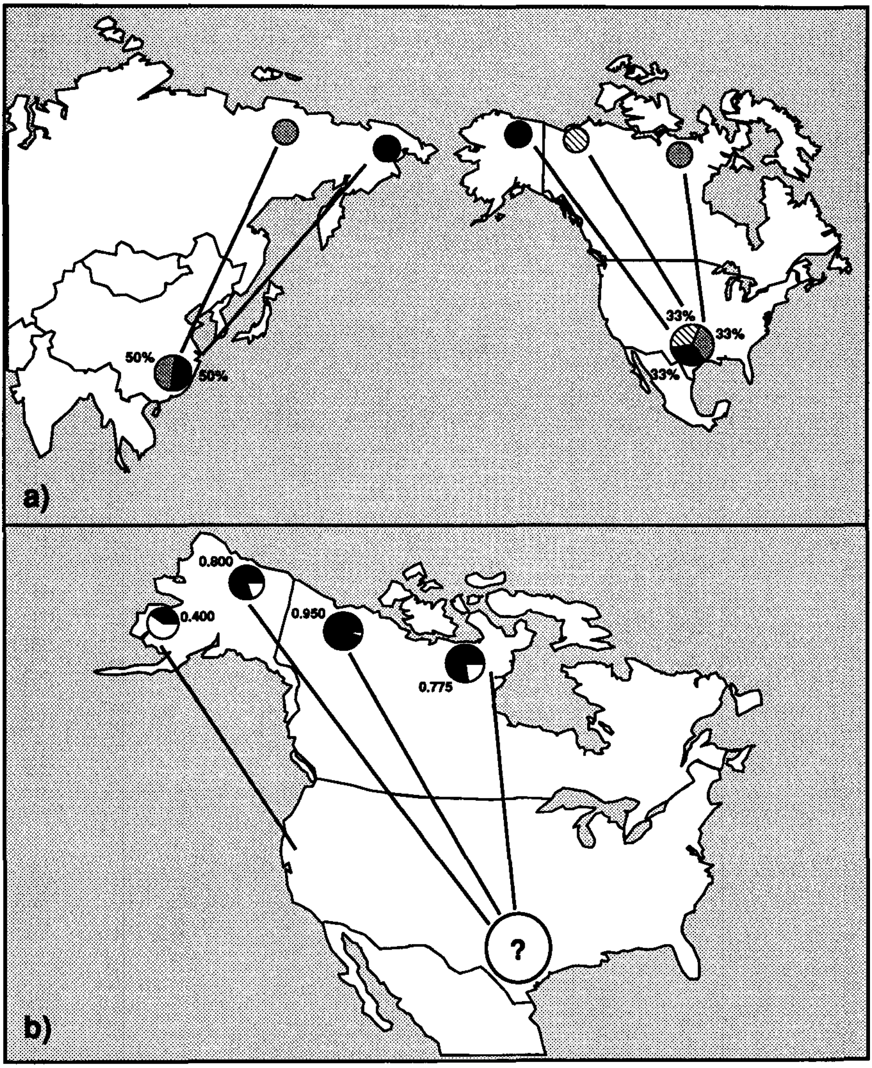

Figure 4. Spatial genetic structuring in arctic-nesting geese, (a) hypothetical distribution of allele frequencies among nesting goose populations. Each population is shown to be fixed for different alleles. Given the estimates of allele frequencies of admixed wintering populations, maximum likelihood estimates of the proportions of birds originating from each nesting area is possible, and (b) actual data of allele frequency differences in a nuclear DNA pseudogene for each of four nesting populations of greater white-fronted geese.

date; Cooke 1987) also suggest a genetic component of underlying traits which vary geographically.

The lack of intra-population genetic studies of arctic nesting geese belies the

Table 1. Selected life history attributes of Pacific-rim geese and propensity for population structuring.

Species of goose	Natal site fidelity	Breeding site fidelity	Sex ratio (percentage female)	Timing of pairing	Pairbond stability	Assort. mating	Family stability	Nesting behavior	Nesting distribution	Molt migration	Inter-flyway dispersal	Potential for structuring
Lesser snow	F-biased	F-biased	49.6	wint-sprg	strong	yes	strong	colonial coastal	interrupted	strong	high	low
	1[a]	1	2,3	4,5	5,6	7,8	5	9	9	10,11,12	9	
Brant	F-biased	F-biased	50.3	wint-sprg	weak	yes	weak	colonial coastal	interrupted	strong	high	mod
	13	13	14	15	16	17	18	9	9	9,12,19	9,19,20,21,22	
Emperor	?	Yes	48.9	sprg-sum	strong	?	mod/strong	dispersed semi-coastal	interrupted	strong	high	?
		23	24		23		23	9	9	9,25	26	
White-fronted	yes	yes	48.5	sprg-sum	strong	yes	strong	dispersed	continuous	low-mod	low	mod-high
	27	27	28,29	27	28	27	28	9	9	12	30,31	
Canada	F-biased	F-biased	49.3	sprg-sum	variable	yes	variable	dispersed	continuous	variable	low	high
	32,33	32,33	34	32,35	36,37	35	36	9	9	12,38,39	9	

References: 1)Cooke et al. 1975; 2) McLandress 1983; 3) Bird Banding Laboratory (BBL)---based on 79,000 adults banded south of breeding areas; 4) Cooke 1987; 5) Prevett 1972; 6) Cooke et al. 1981; 7) Cooch and Beardmore 1958; 8) Ankney 1977; 9) Bellrose 1976; 10) Abraham 1980; 11) Lumsden 1975; 12) Salomonsen 1968; 13) J. Sedinger personal communication: 1993; 14) BBL—based on 16,000 bandings of locals on breeding areas; 15) Einarsen 1965; 16) D. Ward personal communication: 1994; 17) Abraham et al. 1983; 18) Jones and Jones 1966; 19) King and Hodges 1979; 20) Reed et al. 1989a; 21) Boyd and Maltby 1979; 22) Ward et al. 1993; 23) Petersen et al. in press; 24) BBL—based on 3,400 bandings of locals on breeding areas; 25) Palmer 1976; 26) BBL—recovery of Alaska-banded bird on E. side of Bering Sea; 27) C. Ely personal files: 1989; 28) Ely 1993; 29) BBL—based on 14,000 bandings of adults south of breeding areas; 30) Miller et al. 1968; 31) Lensink 1969; 32) *B. c. minima*—C. Ely personal files: 1993; 33) Lessells 1985; 34) BBL—based on 9,300 bandings of adult *minima* south of breeding areas; 35) MacInnes 1966; 36) Johnson and Raveling 1988; 37) Raveling 1988; 38) Davis et al. 1985; 39) Sterling and Dzubin 1967.

tremendous importance of these types of data to questions of population ecology, conservation and behavioral ecology. For example, Quinn et al. (1987, 1989) successfully have used nuclear RFLP variation to document cases of female nest parasitism, multiple paternity and incidence of female-female pairs in lesser snow geese. Other potential applications include establishing the degree of fidelity of local nesting groups to brood rearing and molting areas.

Life History Attributes Influencing Population Structuring: A Goose is not a Goose

Arctic nesting geese have many unique life history characteristics that regulate gene flow and contribute to population structuring (Table 1). Arctic geese exhibit female-biased natal and breeding philopatry, relatively equal sex ratios, extensive male parental investment, and long-term monogamy (Owen 1980, Anderson et al. 1992). Most of these attributes are in stark contrast to ducks (Tribe Anatini) and, hence, broad generalizations regarding the evolution of social systems and genetic structure of waterfowl in general do not necessarily pertain to geese (Rohwer and Anderson 1988, Anderson et al. 1992). Female-biased natal and breeding site fidelity (e.g., Greenwood 1987), and the timing and process of pair formation (Cooke et al. 1988) often are cited as among the most important factors regulating the magnitude and direction of gene flow in geese.

Very little is known about the timing and process of pair formation in most species of geese, but it is likely that mid-winter pairing may not be the general rule for all arctic nesting geese. The most detailed studies of pair formation in geese have been conducted in Europe with captive or semi-captive birds (Lorenz 1959, 1966, Dittami 1981, Choudhury and Black 1993). Studies of wild populations (e.g., Owen et al. 1988) are rare. The evidence is somewhat equivocal, but nearly all studies show that pair formation often occurs in spring and summer. Additional reports implicate the importance of summer liaisons initiated between pre-breeders on the breeding grounds and on molting areas (Lorenz 1959, MacInnes 1966, Raveling personal communication: 1977). The high frequency of pursuit flights (*see* Owen 1980) of cackling Canada and greater white-fronted geese on the breeding grounds, and the relative absence of such flights on wintering and southern staging areas also may indicate that the pair formation process in these species takes place in spring and summer (C. Ely personal files: 1987). Pursuit flights are observed commonly throughout winter in lesser snow geese (Prevett 1972), Ross' geese (C. Ely personal files: 1983) and black brant (Einarsen 1968). Many authors have attributed patterns of genetic structuring and high rates of gene flow in lesser snow geese to the fact that pair formation occurs in winter, when birds from different breeding areas are sympatric (Rockwell and Barrowclough 1987, Cook et al. 1988). If pair formation occurs while populations are segregated on the breeding grounds or on spring staging areas then the potential for gene flow among populations would be greatly reduced. Non-random pairing could be realized even if mate selection occurred in winter if breeding geese exhibit a high degree of fidelity to specific wintering areas (Raveling 1979, Vangilder and Smith, 1985, Novak et al. 1989, Wilson et al. 1992) or females prefer to mate with phenotypically similar males (Cooch and Beardmore 1959, MacInnes 1966, Cooke and McNally 1975, Abraham et al. 1983).

Implications for Management

Of immediate concern to managers of arctic nesting goose populations is protecting threatened species, recognizable subspecies or distinct breeding populations. Under existing guidelines, several criteria must be met for any group or population to be considered "distinct" for purposes of conservation (*e.g.,* Waples 1991). First, recognition of unique subspecies, races or distinct geographic populations implies some degree of reproductive isolation sufficient for evolutionarily important differences to accrue. Second, groups should contribute substantially to the overall diversity of the species, as defined either by morphological or genetic criteria, or by unique ecological or behavioral adaptations to regional environmental conditions.

In the absence of readily identifiable traits, management often becomes problematic, relying on differences in the timing of migration, differences in migration routes or use of wintering areas; characteristics which can show considerable overlap among groups and may vary greatly from year to year. Identification of the proportions of admixed wintering geese which originate from different regions is of particular concern when groups from each breeding region are characterized by different population trends (e.g., different subspecies of Canada geese in Oregon, and the Wrangel Island and Banks Island populations of lesser snow geese in California). Genetics data has been used successfully to identify admixed aggregations (Millar 1987, Rhodes 1993) and already has increased our understanding of the genetic structuring of geographically isolated breeding populations of Canada geese in the Aleutian Islands (Shields and Wilson 1987a).

Use of genetics data in a managerial context has not been restricted to the identification of population subdivisions. Temporal variances in allele frequency can be used to calculate effective population sizes (Waples 1989) and to study the effects of various harvest strategies on population breeding structure (Scribner et al. 1985), and the effects of stocking and translocations on population levels of genetic variability and inter-population differentiation (Scribner 1993).

Administrators may question the practicality of managing populations of migratory birds below the level of species. However, effective subspecies (and species) management of Pacific Rim geese already has been demonstrated as exemplified by the Pacific Rim-nesting Aleutian Canada goose, the cackling Canadian goose and the greater white-fronted goose. These populations all are several times larger than a decade ago (U.S. Fish and Wildlife Service unpublished data) due to management efforts to restrict sport and subsistence harvest.

Destruction and fragmentation of native habitat or changes in land-use practices (Cooke et al. 1988, Avise et al. 1990) also can have a dramatic effect on timing and routes of migration, and the dispersion of birds on wintering areas. Perhaps the most important consequences of habitat change lie in the affects of mixing of birds from different natal origins at the time of pair formation and mating as the magnitude of population genetic structuring appears to be related to life history patterns.

Conclusions

Arctic nesting geese have evolved within a changing landscape. The diversity we recognize and strive to preserve today has evolved over long periods of time relative to past environmental and demographic constraints. Historically, species distributions have expanded and contracted in response to dramatic climatic changes across much of the present breeding areas. In recent times, species distributions and abundance have changed in response to harvest and habitat fragmentation and loss, which are most pronounced in regions occupied during migration and winter.

Genetic markers have led to a greater understanding of geese taxonomy and the extent of geographic population structuring. Analyses have revealed important taxonomic relationships at the species and sub-species levels, though data occasionally are at odds with morphological criteria. Within-species data suggest that the propensity for population structuring at the regional or microgeographic scale is mediated by a number of life history traits, including timing, location, mechanisms of pair formation and degree of philopatry. The utility of using genetic markers to resolve questions of population structuring, dispersal and migratory movements, and for assessing temporal variation in breeding effective population size is similarly dependent on the same suite of species characteristics.

Genetic data have revealed that management of a migratory species as one panmictic population may neglect important genetic differences among unique breeding populations, some of which already are extirpated (e.g., Canada geese on the Commander Islands that wintered in Japan, Palmer 1976). Additional changes are inevitable, but present policies based on sound ecological and genetics research may help minimize human impact. To appreciate the relative importance of the evolutionary processes which underlie observed patterns, behavioral and ecological studies should be conducted in conjunction with studies of population genetics (Lande 1988, Zink and Remsen 1988, Cooke 1987, Cronin 1993).

Acknowledgments

We would like to thank numerous cooperators throughout North America and Asia who have provided samples for our ongoing genetics work with greater white-fronted geese. D. Derksen kindly reviewed an earlier draft of this paper. Research was supported by the National Biological Survey and the U.S. Fish and Wildlife Service.

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Habitat Considerations for Polar Bears in the North Pacific Rim

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Introduction

Polar bears (*Ursus maritimus*) occur in low densities throughout the polar basin and are circumpolar in their distribution (DeMaster and Stirling 1981). They are solitary predators except during breeding and rearing of young, and are characterized by a long life span, late age of sexual maturity and low reproductive rates (Amstrup and DeMaster 1988). The potential rate at which a polar bear population is capable of increasing is low and they are vulnerable to over-harvest if not managed conservatively (Taylor et al. 1987).

In 1973, in response to world-wide concern about reported increasing levels of harvest and an absence of quantitative knowledge of population size, the five nations responsible for managing polar bears (Canada, Denmark/Greenland, Norway/Svalbard, the United States and the former Soviet Union) signed the International Agreement on the Conservation of Polar Bears and Their Habitat (Agreement) in Oslo, Norway. The Agreement came into effect in 1976 after it was ratified by the minimum three countries. The Agreement fostered a large amount of research on population size, discreteness, movement and harvest patterns for the purpose of managing populations within sustainable levels.

Article II of the Agreement stated that ''Each contracting Party shall take appropriate action to protect the ecosystems of which polar bears are a part, with special attention to habitat components such as denning and feeding sites and migration patterns'' (Stirling 1988: 209). This portion of the Agreement has not been responded to extensively by most countries, although progress has been made for protection of some denning areas. For example, the three largest known concentrations of denning polar bears, Wrangel and Herald islands (Russia), Kong Karl's Land (Svalbard), and western Hudson Bay (Canada), are all protected. Denning habitat also is protected in the Northeast Greenland National Park and the Melville Bugt Game Reserve (Greenland), other islands in Svalbard, and in some national and provincial parks in Canada. Denning habitat in Alaska within the Arctic National Wildlife Refuge is incidentally protected, but a portion of this area is under consideration for potential petroleum exploration and development (Clough et al. 1987). To date, no feeding areas or migration routes of polar bears have been specifically protected anywhere in the Arctic.

The U.S. Fish and Wildlife Service is developing a polar bear habitat conservation strategy for the United States as required by provisions of the marine mammal incidental take regulations (50 CFR Part 18, Federal Register 58(219): 60,402–60,412). In light of this requirement, it is appropriate to review the current state of knowledge of polar bear habitat use in general and as it relates specifically to the population in the North Pacific Rim. For the purposes of this discussion, the North Pacific is defined as the northern Bering Sea, the southern Chukchi Sea and the western Beaufort Sea (Figure 1).

General Ecology of Polar Bears in the North Pacific Rim

Two populations of polar bears have been hypothesized to occur in Alaska, a northern population in the Beaufort Sea and a western population in the Chukchi and Bering seas (Lentfer 1974). The Beaufort population is shared with Canada (Amstrup et al. 1986) and the Chukchi/Bering population is shared with Russia (Garner et al. 1990).

Polar bears are predators with a diet consisting primarily of ringed seals (*Phoca hispida*) and secondarily of bearded seals (*Erignathus barbatus*) (Stirling and Archibald 1977, Smith 1980). Atlantic walrus (*Odobenus rosmarus rosmarus*) are a minor component of the diet in Canada (Kiliaan and Stirling 1978, Calvert and Stirling 1990), but Russian scientists believe Pacific walrus (*Odobenus rosmarus divergens*) are an important component of the summer diet in the northern Chukchi Sea (S. E. Belikov unpublished data). The seasonal distribution and abundance of ringed and bearded seals in the Arctic are influenced by sea-ice conditions and water depth (Smith and Stirling 1978, Stirling et al. 1982, Kingsley et al. 1985). Consequently, the seasonal distributions of polar bears are similarly affected (Stirling and Archibald 1977, Stirling et al. 1984).

Data from mark-recapture studies indicate widespread movements of bears within localized areas (Stirling et al. 1975, 1977, Lentfer 1983, Schweinsburg et al. 1983), but data on the specific movement patterns of individual polar bears have not been practical due to the remoteness of polar bear habitats. Satellite telemetry technology (Fancy et al. 1988, Garner et al. 1989) greatly enhanced the ability to study the widespread movements of polar bears. Two general patterns of polar bear movement are evident. Polar bears in the Canadian Arctic have an archipelagic pattern, with extensive use of offshore sea-ice and ice-covered inter-island channels of the Central and High Arctic islands during autumn, winter and spring. During summer, the sea-ice may melt completely and polar bears become stranded on land to await the return of the sea-ice (Derocher and Stirling 199), or retreat to ice-covered bays and later over-summer on land (Schweinsburg 1979, Stirling et al. 1984). In contrast, polar bears in the Beaufort and the Chukchi/Bering seas have a pelagic pattern, remaining on the offshore sea-ice throughout the year, with limited use of land during summer months (Amstrup 1986, Garner et al. 1990, Amstrup and Gardner 1991, Garner and Knick 1991, Garner et al. 1994).

Movements and habitat use patterns of polar bears in the Bering, Chukchi and

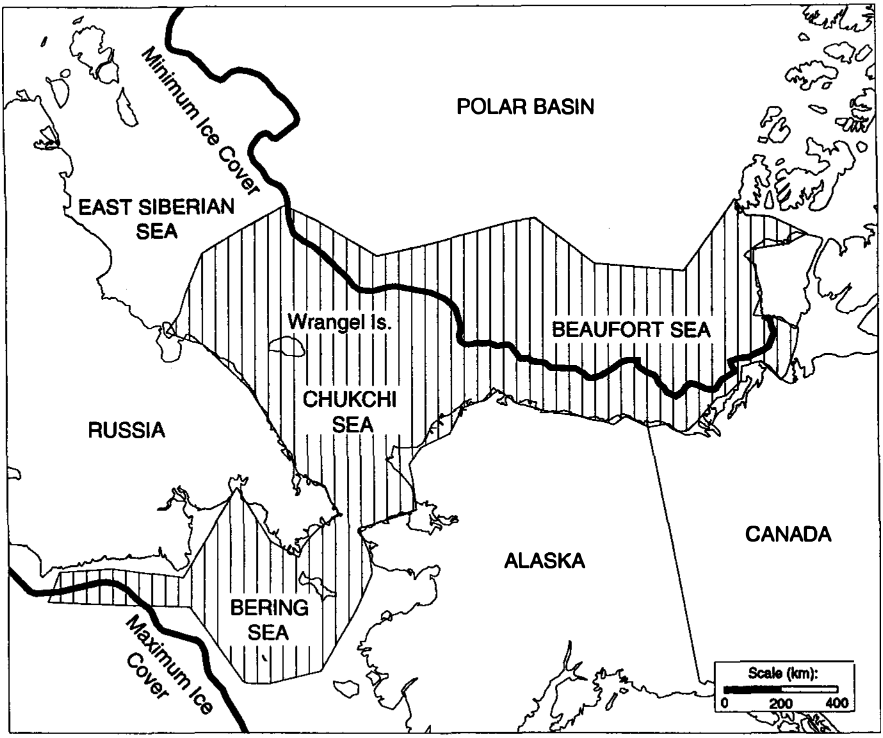

Figure 1. Extent of North Pacific Rim polar bear populations (shaded) and maximum and minimum ice cover in the Bering, Chukchi and Beaufort seas.

Beaufort seas are influenced by large-scale seasonal changes in sea-ice cover (Garner et al. 1990). In the Beaufort Sea, the ice pack normally recedes from the shoreline a distance of 60 to 95 miles (100–150 km) by late summer, while the ice pack in the Bering/Chukchi seas normally recedes approximately 870 miles (1,400 km) from maximum ice cover (Figure 1). Sea-ice habitats consist of three major types: (1) fast ice in the littoral zone near shore, (2) the polar pack ice that covers the central polar basin, and (3) the drifting pack ice that occurs as a zone between the fast ice and the polar pack ice (Lentfer 1972). The polar pack is comprised of large multiyear floes that separate from the polar pack ice, break up to varying degrees and drift south into the Chukchi Sea, but rarely into the northern Bering Sea, which is covered largely with annual sea-ice.

Movement of sea-ice in the Beaufort sea generally is east to west, while sea-ice in the Chukchi Sea generally moves south (Lentfer 1972), becoming compressed as it moves southeast along the Chukotka Peninsula in Russia and southwest along the Seward Peninsula in Alaska until it is extruded through the Bering Straits (Figure 2). Sea-ice in the Bering Sea moves south, but ice that passes through the Bering Straits from the Chukchi Sea accumulates along the northern coast of St. Lawrence Island (Burns et al. 1981). Polar bears occur throughout the sea-ice habitats, and their movements may coincide with or oppose the general pattern of sea-ice movement (Garner et al. 1994).

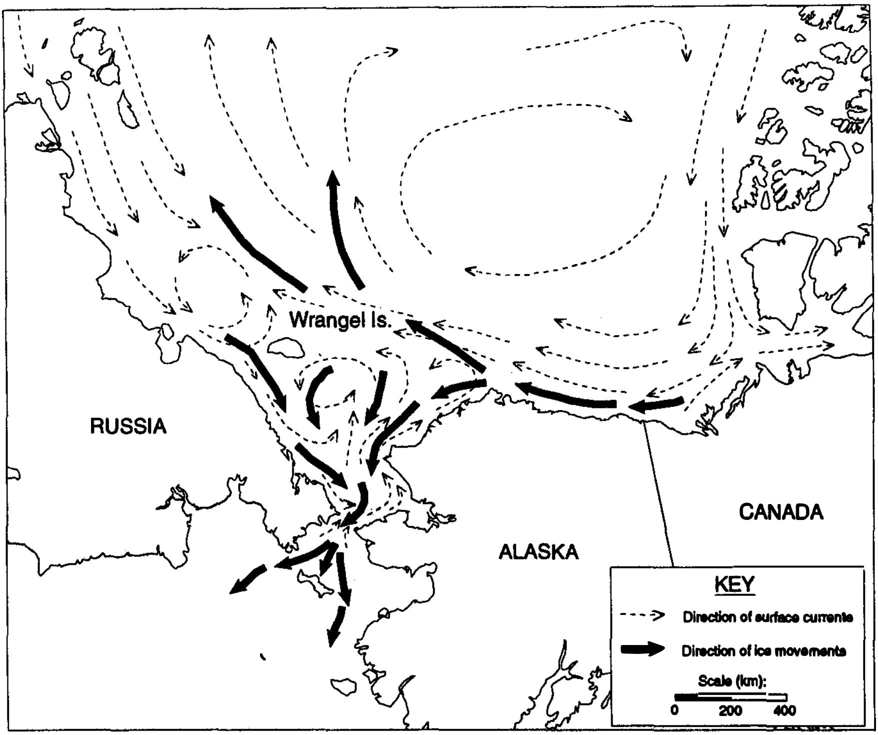

Figure 2. Surface circulation pattern in the polar basin and typical ice movement patterns in the Bering, Chukchi and Beaufort seas (based on Lentfer 1972).

Annual ranges of individual polar bears in the pelagic sea-ice habitats of the Beaufort Sea (3,800–104,000 square miles: 10,000–270,000 km^2 (Amstrup 1986) are more extensive than in the archipelagic habitats of Canada (965–8,800 square miles: 2,500–23,000 km^2 (Schweinsburg and Lee 1982), while ranges of polar bears in the pelagic sea-ice habitats of the Bering and Chukchi seas are the most extensive (57,900–135,100 square miles: 150,000–350,000 km^2 (Garner et al. 1990).

In early winter, pregnant females excavate maternity dens in snow and ice (Harrington 1968, Lentfer and Hensel 1980, Ramsay and Stirling 1990). Unlike black bears (*Ursus americanus*) and brown bears (*Ursus arctos*), non-parturient polar bears do not enter dens but remain active throughout the winter (Stirling et al. 1984), except during periods of extreme cold or inclement weather when they may use temporary dens (Messier et al. 1992). Denning on land by parturient female polar bears is widespread in the Canadian arctic (Harrington 1968, Kolenosky and Prevent 1983, Ramsay and Stirling 1990), and the majority of the denning in the Chukchi Sea also is on land on Wrangel and Herald islands, and along the northern coastline of the Chukotka Peninsula in Russia (Uspenski and Chernyavski 1965, Uspenski and Kistchinski 1972, Stishov 1991). One instance of pelagic denning north of Wrangel Island has been recorded (Garner et al. 1990), and several bears have denned on land in northwestern Alaska (Figure 3). Denning habitat in the Beaufort Sea includes both

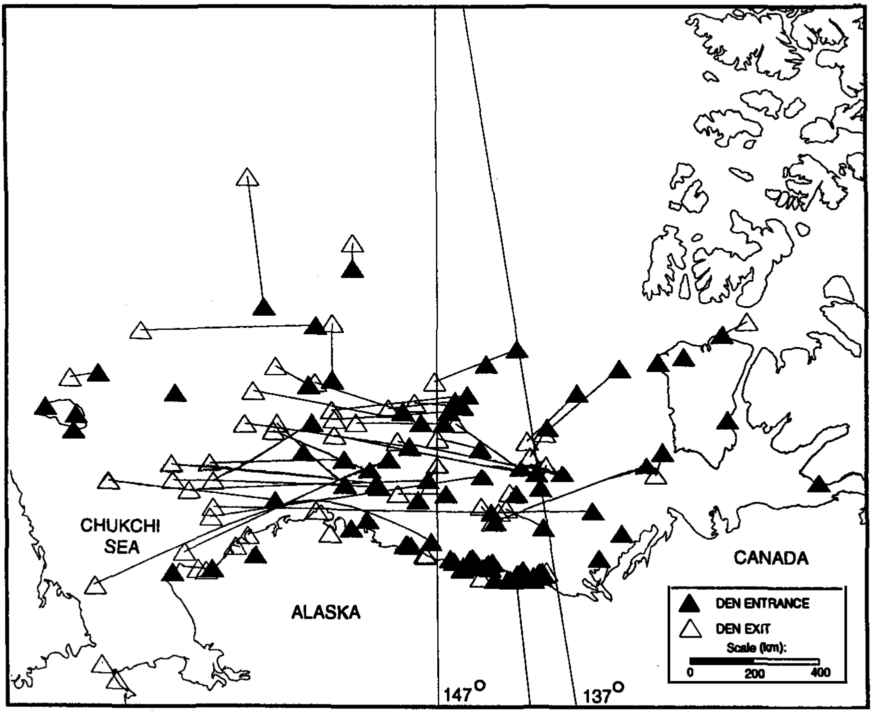

Figure 3. Location of maternity dens for 90 female polar bears, with direction of drift and den emergence locations for pelagic dens, 1981–1991 (modified from Amstrup and Gardner 1994).

land and the pelagic sea-ice (Lentfer 1975, Lentfer and Hensel 1980, Amstrup and Gardner 1994). Radio-collared bears used sea-ice habitats as denning substrate (Figure 3) in 53 percent of dens (48 of 90) located between 1981 and 1991 (Amstrup and Gardner 1994). Dens on the drifting sea-ice moved from 19 to 997 kilometers in a westerly direction between den entrance in October to December and emergence the following March or April (Amstrup and Gardner 1994). Land dens in the Beaufort Sea are concentrated along the coastal region near the Alaskan and Canadian border. Denning on land in northwest Alaska (Figure 3) is rare (Stirling and Andriashek 1992, Amstrup and Gardner 1994).

Habitat Conservation in the North Pacific Rim

Discussion of practical conservation measures for polar bear habitats generally will follow the three categories noted in the Agreement: denning, feeding and migration areas. The environmental disruptions and their potential impacts on polar bears in the Canadian High Arctic described by Stirling et al. (1984) also are applicable to polar bear habitats in Alaska, although the inaccessibility and greater importance of the pelagic sea-ice habitats to polar bears in Alaska may reduce their vulnerability. The level of protection to be afforded polar bear habitats is not specified in the Agreement and is subject to interpretation. Protection could include measures to ensure no disturbance (area closure), to limit disturbance within set criteria or time constraints (temporal closures), or to limit disturbance to levels that result in no detectable detrimental effects upon the population. Criteria probably would have to be established for determining what would be considered detrimental.

Denning

Denning polar bears are thought to be sensitive to disturbance and may abandon maternity dens if the disturbance is prolonged (Belikov 1976, Lentfer and Hensel 1980, Larsen 1985, Amstrup 1993), although Blix and Lentfer (1992) noted that noise and vibration levels generated by petroleum-related activities over 62 yards (100 m) from artificial dens were not detectable above normal background levels. The mainland coast of Canada and Alaska between 137 degrees 00 minutes W and 146 degrees 59 minutes W contained 80 percent (28 of 35) of land dens (Figure 3) recorded by Amstrup and Gardner (1994) and, although the density of dens was low relative to other denning concentrations, this area was considered critical denning habitat for the Beaufort Sea population (Amstrup 1993). Stirling and Andriashek (1992) and Amstrup and Gardner (1994) speculated that land denning may be increasing in the Beaufort Sea. On the basis of unpublished data on hunting polar bears in their dens prior to about 1970, and historical information provided by Leffingwell (1919), Stirling and Andriashek (1922) speculated that polar bears with a tradition of denning along the Beaufort Sea coast were extirpated by hunting after modern firearms were introduced. This extirpation may have been the stimulus that resulted in such a large proportion of denning on the pack ice in the Beaufort Sea documented by Amstrup and Gardner (1994). Protection of pelagic dens on drifting sea-ice in the Beaufort Sea would be difficult because of the large area involved (Figure 3), the difficulty of locating dens, the dynamic nature of sea-ice and possible jurisdiction problems under the Law of the Sea.

Protection. The most concentrated land-denning habitats in northeastern Alaska (Amstrup 1993) are afforded a level of protection from industrial development by being included within the Arctic National Wildlife Refuge (Refuge). However, a portion of the Refuge is under consideration for petroleum exploration and development (Clough et al. 1987). Active management of industrial activities would be needed to protect denning if development occurs (Amstrup 1993). Also, polar bears in Alaska are harvested by Alaskan native subsistence hunters under provisions of the Marine Mammals Protection Act. The harvest is unregulated, unless depletion of the population is documented, and denning females and their young can be taken legally. However, an agreement between northern Alaskan and northwestern Canadian native groups provides protection for denning females and their young (Nageak et al. 1991). Most polar bears that occur in western Alaska den on Wrangel Island, which currently is protected as a state nature reserve in Russia.

Maternity dens that occur in the pelagic sea-ice habitats of the Beaufort Sea are distributed over large areas and are largely inaccessible during the period of occupancy, with little potential for human disturbance. In addition, because the sea-ice drifts so extensively in the Beaufort Gyre, an area cannot be defined except on a scale too large to be practical.

Feeding and Migration

The large distances traveled by polar bears on sea-ice and the dynamic nature of sea-ice habitats have made studies of polar bear habitat selection difficult. Martin and Jonkel (1983) used direct observation of bears on relatively stable near-shore ice to determine a location preference and a preference for rough ice in a localized study area in Barrow Strait, Canada. Aerial reconnaissance of bears and their tracks in seven broad sea-ice habitat types were used to determine habitat preferences of polar bears during late winter and spring in the western Canadian arctic (Stirling et al. 1993). They detected differences in habitat preferences between different age and sex classes of polar bears, with a general preference for floe-edge and the moving ice habitats. Belikov and Gorbunov (1991) used data collected during aerial ice survey patrols to approximate the distribution of polar bear occurrence throughout the Russian arctic and polar basin. Integration of the results of these studies is difficult due to differences in ice habitat classifications between studies and differences in the inference base associated with each study. Also, none of these studies addressed polar bear habitat selection throughout the year.

Recently, satellite-based remote sensing technology has provided daily images of sea-ice concentration over much of the Arctic. This technology, coupled with remotely determined locations of polar bears instrumented with satellite transmitters, offers the opportunity to assess polar bear habitat selection throughout the year. However, the scale of resolution for the remotely sensed sea-ice data is coarse with cell seizes of 240 square miles (625 km^2) (Arthur et al. 1993). Analyses are complicated further because availability of sea-ice habitat types changes continually within and among years, so conventional analytical methods for determining resource selection (Manly et al. 1993) are difficult to apply (Arthur et al. 1993). Existing techniques of evaluating habitat selection assume that availability of habitat types is constant, at least during a defined period (Arthur et al. 1993), and the entire study area is considered available for selection by individual animals. Neither assumption is valid for polar bears using the sea-ice habitats.

Polar bears use seasonal sea-ice habitats of the Bering Sea when sea-ice is present between November and May each year (Arthur et al. 1993, Garner et al. 1994). Considerable spatial and temporal variation in ice coverage and types are common within and among years (Naval Oceanography Command 1986), and application of protective measures within this seasonal context would be difficult. Fast ice habitats in the Beaufort and Chukchi seas where ringed seals have pups and the drifting sea-ice over the continental shelf are important feeding areas for polar bears during spring (Stirling et al. 1993). The distribution and productivity of ringed seals can be highly variable, possibly in relation to changes in biological diversity or ice conditions (Stirling et al. 1977, 1982).

Protection. Polar bear habitats in the pelagic sea-ice of northern and western Alaska occur in remote areas with limited potential for human disturbance throughout most of the year. Because feeding and migration areas are so extensive, it is difficult to define specific areas of a practical seize to apply protective measures. Options for conservation of feeding and migration areas would have to be quite flexible, and based on temporal and spatial restrictions. For example, in late winter and early spring, the shore lead system along the south coast of the Beaufort Sea is an important feeding area and it tends to overlay much of the most likely offshore areas for hydrocarbon reserves (Stirling 1990). Although it is not practical to totally protect this vast area, it might be appropriate to restrict specific activities in particular areas at important times, in order to reduce the risk of direct negative effects.

Conclusions

There are practical limits to what can be done by the United States to meet its obligation for habitat protection under provisions of the Agreement. Land-based denning can be addressed to a certain degree, but the pelagic sea-ice habitats are more difficult to address because of scale and annual variability. Also, the current level of understanding concerning seasonal and annual habitat use patterns of the sea-ice habitats by polar bears is limited.

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A Seabird Monitoring Program for the North Pacific

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Introduction

Seabird monitoring is the accumulation of time series data on any aspect of seabird distribution, abundance, demography or behavior. Typical studies include annual or less frequent measures of numbers or productivity; less commonly, the focus is on marine habitat use, phenology, food habits, survival (as in mark-resighting studies) or mortality (as in beached bird surveys). The key requirement is that observations are replicated over time, and made with sufficient precision and accuracy to permit the meaningful analysis of variability and trends.

Along the Pacific coast of North America, seabird monitoring has consumed substantial amounts of public funding since the early 1970s. The effort stems from various legislative and executive mandates and largely has been uncoordinated among the many entities involved, including provincial, state and federal agencies, some private organizations, university faculty and students. As the demand for seabird monitoring increases and new efforts come on line, particularly on the Asian side of the North Pacific, it is desirable to assess where we have been and to strive for better integration of the work in the future. Our aim in this paper is to reaffirm the rationale for monitoring seabirds, to review briefly the nature and accomplishments of the existing effort, and to suggest some steps that can be taken to improve the effectiveness of seabird monitoring in the Pacific.

Why Monitor Seabirds?

The value of monitoring seabirds is twofold. On one hand, wildlife managers and the public are concerned about the welfare of particular species and populations that may be affected by human use of coastal lands and marine resources. But equally important—and aside from any value placed on this particular group of animals—is the role that seabirds can serve as indicators of change in the marine environment.

Important threats to Pacific seabirds include oil pollution, the introduction of predators to nesting islands, conflicts with commercial fisheries, and disturbance or habitat loss associated with human population growth in coastal areas. In the public perception, pollution is the most notorious of these problems because of the highly visible damage to wildlife that occurs during an oil spill (Bourne et al. 1967, Hope-Jones et al. 1978, Piatt et al. 1990). Less well known or appreciated are the possible demographic effects of chronic low-level pollution by oil at sea (Piatt et al. 1991, Burger and Fry 1993).

Introduced predators and other exotics have caused considerable damage on seabird nesting islands in the past (Moors and Atkinson 1984, Bailey 1993, Bailey and Kaiser 1993). Some of these changes probably are irreversible. Enlightened attitudes generally prevail today concerning planned introductions, but it is difficult to guard against unintentional introductions of exotics, especially rats. The likelihood of further damage is high.

Seabird conflicts with commercial fisheries include direct mortality from drowning in gill nets (DeGange et al. 1993) and competition for shared prey resources (Furness 1982, Furness and Ainley 1984). The incidental take of seabirds by fisherman often is a high-profile issue, yet overfishing and the alteration of marine food webs may have greater significance for seabirds over the long term.

Habitat loss and human disturbance to seabird nesting grounds are serious concerns (Vermeer and Rankin 1984, Litvinenko 1993). These problems likely will intensify with human population growth in coastal areas. In the United States, people inhabiting the coastal zone are expected to number about 127 million by the year 2010, a 60-percent increase since 1960 (Culliton et al. 1990). This undoubtedly will create a variety of management problems for marine and coastal resources, including seabirds.

The idea that seabirds can serve as monitors of changing marine environments is gaining acceptance worldwide (Croxall et al. 1988, Kushlan 1993). In the Pacific, seabirds are known to respond dramatically to El Niño events (Duffy 1990, 1993), but that is but one well-studied example of the kinds of large-scale oceanographic and atmospheric processes to which seabirds are sensitive (Myers 1979, Schumacher and Ried 1983, Royer 1993). Rhinoceros auklet (*Cerorhinca monocerata*) nestlings have shown significant changes in growth rate following small changes in sea temperature (Bertram et al. 1991). In another study, a colony of kittiwakes (*Rissa tridactyla*) exhibited demographic changes associated with long-term trends in weather in the North Sea (Aebischer et al. 1990). Such findings increase the relevance of seabird monitoring in an era when global climate change is a growing concern.

Fishery managers are realizing that seabirds can serve as cost-effective samplers of young year classes of commercial fish stocks, which otherwise are difficult to assess. In Alaska, diet samples from tufted puffins (*Fratercula cirrhata*) provide an early indication of year-class strength in walleye pollock (*Theragra chalcogramma*), a species of enormous commercial importance in the Gulf of Alaska and Bering Sea (Hatch and Sanger 1992). Other promising results are reported from British Columbia (Bertram and Kaiser 1993), California (Anderson et al. 1980, Sunada et al. 1981) eastern Canada (Montevecchi and Berruti 1991), southern Africa (Crawford et al. 1983, Berruti 1985) and Norway (Barrett 1991). Clearly, there is ample justification for seabird monitoring in the Pacific, not only because of concerns for the welfare of the birds themselves, but also because of the contribution these studies can make to fisheries oceanography and management policies for the marine system as a whole.

Designing a Program

Seabird monitoring is most effective when it incorporates planned comparisons. To a large extent, this principle should guide the selection of species, parameters and sites to include in a Pacific-wide program. Among the 86 species that breed in the Pacific north of 20 degrees N (Harrison 1983), the choice of animals is further governed by the specific objectives of monitoring. If the goal is to monitor the health of the marine environment, conventional logic suggests we would want to select species that sample that environment in a variety of ways. For example, we might categorize species as surface feeders or divers, fish or plankton feeders, nearshore or offshore in respect to foraging habitat, then select one or more species from each group for study. On the other hand, we may choose to observe species that are especially valued, rare or vulnerable, without regard to how representative they may be. Other considerations include ease of study (generally greater for open as opposed to concealed nesters) and geographic representation throughout the area of interest.

In principle, any of a large number of variables could be measured at intervals to reveal the effects on seabird populations of natural variability and human activities in the marine environment. A practical list of candidates is presented in Table 1. Measures of population size are arguably the first priority in any monitoring effort because that is the feature of any species' biology we ultimately are trying to conserve. Because seabirds are long-lived, however, other features such as breeding success, feeding or survival rates may give earlier signals of changing conditions than population size itself. The most desirable approach is to examine a suite of responses that integrates and reflects the birds' interaction with their marine environment over a range of temporal and spatial scales (Cairns 1987, Croxall et al. 1988).

The allocation of effort among sites and regions raises possibly the most important issue. Our ability to interpret and apply the results of seabird monitoring is greatly enhanced by having broad geographic coverage for the species we choose to observe. Ideally, a few widespread species should be monitored throughout their ranges in the Pacific, which clearly requires an internationally coordinated effort. A monitoring program in which the effort is broadly distributed geographically also is advantageous because the local decline of a species, even if it is known to be caused by human activity, may be acceptable if the species is known to be secure throughout the majority of its range. For these reasons, we favor a seabird monitoring program in which a few species are monitored at many dispersed colonies at frequent intervals.

Seabird Monitoring to Date

In 1992, the Pacific Seabird Group (PSG) initiated a survey of seabird monitoring effort in the temperate North Pacific. Questionnaires went out to specialists stationed throughout much of the region, with the aim of compiling an inventory of past and present efforts to monitor Pacific seabirds. Respondents were asked to identify—by species, location and year—all measurements (annual indices) of seabird population parameters available from their own work or other studies known to them. The areas

Table 1. Seabird monitoring effort in the North Pacific: results of the PSG survey by parameter group.

Parameter group	Description	Number of observations[a]
A	Population (whole colony or index)	2,310
B	Productivity (young per unit population)	1,045
C	Components of productivity[b]	136
	Laying success (percentage adult population breeding)	
	Clutch size	
	Hatching success	
	Fledging success	
D	Survival (annual return of marked adults)	213
E	Phenology (various indices)	1,024
F	Food habits (various indices)	536
G	Other[c]	224
	Feeding rates	
	Chick growth rates	
	Foraging trip lengths	
	Incubation shift lengths	
	Condition index	
	Parental attendance (time allocation)	
	Egg volumes	
	Etc.	
Total		5,488

[a]An observation is a given parameter measured for a particular species in one location and year.
[b]Incomplete data. Survey requested information on components of productivity only in cases where overall productivity (parameter B) was not measured.
[c]Observations appropriate to the "other" category but not covered in the survey include beached bird censuses and replicated pelagic surveys.

surveyed included five Pacific states (Alaska, Washington, Oregon, California and Hawaii), British Columbia, the Russian Far East and Korea. Not encompassed were Mexico or Japan—important gaps that need to be filled—or the Peoples' Republic of China, for which few, if any, time series data on seabird populations exist (L. Wang and F. Zhang personal communication: 1992).

The PSG survey revealed that upwards of 5,000 observations on seabird population parameters are availabale from North Pacific colonies (Table 1). Because of the geographic omissions mentioned above and the likelihood that no region has yet been fully accounted for, we think the actual total will exceed 10,000 observations. Population size (accounting for 40 percent of the observations reported) has been the most widely studied aspect of seabird population biology, followed by annual productivity (19 percent) and phenology (19 percent).

Effort to monitor Pacific seabirds was minimal prior to 1970 (Figure 1). Since that time, the activity has expanded dramatically, to the point that 400–500 observations now are made annually on seabird population parameters throughout the Pacific. The mid to late 1970s was a period of increased effort associated with the Outer Continental Shelf Environmental Assessment Program (OCSEAP) in Alaska.

At least 57 (55 percent) of the 86 species breeding in the temperate Pacific region have been studied (Table 2). To date, much of the effort has centered on the auks (Alcidae, 42 percent of observations reported), gulls (Laridae, 24 percent) and cor-

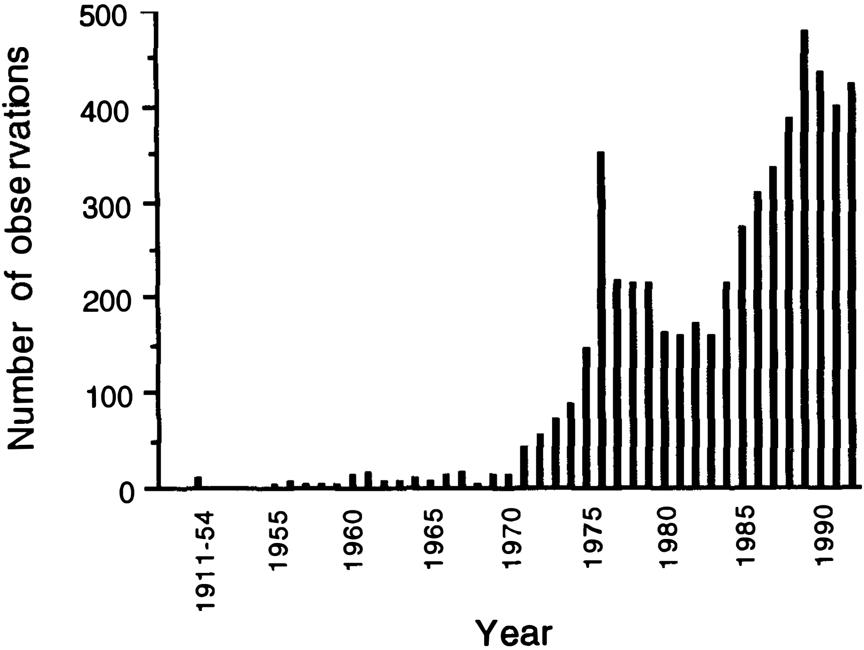

Figure 1. Temporal distribution of seabird monitoring effort in the North Pacific.

morants (Phalacrocoracidae, 21 percent). The single most studied species in the PSG survey was the black-legged kittiwake (*Rissa tridactyla*, 618 observations), followed closely by common murres (*Uria aalge*, 587 observations). Geographically, the current program is weighted heavily toward the west coast of North America, especially Alaska and California, where offshore leasing and other factors prompted increased effort beginning in the mid 1970s (Table 3). In California, a few sites have been studied relatively intensively. By contrast, a large number of sites have been worked more sporadically in Alaska, yielding shorter time series on average.

An important message from the PSG survey is that much information already exists on the population parameters of Pacific seabirds, and additional data are accumulating steadily. However, the lack of ready access to this information, by resource managers and researchers alike, is a continuing problem. Much of the information is never published in the open literature, or publication lags far behind the gathering of data. A comprehensive data management and distribution system is required to put information in the hands of those who need it in a timely manner.

Managing the Data

We envision a microcomputer-based system that consolidates and distributes data quickly, ideally within a few months after each summer field season. Among other benefits, this would allow managers and investigators to formulate and test hypotheses or make decisions about study emphasis in something more nearly approaching ''real

Table 2. Seabird monitoring effort in the North Pacific: results of the PSG survey by taxonomic group.

Taxonomic group	Number of species in region[a]	Number of species observed	Number of observations
Procellariiformes			
Diomedeidae (albatrosses)	3	2[b]	91
Procellariidae (petrels)	11	6	133
Hydrobatidae (storm-petrels)	9	6	152
Pelecaniformes			
Phaethontidae (tropicbirds)	3	1	12
Pelecanidae (pelicans)	1	1	111
Sulidae (boobies)	4	3	84
Phalacrocoracidae (cormorants, shags)	7	4	1,175
Fregatidae (frigatebirds)	2	1	29
Charadriiformes			
Haematopodidae (oystercatchers)	2	2	107
Laridae (gulls, terns)	25	16	1,298
Alcidae (auks)	19	15	2,296
Total	86	58	5,488

[a]Pacific Ocean and adjacent seas north of 20 degrees N.
[b]Excluding short-tailed albatross (*Diomedea albatrus*), for which an undetermined amount of information is available.

time.'' We recognize and understand, however, the reluctance of many investigators to turn over their hard-won data to any kind of central repository in advance of publication. Thus, use of the proposed system would be governed by rules that protect contributors from unauthorized or preemptive publication of their data.

It may be useful at this point to distinguish between the database we are considering and two related efforts in data management already familiar to seabird specialists. First, a *seabird colony catalog,* of which several examples exist for the Pacific coast of North America (Sowls et al. 1978, 1980, Speich and Wahl 1989), is basically a list of all known seabird colonies in a given region, with best available information on species composition and population sizes. It represents the state of knowledge of

Table 3. Seabird monitoring effort in the North Pacific: results of the PSG survey by region.

Region	Study period	Observation number of: Years	Sites	Species	Number of observations	Number of series	$\bar{X}$ observations /series
Korea	1986–1992	4	4	6	13	10	1.3
Russia	1973–1992	11	8	13	205	36	4.0
Alaska	1956–1992	33	52	26	2,685	646	4.2
British Columbia	1982–1992	11	11	12	205	36	5.7
Washington	1974–1992	17	11	12	330	66	5.0
Oregon	1973–1992	20	25	6	281	46	6.1
California	1967–1992	24	4	14	1,374	89	15.4
Hawaii	1911–1992	37	3	18	354	101	3.5
All regions	1911–1992	43	118	58	5,488	1,056	5.2

the distribution and abundance of breeding seabirds. This information is much in demand for land use planning, for damage assessment in the event of oil spills or similar events, and for the general information of everyone interested in seabirds. Estimates are of whole colony sizes and inevitably are crude in many instances. Second, a *pelagic seabird database,* typically with an associated atlas (*e.g.,* Gould et al. 1982, Brown 1986, Morgan et al. 1991), includes all at-sea censuses of seabirds, whether from ships, airplanes, land-based seawatches or small boats working the shoreline. Reasonably standardized techniques have been developed and used for most surveys conducted in the last 20 years or so. Such a database serves the same general purposes as the colony catalog, except it pertains to the pelagic distribution and abundance of seabirds, including the nonbreeding season. Both the colony catalog and pelagic seabird database are essentially descriptive in nature.

In contrast, a *seabird monitoring database* is designed specifically to work with observations on seabird population parameters that are replicated over time. Generally, only a few of the colonies in a given region will be represented, and data usually refer to sample plots rather than whole colonies. This, or companion databases, also can incorporate time series data on the physical and biological environment of seabirds as desired.

Once the sources of seabird monitoring data have been identified and recruited to the effort, the creation of such a database is straightforward. Essentially, we need to replace the parameter codes in our inventory of monitoring effort (Tables 1–3) with real values. A prototype database, currently being developed by the Pacific Seabird Group, includes ancillary information on each observation, such as contacts (names, addresses and phone numbers of persons responsible for the data), documentation (lists of published and unpublished reports that interpret the data or explain the methods used to collect it), sponsors (funding agencies) and comments (where contributors may wish to qualify a particular datum in relation to other values in a series). Commercial software provides the tools for filtering and selecting data efficiently by location, species, parameter type or year. In the most common application, users will want to employ graphics software to generate time series plots that are consistent in design and appearance. A few examples of this type of output are shown in Figure 2. The data also can be linked to statistical analysis software or mapping (GIS) programs for in-depth analysis of temporal and spatial patterns of variation. As a minimum service, the caretakers of this system (PSG or other) should expect to provide users with frequently updated versions of the database and a basic package of data management software on floppy disks or CD-ROM.

Three main obstacles exist to achieving the goal of a comprehensive database for seabird monitoring: (1) professional competition—the reluctance of any work group to allow another's version of a database to emerge as the "standard," thus placing its originators in the position of being "in charge" of all the available data; (2) ethical issues concerning the ownership and distribution of unpublished data; and (3) practical constraints of time and money among those who need to participate.

The first problem requires that we pursue this activity under the aegis of a professional organization like the Pacific Seabird Group. No single nation or government agency has responsibility for seabird research and conservation throughout the Pacific. However, members of the PSG represent all Pacific nations and all seabird interest groups, both public and private. As such, the PSG provides a professional umbrella under which any individual with the time and interest to do so can contribute to the

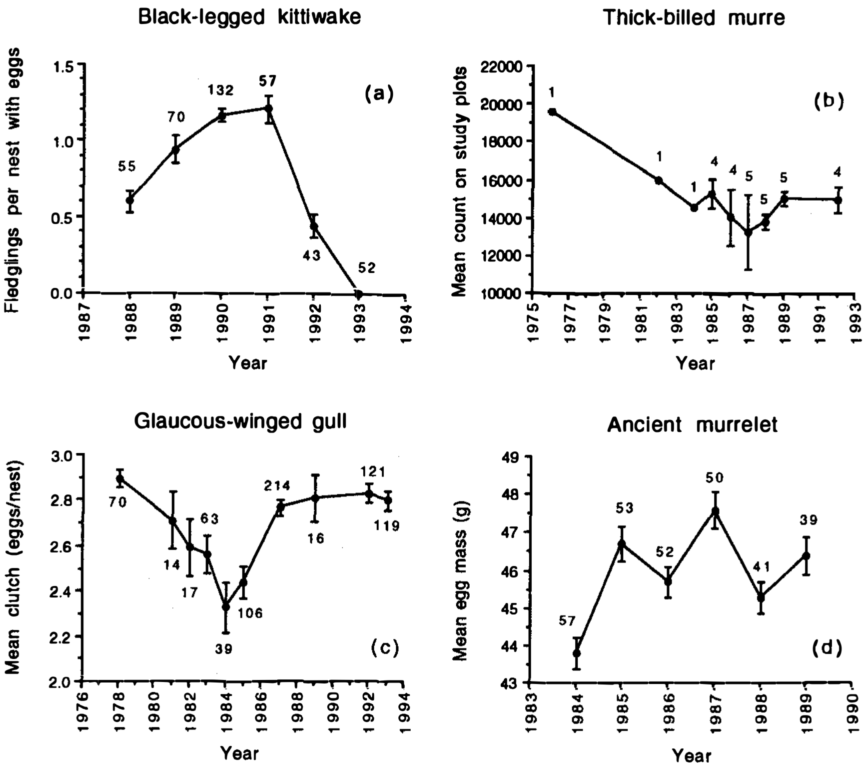

Figure 2. Examples of time series data from seabird monitoring studies in the North Pacific: (a) productivity of black-legged kittwakes on Talan Island, Sea of Okhotsk, Russia (A. Ya. Kondratyev unpublished data); (b) population index of thick-billed murres (*Uria lomvia*) on St. George Island, Alaska (Dragoo and Sundseth 1993); (c) clutch size of glaucous-winged gulls (*Larus glaucescens*) on Middleton Island, Alaska (S. A. Hatch et al. unpublished data); and (d) fresh egg mass of ancient murrelets (*Synthliboramphus antiquus*) on Reef Island, British Columbia (Gaston 1992). Mean, standard error and sample size are shown for each year with data available.

realization of a working database. We also would note that the availability of powerful and affordable microcomputers makes it possible to decentralize access to data in a manner that has not been typical of data-sharing schemes in the past.

Our prototype allows contributors to attach to each observation a data release attribute, specifying the types of use they would consider appropriate in advance of primary publication. Because many important uses of the data do not involve publication, and because most analyses would be synthetic and nonoverlapping anyway, we believe that skeptics will realize they have nothing to lose and much to gain from participation in the program.

Limitations on time and money are very real. We know from experience that even the most enthusiastic of potential contributors find it difficult to follow through when the realities of their jobs and existing workloads tend to relegate this activity to the category of "extracurricular." It is important, therefore, for top administrators to understand and support the effort by mandating participation at all levels and by making the joint venture an integral part of their seabird programs in the future.

Conclusions

We are confident that barriers to such a cooperative effort can be overcome. The benefits of doing so—for seabirds and seabird professionals alike—are clear. Besides the scientific applications of a seabird monitoring database—detection and geographic analysis of trends, hypothesis-testing based on correlation and concordance techniques, the assessment of means and variability in seabird life table statistics, to mention a few—we see the database as an important tool for managing and optimizing the field program. Managers will have a complete inventory of past and ongoing effort—which species are being monitored, which parameters, where and by whom. Updated on an annual basis, that information will permit a continuing assessment of where we are in seabird monitoring, where we would like to be and what we need to do to get there. We view this aspect of the effort as a means to "monitor the monitoring program" for Pacific seabirds.

The charge to natural resource agencies in participating countries is threefold: (1) cooperate in the design and development of the seabird monitoring database so that it fully meets the needs of its users; (2) ensure that all suitable information is incorporated in the database, including existing data as well as future results; and (3) commit the necessary personnel and funds to seabird monitoring on a continuing, long-term basis. With appropriate planning and cooperation, this commitment can be made with the assurances that every small effort contributes importantly to a larger program, and that all resulting data are accessible to resource managers on a timely basis.

Acknowledgments

We thank the many persons who responded to our requests for information in the PSG survey of seabird monitoring effort. We also are grateful to D. V. Derksen and D. L. Hayes for reviewing the manuscript.

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Restoration of Lesser Snow Geese to East Asia: A North Pacific Rim Conservation Project

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Introduction

Lesser snow geese (*Anser caerulescens caerulescens*) are one of the most abundant waterfowl species. The Holarctic population of lesser snow geese (hereafter referred to as snow geese) is about 2.2 million adults following large increases in central and eastern breeding populations in recent decades (Bellrose 1980, F. G. Cooch unpublished data). The origin and historical distribution of snow geese, however, are not well known. A recent genetics study (Quinn 1992) indicates that mitochondrial genome differences are evident among existing populations of snow geese, and the oldest or matriarchal population may be from Wrangel Island on the western edge of their current range. In this paper, we report on historic and current numbers of snow geese in Asia, review relevant literature on restoration projects of migratory birds, and discuss our research and restoration plans for snow geese in the North Pacific Rim.

Populations of Lesser Snow Geese in East Asia

Historical Distribution of Snow Geese Breeding in Russia

Russian records (Pallas 1769) indicated that snow geese or *Bely-gus* were common from the eastern end of the Chukotka Peninsula to the Lena River delta (130 degrees E) prior to the 1800s (Figure 1). Alferaki (1905) and Dement'ev and Gladkov (1952) reported that snow geese were very abundant on the Lena, Yana, Indigirka and Kolyma river deltas. Siberian hunters consumed snow geese through winter and collected goose down to sell in markets on Irkutsk (Alferaki 1905).

By the early 1800s, snow goose populations on the Yana River had profoundly decreased (Hendestrom 1823, Argentov 1861). A decline in snow goose numbers was noted in the Yana and Kolyma river areas in the early 1900s, and snow geese were very rare between the Yana and Indigirka rivers by 1912 (Zhitkov and Zenzinov 1915, Mikhel 1935). The last known, major coastal breeding area was on the Alazeya River near the Kolyma River Delta (Andreev 1994). Although little information has been recorded about the distribution of snow geese on Chukotka Peninsula (Figure 1) (Portenko 1972), no large colonies were found in Chukotka by the 1930s (Bousfield and Syroechkovskiy 1985).

Large numbers of snow geese first were described in 1926 on Wrangel Island, Russia, 140 kilometers north of the Arctic coast of Siberia (71 degrees N, 179 degrees E) and about 500 kilometers west of Alaska. When the first Soviets arrived, they observed several colonies, each exceeding thousands of nesting birds (Mineev 1946). The number of colonies began to decline because of egg collecting and hunting by the settlers, and by the mid-1950s, only two large colonies remained on Wrangel Island. A geological expedition camped near one colony and destroyed it in 1957–58 (Syroechkovsky and Krechmar 1981). The Wrangel Island population exceeded 200,000 adult birds in the early 1960s (Uspenski 1965), but only one colony of

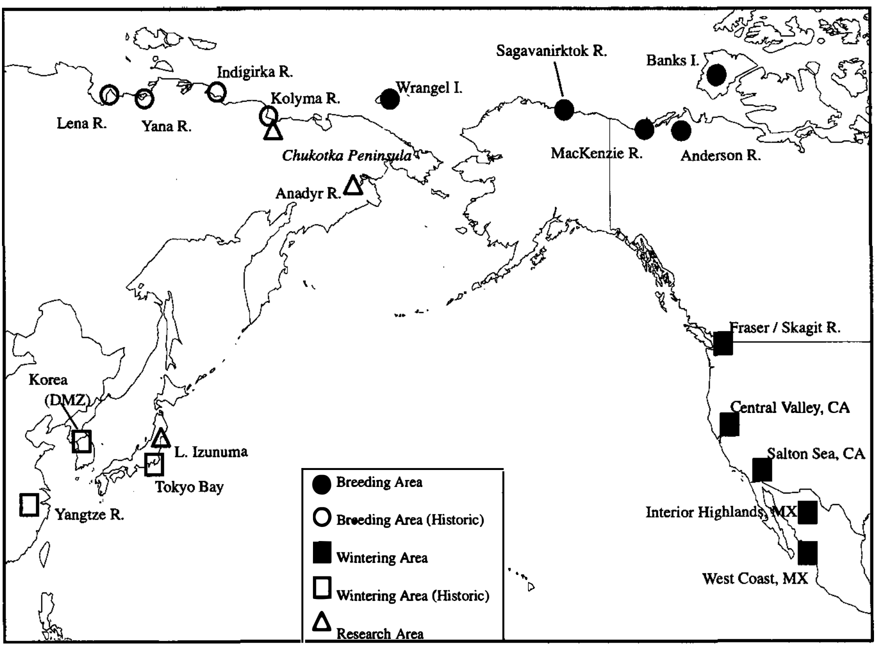

Figure 1. Breeding (circle) and wintering (square) areas of lesser snow geese in the North Pacific Rim. Present (open) and probable historic (solid) use areas are depicted, along with current restoration research areas (open triangles).

120,000 adults was present on Wrangel Island by 1969 (Bousfield and Syroechkovskiy 1985).

Historical Records of Snow Geese Wintering in Japan

Russian researchers surmised that geese breeding in eastern Siberia probably spent the winter in China and Japan (Kistchinski 1973). Although historical populations of snow geese in China are not known, our search of historical records verified that snow geese or *hakugan* formerly were abundant in Japan, especially on the island of Honshu (Seebohm 1890). For example, a painting entitled "Reed and Goose" drawn by Niten Miyamoto (1584–1645) documents snow geese roosting in a marsh, and a mural drawn in the late 1600s shows a flock of snow geese flying near the ancient capitol of Kyoto.

Snow geese were plentiful in the Kanto Plain region surrounding Tokyo until 1895 (Kuroda 1939) and even frequented the pond surrounding the imperial palace (Okada and Takagi 1986). Many snow geese were observed in regions of Hokudo, Musashi, Sagami and Bandou (Hotta 1794), and specimens were collected from Yokohama and Nagasaki (Austin 1949). A book written in the beginning of the Meiji period (1868) stated "at Tsukuda, the reclaimed area of Tokyo Bay which is the center of the Honjyo District now, snow geese appeared and landed flock after flock" (Takatsukasa 1934). Large concentrations of white geese once were common in Japan during winter near Susaki on Tokyo Bay, appearing "like snow" (Blakiston and Pryer 1878, 1882). There were some reports that greater snow goose (*A. c. atlanticus*) specimens were found in museums and Ross' geese (*A. rossii*) were observed in flocks, but most white geese now are presumed to have been *A. c. caerulescens* (Alferaki 1905).

Although numbers of snow geese were not reported in historic references, a review (Kanayama 1985) of Shogun Tokugawa's hunting records (1611–1790) provides an indication. Of 466 geese reported in the harvest from September to May, 177 were identified to species and 32 percent were snow geese. Because goose populations in that era were thought to be at least 10 times larger than current wintering populations of about 35,000 geese (Wild Bird Society of Japan 1988, M. Kurechi unpublished data) and, if we estimate that one-third of the populations was snow geese, populations of snow geese numbered in the tens of thousands. Snow geese remained abundant until 1890 (Austin and Kuroda 1953) when they suddenly disappeared. They since have become rare visitors to Japan, with an occasional individual accompanying flocks of bean geese (*A. fabalis*) in some years (Austin 1949, Brazil 1993).

Breeding Status of the Asian Population

Only one major population of snow geese now remains in the Palearctic (Bellrose 1980). Most breeding snow geese concentrate at Wrangel Island (Bousfield and Syroechkovskiy 1985), but a few small groups have been reported in other areas, including the Kolyma and Chaun Lowlands (Andreev and Dorogoi 1987, Andreev 1994). The population on Wrangel Island was estimated as 100,000 adults in the 1970s (Kistchinski 1973, Syroechkovsky 1981, Subcommittee on White Geese 1992). The colony has varied around 60,000 geese during the past decade (V. Baranyuk unpublished data, Subcommittee on White Geese 1992), but projections suggest that it may stabilize at as few as 24,000 geese (Syroechkovsky 1981). The precipitous

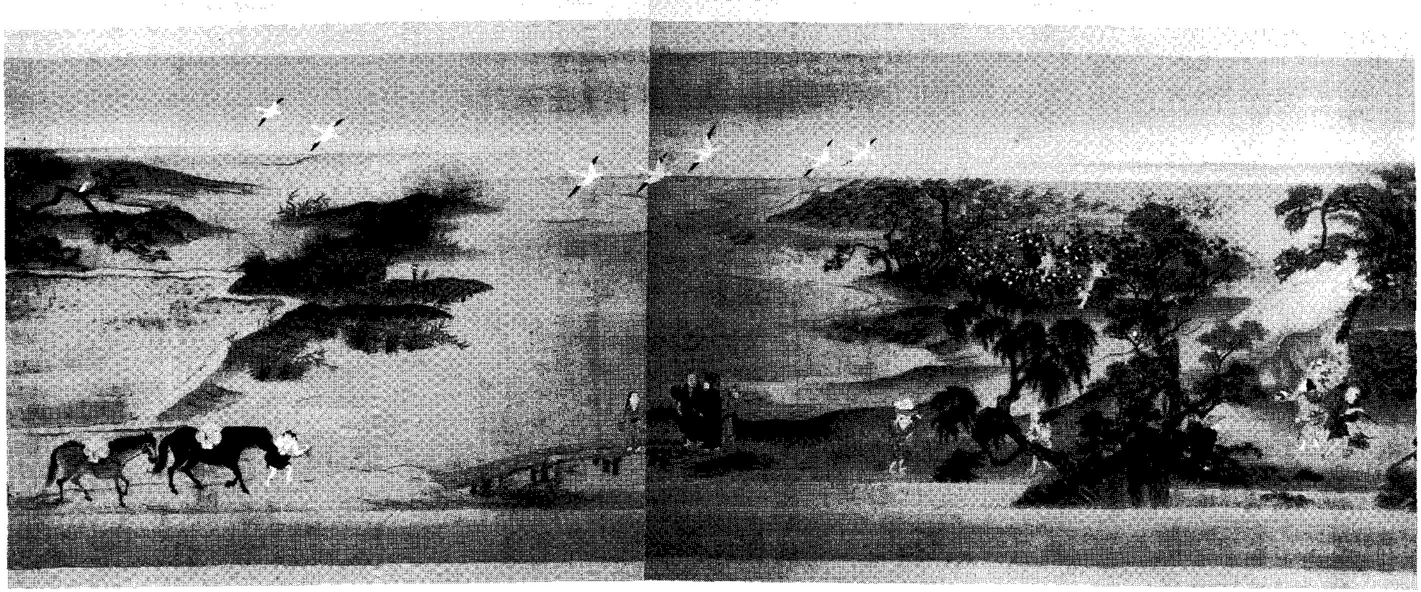

Figure 2. Portion of a mural entitled "Rakuchu rakugaizu-kan" or "Inside and Outside Kyoto" by artist Sumiyoshi Gukei (1631–1705). Flocks of snow geese are depicted in the area of the historic capitol of Edo (Kyoto). Gukei was a buddhist monk whose patron was Tokugawa Sōgun. He drew this large mural (1,368 × 41 cm) in the late 1600s. The original mural is in the Tokyo National Museum.

decline of this remnant colony has increased international concern about its conservation.

Wintering Status of Wrangel Island Snow Geese

The Wrangel Island population concentrates in two North American regions during winter (Subcommittee on White Geese 1992). The northern subpopulation, consisting of about 60 percent of the population, spends the winter on the Fraser River Delta of British Columbia and the Skagit River Delta of Washington. Snow geese from Wrangel Island comprise the major wintering population of geese in this region; thus, management of wintering habitat and hunting regulations can specifically address their requirements.

Most of the southern subpopulation spends the winter in the Central Valley of California, although a few geese migrate to Salton Sea, or to the West Coast or Interior Highlands of Mexico (Figure 1) (Bellrose 1980). In California, management problems are compounded because the Wrangel Island population is mixed with nearly 500,000 snow and Ross' geese from the western and central Canadian arctic. The southern subpopulation from Wrangel Island and other snow geese seem to intermix freely during winter (J. Takekawa unpublished data), resulting in limited options for independent management of this subpopulation.

Rationale for Restoring Lesser Snow Geese to East Asia

Overharvesting seems to be the main reason for the disappearance of breeding snow geese from mainland Siberia and wintering populations from Asia. Settlers and explorers used snow goose colonies for food in northern Russia in the late 1800s (Bousfield and Syroechkovskiy 1985), and overhunting and taking of eggs in the early 1900s destroyed three colonies on Wrangel Island in only a few years (Andreev 1994). Since 1976, Wrangel Island has been protected by the Russian nature reserve system, and the snow goose has been listed in the red data book of endangered animals and is protected from hunting (Bousfield and Syroechknovskiy 1985).

Russian researchers have advocated restoration of lesser snow geese to the Arctic coast of Siberia for more than 20 years (E. Syroechkovsky personal communication). They recommended transplanting snow geese to historic breeding areas to buffer against catastrophic loss of the Wrangel Island populations. Little was known about asian wintering areas; however, alarming declines in populatons of geese wintering in East Asia (Andreev 1994) seemed to indicate high mortality (A. Andreev personal communication).

Our research revealed that large numbers of snow geese formerly spent the winter in Japan. They disappeared in the late 1800s concurrent with widespread distribution of firearms and development on Tokyo Bay during the Meiji Restoration. Geese now are protected by the Japanese Environment Agency and National Cultural Ministry. Goose hunting has been prohibited since 1971 and geese now are recognized as national cultural treasures. Most geese now wintering in Japan concentrate in the area near Lake Izunuma, Miyagi Prefecture, 100 kilometers north of Sendai (Figure 1). In 1989, Lake Izunuma was given protected status under the Ramsar Convention as a wetland of international importance for waterfowl (Finlayson and Moser 1991).

Reestablishing additional breeding areas would ensure survival of snow geese in

Asia given the instability of numbers on Wrangel Island and difficulties managing the southern subpopulation in North America. Recent political changes in Russia have opened paths to cooperative projects with Japan, leading to the first Russian—Japanese Agreement on the Environment in 1993. The Russian Academy of Science and Japanese Environment Agency have agreed to support snow goose restoration as a joint venture, and it should facilitate collaborative research (Morton 1987) among North Pacific Rim countries, including the U.S. and Canada. Development of techniques to restore snow geese to East Asia would benefit other projects directed at restoring populations of arctic nesting geese.

Restoring Snow Geese to East Asia—The First Year

Developing the Restoration Plan

The first meeting for the Restoration of Snow Geese to East Asia was held in Sendai, Japan, during January 1993. Participants from the Japanese Association of Wild Goose Protection (JAWGP), U.S. National Biological Survey (NBS), Russia Academy of Sciences (RAS), Russian Nature Reserves and Eastern Palearctic Wetlands (EPW) discussed potential methods for restoration of snow geese and established a preliminary plan. Three goals were agreed upon for the first year: to review the literature for examples of successful restoration of geese, to examine logistic problems and conduct field trials on preliminary methods, and to locate breeding areas of geese currently wintering in Japan.

Selecting a Restoration Site

The reason for the disappearance of snow goose colonies on mainland Siberia is not known. Siberia has about 500,000 square kilometers of wetlands (Andreev 1994); most goose species nest on tundra plains near the Arctic coast, along shallow rivers or on islands (Andreev 1994). We found no detailed descriptions of suitable characteristics of snow goose colony sites. Most colonies in North America are on river deltas or islands (Kerbes et al. 1983) and range in size from a few hundred to hundreds of thousands of geese (Kerbes 1975). Although female geese exhibit strong fidelity to natal colonies, new areas commonly are pioneered (Geramita and Cooke 1982). Gosling feeding areas may be critical in determining preferred habitats (Cooke and Abraham 1980, Aubin et al. 1993) for colony sites.

Wrangel Island snow geese migrate across seemingly adequate sites on Chukotka Peninsula, yet no mainland colonies have been established there in the past 70 years. The Chukotka Peninsula may not have suitable habitat in an area where snow cover allows adequate time for reproduction (Krechmar and Syroechkovsky 1985, Kerbes 1986). Under favorable conditions, snow goose colonies may grow exponentially (MacInnes and Kerbes 1987), but Wrangel Island is located in the coldest area of Siberia. Breeding is highly synchronous on Wrangel Island, yet the colony often reproduces poorly because snow cover may extend late into summer. Snow geese are unable to nest 42 percent of the time (Litvin and Syroechkovsky 1984) and do not initiate egg-laying after June 10 (Krechmar and Syroechkovsky 1985). Thus, choosing a successful restoration site from historical areas requires selecting a suitable early summer microclimate with suitable gosling feeding areas.

Almost all snow geese from Russia currently migrate to North America, which complicates restoring the East Asian migration of geese that wintered in Japan. Small numbers of snow geese still nest on the Arctic coast at Chaun Bay, Ayun Island, Nolde Lagoon, Kolyuchin Bay and Koolen Lake east of Kolyma River (Lebedev and Filin 1959, Uspenski et al. 1962, Krechmar et al. 1978, Andreev and Dorogoi 1987, Dorogoi 1990b). Geese nesting on the mainland probably migrate to North America because few records of snow geese wintering in Asia have been reported. Hence, establishing a population on Chukotka Peninsula probably would not restore migration to Asia; however, geese nesting west of Kolyma River in historic snow goose areas may winter in southeast Asia.

Techniques for Reestablishing Breeding Populations of Geese

Restoration of giant Canada geese (*Branta candensis maxima*) in the midwestern United States has been successful following three alternatives (Dilland Lee 1970, Bishop and Howing 1973, Lee et al. 1984): releasing hand-reared birds into the wild, holding flightless breeding geese in a new area until their offspring settle in the area or moving second-generation birds to new areas after breeding. However, it is not feasible to overwinter geese at higher latitudes. Restoration techniques tested on western Canada geese (*B. c. moffiti*) at higher latitudes include translocating four- to eight-week-old goslings, moving goslings and adults, releasing captive-reared goslings or yearlings, and releasing captive-reared young with wild foster parents (Wishart 1976). Several researchers (Wishart 1976, Hammer 1982, Lee et al. 1984) found that goslings, especially females, exhibit philopatry to areas where they first learn to fly. Wishart and Hill (1982) reported that goslings returned to areas of first flight while their parents returned to their natal sites.

The nene goose (*B. sandvicensis*) project represents one of the oldest restoration programs (40 years) with releases of captive-reared birds (Kear and Berger 1980). More than 3,000 captive-reared nene have been released, restoring the population from 30 geese in 1951 to a few hundred geese today (Banko 1980, Kear and Berger 1980, Cherfas 1989). Survival of released geese is less than 20 percent (Hoshide et al. 1990), and, although predation rates are high, food resources may limit wild populations. Captive-reared geese also have been used in restoration projects on greylag geese (*A. anser*) in Europe (Ogilvie 1978) and bar-headed geese (*A. indicus*) in India (Qadri 1987).

The most successful restoration of an arctic nesting goose was of Aleutian Canada geese (*B. c. leucopareia*). The population rebounded from fewer than 800 individuals (Springer et al. 1978) to more than 10,000 in 1993 (A. Dahl unpublished data). Initial releases of captive-reared goslings were not successful because these birds lacked a migratory tradition. Results improved when goslings were translocated to alternative breeding islands beginning in 1971 (Martin et al. 1982, Byrd et al. 1991). An experiment with "golden pairs" of captive-reared females and wild males with their young also was successful (Byrd 1994). Releases were conducted simultaneously with increased predator control and hunting regulation, so it is difficult to quantify success of the translocations alone.

Where species no longer are present, researchers have placed goslings with adults of a different species (von Essen 1982, 1991, Fabricus 1991). But cross-fostering may cause goslings to imprint on the parent species, because goslings imprint on almost any moving object (Hinde 1970, Lorenz 1991). Fabricus (1991) found that after replacing all eggs in Canada goose nests with bean goose (BEGO) eggs, the females paired with BEGO, although 26 percent of the males paired with Canada geese. Complete brood replacements seemingly maintained gosling recognition patterns.

Establishing a Migration Tradition

Translocated young geese are incapable of establishing migrations on their own unless they are placed with experienced adults (Matthews 1982). In Sweden, the lesser white-fronted goose (*A. erythropus*) project has been one of the few attempts to reestablish a breeding area and develop migration tradition to a selected wintering area (von Essen 1982, 1991). Eggs of local barnacle geese (*B. leucopsis*) in southern Sweden were replaced with eggs of lesser white-fronted geese (LWFG). These cross-fostered families were taken to Lapland where the fledglings learned to fly. Both barnacle (BRNG) and LWFG migrated to the Netherlands during winter. The following spring, the LWFG followed the BRNG to southern Sweden, but the young LWFG were expected to return to Lapland after nesting was initiated. The LWFG established a new migration, but the first breeding pair was reported in 1987 (von Essen 1991), eight years after the initial release (Table 1). A population of 20–30 breeding LWFG now has been established in Lapland (von Essen unpublished data).

In a novel experiment conducted last autumn, 18 Canada geese were imprinted on an ultralight aircraft (Lishman 1989) and were led to a wintering area at Airlie, Virginia (W. Sladen personal communication). The experiment served to determine whether young geese could learn a migration route led by a surrogate parent (Lorenz 1978). Although these "ultrageese" were successfully led to a new wintering area, more migrations are needed to determine if these imprinted geese follow this tradition on their own. Unfortunately, use of this technique in large remote areas such as Siberia probably is not practical because of logistical constraints.

Restoration of the East Asian migration probably requires cross-fostering snow geese with greater white-fronted geese (*A. albifrons frontalis*) to develop a migration tradition. About 30,000 greater white-fronted geese (GWFG) and 5,000 BEGO now spend the winter in Japan (M. Kurechi unpublished data, Wild Bird Society of Japan

Table 1. Success of reintroduction of lesser white-fronted geese (LWFG) to Lapland (von Essen unpublished data). LWFG goslings were cross-fostered with barnacle goose (BRNG) parents. LWFG were resighted in the nesting area of BRNG (south Sweden), and located in wintering areas in western Europe. Marked LWFG also were observed in north Sweden translocation areas in succeeding years.

			Resighted			
	Released				North Sweden	
Year	BRNG	LWFG	South Sweden	Western Europe	Identified	Unidentified
1981	2	11	9	2	0	0
1982	3	28	10	6	1	0
1983	4	37	26	8	0	0
1984	5	33	22	20	5	6
1985	3	22	15	15	6	14
1986	3	13	9	9	3	2
1987	3	16	0	0	5	6
1988	2	22				11
Total		172	112	60	20	

1988), but their Siberian breeding area is unknown. GWFG probably are reasonable foster parents for snow geese because they are similar in size and often winter together in North America. Timing of breeding also is suitable for restoration because mainland GWFG generally nest later than snow geese on Wrangel Island, facilitating translocation of eggs or goslings.

Results of the First Year

Transplanting Snow Goose Eggs to the Anadyr River

In summer 1993, three teams from the JAWGP traveled to Russia to conduct field trials. Experimental translocations following the methods of von Essen (1982, 1991) were conducted at the Anadyr River research site of the EPW (Figure 1). This site was chosen for field trials because it was an established camp with a population of GWFG. Previous collar-marking studies determined that 20 percent of the GWFG migrate to Japan (Kurechi 1994).

One hundred eggs were obtained from the snow goose colony on Wrangel Island during the first week of June. A single egg was taken from each nest. We were advised by aviculture experts (F. Lee and G. Gee personal communication) in developing a transport system with slings and padded cases to protect the eggs against helicopter vibration. Timing was a crucial consideration for obtaining eggs because egg laying is highly synchronous at Wrangel Island (Syroechkovsky 1976, 1979) and unincubated eggs were expected to lose viability at a rate as high as 10 percent per day (G. Gee personal communication). Eggs were collected in the first week of June and transported as quickly as possible.

We replaced eggs of GWFG in seven nests with 41 snow goose eggs (Table 2). An additional 43 snow goose goslings were hatched in incubators, marked with leg bands and released in a lake with molting flocks of GWFG. Although bad weather delayed transport for ten days, 86 percent of the eggs hatched. First-laid eggs had the best hatching rate (Table 2), which was expected (Litvin and Syroechkovsky 1984). Success cross-fostered families were observed in August prior to autumn migration.

An aerial survey also was conducted on the Lower Kolyma River on July 17 to examine historically used areas that may be suitable for future translocations. The survey recorded GWFG and BEGO including 3,720 adults and 60 goslings in 29 flocks and unanticipated sightings of 32 adult and 20 gosling snow geese. A camp (70 degrees 23'N 159 degrees 55'E) was established near the snow goose sightings, and 25 GWFG and 104 BEGO were captured and marked with neck collars. Eighteen snow geese were observed in molting flocks.

Three juvenile and two adult snow geese were observed in Japan during winter of 1993, an increase over previous years. It is not known whether these geese were part of the release program because they were not with GWFG parents. GWFG or BEGO marked with collars on the Kolyma River (4) and Anadyr River (1) were resighted in Korea during January 1994. An unusual sighting of eleven snow geese also was reported in Korea along the demilitarized zone.

Satellite-marking Greater White-fronted Geese

Six greater white-fronted geese were marked with satellite transmitters (*see* Higuchi et al. 1991, Higuchi et al. 1992, Ely et al. 1993) at the Anadyr River in August 1993.

Table 2. Hatching success of lesser snow goose eggs collected on Wrangel Island, Russia, in 1993. Clutch size when the eggs were collected and eggs collected from outside of nests were indicated.

Clutch size	Collected	Hatched	Percentage success
1	33	33	100
2	37	30	81
3	24	19	79
4	1	0	0
Out-of-nest	5	4	80
Total	100	86	86

Four larger transmitters (55g, NTT, T2038) were attached on neck collars and two smaller transmitter (40g, T2050) were attached with tail mounts. Only one of the smaller transmitters worked through autumn migration, and the bird with that transmitter migrated to the Naoli River valley in Heilongjiang Province, China (46.7 degrees N, 132.5 degrees E) when the radio stopped working.

In January 1994, National Biological Survey (NBS) biologists joined JAWGP members in the first study to cooperatively mark geese with radio telemetry in Japan. Thirty-six GWFG were captured with rocket nets near Lake Izunuma (Figure 1) on February 4. Ten geese were marked with satellite transmitters (40g, T2050), five backpack and five tail-mount attachments. All transmitters were working after the first three weeks. This spring, we will track these GWFG back to their Siberian breeding areas to identify potential restoration sites for snow geese.

Discussion and Conclusions

Temple (1983) suggested that four points be considered in restoration of birds: availability of suitable areas for release, a viable population for reintroduction, a positive survival rate of released birds and project leader who can sustain the program for several years. We feel that all of these points were met for returning snow geese to East Asia. There appear to be extensive areas suitable for a colony on the Siberian mainland. Restoring snow geese to western Russia where populations may migrate to Europe could be detrimental to crops. Large numbers of wintering geese in western Europe already cause significant crop damage (Greenwood 1993). However, historical evidence seems to indicate that geese breeding east of the Lena River winter in Asia.

Eggs of snow geese are easy to hatch, and eggs and goslings are readily available from Wrangel Island. Survival rates of young birds are not known, but predation rates are not as high as those in restoration projects of species such as the Aleutian Canada goose where breeding islands are heavily populated with introduced foxes (Byrd et al. 1991), or nene geese that must avoid mongooses and several other predators (Kear and Berger 1980). Additional protection against predators may be obtained by introducing geese near snowy owl (*Nyctea scandiaca*) nests, which may increase gosling survival (Litvin et al. 1985, Dorogoi 1990a).

Although every participating group has made significant contributions to the project, longevity of the restoration depends on the JAWGP. The JAWGP is a volunteer organization, but members have shown great dedication to the work. Funding of this project in Japan has been supported because snow geese are a popular symbol in

Japan. Furthermore, the publicity generated by the project may benefit conservation education in Japan.

Cooperative work on this restoration project may initiate studies to compare Arctic nesting goose populations on both continents, contrasting their ecological differences to improve our understanding of their migratory behavior. For example, hunting restrictions in Japan provide an opportunity to study behavior of a nonhunted wintering population of geese for comparison with North America populations. Japan is the easternmost wintering area in Asia. It is possible that breeding areas of geese from this population may separate those migrating to Asia and North America. Thus, we can learn much about migration of geese and arctic nesting populations by supporting conservation projects on both sides of the Pacific Rim.

Recommendations

1. The next restoration meeting should be held with the upcoming North American Arctic Nesting Goose Conference. A final restoration plan should be completed within two years, and include a detailed work schedule.
2. Restoration will continue with egg and gosling transplants. Surveys will be conducted at breeding areas located by the satellite-marked GWFG to locate suitable restoration sites. One experiment will include transporting goslings to the Japanese wintering area and releasing them during winter.
3. The Japanese Environment Agency and the Canadian Wildlife Service should be invited to join the restoration committee, and the restoration project should be submitted for inclusion to all participants Environments Agreements.
4. The rapid decline in East Asian populations of arctic nesting geese should be investigated in conjunction with this project, including studies of overwinter survival of geese in Korea and China.
5. Travel continues to be a serious barrier to effective research in Siberia. An effort should be made to make small aircraft available for transportation and surveys.

Acknowledgments

We thank G. Wylie and C. Ely (NBS), and W. S. Boyd of the Canadian Wildlife Service (CWS) for reviewing drafts of the manuscript. K. Lahlum (NBS) conducted literature searches and S. Warnock (NBS) prepared figures. L. von Essen of the Swedish Hunter's Association, R. Kerbes and W. S. Boyd (CWS), and S. Iwabuchi, S. Uemura and T. Ikeuchi of the Japanese Association for Wild Goose Protection contributed valuable information. F. B. Lee, G. Gee (NBS), and W. Eldridge (FWS) assisted in restoration planning. S. Kohl (FWS) provided assistance with international arrangements. T. Suganuma, M. Kuriki, M. Arisawa and Y. Nakai of Nippon Telegraph and Telephone provided satellite transmitters and financial support for the project.

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Special Session 2. *Wildlife Population Estimation*

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Introductory Comments

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The initial task facing my cochairman and me as we began to put together this session was to focus the broad subject of wildlife population estimation on a topic that is contemporary, relevant and, in our opinion, in need of attention. Although biometricians continuously expand the statistical capabilities and robustness of traditional estimation techniques such as capture/recapture and transect sampling, and develop new ideas for approaches or refinements based on advances in technology, we chose not to emphasize this theme in our session. More specialized conferences, such as the recent Wildlife : 2001 Populations Symposium, and technical workshops both provide adequate opportunity to stay abreast of these kinds of advances in population estimation.

Rather, the theme of this morning's session will be the challenges of estimation of population size on a large scale, and the associated assessment of population trends, either temporally or spatially. It seems we are now in a time of increased emphasis on assessment of status and trends of wildlife populations and communities. Not that the notion of large-scale assessment of wildlife status and trends is a new concept; we need only to remember the U.S. Bureau of Biological Survey of a century ago. However, as evidence of renewed emphasis, we now have the National Biological Survey, and a fundamental objective of this new agency is to generate information on current status and trends of animal, as well as plant, species in an effort to establish an earlier warning system that can be trusted to detect signs of populations or other natural resources that are about to be in trouble, rather than relying on current systems that either are unreliable or only have the sensitivity to detect near catastrophic decreases in population levels.

On more regional or state levels, natural resource agencies have increasing needs for more accurate and reliable information about the population status of species entrusted to their management. State agencies must satisfy themselves, their constituents and public interest groups that their management strategies are based on sound information. For example, population level of game species are assessed to ensure that harvest levels are appropriate or to evaluate effects of other management practices. State and federal agencies must monitor populations subject to subsistence harvesting or international treaty mandates. In many instances, the database on which our management decisions are based must be defendable in court. Thus, many traditional estimation and survey analysis methods are being or should be reviewed, reevaluated and redesigned to meet the challenge generated by new priorities and evaluation criteria.

Consider for a moment the tremendous influence that population estimates and trends have in setting priorities and agendas in wildlife agencies. For example, the North American Waterfowl Management Plan and the Partners in Flight program can trace their beginnings to results of surveys that indicated unacceptable declines in waterfowl and neotropical migrant populations. These programs have had a significant influence on not only agency management programs, but also have influenced research agendas and funding priorities. Of course, population estimates are a fundamental component of the decision-making process in our threatened and endangered species programs. State agency priorities also often are driven by results of monitoring efforts conducted on species of concern or high economic importance. Mitigation for impacted resources can be driven by biomonitoring programs that involve assessment of key populations. Yet, paradoxically, it often happens that the estimation techniques and survey designs upon which we rely so heavily produce results with confounded interpretations (most often due to use of indices that are themselves uninterpretable), nonquantifiable precision and lack of sensitivity to detect biologically significant changes. This afternoon, our session participants will discuss and illustrate prescriptions for planning and evaluating surveys, so that the probability of expensive disappointments is minimized.

Finally, I want to comment on the limitations of our capabilities, and the need to recognize them. In spite of continuous advances in statistical techniques and an increasing appreciation of the need for rigor in our science, the very nature of our subject matter, i.e., wild free-ranging populations of animals, constantly works against us and places relatively severe limitations on our ability to transfer theory into practice. We are constrained by the cruel fact that "if there aren't many of them out there and/or they're hard to find," our methods break down. The degree of difficulty increases as distribution patterns become more fragmented and movement patterns become more dynamic. Add to this mix a component of long-term temporal variation and a desire to detect population trends, and the task becomes more challenging. In the limit, the desired objectives of the study may not be achievable, given any reasonable amount of available resources. Equivalently, the accuracy and precision of our estimates and the sensitivity of comparisons sink to noninformative levels if we "do the best we can with what we have." For those involved in these programs, it is a responsibility to realistically evaluate, perhaps with the aid of the information provided by our authors, the limitations of our capabilities and to thereby delineate the boundary between wishful thinking and wise use of limited resources.

The Principles and Practice of Large-scale Wildlife Surveys

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Introduction

The role of large-scale surveys in the management of wildlife resources is receiving increasing recognition from government and international agencies. In this paper, I address issues relevant to my own experience in designing, analyzing, reviewing or advising on large-scale surveys. I start by listing what I consider to be the key components to be addressed when setting up a large-scale survey; too often, many of the key issues are not addressed until some time after setting it up, perhaps when it becomes apparent to the analyst assigned the task of modeling the survey data that the objectives (if well defined) cannot be met given the design of the survey. More detailed discussion of this problem is given by Conroy and Smith (in preparation). I then briefly discuss the main options for carrying out large-scale surveys: questionnaire surveys; atlas surveys; complete counts at sample sites; transect surveys; mark/recapture; and indirect survey methods. Short discussions follow on sampling strategies, the role of spatial modeling and models for change over time.

Key Components of a Large-scale Survey

Numerous factors must be taken into account when designing and implementing large-scale wildlife surveys. The following lists several aspects of setting up a large-scale survey that are often neglected to the detriment of the project:

- Define management objectives clearly at the outset.
- Establish an effective steering group, responsible for collaboration between participating bodies and for meeting management objectives.
- From the start, formally involve analysts/statisticians with a proven track record.
- Identify survey methods appropriate for the species and associated habitats.
- If there are doubts about the appropriate choice of methods, or about whether adequate sample sizes will be achieved, carry out an initial experimental or pilot survey.
- Design the survey to make efficient use of resources.
- Train and equip participants to ensure adequate data quality.
- Carry out analyses promptly and to a high standard.
- If the surveys are ongoing, publish results regularly.
- Review methods continually but, whenever possible, revise them in ways that do not compromise the need to monitor change over time.

The methodology for analyzing the survey data should be identified before data collection starts, to ensure that relevant data are collected to a sufficiently high standard. When large-scale surveys are first implemented in new circumstances, data quality is often poor. For example, when the International Whaling Commission started its series of sightings surveys in the Antarctic, it was thought that mark/recapture would be the main method of stock assessment, but line transect surveys were carried out in addition. It was soon realized that abundance would be more reliably estimated from the line transect data. However, it was not understood how sensitive the line transect method is to errors in sighting angles when the distance of a whale pod from the transect line is calculated as $r.sin\theta$, where r is the sighting distance of the pod from the vessel and θ is the sighting angle. Lack of proper training and equipment for observers rendered the task of the analysts far more difficult than was necessary. Consequently, abundance estimation was compromised in that bias was high and precision was low relative to later surveys, when angleboards were used to improve angle estimates.

Questionnaire Surveys

For terrestrial surveys, when resources are limited, or when an inexpensive pilot survey is planned in advance of a main survey or monitoring program, a targeted questionnaire survey may prove useful. Indeed, for scarce but readily recognized species, it may sometimes be the only economically feasible method. Additionally, it allows retrospective assessment of change in unmonitored populations, although responses are likely to be subject to bias to an unquantifiable degree. This option was used in Scotland in an attempt to quantify past and current distribution of adders (*Vipera berus*), the only poisonous snake found in the United Kingdom. Following an amendment to the Wildlife and Countryside Act, the adder was given special protection against intentional killing and injuring, but little was known about the status and the need for protection in Scotland. The questionnaire was mailed to the nearest farm to the center of each 5-kilometer in Scotland, except in cases where no farm was within the 5-kilometer square. In addition, it was sent to landowners, gamekeepers, foresters, reserve wardens, hill walkers and other users of the countryside. The response rate among farmers was 67 percent, and the pooled response rate for other categories was 60 percent. These high return rates reflect strong public interest in the survey, enhanced by effective publicity, together with good questionnaire design and effective follow-up of nonrespondents. The questionnaire asked respondents whether they had seen adders in their area in three different time periods—the last 5 years, the previous 5 years or more than 10 years ago. Comparison of the two five-year periods allows an assessment of whether adder distribution is changing; as a check of consistency, the respondents who noted the presence of adders were also asked whether they were becoming more or less common (if either) in their area. Logistic regression is currently being used to model the presence/absence data for each time period, to allow estimation of the full distribution and assessment of change over time. In addition, a monitoring program is being established in selected sites, to assess future change. Assessment of bias in respondents' answers was made by comparing responses of those keen to see fewer adders (e.g., gamekeepers) with those who might be expected to favor protection (e.g., reserve wardens).

Atlas Surveys

Atlas surveys generally attempt to map the distribution of species over a wide area. The region of interest is usually divided into grid squares, but sometimes into irregular sites within which habitat is relatively homogeneous. Records are collated by square (or site) and, in most surveys, an effort is made to secure records from every square in the region. No attempt is made to record everything in a square. Often, amateurs gather the bulk of the records, and atlases are notorious for the heterogeneity in effort and observer ability. The former is often reduced by restricting the time spent recording in a given square, or by requesting that observers note recording time. Variation in observer ability is seldom addressed.

One useful strategy for dealing with heterogeneity is to select a (possibly restricted) random sample of squares for more detailed cover by professional, trained observers. Absolute or relative density can then be assessed using data from these random squares, while amateur records provide distributional information from all squares.

Högmander and Møller (in preparation) provide methods based on image analysis for inferring distribution from atlas data with heterogeneous effort. Those methods require that habitat is relatively similar in neighboring squares.

Probably the most useful information gained from atlas surveys is distributional change between one atlas and the next. Surveys tend to be separated by a number of years, during which fashions and methodology evolve. It is therefore tempting to adopt more sophisticated methods than in the previous atlas survey. Unless done with great care, this will compromise comparability of the surveys, and hence assessment of change. In short, it is inevitable that successive bird atlases in the same region will be used to assess distributional change, whether or not field methods were comparable, so ensure that they are!

Complete Counts at Sample Sites

For highly gregarious species that concentrate at a few known sites, complete counts of the population may be possible. This approach is adopted for goose counts in the United Kingdom, although it is inevitable that some sites are missed. Another example that comes close to the strategy of complete counting is the assessment of red deer (*Cervus elaphus*) numbers in the Scottish Highlands. A coordinated team of Red Deer Commission stalkers counts deer numbers by management blocks. The method is effective in areas of open moorland, but undercounting may occur if the deer have access to extensive areas of forestry plantation. All management blocks are counted, at intervals varying between 1 and more than 10 years. However, it cannot really be considered a complete count, because only a sample of blocks is counted in any one year. Thus, modeling is required to estimate the number of deer in the Scottish Highlands in any given year.

If complete counts are attempted in large-scale surveys, generally only a subset of sites within the study region is selected. This subset should be representative of the region, which usually requires some form of randomization (see below). Complete counts might be made within selected sites by counting visible animals from a convenient vantage point or from a predetermined point, by traversing the plot systematically or otherwise, or by having a line of beaters drive the animals past counters. Special cases of complete counts at sample sites include quadrant counts, strip transects and circular plots. Mapping censuses, frequently favored by birders, are another example. The Common Birds Census (CBC) of the British Trust for Ornithology (BTO) is a large-scale "mapping census" survey of the United Kingdom. Amateur observers map songbird locations, which are subsequently interpreted to determine number of territories. The CBC suffers from being based on woodland and farmland sites selected subjectively by the observers. Thus, there is a bias in favor of more interesting sites, which may exhibit atypical trends in abundance for many species, and there is a tendency for sites that deteriorate to drop out of the scheme, which may bias estimated trends in abundance. The method also is expensive in terms of staff time. For these reasons, the BTO hopes to phase out the CBC in favor of the Pilot Census Project (below).

Transect Surveys

Transect surveys in various guises (Buckland et al. 1993a) provide the most widely used approach for assessing abundance of terrestrial vertebrates and marine mammals from large-scale surveys. The simplest form is the strip transect, which is also a special case of complete counts at sample sites. In terrestrial surveys, there are often considerable practical advantages to adopting a design in which transects are positioned along roads and tracks. Indeed, many birders in particular argue that large-scale surveys cannot be implemented unless such a strategy is adopted. In common with Conroy and Smith (in preparation), we stress that reliable estimates of abundance *cannot* be obtained from such surveys. Furthermore, even if abundance estimates are considered to be merely relative abundance for monitoring change over time, bias should be anticipated. For example, increased traffic over time may lead to an increasing trend in disturbance. Further, the road or track gives easy access to habitat alongside it. This may result in deterioration of that habitat. Conversely, the presence of the road or track, or of human activity alongside it, may increase the diversity of habitat. Any of these effects can cause abundance, and trends in abundance, along roads or tracks to be atypical. Line transect surveys of capercaillie (*Tetrao urogallus*), a large turkey-like grouse, carried out along tracks in two Scottish forests, Monaughty and Culbin, indicated that densities in both forests were similar, even though drive counts showed that densities in Monaughty were around three times those in Culbin. Subsequent off-track transects showed good agreement with drive counts. Investigation showed that capercaillie used tracks for display and as a source of grit in both forests. However, Culbin was located on sand with no natural water sources, whereas water was plentiful in Monaughty. Capercaillie congregated far more on or near tracks in Culbin because small water reservoirs, for fighting fire, were located alongside tracks for ease of access.

In the Pilot Census Project of the BTO, a stratified random sample of 1-kilometer squares throughout the United Kingdom has been identified for monitoring. In each square, an idealized transect route is defined. Observers keep as close as possible to the idealized route, and record on the form the extent of departure forced on them by topography, habitat or access difficulties. Most observers are volunteers, so methods are kept as simple as possible. All species are recorded and assigned to one of three distance categories, or to a separate category for birds flying over. Habitat along the route is also recorded on the first visit. The scheme started in 1992, and squares are to be covered each breeding season. The primary purpose of the project is to monitor changes in abundance over time of approximately 100 species of birds.

Mark/recapture

Skalski and Robson (1992) note that mark/recapture should have a role to play in large-scale surveys, but claim that, as yet, it has not been used. In fact, several examples exist. The Constant Effort Sites (CES) scheme of the BTO involves use of a constant length of mist net at regular time points through the breeding season at a number of sites throughout the United Kingdom. Passerines are banded, and the data are used to monitor annual changes in adult numbers and productivity (juvenile to adult ratio). In addition, adult survival rates are estimated by mark/recapture methods. Large-scale mark/recapture exercises have also seen use in the marine environment. Large tagging studies of tuna in the eastern tropical Pacific and more recently in the South Pacific have provided rapid assessments of stock size at times of sudden expansion of fishing activity, thus allowing assessment of whether the fishery is sustainable. In this example, ongoing monitoring is best achieved through monitoring the catch rather than continuing the tagging program indefinitely. Historically, assessment of pelagic whale stocks was attempted through firing Discovery marks into animals. This method had very limited success (Buckland and Duff 1989), but recently, there has been considerable enthusiasm for the use of natural markings data. Extensive libraries of fluke photographs have been compiled for humpback whales in the North Atlantic and North Pacific, and work is ongoing to assess movements, abundance and survival rates from these. The method is being used or investigated for a number of other whale populations.

Indirect Methods

Indirect methods are sometimes used in large-scale surveys. For example, it is nearly impossible to assess deer numbers in the dense forestry plantations in Scotland. Instead, it is more cost effective to carry out dung counts. If plots are cleared initially, the counts can be converted to estimates of deer numbers by estimating the average number of pellet groups produced per deer per day. In areas of low density, it may not be practical to use the clearance method. In that case, counts are made on plots that were not previously cleared. It is then necessary to estimate the decay rate of pellet groups. This method is less robust, so more careful experimental design is required when determining where to locate the sample plots.

In the Pinewood Bird Survey, carried out principally by the Royal Society for the Protection of Birds, line transect methods are being used to assess the numbers of capercaillie, crested tits (*Parus cristatus*) and crossbills (*Loxia curvirostra* and *Loxia scotica*) in Scotland. However, crossbills are particularly difficult to survey in this way, so that at selected points, pine cones dropped by crossbills are being counted, to provide an alternative method for assessing relative abundance.

Sampling Strategies

There are generally many options for the design of a sample survey, and the choice can have a dramatic impact on precision of estimates for a given cost. Before a sampling scheme can be determined, a decision must be made on the sampling unit. This might be grid squares, homogeneous blocks of habitat or, if point counts are to be made, intersections of grid lines. In the Pinewood Bird Survey in the United Kingdom there was considerable discussion of whether the sampling unit should be squares of the national grid or pinewoods. The latter is the more natural unit, but is variable in size and shape, and sometimes poorly defined. It is simpler to form a sampling scheme, and to estimate abundance for Scotland, if the sampling unit is the grid square, but the nominal transect lines may pass through unsuitable habitat in squares that are not entirely pinewood. Some pinewood squares may contain very little pine, so that little if any lies on the transect. If sampled pinewoods or grid squares are given equal weight in subsequent analysis (rather than weights in proportion to pinewood area), then effort can be concentrated in large pinewood blocks by sampling pinewoods with probability proportional to size, or grid squares with probability proportional to the area of pinewood within them.

If the sampling unit is grid squares or intersections of grid lines, there is necessarily a systematic element to sampling. (True random sampling would allow some sampled squares to overlap partially.) There are merits to choosing a fully systematic sample, in which the first square is chosen at random, but subsequent squares are selected as every k^{th} square in each direction, where k is chosen to give the desired number of sampling units. This has the advantage of giving a better spread and a more representative sample than a random sample, leading to better estimator precision. The classical disadvantage that a systematic sample might pick up systematic variation in the region is unlikely to occur in large-scale wildlife surveys if the number of units sampled is high. However, another disadvantage is that, to estimate the (improved) precision, the sample is generally assumed to be random! This disadvantage can be removed by resorting to spatial modeling (below), but then another difficulty arises. If there is spatial correlation in the residuals of a fitted spatial model, then to estimate the extent of that correlation effectively, the sample units should have variable spacings, exactly the property they do not have if the sample is systematic. A modification to simple random sampling that ensures a reasonable geographic spread in the sample is to constrain the randomization in some way. For example, if a 2-percent sample of 1-kilometer squares is required, an option is to select two squares at random in each 10-kilometer square. This is a stratified random sampling scheme, but with an even sampling rate across strata, and the analyst might be better to assume that the sample was fully random, as replication in each stratum (10-km square) is minimal.

Stratified sampling can yield large cost savings over random sampling. Thus, the strata might be chosen according to the costs of surveying different units, with sampling rates assigned to strata that maximize precision of say an overall abundance estimate subject to fixed costs. In the Pilot Census Project of the BTO, stratification was carried out by land class and by density of observers.

When the cost of getting to a sampling unit is high, but the marginal cost of sampling neighboring units is low, cluster sampling is useful. The clusters themselves might be selected according to a stratified scheme.

Thompson (1992) advocates use of adaptive sampling. Its main disadvantage is that substantial knowledge is required in advance to ensure that the number of units to be sampled is of the desired order. Thompson (1992) and Conroy and Smith (1994) discuss sampling issues in greater detail.

Spatial Modeling

Data from large-scale surveys invite the analyst to attempt spatial modeling. Potentially, this allows animal density to be estimated as a surface. Several advantages may accrue: abundance may be estimated by any subregion of interest, simply by integrating under the surface across the subregion; the spatial distribution of the surveyed species may be related to the distribution of habitat, either by means of maps or through fitting covariates to the observed data; changes over time in the spatial distribution of wildlife can be shown through a sequence of maps of the surface; precision of abundance estimates may be improved by modeling the spatial component of variation. The value of spatial modeling can be enhanced considerably when a geographic information system is available that provides topographic, climatic, habitat and other covariates, although these covariates are likely to be at different resolutions from that used to record wildlife distribution. Several options for spatial modeling of large-scale survey data exist, and are discussed by Buckland and Elston (1993). If spatial correlation in the residuals is not a problem, generalized linear and generalized additive models provide a useful framework. When strong spatial correlation is present, possible methods include: Kriging in its various forms; Högmander and Møller (in preparation) show how to adapt image analysis methods for modeling wildlife distribution in a homogeneous habitat; Augustin et al. (in preparation) use autologistic regression to model presence/absence data in a heterogeneous habitat. In the latter case, the Gibbs sampler is used to allow fitting of the autologistic model when only a sample of sites was surveyed.

Models for Change

A common aim of most large-scale surveys is to monitor change over time. Conroy and Smith (in preparation) note the desirability of surveying the same sites in different years when the primary aim is to model change. In annual surveys, one method of quantifying change between successive surveys is ratio estimation. Consider $r_i = \frac{a_i - a_{i-1}}{a_{i-1}}$ where a_h is the number of animals recorded in year h, $h = i - 1$ or i, summed across all sampling units surveyed in both years. Then,

$$se(r_i) = \sqrt{\frac{n\sum_{j=1}^{n}(d_{ij}-r_i a_{(i-1)j})^2}{(n-1)(\sum_{j=1}^{n} a_{(i-1)j})^2}}$$

where $d_{ij} = a_{ij} - a_{(i-1)j}$,

n is the number of units (sites) sampled,

and a_{ij} is the number of animals recorded in year i at unit j

The estimate and standard error are valid if some animals are recorded in both years (i.e., counts are not independent between years) and when the underlying change is different in different sampling units. The above approach is used by the BTO, both in their Common Birds Census and in their Constant Effort Sites scheme, to quantify changes in abundance of breeding birds between successive years.

Often, change over a time period longer than two years must be quantified. By adding one to the above index, it may be chained, so that the index for change from year i − 1 to year i + 1 for example would be $(1 + r_i)(1 + r_{i+1})$. A theoretically superior method due to Mountford (1982) has proved problematic to implement. Use of loglinear Poisson regression to improve upon the chain method is currently being investigated (van Strien et al. in preparation).

If abundance (relative or absolute) is estimated annually, Buckland et al. (1992) provide a method for quantifying change over time that was designed to be easily understood by wildlife managers. We illustrate it here in Figure 1, which is modified from Figure 1 of Anganuzzi et al. (1993). In this case, the population of interest is offshore spotted dolphins (*Stenella attenuata*) in the eastern tropical Pacific. The figure allows the user to determine at a glance whether abundance has changed significantly between any two years. In this example, confidence intervals were obtained by bootstrapping the full estimation procedure and applying the percentile method at each time point. In some studies, it is difficult to quantify the precision of the annual estimates. In this case, the sequence of annual estimates might be modeled, for example, using generalized additive models (Hastie and Tibshirani 1990) to give sufficient flexibility in estimating the underlying trend in abundance, and, hence, to quantify precision from the residuals of individual estimates about the estimated trend. An example of this approach, but assuming the trend followed an exponential curve, is given by Buckland et al. (1993b), who estimate the rate of increase of the California gray whale (*Eschrichtius robustus*) from migration watch point data.

A stochastic model for change for use on data from successive atlas surveys is given by Buckland and Elston (1993). It models the probability of occupation of an atlas site as a function of habitat suitability and of its distance from sites occupied in the previous atlas survey. In a later paper (Buckland et al. in preparation), it is shown how this method may be used to quantify probability of extinction in a way that takes account of habitat heterogeneity.

Conclusions

Many researchers are prone to be too optimistic and insufficiently critical when deciding on appropriate methods for a large-scale survey. Many funding bodies fail to review adequately the rationale and objectives of a proposed survey, and to monitor the implementation of the survey. As a consequence, large-scale surveys frequently make inefficient use of resources, fail to meet key objectives, are later required to meet objectives not specified at the outset, and generate data of low quality and dubious relevance. Major failings that I have encountered in large-scale and, in many cases, generously funded projects include the following.

- Adoption of fashionable techniques (such as photo-ID or DNA fingerprinting mark/recapture methods) in circumstances when better developed but less inspiring methods would meet the stated objectives at lower cost.
- Implementation of a series of surveys for estimating trends in abundance when it is clear from the outset that precision will be inadequate even to determine whether any trend is up or down.

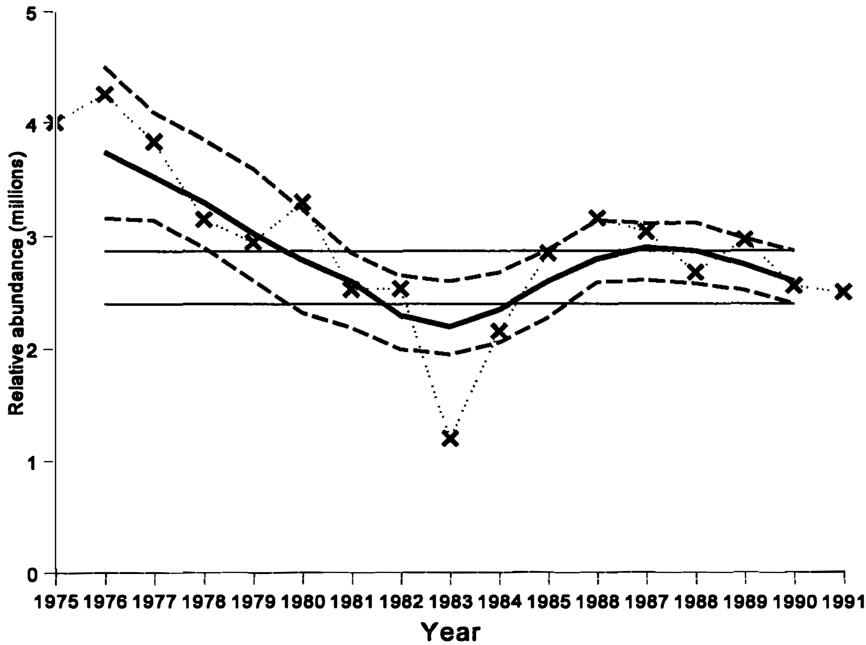

Figure 1. Smoothed trends in abundance of the northern offshore stock of spotted dolphin in the eastern Pacific Ocean. The solid line results from applying a compound running median smoother to the individual year estimates, shown by crosses. The broken lines indicate approximate 85-percent confidence limits. The horizontal lines correspond to 85-percent confidence limits for the 1990 estimate. If both the 1990 confidence limits lie above the upper limit for an earlier year, abundance has increased significantly between that year and 1990 ($p < 0.05$); if both limits lie below the lower limit for an earlier year, abundance has decreased significantly.

- Monitoring a large sample of subjectively chosen sites, or of sites for which access is easy (e.g., roadsides), when the stated objective is to estimate trends in abundance (by habitat) throughout a region.
- Changing methodology between surveys on the grounds that comparisons between data from the two surveys will not be of interest, then in the final publication, giving equal status to looking at changes in distribution between the surveys as to estimating the actual distribution at the time of the second survey.
- Selecting a simple random sample of sites for monitoring when the stated objectives can clearly be met in a more cost effective way by using a modified (e.g., stratified) sampling scheme.
- Analyzing the survey data as if they were obtained from a simple random sample, when a complex sampling scheme had been adopted to increase sample sizes on scarce species; the resulting bias was impressive!

In the initial enthusiasm for setting up a new project, it is easy to make errors that are extremely costly. Be warned, and seek advice of experienced practitioners and analysts. Take heed of the above examples and of the paper by Conroy and Smith (1994).

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Designing Large-scale Surveys of Wildlife Abundance and Diversity Using Statistical Sampling Principles

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Introduction

Surveys of the kind, abundance and distribution of fauna have played an important role in the development of wildlife management in North America, beginning with the earliest natural history surveys of Audubon, Bartram and others. Current national surveys of wildlife include waterfowl breeding ground (Pospahala et al. 1974) and winter (Conroy et al. 1988) surveys, breeding bird surveys (Robbins et al. 1986), and the Christmas Bird Count. More recently, concern over losses of species richness and abundance have led to efforts such as the Nature Conservancy's Natural Heritage Inventory program, which seeks to catalog the occurrence of important species and habitats within states, and gap analysis (Scott et al. 1993), which uses geographic information systems to delineate habitats, land use and species occurrence nationally.

One goal of surveys is the *description* of abundance patterns over time and space. For example, what is the current distribution and abundance of neotropical migrant birds? Are populations increasing or declining in abundance? Typically, surveys also are directed toward *understanding* and *predicting* patterns of abundance. For example, if forest bird abundance or species richness is observed to be declining over time, can the causes be controlled partially by managers, e.g., through modified silviculture? Thus, surveys can be used for making two types of decisions: (1) what to believe about the state of the resource, and, given this belief, (2) what to do about it.

Whether or not survey data are available, managers and policy makers must make decisions on the best available information and current understanding. Even failure to make a decision is a decision by default. Many of these decisions will be "wrong," but some "wrong" decisions are worse than others, i.e., they result in a greater *loss* (money, resources, opportunity, credibility). There are a number of reasons why survey data might lead to an incorrect decision: (1) the data contain sampling error; (2) biases exist in analyses because of invalid survey designs or models; (3) the understanding of the system (population, community, ecosystem) is incomplete, so that regardless of how well we measure the current state of the system, it is difficult to predict the consequences of management; (4) not all factors relevant to decision making are measured (Walters [1986] calls this a failure to "bound the problem" correctly); and (5) the data are gathered with the intent either to avoid making a decision, or to obfuscate the problem. Our presentation deals with the first four factors and shows how survey data can assist with decision making, increase understanding and provide more options for management. To do so, we must define our objectives, delineate our target population, establish measures of uncertainty, determine the costs of collecting data, and establish rules for optimizing sample design and for decision making. The focus of this paper is to show how statistical sampling principles, in conjunction with decision theory, can provide guidance in these areas.

Defining the Objectives of a Survey

The importance of designing a survey based on clearly defined objectives cannot be overstated. Survey objectives determine other critical components of the survey (i.e., the target population, the duration and spatial extent of the survey, and the temporal and spatial size of the sampling unit), and guide the design so that questions of interest can be answered. Cochran (1977: 5) advises that without a lucid statement of survey objectives, "it is easy in a complex survey to forget the objectives when engrosses in the details of planning, and to make decisions that are at variance with the objectives."

Even if remembered, loosely defined objectives will yield unsatisfactory results. Problems arise when objectives are broadly defined during survey design, but then narrowed at the analysis stage. Problems also arise when the objective of a survey merely is to collect a large amount of data with the mind that *something* will be learned later by exploratory analysis—a situation we believe is common in bio-monitoring programs.

Surveys can be used to describe either the status of, or trends in, a wildlife population; these, in turn, require different designs. *Status* is best assessed by collecting data on a probability sample of sites selected from the area of interest, and selecting new sites every time status is assessed. In contrast, *trend* is best assessed by revisiting a single probability sample of sites through time. Assessments of both status and trends may be obtained by an inter-penetrating sample similar to that implemented by the EMAP program (Messer et al. 1991). The duration of a trend survey must be adequate to detect the trend amidst variation due to population fluctuations and the sampling process. In contrast, a status survey must occur over a short enough interval so that changes in status do not occur, rendering the data gathered meaningless as a "baseline" or "benchmark."

The descriptions of status and trend are incomplete without additional information on underlying processes. For example, managers want information on *why* a population is declining or what underlying relationship links habitat to demography and to observed abundance. Rapid assessment of ecosystem integrity favors designs that provide unambiguous answers based on data collected across compact spatial and temporal scales, and thus experimental or quasi-experimental approaches. In contrast, comparisons and contrasts made without experimental perturbation of the system provide inference about associations, but not about cause-and-effect.

Understanding processes may be highly dependent both on proper *metrics* and appropriate *scales* in time and space. For example, simple abundance or density measures among habitats may poorly represent process and lead to false inference about habitat "benefits" (Hobbs and Hanley 1990, Van Horne 1983, Pulliam 1988).

As noted above, the resolution or "size" of the sampling unit in time and space is determined by the objectives of the survey, and may influence the utility of chosen metrics. For example, species richness as a metric is highly scale-dependent and is determined by processes that differ dependent on scale (e.g., Allen and Starr 1982, Harrison et al. 1992, Levin 1992, Maurer and Heywood 1993). Thus, it is non-informative to describe the relative "diversity" (i.e., species richness) of sampled ecosystems, without first clarifying the spatial and temporal context of the target population and sampling units. Finally, we agree with Davis et al. (1990) that "until we conduct an *unbiased* survey of biological diversity *at the appropriate level of resolution* we simply do not know what percentage of biodiversity is protected nor what remains to be done [emphases added]."

Defining the Target Population

In order for statistical inference to be possible from sample survey data, the *target population*, that is, the entity to which inference will be applied, first must be defined. For example, it is of no use to obtain an estimate of abundance of a species if the geographic area and time interval to which the estimate applies are unknown. Definition of the target population can be complicated if survey objectives include several species or an entire community. Thus, a first step is in deciding the *level of ecological organization* at which inference is targeted; for example, a single population, populations of several species, a community, a landscape, and so forth. Further, a survey design that is optimal at one level of organization may not be optimal at others. Thus, while a survey targeted at a community may be disaggregated to provide data for the constituent populations, it is unlikely that these will optimal for any single species. Similarly, several single-species surveys simply cannot be aggregated to obtain a survey that is optimal for a community. Related to this, a multi-species population survey will estimate density of common species well, but rare species poorly; sampling of rare species may require special methods (e.g., Sanders 1968, Hurlbert 1971, Heck et al. 1975, Smith and Grassle 1977, Sudman et al. 1988, Thompson 1990).

Likewise, the *spatial and temporal resolution* of the target population must be established. Thus, statewide surveys cannot be decomposed into county surveys without loss of precision, and aggregated (and individually optimal) state surveys will not be optimal nationally. Similarly, depending on objectives, single point in time, seasonal, yearly, multi-year average and long-term trend information may be of interest. Selection of a particular time scale will determine whether inferences at other scales are appropriate.

In most surveys, these decisions will be complicated by the fact that there are multiple resources and multiple goals; implicitly there will be trade-offs. Thus, if there are multiple species, communities or levels of spatio-temporal resolution that are of interest, survey designs that are optimal at one level are unlikely to be optimal at another.

Evaluation of Uncertainty

The terms "inventory" or "census" sometimes are used by natural resource managers implying an exact knowledge of the kinds and numbers of resources under management. Even if theoretically possible (for example, total counts of trees in a forest stand), these seldom are practical. They almost never are possible for wildlife resources, because wildlife are mobile and typically difficult to detect and enumerate. Thus, even if all habitats on a management area could be observed continuously, not all wildlife within these habitats (the target population) would be observed.

Except in the very special case where a complete inventory is possible, sample data must be used to make inferences about the target population, and these inferences will be *uncertain* (i.e., there will be a probability of them being "wrong"). The sampling process first involves definition and selection of units in time and space, then detection of individual animals within sampling units. Finally, estimates are made of population parameters (e.g., species density or diversity) derived from the design or model underlying the sampling process. Ultimately, management decisions are made and based (at least in part) on these estimates and subsequent analyses, e.g., trend analyses. Because of this sampling process, managers must take into account several sources of variability that affect the probability that their decision is correct. We are very much concerned about extant surveys that are treated as inventories, but in fact involve sampling with error, including unknown, confounding factors. Surveys that either ignore sampling error, or treat it in an ad-hoc way, are not amenable to scientific standards of repeatability and reliability.

The error of an estimated population parameter (e.g., density) will be a function of demographic and environmental fluctuations, random errors due to sampling, and systematic errors due to imperfect detectability. The fluctuation of density due to demographic and environmental stochasticity may be of interest and valuable in management decisions. However, that information is confounded by random and systematic errors. Therefore, it is important to carefully design the survey to minimize, and perhaps estimate, both random and systematic errors, in order to separate important "signals" (e.g., environmental trends) from background "noise" (sampling and systematic errors).

Systematic errors cannot be eliminated completely by design, and auxiliary information is needed to model detectability so that indefensible assumptions can be avoided. To illustrate the effect of systematic errors, consider a survey where the objective is to compare butterfly density across two land-use types. Let y_{ij} denote the density (or perhaps the log transformation of density) on the *j*the plot of the i^{th} land-use type, $i = 1,2$ $j = 1,...,n$. In the absence of systematic errors the underlying model of the data could be represented by:

$$y_{ij} = \mu + \tau_i + \varepsilon_{ij} \quad \textbf{(1)}$$

where μ is the overall mean density, τ_i denotes the effect due to land use and ε_{ij} denotes random error. We are interested in testing the hypothesis that the densities are equal between two land-use types (H_o: $\mu_1 = \mu_2$ versusH_a: $\mu_1 \neq \mu_2$. This suggests the contrast $\bar{y}_1 - \bar{y}_2$. In this case, the contrast, $\bar{y}_1 - \bar{y}_2$ tests the hypothesis of interest, i.e., $\tau_1 = \tau_2$ or equivalently $\mu_1 = \mu_2$. However, suppose that detectability of the butterfly differs between the land-use types, perhaps due to differing amounts of cover. Then the underlying model is best represented by:

$$y_{ij} = \mu + \tau_i + \delta_i + \varepsilon_{ij} \quad \textbf{(2)}$$

where δ_i denotes the detectability effect in the i^{th} land-use type. Consequently, the contrast no longer tests the hypothesis of interest, but instead includes effects due to detectability, i.e., $(\bar{y}_1 - \bar{y}_2) = (\tau_1 - \tau_2) + (\delta_1 - \delta_2)$. Land-use effects are confounded by detectability effects. If detectability cannot be assumed equal across land-use types, it must be estimated and removed from the contrast before testing for a land-use effect.

Optimal Survey Design

Sampling Effort

As seen above, there are a number of factors that compel greater effort in sample surveys. Thus, to increase the scope of inference, the target population should be as encompassing as possible in both time and space. To reduce variability in estimates and increase the statistical power of detecting patterns in time and space, an adequate number of sampling units must be used and detection probabilities should be as high as possible. However, a number of practical considerations often lead to reduced survey effort. Most obvious is the cost of surveys in terms of personnel, transportation, and compilation and analysis of data. Agencies rightly see these expenditures as involving trade-offs: resources spent on one survey are not available for a competing survey, or for management.

Sampling theory (e.g., Cochran 1977, Thompson 1992) can be used to evaluate the trade-offs between the costs of collecting survey data, versus the losses associated with the commission of errors, assuming that the objectives and target population are well defined, and that a dollar or other value can be assigned to the respective costs and losses. Suppose that the total cost, C, of conducting a wildlife abundance survey is given by the expression:

$$C + C_0 + C_1 n \tag{3}$$

where C_0 represents the fixed cost (overhead or other costs that do not change, regardless of sampling effort) of conducting the survey, and C_1 is the per-unit cost of sampling. Suppose further that as a consequence of errors in estimating N (true abundance) from $\hat{N}$ (the sample survey estimate of N), the agency incurs costs of D dollars, times the absolute difference in N and $\hat{N}$. Then, as a special case of a "cost plus loss function" we have (Schreuder et al. 1993):

$$\Delta + C_0 + C_1 n + 2DS_Y(2\pi n)^{-1/2} \tag{4}$$

where S_Y is the standard deviation of $\hat{N}$. This expression is minimized at:

$$n = \left(\frac{D^2 S_y^2}{2\pi C_1^2} \right)^{1/3} \tag{5}$$

For example, if the per-unit costs (C_1) are \$5, accuracy loss (D) is \$10 and $S_Y = 50$, optimal sample size is $n = 12$ units.

Multi-stage, Stratified and Cluster Sampling

Wildlife can be viewed as elements in a hierarchy of increasingly coarser spatio-temporal scales. Thus, a songbird nesting in a forest stand is an element of a population of breeding songbirds which is an element of a regional aggregation of songbird populations, which can be aggregated further with other regional populations to comprise a national population. Surveys of wildlife populations thus are inherently *multi-stage* in design. For example, a two-stage design could be used to estimate the abundance of forest birds statewide, with the first stage comprised of a primary sample of mapped forest stands selected and a second stage comprised of survey lines or points located randomly in each selected stand. Generally, land units will be sampled at all but perhaps the last stage, at which individual animals or groups of animals are sampled. Design-based sampling plans are used to select land units and provide estimates appropriate for the target population (e.g., statewide abundance by areal expansion from sample plot estimates). Sampling animals is complicated by the fact that within a selected unit of land some animals will be undetected. Model-based sampling, such as capture-recapture and line transect methods, then is used in conjunction with design-based sampling. In the latter, properties of the estimator, such as unbiasedness and precision, depend on the random selection of sampling units, whereas in the former these properties depend on the selection of an appropriate model. Seber (1982, 1986, 1992) provides reviews of model-based sampling for wildlife, while Steinhorst and Samuel (1989) and Hansen et al. (1983) discuss combination of design-and model-based sampling designs.

At each stage of sampling, a variety of sampling selection plans can be employed. When sampling units of land, the advantage of each sampling plan will depend on its cost effectiveness and on the spatial distribution of the target population. The simplest possible design-based approach involves selection of sampling units, preferably at random but sometimes systematically, from a list of possible units, with equal probability. Several modifications of simple random sampling approaches exist for reducing the sample variability or increasing the efficiency of the resulting estimates (Cochran 1977). Stratified random sampling results in estimates with higher precision than simple random sampling when the variation among units within strata is small, but the variation among stratum means is large. In some situations sampling units form natural clusters, and it then may be more convenient or cost-effective to first select a sample of clusters, then take observations (e.g., counts of ducks) on all sampling units (e.g., individual ponds) within a selected cluster (e.g., a network of ponds). In contrast to stratified random sampling, cluster sampling provides higher precision when variation among units within a cluster is high while variation among the cluster means is small.

Wildlife populations often are distributed unevenly, and a randomly selected unit of land in some cases may contain no animals, while in others contain only a portion of a large aggregation. Biologists aware of the tendencies for wildlife to aggregate are tempted in such circumstances to sample units adjacent to selected ones where large counts occur, in the hopes of getting as large (and presumably accurate) a count as possible. However, including these additional observations into conventional estimators will result in seriously biased estimates of abundance. Thompson (1990, 1992) developed adaptive cluster sampling designs that allow for increased sampling effort in the vicinity of randomly selected units meeting a criterion (i.e., presence or threshold abundance of a target species). Despite its intuitive appeal, adaptive cluster sampling does not always result in estimates with smaller variance, and its efficiency relative to simple random sampling depends critically on the distribution of the target population, the sampling unit size and the criterion that determines when to adapt sampling (Thompson 1990). These factors should be considered carefully, perhaps through simulation, before large-scale implementation of adaptive sampling (Smith 1993).

Finally, auxiliary information frequently is available that can be used to predict observations on sampling units on which observations of wildlife are not attempted. If a predictive relationship between the auxiliary data and observations of wildlife is justified, the use of auxiliary data will increase sampling efficiency, particularly if these data are cheaper or easier to obtain than direct information on abundance. For example, structural features such as type and basal area may crudely predict abundance of forest birds, and can easily be quantified from aerial photographs. An empirical relationship then can be established on a sample of stands for which both bird abundance is estimated and aerial photographs obtained, and used to predict abundance on stands where only aerial photographs are available (Cochran 1977, Eberhardt and Simmons 1987, Thompson 1992).

Using Surveys to Make Decisions

As alluded to earlier, management decisions cannot wait for survey results, and managers may question the marginal value of increased survey effort to their decision-making process, if, indeed, survey data are even formally considered in decision making. This points to a problem we seen in many survey efforts: the lack of a formal connection of surveys and research to management decisions. Johnson et al. (1993) observed that many waterfowl managers consider "research (the *accumulation* of information and understanding) and management (i.e., the *application* of information) as mutually exclusive pursuits." Unfortunately, this view of research and management as disjoint activities is common in natural resource management. We particularly are concerned that recent initiatives toward large-scale inventory and monitoring of biodiversity (e.g., Scott et al. 1993, National Research Council 1993), although motivated by widely recognized problems such as loss of habitats and species abundance and richness, need stronger connection to objectives and decision making.

Statistical decision theory provides a powerful tool by which to evaluate the importance of survey information in the context of decision making. The procedure for making decisions can be summarized as follows (Lindley 1985): (1) list all the possible decisions that can be made $\{d_1, d_2, ...d_m\}$; (2) list the uncertain events or outcomes that can occur $\{\theta_1, ...\theta_n\}$; (3) assign prior probabilities to the events $\{p(\theta_1)....., p(\theta_n)\}$; as will be seen, these probabilities may be based on survey information; (4) assign utilities to the consequences; and (5) choose the decision that maximizes expected utility:

$$\bar{u}(d_i) = \sum_{j=1}^{n} u(d_i, \theta_j)\, p\,(\theta_j) \qquad \textbf{(6)}$$

For example, suppose that we are responsible for the management of a population of a threatened species, and there are two possible decisions that can be made: either take no action (d_1) or implement conservation effects such as habitat restoration (d_2). Our criteria for making a decision includes the status of the species: whether the population is increasing or stable (θ_1), or is decreasing (θ_2). Under this scenario there are four possible consequences (C_{ij}, where i represents a decision d_1 and j represents outcome θ_j) to a decision: the population is stable and we take no action (C_{11}); the population is stable but we nevertheless decide on conservation action (C_{21}); the population is declining but we take no action (C_{12}); and the population is declining and we take action (C_{22}). Each of these consequences can be assigned utilities, which represent the desirability (scaled as probabilities) of each of the consequences (Table 1). Clearly, C_{11} is the most desirable outcome, because it avoids either an unnecessary conservation action or a deterioration of the population's status; therefore its utility, $u(C_{11})$, is 1.0. Conversely, C_{12} is the worst possible outcome, because it involves taking no action in the face of a declining population, and we have assigned it a utility of zero. The other two consequences are of intermediate utility, and we have chosen two scenarios. In the first (Table 1a), we are neutral about the utility of taking conservation action without a need, thus $u(C_{21}) = 0.5$. However, we wish to keep the probability of taking warranted conservation measures high, so $u(C_{22}) = 0.75$. In the other scenario (Table 1b), we maintain a high probability of taking conservation action if it is needed, but now the cost of taking unwarranted action, for example because of constraints on other resource use or other considerations (e.g., agency credibility), causes the utility of C_{21} to be much lower (e.g., 0.1). Finally, suppose we can assign probabilities to the two outcomes, θ_1 and θ_2, based on survey data or other information. It can be shown that management action (d_2) should be taken if:

$$[p(\theta_2)]\ [u(C_{22}) - u(C_{12})] > [1 - p(\theta_2)]\ [u(C_{11}) - u(C_{21})] \qquad \textbf{(7)}$$

noting that:

$$p(\theta_1)+p(\theta_2)=1 \qquad \textbf{(8)}$$

because θ_1 and θ_2 are mutually exclusive and exhaustive events.
Decision d_1 (no action) will be chosen if this inequality is reversed; if the two sides of the expression are equal, the decisions have equal utility and the choice is arbitrary. For the utilities in Table 1a, d_2 always will have higher utility than d_1 if probability of decline ($p(\theta_2)$) is > 0.40, whereas for those in Table 1b, the probability of decline must be > 0.55.

Survey data can be incorporated using Bayes' Theorem (e.g., *see* Maritz and Lwin 1989) we have:

$$\frac{p(\theta_1|X)}{p(\theta_2|X)} = \frac{p(X|\theta_1)}{p(X|\theta_2)} - \frac{p(\theta_1)}{p(\theta_2)} \qquad \textbf{(9)}$$

where X represents the survey data and $P(X|\theta i)$, $i = 1, 2$ represents the likelihood of X under the alternative hypotheses θ_1 (stable or increasing) and θ_2 (declining). The data might be a series of annual estimates of abundance, X_t for years $t = 1$ to n. A simple trend model such as:

$$\ln X_t = \ln X_0 + \beta_t,\ t = 0,1,2,\ldots,n \qquad \textbf{(1)}$$

could be fit to these data, and under assumptions of normal and independent errors, the likelihood ratio evaluated under $H_0{:}\beta = 0$ (corresponding to θ_1) and the one-sided $H_a{:}\beta < 0$ (corresponding to θ_2) computed as:

$$\frac{p(X|\beta=0)}{p(X|\beta<0)} = \left[\frac{n-2}{F+(n-2)}\right]^{n/2} \qquad \textbf{(11)}$$

Table 1. Utility table for hypothetical problem involving decision to either take no action (d_1) or conservation action (d_2), given uncertainty about whether population is increasing or stable (θ_1), or declining (θ_2). Cell entries are utilities ($u(C_{ij})$; see text). In both cases (a and b) the utility is highest ($u(C_{ij}) = 1$) for (correctly) taking no action if population is stable or increasing; next highest ($u(C_{ij}) = 0.75$) for (correctly) taking conservation action if population is decreasing; and lowest ($u(C_{ij}) = 0$) for (incorrectly) taking no action if population is declining.

a. Indifferent ($u(C_{ij}) = 0.5$) utility to incorrect decision of taking conservation action if population is actually increasing or stable.

	θ_1: increasing/stable	θ_2: declining
d_1: no action	1.0	0.0
d_2: conservation	0.5	0.75
Probabilities	$p(\theta_1)$	$p(\theta_2)$

b. Low ($u(C_{ij}) = 0.1$) utility to incorrect decision of taking conservation action if population is actually increasing or stable.

	θ_1: increasing/stable	θ_2: declining
d_1: no action	1.0	0.0
d_2: conservation	0.1	0.75
Probabilities	$p(\theta_1)$	$p(\theta_2)$

where F is the computed F (or t^2) statistic for the test of a model effect (Graybill 1976: 187–188). For example, if $n = 10$ years of data are used to fit the model and $F = 3.5$, the likelihood ratio would be $(8/11.5)^5 = 0.16$. Given non-informative prior probabilities (e.g., $p(\theta_1) = p(\theta_2) = 0.5$), this reflects our relative belief (based on current data) in the two hypotheses; in this case, $p(\theta_1)/p(\theta_2) = 0.23$, or $p(\theta_2) = 0.86$, well in excess of the threshold of evidence needed for decision d_2 given either set of utilities in Table 1. In contrast, $F = 0.07$ would suggest $p(\theta_2) \approx 0.50$, and decision d_2 for the utilities in Table 1a, but d_1 for those in Table 1b. This example illustrates that the value of information (whether it is from surveys or other sources) derives from its decision-making context. In the first case, it required a lower threshold of "proof" of a decline to justify a management action, and in fact that action can and should be taken even if the evidence from data favors "no decline." On the other hand, if the utilities are as in Table 1b, then there is a higher "burden of proof" on demonstrating a decline. Thus, additional data that may improve upon estimates of trend, for example, may be more important in the latter case than in the former. Decision theory thus can be used to evaluate the expected gain in utility with improvements in information, up to the maximum such gain possible (Lindley 1985: 120–130).

Finally, this approach formalizes the mental process that managers hopefully go through: make a provisional statement about the state of the systems (based on previous data, models, biology or guesswork), collect new data and reevaluate one's prior knowledge, based on current information (*see also* Walters 1986, Conroy 1993). The process is fundamentally sequential and iterative; consequentially, both decisions, and the collection of data, need to be considered in a long-term manner. Recently, Johnson et al. (1993) described an optimal decision-making approach for harvest management of waterfowl, in which Bayesian decision-theory is used in conjunction with adaptive dynamic programming. Decision makers use current information from waterfowl population surveys, in conjunction with alternative hypotheses about the effects of harvest on population dynamics, to forecast outcomes under alternative management scenarios (i.e., decisions). Each year, new survey data are obtained which can be used to adjust the prior probabilities of the alternative models, and, if necessary, to modify decision making. This approach encompasses what we think are ideal features of a natural resource survey: it is based on a specific, quantifiable objective (in this case, optimal long-term harvest), it uses statistically sound survey data as an adjunct to decision making and it is adaptive. These features also are compatible with stated priorities of the newly formed National Partnership for Biological Survey: "To provide a better and more efficient information base from which to make planning and operational *decisions*, thereby ... improving the management of biological resources" (National Research Council 1993: 54 [emphasis added]).

Conclusion

Many surveys of wildlife abundance or diversity lack either clearly stated objectives (abundance? trend? diversity?), a sampling design to meet these objectives or a means of determining when the objectives have been met. Thus, it often is asserted that surveys are "needed" and "useful for management" when there are no objective grounds for the assertion. This provides neither a good justification for spending dollars that could be used for other purposes ("a dollar spent on a survey is a dollar that could have been spent saving the species"), nor a rational context for the use of survey information in management decision. We suggest that managers closely examine the goals of extant and contemplated survey efforts, and justify them in the context of objectives and decision making. Statistical sampling and decision theories provide a formal mechanism to do so.

Acknowledgments

We appreciate earlier useful comments and suggestions by D. L. Otis and B. K. Williams. The Georgia Cooperative Fish and Wildlife Research Unit is sponsored jointly by the National Biological Survey, the University of Georgia, Georgia Department of Natural Resources and the Wildlife Management Institute.

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Horvitz-Thompson Survey Sample Methods for Estimating Large-scale Animal Abundance

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Introduction

Wildlife management depends on a fundamental knowledge of species population dynamics and on the ability to monitor population changes or responses to management. Animal abundance and the rate of change are two of the principal parameters for assessing the status of wildlife populations and determining the need for management. Survey sampling (or descriptive sampling) methods typically are used to obtain large-scale abundance estimates because the financial and logistic constraints allow sampling on only a fraction of the area occupied by the population (Cochran 1977, Eberhardt and Thomas 1991). However, most survey sampling methods fail to account for the fact that many animals on the sampled plots are not detected. In contrast, numerous methods have been developed to estimate the probability of detecting animals in small areas (Seber 1982, Lancia et al. 1994), but most have not been extended to large-scale surveys.

In addition to providing essential management information, considerable progress in large-scale ecological and environmental research can be made by combining field observations with controlled experiments (Eberhardt and Thomas 1991). These large-scale experimental field studies (Sinclair 1991) and evaluations of wildlife management (Macnab 1983) also should be conducted with a sampling framework that encompasses the population of concern (Eberhardt and Thomas 1991). To realistically evaluate landscape- and ecosystem-scale research, the spatial design of the sampling effort must be at least as meticulous as the design of experimental manipulations.

The basic components that must be included in large-scale animal abundance surveys are the delineation of the species (or population) range, the characterization of the geographic distribution of abundance, the selection of a spatial sampling design and the application of a survey technique to determine the abundance on each surveyed plot. These factors must be integrated to produce a statistically based estimate of the population size that meets the required level of precision. In this paper, we describe these basic survey components and present a Horvitz-Thompson survey sample method for wildlife populations. This method requires that the survey design consider the spatial distribution of animals and the effort needed to estimate animal abundance on sample plots.

Elements of Large-scale Surveys

Survey Objectives and Overview

The first step in planning a survey is to establish a clear set of survey objectives, including the goals of the survey, anticipated uses and level of desired precision. The precision of the abundance estimate is a critical component of planning a survey and evaluating its success. Robson and Regier (1964) provided some general guidelines for setting precision and accuracy objectives, but the circumstances and goals of each survey may be unique. When the objectives are established, a preliminary survey can be designed to meet the precision, accuracy, cost criteria and other critical features of the survey. Calculations can be made to determine the number of sample plots and the effort required to detect animals on sample plots. Because survey accuracy is affected by errors in spatial sampling (plot to plot variation) and the detection of animals on a sample plot, trade-offs between the number of sample plots and the effort spent detecting animals on those plots is necessary (Alho 1992). Finally, survey planning and optimization require informed guesses about the survey design characteristics, probability of detecting animals and survey costs. Because additional knowledge and experience are acquired each time the survey is conducted, a survey may require a recursive approach using previous results to further improve and optimize the design.

Population Range and Survey Boundary

After the survey objectives are established, the population boundary or species range must be determined. Because most survey methods require a geographic framework for estimating population abundance, a limit on the geographic area for the survey must be established. Precise delineations of the geographic boundaries for a population may be difficult for large-scale surveys when exact boundaries are unknown. Species range maps, broad ecosystem boundaries, habitat characteristics and other factors may provide useful guidelines for establishing the geographic scope of the survey. In the absence of such guidelines, a pre-survey may be necessary to determine the population boundary. The geographic boundary defines the extent of the population that will be surveyed. Therefore, changes in the geographic boundary have a direct effect on estimates of abundance.

Two practical problems in conducting large-scale surveys within fixed geographic boundaries are inadvertently excluding animals outside the boundary and wasting survey resources by sampling in areas with no animals. Animals that occur outside the survey boundary will not be included in the population estimate. Difficulties in identifying adequate geographic boundaries may be compounded for highly mobile species that respond primarily to suitable environmental conditions. For example, pintail ducks (*Anas acuta*) may drastically change annual breeding distributions in response to environmental conditions (Bellrose 1976: 267). Less obvious problems can arise when specific portions of the population have different use patterns by age or sex. If these problems exist, survey boundaries may have to be adjusted annually to account for changes in distribution.

Because large-scale surveys typically are expensive, excluding areas that have no animals within the geographic boundary of the survey may be cost effective. There are no conceptual reasons why the survey areas must be contiguous or why some areas within the geographic boundary cannot be excluded. Exclusions may be based on known geographic distributions, unsuitable habitats or other reliable indicators that no (or relatively few) animals will be present. Again, animals outside the survey boundaries will not be included in the population estimate.

Spatial Sampling Patterns

Few species of animals have a uniform distribution of abundance across the landscape. Most species respond to the favorable or unfavorable distribution of environmental characteristics, thereby creating patterns or gradients in species abundance. These patterns of abundance can be used to improve the reliability (precision) of surveys through stratification. The goal of stratification is to produce survey areas with similar levels of abundance so that variance among sample plots within each strata is minimized. However, it is not commonly recognized that the estimated abundance of animals after corrections for detection probability is used to calculate the variance within each strata. Assignments of plots to different strata must be based on how detection will affect the estimate of abundance on each plot and is especially important when habitat characteristics, which may affect detection probabilities, are used to define different strata. Usually at least three to six strata are defined (Eberhardt and Thomas 1991); a higher number of strata levels can facilitate reliable population estimates. For most wildlife surveys, the a priori knowledge of population distribution is too coarse to define more than three strata. On a landscape-scale, predictive habitat or environmental characteristics may be useful for identifying potential strata (Ratti and Garton 1994). The revision of strata boundaries or even levels as more knowledge is gained about the actual species distribution patterns is not unusual. Similar to the problem of population boundaries, highly mobile species may require a flexible annual adjustment of strata boundaries, depending on changing distribution patterns.

If populations are distributed randomly or information on population distribution is limited, a simple random sample may be the best choice. This approach requires that every sample plot in the population has an equal chance of being selected and that the procedure for selecting plots must truly be random (Ratti and Garton 1994). A more general form of random sampling arises when sample plots are selected with unequal probabilities. This usually occurs when plots have different sizes and random coordinates are used to select plots. In this case, each plot has a sampling probability proportional to the size of the plot (PPS sampling).

When population abundance follows well-defined gradients, other survey designs, including systematic sampling, may be more appropriate. Systematic sampling also may be a useful survey design when the objective is to determine the pattern of abundance. For randomly distributed populations, systematic sampling may provide estimates that are similar to a simple random sample (Ratti and Garton 1994). Other survey designs such as cluster sampling may be useful when the logistics of estimating abundance make travel between survey units expensive and when the population densities on adjacent sample plots are heterogeneous. If densities on adjacent plots are similar, cluster sampling will not increase the precision of the population estimate, compared to a simple random sample. Additional information on multi-stage sampling for wildlife studies can be found in Bart and Notz (1994) and more advanced statistical details are available in Cochran (1977) and Scheaffer et al. (1990).

Abundance on Sample Plots

The fundamental problem with determining animal abundance, even in relatively small areas, is that many animals will not be detected. In many circumstances, there is clear evidence that a large portion of the population will not be detected even with the most sophisticated methods (Caughley 1977: 35). Our inability to determine the actual number of animals in a particular area has given rise to numerous survey methods to estimate abundance when only a portion of the animals actually are observed. These methods include a plethora of popular survey techniques, such as capture-recapture, line transect, point counts, aerial surveys and catch-effort (Bibby et al. 1992, Lancia et al. 1994). This variety of survey techniques has developed to accommodate differences in species biology and behavior, habitats used, logistic considerations, seasons, sample plot size, and even researcher preferences. Further complications can arise when animals occur in groups because an assumption for many survey techniques is that each animal is observed independently. Group size also may be confounded with detection probabilities (Cook and Martin 1974, Samuel and Pollock 1981, Drummer and McDonald 1987, Samuel et al. 1987), producing biased population estimates when this effect is not considered. Few survey techniques currently permit the estimation of detection probabilities of groups of animals.

In general, most survey techniques attempt to determine the probability of detecting animals in an area and convert this probability and the number of observed animals into an estimate of actual abundance. Ideally, the survey design accommodates a variety of methods for determining animal detection probabilities on sample plots. Surveys where more than a small portion of the animals are undetected cannot provide estimates of abundance unless detection probabilities are determined. Techniques that do not account for undetected animals should be considered only indices of abundance.

Horvitz-Thompson Population Estimator

In the standard sample survey methods, the probabilities of selecting sample plots must be predetermined. This implies that an exhaustive sampling frame of non-overlapping units can be listed and randomly selected (*see* selection schemes above) with know probabilities for each plot. For most wildlife surveys, this requirement may be met for sample plots of land that can be completely and uniquely identified. In contrast, a sampling frame of individual animals cannot be developed without a priori knowledge of the number of animals on each sample plot. To overcome the requirement of a sampling frame for animals, Steinhorst and Samuel (1989) and Samuel et al. (1992) developed a modified Horvitz-Thompson sample survey estimator of animal abundance that incorporates the probability of detecting animals during aerial surveys. In the original development, the term "sighting probability" is used for aerial surveys; however, this approach applies to the general probability of detecting animals on a sampled plot. By using this general approach to animal detection, the modified Horvitz-Thompson method provides a comprehensive framework for the design of large-scale abundance surveys. The population estimator (Steinhorst and Samuel 1989) uses the detection probabilities to provide an unbiased estimate of abundance. The general abundance estimator is:

$$t = \sum_{k=1}^{l} \frac{1}{p_k} \sum_{i=1}^{n_k} \frac{m_{ik}}{\pi_{ik}}, \tag{1}$$

where

t = the estimated total population,

p_k = the probability of selecting the kth sample plot,

n_k = the number of groups (≥ 1 animal) detected on sample plot k,

m_{ik} = the number of animals in the ith detected group on sample plot k, and

π_{ik} = the probability of detecting group m_{ik} during the survey.

In this general form, the Horvitz-Thompson estimator allows for unique probabilities of sampling each plot and different probabilities of detecting each animal (or group) on a sample plot. The number of detected animals are adjusted by the detection probability and the probability of sampling a plot to produce an estimated total abundance. Lancia et al. (1994) presented a simplified version of Equation 1 and discussed adjustments for the detection probability and the sampling proportion.

The Horvitz-Thompson estimator incorporates three sources of survey error (Steinhorst and Samuel 1989, Samuel et al. 1922) from not surveying all the sample plots, not detecting all animals on a sample plot and from estimating the probability of detecting animals. The general equation for the variance is:

$$S_t^2 = S_{Dt}^2 + S_{Vt}^2 + S_{\pi t}^2 \tag{2}$$

where

S^2_t = the variance of the estimated population,

S^2_{Dt} = variance attributed to the sampling design,

S^2_{Vt} = variance attributed to not detecting all animals (visibility), and

$S^2_{\pi t}$ = variance attributed to estimating the probability of detecting animals.

The variation in spatial sampling (S^2_{Dt}) often is the largest portion of the total variance (S^2_t). In the Horvitz-Thompson approach, any sampling design for plots can be accommodated, but designs that reduce the variability in spatial sampling (e.g., stratified sampling) are more efficient because they provide more precise estimates of the population. Spatial variation also can be reduced by increasing the proportion of sample plots. In a similar manner, population variance (S^2_t) can be reduced with survey techniques that maximize the probability of detecting animals (reducing S^2_{Vt}) and minimize the variation from detection probabilities ($S^2_{\pi t}$).

In the Horvitz-Thompson method, variance for the probability of detecting animals ($S^2_{\pi t}$) must be based on the model for estimating detection probability. This method allows flexible estimators with separate detection probabilities for each observed animal; however, such heterogeneity may increase variance in the $S^2_{\pi t}$ component and in the population estimate. A variety of survey techniques for estimating detection probabilities are available to biologists and include line transect (Burnham et al. 1980), capture-recapture (Otis et al. 1978), circular plots (Reynolds et al. 1980), visibility models (Samuel et al. 1987, Otten et al. 1993), catch-effort (Alho 1992) and other approaches (*see* Lancia et al. 1994). Care should be used in selecting the most efficient methods(s) for detecting animals on the selected plots. Improved precision can be achieved with survey methods that can incorporate homogeneous detection probabilities of animals in a sample plot or, better yet, of animals in many sample plots. The need to improve efficiency in estimating detection probabilities was a principal motivation for developing general visibility models for elk (*Cervus elaphus*) surveys (Samuel et al. 1987).

Some survey techniques evaluate heterogeneity in detection probabilities during surveys on single sample plots (e.g., heterogeneous capture-recapture models). However, methods for evaluating and combining detection probabilities across multiple plots have not received much attention. One exception is the development of statistical methods for testing capture-recapture models among different populations (Skalski and Robson 1992). These tests also may be applicable to testing capture-recapture model similarity among sample plots. When models among plots are similar, more precise detection probabilities can be estimated by pooling results across plots for more precise population estimates. Similar improvements may be achieved with general methods to model capture-recapture probabilities (Alho 1990), catch-effort models (Alho 1992) or line transect methods (Burnham et al. 1980). Whatever approach is used, alternative survey techniques and detection probability estimates must be considered thoroughly during planning and analyzing large-scale abundance surveys. Special care also must be given to ensure that the assumptions (e.g., closed population, homogeneous detection probabilities, tag loss, etc.) for the selected survey technique can be met (Seber 1982). If detection probabilities cannot be determined in a timely manner (violating the closure assumption), open-population models (Pollock et al. 1990, Lancia et al. 1994) may have to be used. In the latter case, models that incorporate movement between sample plots (Hestbeck et al. 1991) also should be considered.

Survey Examples

In this section, we provide brief examples of some of the problems that may be encountered in large-scale surveys. We use elk population estimates to illustrate the importance of spatial and temporal variation in detection probability. Preliminary results from Canada goose surveys are used to illustrate some of the recursive aspects of survey design. Experiments on detection probability for duck surveys are used to speculate about the effects of animal behavior on survey results.

Elk populations have been monitored in portions of northcentral Idaho by helicopter survey during the last 10 years. Detection probabilities have been estimated with a visibility model (Samuel et al. 1987), with additional refinements as further data were collected (E. O. Garton unpublished data). Average visibility rates of bull and cow elk have differed (Samuel et al. 1992), primarily because bulls occur in smaller groups and more dense cover that makes them less visible than cows during aerial surveys. Incorporating heterogeneous visibility based on group size and vegetation allowed us to more accurately assess the total bull population and bull:cow ratios for improved herd management. In addition, winter conditions have varied considerably during the decade of conducting surveys. In particular, annual changes in snow conditions influenced the spatial distribution, habitat use and grouping behavior of elk in the survey area. During mild winters, animals are more dispersed, in smaller groups and in denser vegetation. These annual changes influenced the average visibility of animals. The spatial and temporal changes in detection probabilities are beyond the control of wildlife biologists and emphasize the danger in assuming a constant rate of detection. Use of an average detection probability would have severely biased population estimates of bulls and decreased the probabilities of detecting population changes from elk harvest and habitat management.

At one time, giant Canada geese (*Branta canadensis maxima*) were believed to be extinct (Bellrose 1976). However, the race was rediscovered and increased under protection, propagation and vigorous transplant programs. Recently, aerial surveys were initiated to assess the population of these birds in the Mississippi Flyway, where they have become a nuisance in some locations. Intensive helicopter surveys of sample plots were conducted during the nesting season to maximize detection of breeding pairs, nests and nonbreeding groups. Initially, 1-square mile (2.59^2 km) sample plots were surveyed in a stratified random design and random plots that did not contain viable goose habitat (absence of water on aerial photos) were not sampled. Subsequent survey refinements were attempts to reduce population variation with 2.25-square mile (5.83^2 km) sample plots and reduce helicopter transport costs by sampling additional plots from a surrounding cluster. Preliminary results indicate the number of geese are more consistent on larger plots than on smaller plots. However, cluster sampling has not proved effective because the densities of geese on sample plots in the surrounding area are similar. Thus, sampling nearby areas provides little new information about goose abundance. Results from the 1993 survey indicate that ≥ 800,000 giant Canada geese now are present in the Mississippi Flyway. Further survey refinements are needed to improve the precision of population estimates, determine detection probability, and improve the efficiency of the survey design and conduct.

Smith (1993) recently conducted experiments with decoys to estimate duck detection probabilities during helicopter and fixed-wing aerial surveys. They concluded that visibility varied by habitat characteristics, distance from the transect and group size. From simulations, they concluded that changes in habitat-use patterns could produce large changes in overall visibility and confound population monitoring.

Future Needs

Little attention has been given to either the practical problems of developing large-scale surveys of wildlife species or the unique statistical problems associated with wildlife population estimation. In general, practical and theoretical work are needed in at least three areas. First, new survey techniques or modifications are needed to assess spatial heterogeneity in detection probabilities. These methods should incorporate testing for heterogeneous detection and the means to efficiently combine detection probabilities across sample plots. Comprehensive development of statistical methods may be difficult because many different survey techniques currently are in use; however, for the capture-recapture techniques, model tests among populations (Skalski and Robson 1992) may be adaptable to testing among sample plots.

Statistical procedures for selecting the most appropriate model from the detection data have received considerable attention (Burnham and Anderson 1992). However, the practical effects of different detection models need further consideration in the context of population estimation. In general, detection models with more heterogeneity produce less biased but more variable population estimates. Although completely unbiased estimates of wildlife populations may be impractical, decision rules are needed to evaluate the relative merit of biased, more precise estimates compared with less biased, less precise estimates. One possible approach is to compare the population mean square error (MSE = bias^2 + variance) of different detection models. Because MSE consists of variance and bias^2, comparisons can be made among uncorrected population estimates and estimates corrected for different amounts of detection heterogeneity (Figure 1). These MSEs may be scaled by estimated population size (e.g., CVs) to standardize the comparisons. In addition, biologists should consider the importance of improving accuracy and precision by devoting more resources to increasing detection and estimating detection probabilities.

A third potential area for improvement in wildlife surveys is the development of predictive associations between animal abundance and environmental characteristics. On a landscape-scale, species abundance consistently may be related to particular habitat characteristics that are favorable to the species. Potential relationships between landscape patterns and population abundance could be evaluated on a portion of the sample plots. Consistent relationships could be used to develop regression methods to predict abundance on sample plots and to incorporate predictions into an overall population estimate. Data from large-scale geographic information systems should be useful for investigating landscape patterns (Turner 1990) and evaluating species relationships (Palmeirim 1988). Large-scale approaches based on techniques such as cokriging (Stein and Corsten 1991) also deserve investigation.

Summary

Large-scale surveys to estimate animal abundance can be useful for monitoring population status and trends, for measuring responses to management or environmental alterations, and for testing ecological hypotheses about abundance. However, large-scale surveys may be expensive and logistically complex. To ensure resources are not wasted on unattainable targets, the goals and uses of each survey should be specified carefully and alternative methods for addressing these objectives always should be considered. During survey design, the importance of each survey error component (spatial design, proportion of detected animals, precision in detection) should be considered carefully to produce a complete statistically based survey. Failure to address these three survey components may produce population estimates that are inaccurate (biased low), have unrealistic precision (too precise) and do not satisfactorily meet the survey objectives. Optimum survey design requires trade-offs in these sources of error relative to the costs of sampling plots and detecting animals on plots, considerations that are specific to the spatial logistics and survey methods. The Horvitz-Thompson estimators provide a comprehensive framework for considering all three survey components during the design and analysis of large-scale wildlife surveys.

Problems of spatial and temporal (especially survey to survey) heterogeneity in detection probabilities have received little consideration, but failure to account for heterogeneity produces biased population estimates. The goal of producing unbiased population estimates is in conflict with the increased variation from heterogeneous detection in the population estimate. One solution to this conflict is to use an MSE-based approach to achieve a balance between bias reduction and increased variation. Further research is needed to develop methods that address spatial heterogeneity in detection, evaluate the effects of temporal heterogeneity on survey objectives and optimize decisions related to survey bias and variance.

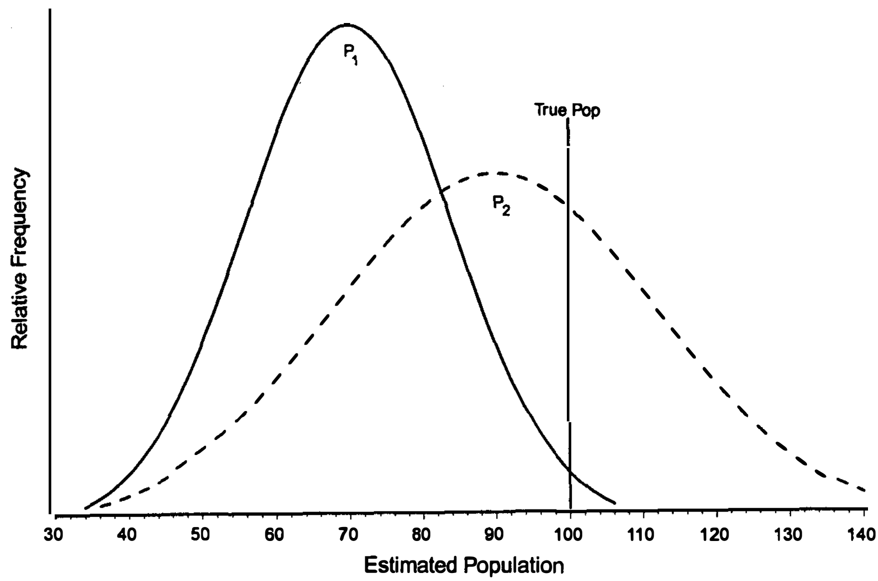

Figure 1. Comparison of hypothetical population size estimates of a true population of 100 animals. Estimate P_1 ($t = 70$, $S^2_t = 196$), which is significantly lower than the true population, includes sampling variance (S^2_{Dt}) but ignores variation related to detecting animals. Estimate P_2 ($t = 90$, $S^2_t = 400$) includes sampling variance and variance components related to animal detection (S^2_{Vt} and $S^2_{\pi t}$). Although the two estimates are not significantly different, mean square error is greater for P_1 (bias^2 + $S_t^2 = 1{,}096$) than for P_2 (500). With the mean square error approach, correction for animal detection provides improved accuracy with undue sacrifice of precision.

Finally, managers and researchers involved in the survey design process must realize that obtaining the best survey results requires an interactive and recursive process of survey design, execution, analysis and redesign. Survey refinements will be possible as further knowledge is gained on the actual abundance and distribution of the population and on the most efficient techniques for detection animals.

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Challenges in the Estimation of Population Size for Polar Bears in Western Alaska

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Introduction

Polar bears (*Ursus maritimus*) that occur seasonally in Alaskan waters are thought to belong to two separate populations, one in northern Alaska and one on western Alaska (Lentfer 1974). The northern Alaska area encompasses the entire Beaufort Sea and extends into Canadian waters (Amstrup et al. 1986), while the western Alaska area includes the northern Bering Sea and the entire Chukchi Sea, including Russian territory (Garner et al. 1990). Polar bears in both areas are subject to harvest by native Alaskan subsistence hunters as allowed by provisions of the 1972 Marine Mammals Protection Act (MMPA). However, from 57 to 81 percent of the 1980–1988 average annual harvest of 128 bears has occurred in the western Alaska area (Schliebe 1986, 1990). The U.S. Fish and Wildlife Service is required by provisions of the MMPA to manage polar bear populations at the optimum sustainable population level. This mandate necessitates an evaluation of the effects of the annual subsistence harvest upon the affected polar bear populations. Limited data for the Beaufort Sea (Amstrup et al. 1986) are available to preliminarily address this problem, however, few data are available for the western Alaska area and the effects of current harvest levels cannot be adequately evaluated.

Russia banned hunting of polar bears in 1956, however, they have recently expressed interest in reopening portions of the Russian Arctic to polar bear hunting, including the Chukotka region of the Chukchi Sea. The 1976 International Agreement on the Conservation of Polar Bears requires nations sharing populations to manage them on a cooperative basis (Stirling 1988: 210). Preliminary discussions indicate an allocation-of-take agreement may be an appropriate management protocol between the two countries. However, scientifically based estimates of population size or status are not available for the Chukchi and Bering seas on which to base such an agreement. The expanse of the area of concern, the low density of polar bears and the international aspects of this population make application of current survey procedures to estimate population size extremely difficult to apply.

Area of Concern and Window of Opportunity in Western Alaska

Ongoing research by U.S. and Russian scientists in the Bering and Chukchi seas is defining the population bounds of the shared polar bear population and other aspects of polar bear life history and ecology. These studies indicate that polar bears are dispersed widely throughout the sea-ice habitats of the southern Chukchi Sea and northern Bering Sea during autumn, winter and spring months (Garner et al. 1990, Garner and Knick 1991, Garner et al. 1994). The sea-ice recedes approximately 870 miles (1,400 km) from its maximum extent in early spring to its minimum extent in early autumn, when the Bering Sea and a majority of the Chukchi Sea are ice-free (Figure 1). Unlike polar bears in the Canadian arctic (Stirling et al. 1984, Derocher and Stirling 1990), western Alaska polar bears do not use summer retreats on land during the minimum ice period, but remain on the sea-ice throughout the year (Garner et al. 1990, Garner and Knick 1991, Garner et al. 1994). The total area used, exclusive of mainland areas, by 162 female polar bears fitted with satellite transmitters between 1986 and 1993 encompassed approximately 570,000 square miles (1.5 million km^2, Figure 1).

Polar bears in the Chukchi Sea appear to concentrate along the ice edge and normally do not range over 125 miles (200 km) into permanent pack ice during mid-August to mid-October (Garner et al. 1994), when the ice pack is at its minimum coverage (Naval Oceanography Command 1986, Figure 1). During this period, the polar bears' main prey (ringed seals, *Phoca hispida*) concentrate in the unconsolidated ice edge and along leads extending into the permanent pack ice.

The exception to this distributional pattern is the maternity denning component of the population, which is concentrated on Wrangel and Herald islands, and along the northern coast of the Chukotka Peninsula in Russia (Garner et al. 1994). Unlike black (*Ursus americanus*) and brown bears (*Ursus arctos*), non-parturient polar bears do not enter winter dens but remain active throughout the year (Stirling et al. 1984). Therefore, population surveys occurring between November and March would not encounter the maternity denning component of the population.

To maximize survey efficiency (i.e., maximize bear concentration and minimize study area size), a survey of the Chukchi/Bering seas polar bear population should be performed during a period of minimal ice cover. Therefore, the window of opportunity is during mid-August through mid-October along the sea-ice edge between approximately 156 degrees W and 170 degrees E longitude (Figure 1). At this time, the areal extent of the population is reduced to a 680 by 110 mile arc encompassing approximately 76,000 square miles (1,100 by 180 km arc, 198,000 km^2) lying along the 72 degree N parallel of latitude.

Polar Bear Population Estimation

Multiple-year Mark-recapture Procedures

The use of multiple-year mark-recapture data for estimating polar bear population size and status has become the standard procedure used in many studies (DeMaster et al. 1980, Stirling and Kiliaan 1980, Schweinsburg et al. 1982, Furnell and Schweinsburg 1984, Amstrup et al. 1986). This technique requires ongoing mark-recapture programs that can incur high annual operational costs. These costs limit the

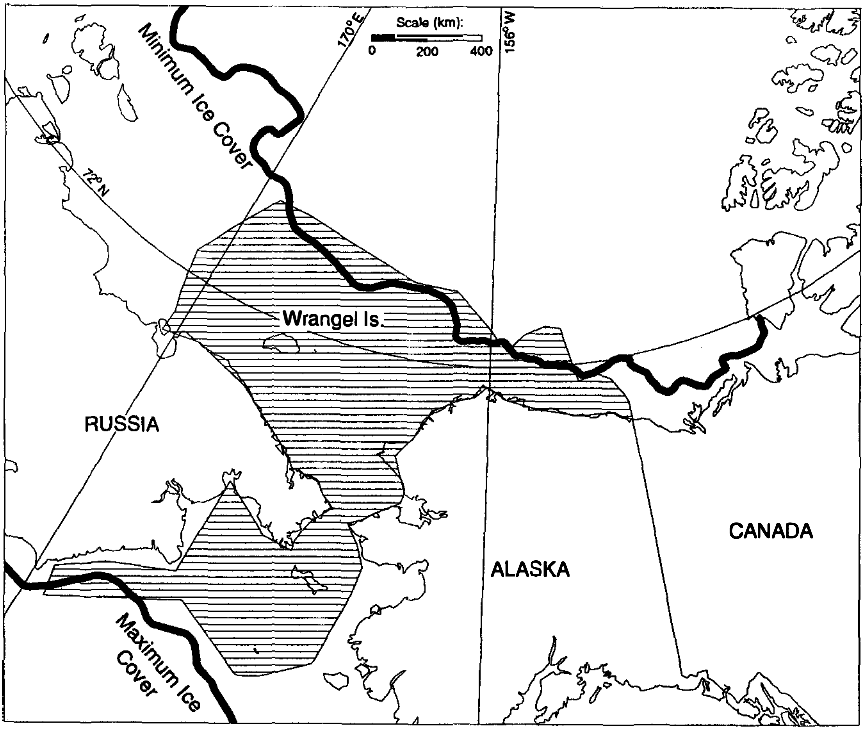

Figure 1. Western Alaska polar bear population area (shaded) showing extent of maximum and minimum ice cover.

practicality of using the procedures for the routine monitoring of polar bear populations in western Alaska. Also, the application of the methodology is confounded by the extensive movement patterns of bears in this area and the international nature of the population (Garner et al. 1900, Garner and Knick 1991, Garner et al. 1994). Recent capture and marking of polar bears in western Alaska has been limited to those bears present during spring, while capture and marking of polar bears in eastern Russia has been confined to the denning concentration on Wrangel Island. Therefore, marks have not been deployed throughout the population area, and marked bears do not appear to redistribute in a random manner throughout the population area. Derocher (1987) also noted that the methodology can give imprecise results if sample size is small and marked animals have low probabilities of recapture. Between 1986 and 1993, a total of 297 polar bears have been marked in western Alaska and eastern Russia, but only four non-telemetry aided recoveries have been recorded to date. For these reasons, use of DeMaster et al.'s (1980) multiple-year mark-recapture technique or recent modifications do not appear to be feasible for estimating the size of the polar bear population in the Chukchi-Bering Sea.

Taylor and Lee (1994) have considered using tetracycline as a biomarker that can be detected in the teeth of harvested polar bears. This multiple-year mark-recapture methodology has potential for use in estimating population size for polar bears in western Alaska if large numbers of individuals can be marked during periodic surveys and if sufficient numbers of marked individuals are identified in the harvest.

Aerial Survey Procedures

Polar bears occur in low densities across extensive areas of polar ice (Amstrup and DeMaster 1988), and this fact leads to difficulty in design of aerial surveys. Larsen (1972) used ship- and aircraft-based surveys of polar bears in Svalbard, Norway, but indicated both yielded inaccurate results. Strip transects were used to assess population size for polar bears in Alaska (Tovey and Scott 1957, Scott et al. 1959), however, large sample sizes were needed to obtain sufficiently narrow confidence limits on the estimate (Eberhardt 1978). Recently, Wiig and Bakken (1990) used aerial strip surveys of polar bears in portions of the Barents Sea, but did not expand their density estimates to the entire Barents Sea due to low confidence in the sample survey design.

Gilbert (1976) considered several single-season aerial procedures (including closed population mark-resight models) for estimation of population size in a 68- by 68-mile study area (110 by 110 km) off the north coast of Alaska. Closed population models assume negligible ingress, and that marked and unmarked bears have the same rates of egress during the sample period, but Gilbert found that movement of bears through the area was an important factor and suggested an open population model would be more appropriate. However, Garner et al. (1992b) indicate that open population models can be biased if movements result in bears leaving and returning to an area. Recent data from satellite instrumented polar bears indicate that extensive movements are common throughout most of the year, especially in the Bering and Chukchi seas (Garner et al. 1990, Garner and Knick 1991, Garner et al. 1994).

Although design and execution are difficult as indicated above, the potential methodologies for estimation of polar bear numbers in western Alaska seem to be in the area of aerial surveys; perhaps combined with short-term intense mark-resight methods applied over relatively small strata. Strip survey and line transect methodologies were summarized in landmark publications about 1980 (Gates 1979, Burnham et al. 1980, Seber 1982). Since 1980, advances have been made in refinement of the line transect technique (Buckland et al. 1993) and in improvements in double sampling procedures for strip surveys (Graham and Bell 1989, Crete et al. 1991).

Double counting procedures for correction of the proportion of polar bears missed during aerial strip surveys require independent counts made by tandem observers of polar bear groups detected within the same strip. Data are recorded to determine groups detected by both observers, groups detected by observer 1 and not by observer 2 and groups detected by observer 2 and not by observer 1 (Crete et al. 1991). Correction for the proportion of groups missed is made by an adaptation of the Lincoln-Petersen estimate (Seber 1982: 59). Garner et al. (1992b) note that the standard confidence intervals for the Lincoln-Petersen estimate sometimes exhibit poor performance and they developed an alternative procedure. The double counting procedure adapted to survey of polar bear by Crete et al. (1991) does not consider the influence of group size on detection probability.

Group size typically has a strong influence on the probability that both observers will miss a given group, prompting some researchers to stratify their data on group size and estimate density of groups within each size category (e.g., Cook and Jacobson

1979). Also, if the probability of detection for a group of bears depends on the size of the group, then there is danger of overestimating the mean group size because large groups are detected more often than small groups at the same perpendicular distance from the transect line.

Line transect sampling refers to those surveys where perpendicular distances to detected groups are recorded and where an assumed model is fitted to the probability of detection as a function of the perpendicular distance from the flight line. Given an estimate of the mean probability of detection of a group, the observed density of polar bears is corrected for the proportion of the population which was missed. One of the primary new developments in line transect methodology is development of procedures for estimation of the proportion of individuals missed on the transect line (g(0) in the notation of line transect sampling, Buckland et al. 1993). Data collected are the same as in the double counting method discussed above which implies that both methodologies can be used simultaneously in future polar bear surveys. In fact, relationships exist between estimates of g(0) and the two-sample mark-recapture Lincoln-Petersen estimate. The methodology has been used with apparent success in estimation of abundance of cetaceans (e.g., Butterworth and Borchers 1988) and must be considered in planning of any future aerial surveys for polar bear.

The most straightforward procedure for correction of group size-biased line transect data is to truncate groups that are detected far from the line to eliminate group detection bias. However, in polar bear surveys, minimal numbers of sightings are expected and truncation of groups to eliminate group size-bias seems wasteful of data. Ad hoc recommendations exist for dealing with group size-bias in line transect surveys (Buckland et al. 1993). Drummer and McDonald (1987) treat the size of the group as a covariate and incorporate size as a covariate in the model for probability of detection of a group. Quang (1991) has extended the method to allow for nonparametric estimation of size-biased line transect data.

Single-season Lincoln-Petersen Index in a Stratified Study Region

Mark-recapture closed population model estimates may be applicable in surveys conducted over a short time period of a single season with relatively small strata if excessive movement does not occur during the term. A primary advantage of the mark-recapture approach relative to aerial surveys for estimation of polar bear abundance is that information is collected on other population parameters which may be as important or more important than knowledge of the number of animals present.

A Lincoln-Petersen type estimator of population size can be utilized. The following assumptions are required: (1) the population is closed (there are no immigrants and no deaths or emigrants); (2) marked individuals always are recognized by the observer and no marks are lost; and (3) all members of the population have an equal probability of being captured. Robson and Regier (1964) provide recommendations concerning sample size requirements and in Garner et al. (1992b) recommendations are given for computing confidence intervals based on the Lincoln-Petersen method.

Consider a survey in which the area is stratified into several strata and a Lincoln-Petersen type estimate is planned for each strata. A typical problem with capture-recapture statistics applied to a sparse population of animals is that few animals are marked in each subregion and still fewer marked animals are captured in the recapture phase of the study. This results in high variance of estimated abundance. One adjustment to improve the estimate of abundance in each stratum is to design the recapture phase so that it is reasonable to assume that the probability of resighting a group is the same within each stratum. For example, one might apply effort for resighting polar bar in proportion to the area of each stratum. If M_i denotes the number of marked groups in the ith region, n_i denotes the number of groups resighted in the recapture phase and m_i denotes the number of marked animals resighted, then the probability of detection of a group could be estimated by

$$P = \sum m_i / \sum M_i.$$

Give P, the abundance in the ith stratum would be estimated by

$$\hat{N}_i = n_i / P.$$

Adaptive Aerial Survey

Adaptive sampling is a recently developed procedure to concentrate additional sampling effort in potential high-density areas (Thompson 1990, 1991). Computer simulations show that dramatic improvements in precision of estimates can be achieved using adaptive sampling relative to non-adaptive procedures for some rare clustered populations. There is evidence in the polar bear literature that the bears tend to have higher densities in certain areas of habitat. However, prediction of the location of these concentration areas is difficult because of changing sea-ice conditions. An additional benefit of the procedure is information on the size and shape of networks of units which satisfy the criteria for adaptive sampling. A primary disadvantage of the procedure is that it is not possible to know the exact sampling effort which will be required before a study begins.

Joint U.S./Russian Chukchi Sea Polar Bear Survey

A joint U.S./Russian survey of polar bears in the Chukchi Sea has been proposed and has agreement in principal between U.S. and Russian scientists (Garner et al. 1992a). The survey would be conducted in the ice-edge region of consolidated and unconsolidated ice in the northern Chukchi Sea during minimum ice cover in early autumn 1995 or 1996. The southern boundary of the survey area will be the interface between open water and unconsolidated ice; consolidated permanent pack ice will constitute the study area's northern boundary. The anticipated method of operation is a ship-based helicopter aerial survey along the ice edge. Survey approach will involve a sequential sampling of approximately 115- by 68-mile (185 by 110 km) strata (primary units) along the ice edge (Garner et al. 1992a).

Pilot Polar Bear Survey: Beaufort Sea 1994

Prior to finalization of the survey protocol for the survey of the Chukchi Sea polar bear population, a pilot study of potentially applicable methodologies will be tested in the Beaufort Sea during spring 1994 (Garner et al. 1992a). This study simultaneously will evaluate field procedures for collecting data from: (1) standard aerial line transect and strip surveys using independent tandem observers (fixed-wing aircraft), (2) single-season mark-resight methodology, (3) tetracycline marking for multiyear mark-recapture methodology, and (4) aerial line transect and strip surveys using independent tandem observers and adaptive sampling (helicopter).

The pilot study will survey one stratum and an expanded area block (230 by 115 mile; 370 by 185 km) containing the stratum centered north of the Prudhoe Bay, Alaska airport. All survey methods used will follow the anticipated protocol for the 1995 Chukchi Sea Survey as closely as possible, with the exception that a standard aerial transect and strip survey will be conducted in the expanded area block using a fixed-wing aircraft. This standard aerial method from a fixed-wing aircraft will be used exclusively during the 1994 Pilot Study to study movement of marked individuals and to determine the feasibility of this method for use during the autumn season if logistics of a ship-based survey in the Chukchi Sea prove impossible (Garner et al. 1992a).

Standard aerial line transect survey lines within the sample stratum will be approximately 68 miles (110 km) long and require approximately 30 minutes flying time. A maximum of 8 hours of flying time is expected each day. With a total of 20 hours of survey time per stratum, a total of 40 transects (2,762 miles; 4,445 km) can be flow during a three-day line transect survey period. Encounter rate of polar bears during the aerial resighting phase, based on unpublished data from the Russian portion of fixed-wing aircraft aerial surveys of walrus along the ice edge (Gilbert et al. 1992), is expected to approach 1 bear/174 miles (280 km) of survey line. Therefore, a total of 16 bears may be sighted during the standard aerial transect surveys within a stratum. The number of sightings of polar bear in any one stratum may be marginal for estimating correction factors for density, however, in the planned 1995 or 1996 Chukchi Sea survey, data may be pooled across strata. Also, adaptive sampling may provide an advantage in estimation of visibility correction factors if the encounter rate is increased.

The pilot survey will occur in two phases, with phase 1 activities occurring during the first thee days (Garner et al. 1992a). Objectives for phase 1 are to: (1) mark bears with dye spots for a single-season mark-resight estimates of population density during phase 2, (2) obtain information for stratifying the primary sample unit into high- and low-density polar bear areas for phase 2 resighting/line transect/strip surveys, (3) obtain supplemental sightings using line transect and double counting methodologies to increase the sample size for estimating visibility correction factors in phase 2, and (4) conduct standard fixed-wing line transect and strip surveys in the primary unit. Primary objectives for phase 2 are to: (1) obtain estimates of polar bear density using aerial line transect sampling with adjustments for visibility bias on the transect line, (2) obtain estimates of polar bear density using aerial strip sampling with adjustments for visibility bias using double counting methodology, (3) evaluate the feasibility of using the adaptive sampling strategy developed by Thompson (1990, 1991), and (4) study movements of marked polar bears during the survey period.

Data analysis will follow standard line transect methodology and mark-resight methodology (Burnham et al. 1980, Seber 1982, Garner et al. 1992a, Buckland et al. 1993) Additionally, during the fixed-wing aerial transect survey there will be an adjustment for proportion of bears missed on the transect line using two independent observers on the same side of the aircraft following methods in Buckland et al. (1993). Results will be compared with the double sampling methodology of Crete et al. (1991) because data are available for both analyses.

Overall detection probability of a polar bear group, P, and mean group size ($\bar{x}$)

will be estimated from all phase 2 line transect data using standard line transect theory applied to the sightings of polar bear groups (Burnham et al. 1980, Drummer and McDonald 1987, Buckland et al. 1993). During the planned 1995 survey of the Chukchi Sea, it may be necessary to pool data across primary units and from both phases of the survey to estimate the overall detection probability and mean group size; however, if sufficient sightings are available from smaller geographic areas, then separate estimates of these parameters will be given (Garner et al. 1992a).

Because of limited visibility caused by inclement weather, the altitude of the survey aircraft may vary, and the maximum half-width of the line transects and strip surveys will vary accordingly. At least two standardized survey altitudes will be selected and visibility correction factors will be developed for each to address effects of inclement weather on survey parameters. Further details on data analysis and formulas are given in the appendices of Garner et al. (1992a).

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Adapting New Techniques to Population Management: Wyoming's Pronghorn Experience

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Introduction

One of the biggest challenges facing natural resource agencies is making defensible, objective decisions in the presence of uncertainty. Population sizes are rarely known for most wildlife populations although many management prescriptions are designed to influence populations levels. Wildlife management may be severely hampered by the lack of reliable estimates (Gasaway et al. 1986). Objective estimates with measures of reliability offer a number of benefits to wildlife managers including the following: (1) confidence intervals help judge risk in management alternatives, (2) population estimates with defensible confidence intervals can be used to realistically determine if population objectives are met, and (3) confidence intervals help wildlife managers perceive the relative reliability of population estimates (Czaplewski 1986). Resource managers must learn to accept a substantial amount of uncertainty (Walters 1986).

It is highly desirable for agencies to adopt cost-efficient techniques that accurately assess the status of populations and quantify the reliability of these estimates. Several major advances in estimating population parameters have been made since the late 1970s (e.g., Otis et al. 1978, Burnham et al. 1980, Lebreton et al. 1992). The more reliable population estimation techniques often remain within the research domain. Wildlife agencies have been relatively slow to adopt new techniques and even slower at replacing outdated or poor procedures.

The purpose of this paper is to illustrate how wildlife management agencies can improve management of a particular species by adopting a more scientific and efficient method for population estimation. I describe the implementation of aerial line transect sampling to estimate pronghorn (*Antilocapra americana*) populations in Wyoming and the impacts this has had on the management of this species. The reader may refer to Burnham et al. (1980) and Buckland et al. (1993) for a comprehensive discussion of the theory underlying line transect sampling, and White et al. (1989), Johnson et al. (1991) and Buckland et al. (1993) regarding the application and testing of aerial line transect sampling of terrestrial wildlife. The conclusions and interpretations presented in this paper do not necessarily represent an official agency position. I assume sole responsibility for the content of this paper.

Pronghorn Population Management in Wyoming

In 1974, the Wyoming Game and Fish Department (WGFD) initiated strategic planning for the management of wildlife (Crowe 1983). This process included the definition of individual herds (also called "herd units" or "data analysis units"). Herds are assumed to be relatively discrete populations (Pojar 1981). Currently, the WGFD manages 51 pronghorn herds (Figure 1). Herds include one or more of

Table 1. Pronghorn herds in Wyoming.

Herd number	Herd name	Hunt areas	Total occupied habitat miles^2	(km^2)
202	Crystal Creek	79	492.1	(1,274.5)
203	Copper Mountain	76,114,115	1,546.2	(4,004.7)
204	Fifteenmile	77,83,110	2,297.5	(5,950.5)
205	Carter Mountain	78,81,82	1,087.0	(2,815.3)
207	Badger Basin	80	608.4	(1,575.8)
308	Clearmont	15	1,119.7	(2,900.0)
309	Pumpkin Buttes	23	1,506.8	(3,902.6)
310	Upper Powder River	20	393.3	(1,020.2)
316	Highlight	24	1,016.3	(2,632.2)
318	Crazy Woman	23,113	1,157.8	(2,998.7)
339	North Black Hills	1–3,18,19	2,080.4	(5,388.2)
351	Gillette	17	1,362.4	(3,528.6)
352	Middle Fork	21	534.9	(1,385.4)
353	Ucross	10,16	834.5	(2,161.4)
354	Buffalo	102	146.1	(378.4)
355	Beckton	109	81.8	(211.8)
401	Sublette	85–92,96,107	6,695.7	(17,341.9)
411	Unita-Cedar Mountain	95,99	1,859.0	(4,814.9)
412	South Rock Springs	59,112	1,212.9	(3,141.4)
414	Bitter Creek	54,57,58	2,914.7	(7,549.1)
417	West Green River	93	1,398.4	(3,621.9)
419	Carter Lease	94,98,100	1,979.2	(5,126.1)
438	Baggs	53,55	1,152.8	(2,985.8)
520	Chalk Bluffs	111	344.7	(892.8)
521	Hawk Springs	34–36	2,691.4	(6,970.7)
522	Meadowdale	11–14	1,722.6	(4,461.5)
523	Iron Mountain	38–40,104	2,280.1	(5,905.5)
524	Dwyer	103	752.3	(1,948.5)
525	Medicine Bow	41,42,46–48	3,201.7	(8,292.4)
526	Cooper Lake	43	458.2	(1,186.7)
527	Centennial	37,44,45	1,153.1	(2,986.5)
528	Elk Mountain	49,50	607.2	(1,572.6)
529	Big Creek	51	206.1	(533.8)
615	Red Desert	60,61,64	3,358.2	(8,697.7)
630	Iron Springs	52,56,108	1,100.4	(2,850.0)
631	Wind River	84	152.4	(394.7)
632	Fremont	65–67,74	2,310.0	(5,982.9)
633	Sweetwater	68,69,106	1,590.2	(4,118.6)
634	Badwater	75	1,041.3	(2,697.0)
635	Project	97	155.4	(402.5)
636	North Ferris	63	474.0	(1,227.7)
637	South Ferris	62	922.4	(2,389.0)
740	South Black Hills	4,5	930.1	(2,339.0)
741	Thunder Basin	7	1,037.3	(2,686.1)
742	Lance Creek	6,8,9,27	4,329.4	(11,213.2)
743	LaPrele	30	568.0	(1,471.1)
744	Bates Hole-Hat Six	31–33	830.1	(2,150.0)
745	Rattlesnake	70–72	1,025.3	(2,655.5)

Table 1. *Continued.*

Herd number	Herd name	Hunt areas	Total occupied habitat	
			miles^2	(km^2)
746	North Natrona	73	1,349.6	(3,495.5)
747	Ormsby	25	729.3	(1,888.9)
748	Bear Creek/Sage Creek	26,28	1,856.1	(4,807.3)
Total			70,628.4	(182,927.6)

Wyoming's 113 pronghorn hunt areas (Table 1). Pronghorn occupy a total of 70,628 square miles (182,928 km^2) or about 72 percent of the state's land area. Occupied habitats for individual herds range from 82 to 6,696 square miles (112–17,342 km^2) as shown in Table 1.

The WGFD manages pronghorn for publicly approved population objectives. Objectives are set for postseason (wintering) populations for each herd. Initial objectives were based on public comments about desired population levels relative to initial estimates. Once objectives are established, harvests and other management practices are adjusted to direct herds toward those levels. Objectives periodically are revised, based on changes in public desires, new information about population levels and other factors. The current population objective for the state is 395,260 wintering

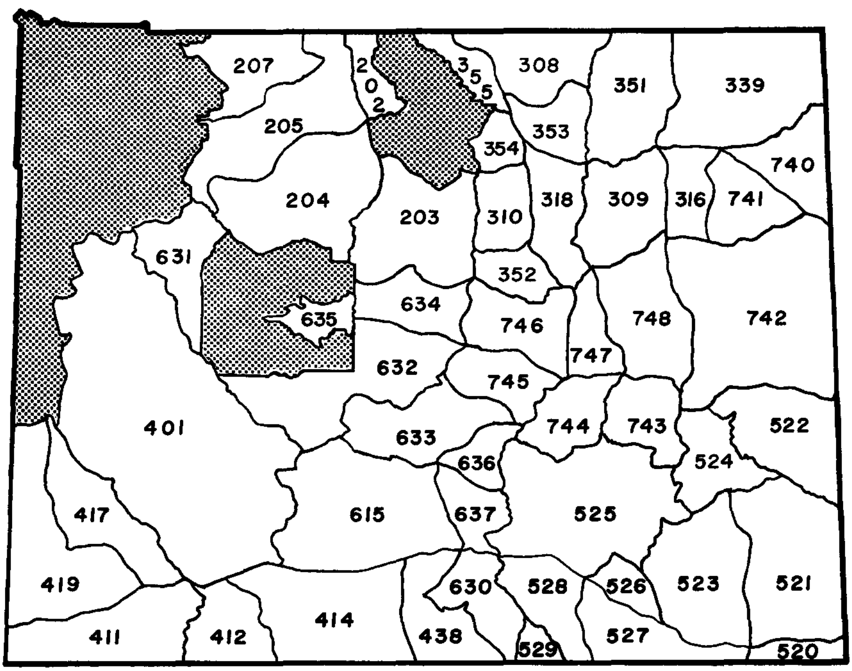

Figure 1. Pronghorn herds managed by the Wyoming Game and Fish Department. Shaded areas include vast regions of unoccupied habitat and pronghorn herds under other jurisdictions.

Table 2. Status of pronghorn populations and line transect (LT) surveys in Wyoming.

Herd number	Population objective[a]	1992 estimate[a]	Number of LT surveys by biological year (June 1–May 31)[b]					
			1988	1989	1990	1991	1992	Total
202	160	96	0	0	0	0	0	0
203	2,750	3,577	0	0	0	0	0	0
204	4,600	7,919	0	0	0	0	0	0
205	7,000	7,100	0	0	0	0	1	1
207	650	1,200	0	0	0	0	0	0
308	3,000	4,682	0	0	1	1	0	2
309	18,000	27,514	0	1	1	0	0	2
310	3,000	6,744	0	1	1	0	1	3
316	11,000	12,520	0	1	1	0	0	2
318	7,000	9,479	0	1	0	1	1	3
339	14,000	14,690	0	0	0	1	0	1
351	11,000	16,314	1	0	1	1	0	3
352	2,100	4,085	0	0	1	0	0	1
353	2,500	5,235	0	1	1	0	0	2
354	1,000	1,963	0	1	0	1	0	2
355	100	383	0	0	0	0	0	0
401	30,000	32,811	0	1	0	1	0	2
411	7,000	9,718	0	0	2	0	3	5
412	4,000	4,984	1	0	1	0	1	3
414	11,000	22,000	0	0	1	1	1	3
417	3,000	10,731	1	0	1	1	1	4
419	3,600	8,729	0	1	1	1	1	4
438	7,200	11,000	0	0	0	2	1	3
520	450	1,268	0	0	1	0	0	1
521	5,000	7,025	0	0	0	0	1	1
522	5,000	10,368	0	0	1	0	0	1
523	8,000	23,997	0	0	0	1	0	1
524	2,500	3,932	0	0	0	0	1	1
525	45,000	37,152	0	0	0	0	1	1
526	3,000	5,603	0	0	0	1	1	2
527	6,000	15,229	0	0	0	1	1	2
528	5,000	7,150	0	0	0	1	0	1
529	600	390	0	0	0	0	0	0
615	12,000	12,800	0	1	0	0	1	2
630	12,000	11,667	0	0	1	0	0	1
631	300	436	0	0	0	0	0	0
632	10,000	17,230	0	3	1	0	2	6
633	10,000	9,177	1	1	0	1	1	4
634	3,000	3,319	0	0	0	1	0	1
635	250	548	0	0	0	0	0	0
636	5,000	3,970	1	0	0	0	0	1
637	6,500	6,947	0	0	0	0	0	0
740	3,000	4,067	0	1	0	0	1	2
741	8,000	11,714	0	0	1	1	0	2
742	27,000	23,566	0	0	0	1	0	1
743	3,500	4,845	0	0	0	1	0	1
744	11,500	7,537	0	1	1	1	0	3

Table 2. *Continued.*

Herd number	Population objective[a]	1992 estimate[a]	Number of LT surveys by biological year (June 1–May 31)[b] 1988	1989	1990	1991	1992	Total
745	12,000	12,332	1	1	1	1	0	4
746	9,000	14,107	0	1	0	1	1	3
747	8,000	12,860	0	1	1	1	1	4
748	20,000	22,947	0	1	0	0	1	2
Total	395,260	515,477	6	19	21	24	24	94

[a]Data from Wyoming Game and Fish Department (1993).
[b]Data from Wyoming Game and Fish Department (1988–1992) and Wyoming Game and Fish Department files.

pronghorn (Wyoming Game and Fish Department 1993). Objectives for individual herds range from 100 to 45,000 animals (Table 2).

Population Monitoring

Monitoring population levels is extremely important in order to determine if population objectives are being met. Three techniques commonly are used by the WGFD to estimate pronghorn populations: population models, aerial trend counts and aerial line transects. Pronghorn are ideally suited to aerial surveys because of their wide distribution and the relatively open habitats they occupy. The statewide population was estimated at 515,477 pronghorn in the 1992 postseason population (Wyoming Game and Fish Department 1993). This estimate is about 30 percent above the statewide objective. Individual herd sizes ranged from 96 to 37,152 in 1992 (Table 2). The statewide population since has declined due to high mortality resulting from the 1992–1993 winter, increased harvests and other factors.

Population Models

The WGFD has been using simulation models to estimate pronghorn populations since 1976 (Strickland 1979). The WGFD uses POP–II, an interactive, deterministic computer program (Bartholow 1990a). This model incorporates data routinely collected by wildlife management agencies. The model provides estimates of population sizes given harvest, observed population ratios and assumptions about natural mortality and other characteristics. POP–II simulations help determine progress toward objectives and evaluate alternative harvest strategies (Gasson and Wollrab 1986).

Population models are calibrated by aligning simulated values with independent estimates. In Wyoming, trend counts and, more recently, line transects have been used for validation and alignment. Modeling helps to critically analyze existing data, survey methods and the sensitivity of input (Strickland 1979, Pojar 1979, Gasson and Wollrab 1986, Bartholow 1990a). Awareness of data limitations has prompted agencies to evaluate and obtain more reliable data for model input, such as herd composition (Czaplewski et al. 1983, Bowden et al. 1984, McCullough 1993, Woolley and Lindzey 1994). Model limitations also need to be considered (Conroy 1993, Czaplewski 1986).

Models should be viewed as testable hypotheses (Pojar 1981, Conroy 1993). Predictions should be tested against independent data to evaluate their reliability (Pojar

1981). Unfortunately, most models have low statistical power to reject the null hypothesis of no difference between model predictions and observations (Conroy 1993). Some procedures are available to address uncertainty in population modeling. Czaplewski (1986) applied the Kalman filter to some of Wyoming's pronghorn populations to obtain confidence intervals for population estimates. Bartholow (1990b) developed POP–III as a companion to POP–II to help address the problem of future uncertainty using a Monte Carlo technique. These options have not received much attention in pronghorn management. Lack of reliable estimates of population sizes has been the weakest link in the modeling process (Pojar 1979). In Wyoming, underestimation contributed to pronghorn populations exceeding objectives (Czaplewski 1986). The inability to determine the rate of population decline also may have serious ramifications for managers (Gasaway et al. 1986).

Trend Counts

Trend counts have been used on pronghorn herds in Wyoming to determine relative population changes and also to base estimates of population sizes (Wyoming Game and Fish Department 1982). Parallel strips are flown throughout a herd to obtain complete coverage. Standardized procedures and acceptable conditions have been established in order to use these surveys as indices (Wyoming Game and Fish Department 1982). Most herds are surveyed about every three years because of budget and time constraints. Trend counts are used as independent estimates of herd size for evaluating the adequacy of simulations (Czaplewski 1986). Models are aligned on trend counts by subjectively assuming that some proportion of the herds are missed during surveys. Accuracy of trend counts may range from 50 to 80 percent (Wyoming Game and Fish Department 1982, Firchow et al. 1990, Johnson et al. 1991). One common problem of trend counts is that portions of the area to be surveyed are not counted (Pojar 1981). Reexamination of results for some Wyoming herds indicated the areas to be covered could not have been surveyed in the time taken to complete the trend counts (Guenzel 1991a and 1991b). The utility of trend counts is limited because unknown proportions of populations are missed and measures of reliability are unavailable. Trend counts still are conducted on a few herds.

Line Transects

Aerial line transect sampling was initially tested for estimating pronghorn populations in Wyoming in 1987 and 1988 (Johnson et al. 1991). Since then, the technique has been refined and incorporated into pronghorn management in Wyoming and elsewhere (e.g., Killaby et al. 1992). The technique does not require marking animals or double sampling. Line transect sampling offers several advantages over trend counts. The technique allows animals that are away from the line to be missed (Buckland et al. 1993). Population estimates are adjusted for missing animals that should have been counted (i.e., visibility bias) as a function of perpendicular distance from the line (Burnham et al. 1980, Buckland et al. 1993). Line transects are more robust to varying survey conditions than trend counts (Buckland et al. 1993). Complete coverage of the area is not required. Line transect sampling provides measures of the precision of population estimates on which to assess reliability.

Quality control is crucial. Training observers is extremely important to the successful implementation of line transect sampling (Buckland et al. 1993). A manual

(Johnson and Lindzey 1990) and video were prepared to help train WGFD personnel. Measurement error is controlled as much as possible during surveys using window and wing strut markers. A radar altimeter and GPS-aligned LORAN-C are linked to an onboard computer to help the pilot fly transects and to record height above ground and location for each observation. Heights are used later to adjust perpendicular distances for variation in altitude. Typically, aerial line transect surveys have been designed using systematically spaced transects throughout the occupied habitat with a random start (Johnson and Lindzey 1990). Designs for some herds subsequently have been modified to improve precision and accommodate changes in expected cluster sizes (e.g., Christiansen 1992a), or to address other management questions (Christiansen 1992b, Lockwood 1992).

Population estimates with confidence intervals are obtained using the program DISTANCE (Laake et al. 1993, Buckland et al. 1993). The program has the capability to correct estimates for cluster-size bias, provide bootstrapped confidence intervals, and allow for stratification and more complicated survey designs.

Status of line transect sampling. Since 1988, 42 of Wyoming's 51 pronghorn herds have been surveyed using the aerial line transect technique (Table 2). This represents over 90 percent of the total habitat occupied by pronghorn in the state and about 96 percent of the statewide population. Most of the pronghorn herds that have not been surveyed using aerial line transects are small and have relatively low densities (cf., tables 1 and 2). The line transect technique has enabled individual herds to be sampled more frequently than had occurred using trend counts. Ninety-four line transect surveys were conducted during the 1988-1992 biological years (June 1–May 31) with individual herds being surveyed up to six times over this period (Table 2). Nineteen herds have been surveyed in consecutive years with six herds being surveyed for at least three consecutive years.

Cost savings. Implementation of line transect sampling has resulted in tremendous savings of time and money over trend counts, depending on the sampling intensity of surveys. Line transect surveys can be completed in as little as 20 percent of the time needed to conduct trend counts (Johnson et al. 1991), resulting in savings in manpower and money (30–50 percent of costs for trend counts). Table 3 compares time and costs (in 1993 U.S. dollars) between trend counts and line transects for selected herds. Small, low-density herds require relatively higher sampling intensity compared to larger herds. One trend count took more than a month to complete (Rudd 1988), whereas a line transect survey of the same herd was completed in two days (Thomas 1992). These savings allow more flight time for other wildlife surveys. Such savings become more important when agencies face reduced budgets (Killaby et al. 1992).

Reliability. For most pronghorn herds, the true population size is unknown. Wildlife managers are faced with the dilemma of determining which, if any, of the available techniques is reliable enough for management. Initial estimates of pronghorn herd sizes using line transect surveys often were much higher than estimates based on simulation modeling or trend counts. Although these results would be expected because of the statistical correction for sightability, the magnitude of these differences was greater than some biologists expected. Many biologists were skeptical of the

Table 3. Comparison of survey time and costs between trend count and line transect surveys for selected pronghorn herds in Wyoming.

	Trend count			Line transect					
							Percentage of trend count		
Herd number	Time (hours)	Cost/ hour[a]	Survey cost[a]	Time (hours)	Cost/ hour[a]	Survey cost[a]	Time	Cost	Source
525	33.9	$150[d]	$5,085	8.5	$170[c]	$1,445	25.1	25.1	Rudd 1988 WGFD file data
615	51.1	$150[d]	$7,665	8.6	$170[c]	$1,462	16.8	19.1	Hiatt 1992
742	26.3	$ 90[b]	$2,367	4.8	$170[c]	$ 816	18.2	34.5	Lanka 1990
745	29.5	$ 90[b]	$2,655	5.5	$170[c]	$ 935	18.6	35.2	Guenzel 1987 Thiele 1990

[a]Standardized for 1993 cost in U.S. dollars.
[b]Two-place aircraft (e.g., Piper Supercub, Bellanca Scout, etc.).
[c]Maule M-5 equipped for line transect surveys with onboard computer, GPS, LORAN and radar altimeter).
[d]Four-place aircraft (e.g., Cessna 182).

initial line transect estimates. The situation was similar to "sticker shock"—it was hard for personnel to relate to densities that were higher than they were used to. This forced biologists to critically examine line transect estimates.

Population estimates from models aligned on trend counts were difficult to defend because these estimates were based on subjectively determined accuracy with no measure of reliability. Therefore, biologists had to evaluate the accuracy of the line transect estimates based on the quality of the estimates themselves.

Line transect theory is well founded (Burnham et al. 1980, Buckland et al. 1993). Johnson et al. (1991) demonstrated that assumptions could be met adequately during aerial surveys. Survey procedures help to meet critical assumptions such as seeing all animals on the line. Line transect estimates also control for the effects of cluster-size bias and variation in altitude. Estimates from line transect surveys generally are repeatable (Emmerich 1990, Johnson et al. 1991). However, estimates have been more variable in some herds (e.g., Christiansen 1992b). Movements or other real phenomena may explain some of those variations.

Coefficients of variation for herd sizes estimated by aerial line transects typically range from 15 to 25 percent using present survey designs. However, these have exceeded 40 percent for a few surveys. Most of the variation in population size estimates is attributable to variation in encounter rates (the mean number of pronghorn clusters observed per unit length of transect).

In a number of herds, 95-percent confidence intervals for the higher line transect estimates did not capture the estimated population sizes using POP–II. The first line transect estimate for the Iron Mountain Herd was 21,125 with a confidence interval of 14,859 to 30,052, compared with the POP–II estimate of 7,352 pronghorn (Guenzel 1991a). Similar results were obtained for the Centennial Herd where the estimate of 13,653 with confidence limits of 8,794 to 21,196 did not include the POP–II estimate of 7,585 animals (Guenzel 1991b) and the Cooper Lake Herd where the estimate of 5,143 with confidence intervals of 3,481 to 7,898 exceeded the POP–II estimate of 2,486 (Guenzel 1991c). These patterns are typical of results in many herds and support observations that population models aligned with trend counts underestimated population sizes for those herds. This is not surprising given the previously stated concerns about trend counts and the reliability of composition ratios which are used in models (Bowden et al. 1984, McCullough 1993, Woolley and Lindzey 1994).

Under current survey designs, statistical power of line transect estimates may be relatively low for detecting subtle changes in population sizes in some herds. However, power seems adequate to detect larger effects such as high mortality from severe winters. The line transect estimates listed above for the Centennial and Cooper Lake herds at the end of the 1991 biological year were compared with line transect estimates following the severe 1992–1993 winter. The Centennial Herd had declined to 9,262 pronghorn with 95-percent confidence limits of 6,355 to 12,790 (Guenzel 1992a). The Cooper Lake Herd declined to 2,363 pronghorn with a confidence interval of 1,638 to 3,409 (Guenzel 1992b). Statistical power for these population changes was calculated using the PASS program (Hintze 1991) for a two-sample t-test with unequal variances (two-tailed test; $\alpha = 0.05$). In the Centennial Herd, the power to detect a 5-percent change from the 1992 population was 0.851, whereas the power to detect a 10-percent change was 0.999. In the Cooper Lake Herd, power to detect 5-, 10- and 15-percent changes from the 1992 population was 0.182, 0.552 and 0.880, respectively. Improved survey designs may help improve long-term monitoring of pronghorn populations in Wyoming.

Impact on pronghorn management. Aerial line transect sampling has profoundly influenced pronghorn management in Wyoming. The higher population estimates forced biologists to critically examine line transect surveys, and then to critically examine data quality and reliability for population models and other management criteria. In retrospect, it appears that past harvests may not have had as much influence on population growth as predicted. Biologists have been reconsidering assumptions about the dynamics of many herds (e.g., population regulation, variation in natural mortality, population closure, etc.) and data adequacy. Additional research has been initiated to address some of these questions (Christiansen 1992b, Woolley and Lindzey 1994). A side benefit of implementing line transect surveys has been increased safety. Personnel spend less time conducting low-level surveys.

Line transect estimates are used to realign and validate population models. Confidence intervals help biologists to consider the reliability of estimates in management decisions. Population objectives are being reviewed for a number of herds. In some populations, harvests were increased. Reduced survey costs have allowed additional surveys to be conducted. Line transects have been used to help quantify effects of severe winters on herds (Guenzel 1992a, 1992b).

The use of aerial line transect sampling should contribute to greater understanding of the dynamics of pronghorn populations. This, in turn, should help improve predictions from population models. Predictions can be tested through management actions to help further refine models and understanding, resulting in an adaptive management program for pronghorn in Wyoming (Walters 1986, Conroy 1993).

Problems encountered with line transect surveys. Aerial line transect surveys should not be viewed as a final product but as an ongoing experiment. The technique is merely the leading candidate among existing methods. It appears to offer several advantages over other techniques. The limitations of aerial line transect surveys still are being evaluated under varying population densities and landscapes through routine management and additional testing. Some questions remain about the suitability of the technique in rough topography. One problem is reconciling differences among estimates using various techniques.

The need for continuing training, quality control and oversight cannot be overstated. At least some of the problems encountered with the use of line transect surveys have been due to incorrect application of the technique or errors during analysis. Manuals and other training aids need to be updated with current procedures. Some personnel still emphasize point estimates without adequately considering the reliability of these estimates. Precision of estimates for some herds is inadequate for management purposes, but improved survey design and higher sampling intensity can increase precision.

Conclusions

Wildlife management agencies can benefit greatly by adapting more reliable and efficient techniques to population management. The implementation of aerial line transect sampling for pronghorn management in Wyoming has forced the WGFD to critically evaluate existing data, management techniques and population models. Line transects provide wildlife managers with improved estimates. The quality and reliability of line transect estimates can be assessed objectively. In retrospect, trend counts and herd simulations appear to have underestimated populations. The adoption of line transect sampling has resulted in realignment of simulations, increased harvests and initiation of raising objectives in several herds. Aerial line transect surveys save a substantial amount of time and funding, enabling additional population monitoring within existing budgets.

The adoption of line transect surveys also has presented some problems. Line transect sampling requires increased training and quality control. The limitations of the line transect method still are being evaluated through routine management and additional testing. The implementation of line transect sampling is helping to direct Wyoming toward adaptive pronghorn management. Wyoming's line transect experiences demonstrate the practical advantages to wildlife management agencies of adopting more scientific population estimation procedures.

Acknowledgments

Several individuals deserve special thanks for their contributions to aerial line transect sampling in Wyoming. These include Bruce Johnson, Fred Lindzey, Fred and Linda Reed, Jeff Laake, David Anderson, John Emmerich and Tom Christiansen. Administrators and field personnel in the WGFD Wildlife Division deserve special acknowledgment for their support and cooperation in implementing line transect surveys. Pat Hnilicka provided some of the statewide data on pronghorn herds for this report. Finally, I thank Bill Hepworth, Fred Lindzey and David Otis for their helpful discussions and review of this manuscript.

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Detecting Pacific Walrus Population Trends with Aerial Surveys: A Review

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Introduction

Estimates of population size are needed for Pacific walruses (*Odobenus rosmarus divergens*) to set conservation policy in Russia and to comply with the U.S. Marine Mammal Protection Act requirement that populations be maintained within Optimum Sustainable Population (OSP) range, usually above some threshold number (U.S. Fish and Wildlife Service 1993). The direction of change in population size and a measure of the population's status relative to the current carrying capacity of its environment also are needed for reasonable management of a harvested population (Eberhardt and Siniff 1977, Eberhardt 1978, DeMaster 1984, Croxall 1989, Fay et al. 1989, Fowler and Siniff 1991).

Aerial surveys have been the primary method of estimating the size of the Pacific walrus population since 1958 (Buckley 1958, Fedoseev 1962). The only surveys over the entire range of the Pacific walrus are the cooperative efforts by the United States and the Soviet Union that have been conducted every five years since 1975. Several sources of bias and imprecision were apparent in the first aerial surveys and continued to be noted in subsequent surveys (Kenyon 1960, 1961, 1968, 1972, Estes 1974, Estes and Gilbert 1978, Johnson et al. 1982, Gilbert 1989, Gilbert et al. 1992). Lack of precision was noted on most surveys, but additional surveys were supported because they were believed to indicate trend and because no alternate method of assessing population size was available (Johnson et al. 1982, Gilbert 1986). Because the area to be surveyed is large and remote, logistics and costs have been significant factors in survey planning. Before committing resources for additional surveys, the effectiveness of aerial surveys to detect changes in the size of the walrus population should be considered carefully.

Our first objective was to evaluate the utility of data from past aerial surveys to detect a trend in the size of the Pacific walrus population. Our second objective was to estimate the number of surveys necessary to detect a trend with a given precision. Our final objective was to determine how the fraction of the population not visible to the surveyors might influence the accuracy and precision of the estimates.

Use of Previous Aerial Surveys to Detect a Trend

Before 1975, eleven aerial surveys were conducted over various parts of the range of Pacific walruses (Table 1) by Soviet or American researchers. Because these efforts

Table 1. Summary of aerial surveys for Pacific walruses, 1958–1990.

Year	Month	Region	Reference
1958	May	American waters	Buckley 1958
1958	Aug-Sept	Soviet waters	Fedoseev 1962
1960	Feb-Mar	American Bering	Kenyon 1960
1960	April	American Bering	Kenyon 1960
1960	Sept-Oct	Soviet Chukchi	Fedoseev 1962
1961	March	American Chukchi	Kenyon 1961
1964	Sept-Oct	Soviet Chukchi	Gol'tsev 1968
1968	April	American Bering	Kenyon 1968
1970	Sept-Oct	Soviet Chukchi	Gol'tsev 1972
1972	April	American Bering	Kenyon 1972
1974	Sept	American Chukchi	Estes 1974
1975	Sept-Oct	Soviet Chukchi	Gol'tsev 1976
1975	Sept-Oct	American Chukchi	Estes and Gilbert 1978
1976	April	American Bering	Braham et al. 1984
1980	Aug-Sept	American Chukchi	Wartzok and Ray 1980
1980	All year[a]	Bristol Bay	Fay and Lowry 1981
1980	Sept	American Chukchi	Johnson et al. 1982
1980	Sept-Oct	Soviet Chukchi	Fedoseev 1981
1985	Sept	American Chukchi	Gilbert 1986
1985	Sept	Soviet Chukchi	Fedoseev and Razlivalov 1986
1985	Sept	All Chukchi	Gilbert 1989
1987	Spring	Soviet Bering	Fedoseev et al. 1988
1987	March	Gulf of Anadyr	Mymrin et al. 1990
1989	June	Bering Strait	Gilbert unpubl. data
1990	Sept-Oct	All Chukchi	Gilbert et al. 1992

[a]April 1980–May 1981.

did not cover the entire range of the walruses, no population size or changes in population size could be extracted from the results.

In autumn 1975, the United Sates and the Soviet Union conducted the first cooperative range-wide survey for walruses (Estes and Gilbert 1978, Estes and Gol'tsev 1984). The effort was repeated in 1980 (Johnson et al. 1982), 1985 (Gilbert 1989) and 1990 (Gilbert et al. 1992). Each estimate of the total population size combined estimates of the number of walruses on (1) American terrestrial haulouts, (2) Soviet terrestrial haulouts, (3) sea ice in the American sector of the Chukchi Sea, and (4) sea ice in the Soviet sector of the Chukchi Sea (Figure 1).

With each cooperative survey, the procedures for surveying the American sea ice sectors were changed in an effort to increase the precision of the estimate; all procedures were statistically valid samples. For the 1975–1985 surveys, variance was calculated only for the American sector of the pack ice; in 1990, all ice sector data were analyzed together and a variance was calculated (Gilbert et al. 1992). At most of the terrestrial haulouts only one count was made, precluding any estimate of variance. The coefficient of variation (CV) for the ice portion of the cooperative surveys ranged from 0.23 to 1.39 and averaged 0.62 (Table 2). At land haulouts in Bristol Bay in 1987–1991, CVs averaged 0.6 for peak day counts (Hills 1992). Therefore, we used a CV of 0.6 to calculate confidence limits for the overall population estimates for the cooperative surveys.

Throughout these surveys, it was recognized that the fraction of the walrus population available to be counted varied because the number of animals hauling out on land or ice varied significantly from day to day (Estes and Gilbert 1978, Gilbert 1989, Gilbert et al. 1992). Those individuals in the water often were below the surface and

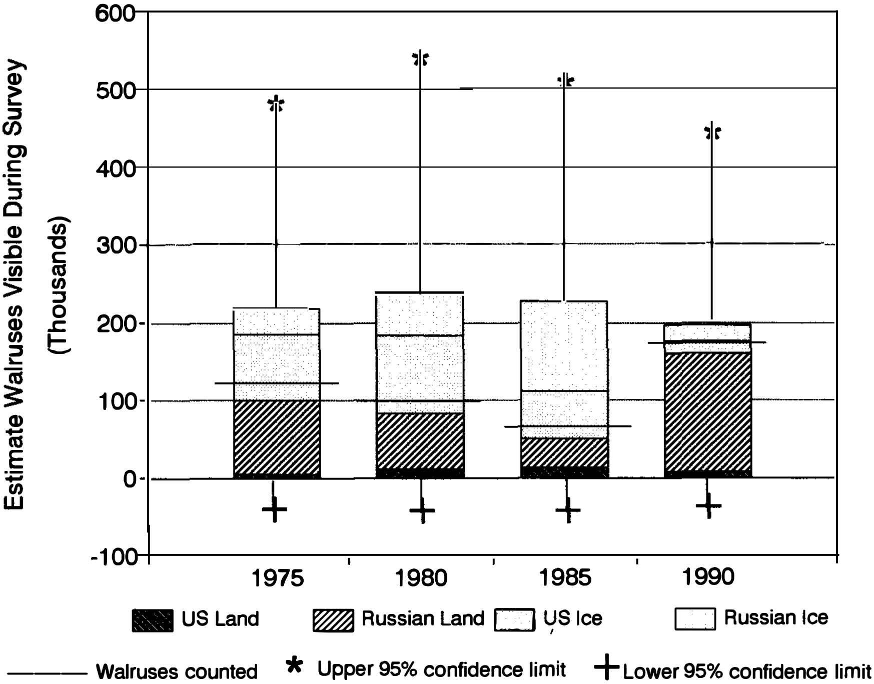

Figure 1. Estimates of the size of the Pacific walrus population from cooperative U.S./U.S.S.R. aerial surveys conducted in 1975–1990, showing the source of the four portions of the overall estimate, actual counts, upper and lower 95-percent confidence limits. The 1990 survey was analyzed cooperatively and ice portions were not divided into U.S. and U.S.S.R. portions. The portion of the 1990 bar labelled "U.S. ice" corresponds to all walruses seen on ice, the portion labelled "Russian ice" corresponds to walruses seen in open water. The CV used to calculate the confidence limits was 0.6.

less likely to be seen than those on land or sea ice. Sometimes efforts were made to repeat counts over several days and use the highest count for estimating numbers, other times an area was counted only once. None of the surveys were corrected for a "fraction not observable" nor did any variances consider this fraction.

The point estimates for each of these surveys (Figure 1) have been considered the best information available on the status of the Pacific walrus population. The question is whether the decline in the point estimates between 1975 and 1990 reflects a real decline in the population. We attempted to calculate a linear regression over population size estimates of Pacific walruses using summary data as described by Draper and Smith (1981). However, we could not estimate the linear regression of population size estimates among years using summary data because, although we could estimate the variance associated with each total population estimate assuming a standard CV, we did not have any means of selecting a justifiable sample size for each survey. However, the lower 95 percent confidence limit included zero in each case (Figure 1).

Information Needed to Detect a Trend

We then asked how many surveys at what precision would be necessary to detect any trend (two-tailed test) and a negative trend (one-tailed test) in the walrus popu-

Table 2. Coefficients of variation (CV) for selected aerial surveys for Pacific walruses, 1958–1990. Estimates and Cvs for surveys in Soviet waters were recalculated using Soviet data but strip transect methods (Gilbert unpublished data).

Year	Month	Region	Survey estimate	Coefficient of variation
1975	Sept-Oct	Soviet Chukchi	3,527	0.31
1975	Sept-Oct	American Chukchi	2,475–100,568	0.62 (0.26–0.99)
1976	April	American Bering	33,300–80,700	0.21
1980	Sept	American Chukchi	116,240	0.20–0.38
1980	Sept-Oct	Soviet Chukchi	68,250	0.26
1985	26 Sept	Soviet waters	1,693	0.48
	27 Sept	Soviet waters	17,691	0.23
	29 Sept	American waters	445	0.88
	30 Sept	American waters	60,818	0.62
1990	24 Sept	All Chukchi	256	0.48
	25 Sept	All Chukchi	1,639	0.81
	26 Sept	All Chukchi	48	1.39
	27 Sept	All Chukchi	3,352	0.64
	30 Sept	All Chukchi	402	1.16
	1 Oct	All Chukchi	3,603	0.58
	3 Oct	All Chukchi	7,189	1.20

lation. This is best evaluated using trend analysis (Peterman 1990, Gerrodette 1987, 1991). Such an analysis requires identification of the rate of change in population size that one wishes to detect, the risk one is willing to take that a non-significant trend will be judged significant (Type I error), and the risk one is willing to take that a significant trend will be judged non-significant (Type II error) (Gerrodette 1987, 1991). In general, an inverse relationship exists between α, the risk of a Type I error and β, the risk of a Type II error; as α is increased, β decreases and power $(1 - \beta)$ increases (Toft and Shea 1983, de la Mare 1984, Rotenberry and Weins 1985, Peterman 1990, Forney et al. 1991).

The rate of change in size of an increasing population used in the trend analyses was based on DeMaster's (1984) model of a hypothetical walrus population from which he concluded that an annual growth rate $(\lambda - 1)$ of 0.03 – 0.05 was reasonable. For our estimates of the number of surveys required, we evaluated populations increasing 5 percent/year, decreasing 5 percent/year and decreasing 10 percent/year. For this evaluation, we used the formula that assumes an exponential change in population size and a CV proportional to $1/(\text{Population Size})^{0.5}$ (Gerrodette 1987: 1366). We evaluated one- and two-tailed tests with CV for individual surveys of 0.6 and 0.3. We calculated the number of surveys needed to detect a trend for α and β ranging from 0.05 to 0.30.

In general, the number of surveys needed to detect a change increased with increasing CV, lower rates of change, decreasing risk of Type I and Type II error, and negative rates of change (figures 2 and 3). Reducing the CV from 0.6 to 0.3 reduced the number of surveys substantially, but selecting a lower desired precision or power also reduced the number of surveys. More surveys are required to detect a negative trend than a positive trend (Figure 3). Under the most liberal conditions we considered (CV = 0.3, one-tailed test, and $\alpha = \beta = 0.3$), at least 11 annual surveys would be necessary to detect a decline of 5 percent/year in the population size.

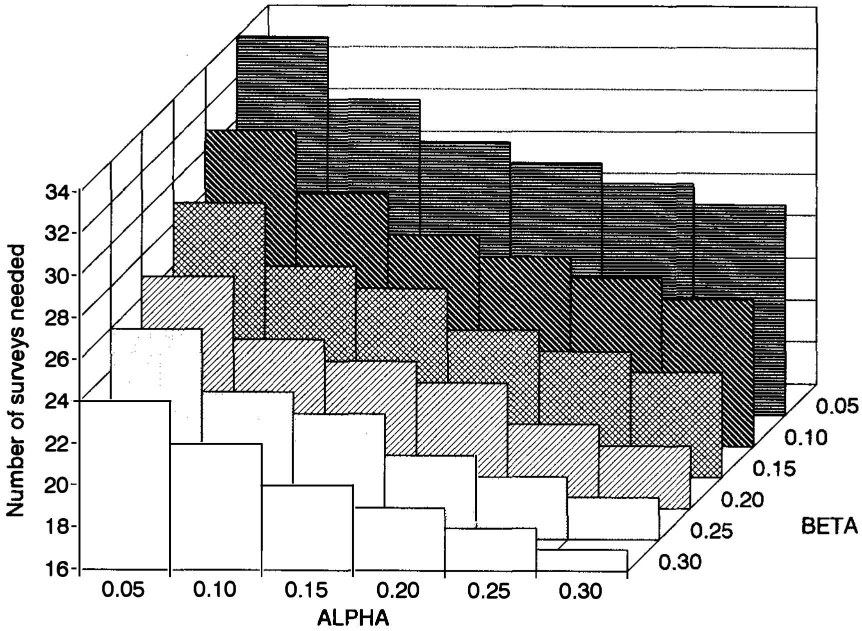

Figure 2. The number of surveys needed to detect a trend for $\alpha = 0.05 - 0.3$, $\beta = 0.05 - 0.3$ while holding constant the rate of change of the population at – 0.05, coefficient of variation at 0.6, for a two-tailed test using the z-distribution, with exponential change, from equation 15 in Gerrodette (1987).

For Pacific walruses, the point estimates of the population size may appear to indicate a trend, but despite efforts to increase the precision of the estimates, the large variation in each estimate negates any conclusion (Figure 1). Given the conditions of the past range-wide surveys (CV = 0.6, five-year survey intervals) and $\alpha = \beta = 0.1$, the maximum negative (one-tailed) rate of change detectable statistically was 78 percent. Under the same conditions, 13 surveys over a 60-year period would be required to detect statistically a population declining at 5 percent/year. The variability of population estimates is large, the rate of change of the population is likely to be small (DeMaster 1984), and surveys are infrequent because they are expensive and require international coordination. The result of these factors is that statistical detection of a real trend in the walrus population is highly unlikely given realistic levels of effort and funding.

Consideration of the Fraction Not Available to be Counted

A source of variation that has been recognized but not included in the estimates of population size for Pacific walruses is the unknown proportion of the population that is not available to be counted during the survey (Estes and Gilbert 1978, Eberhardt et al. 1979, Gilbert et al. 1992). Each estimate of population size is derived from a unknown and probably variable proportion of the entire population. By chance, a large proportion of the population might be seen on a given survey, and a small proportion on the next survey.

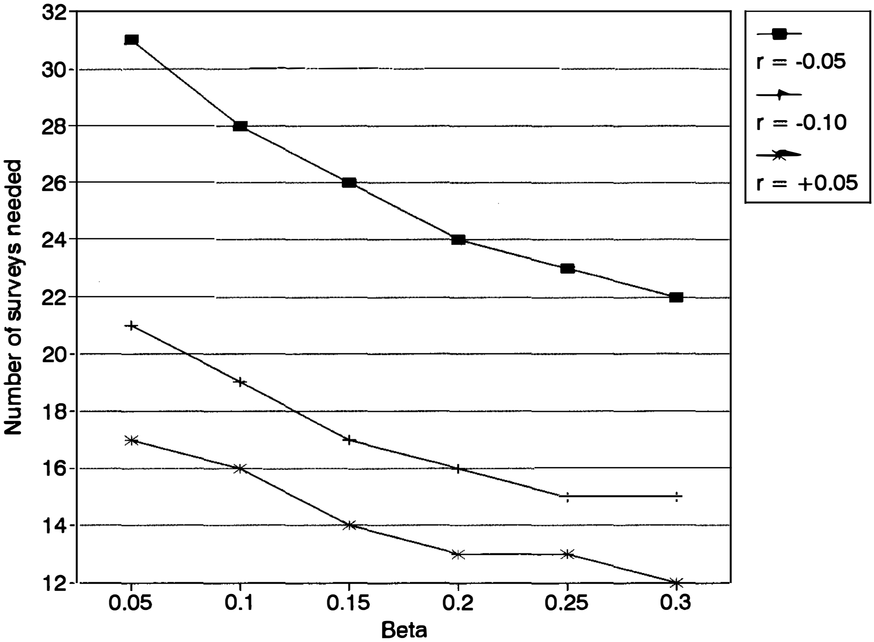

Figure 3. The number of surveys needed to detect a trend for three rates of change of the population: − 0.1, − 0.05, + 0.05 for $\beta = 0.05 - 0.3$ while holding constant $\alpha = 0.10$, the coeffiecient of variation at 0.6, for a two-tailed test using the z-distribution, with exponential change, from equation 15 in Gerrodette (1987).

The total number of walruses could be calculated using the following formula:

$$W = (w)\ (1/p) \tag{1}$$

where

W = total walrus population,

p = the proportion of walrus population hauled out, and

w = number of walruses available to be counted.

The factor 1/p is an "availability correction factor," analogous to Gasaway et al.'s (1986) sightability correction factor. The variance of W can be calculated using the following formula (Goodman 1960):

$$V(W) = V(w) * (1/p)^2 + V(1/p) * w^2 - V(w) * V(1/p) \tag{2}$$

Little is known about p and how it varies with environmental variables, age-sex composition of the population, season or geographical area. Twelve male walruses fitted with satellite transmitters in Bristol Bay, Alaska, spent 46– 89percent of the time in the water; in spring and summer, 15 females in the Bering and Chukchi sea spent 56– 89percent of the time in the water (Hills 1992).

If the final term in Equation 2 is ignored (it generally is an order of magnitude smaller than the other terms), the relative contributions of the variability in the estimate of the available population and the variability in the estimate of the correction factor to the CV of W can be calculated as:

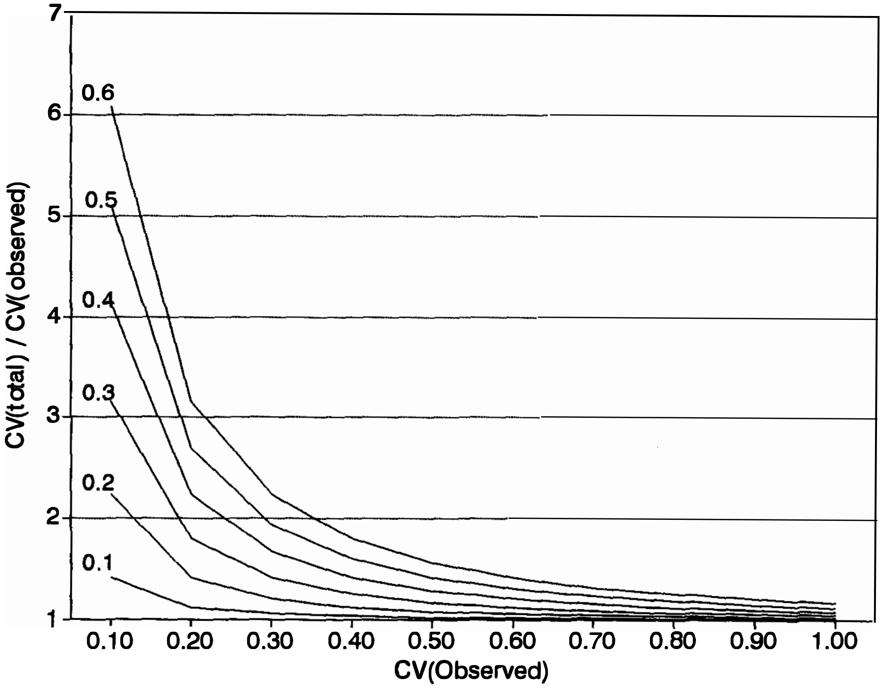

Figure 4. The effect on the coefficient of variation (CV) of the estimate of total population size of different CVs for (1) the estimated number of walruses observed on an aerial survey, and (2) a correction factor for walruses unavailable to be observed during an aerial survey.

$$CV(W) = [CV^2(w) + CV^2(1/p)]^{0.5} \quad (3)$$

The CV(1/p) best would be estimated by having some of the population fitted with satellite transmitters that could be monitored at the time of the survey. If the transmitters were distributed among sex, age and reproductive classes in a representative fraction, the proportion of transmittered animals that are available to be counted would be an unbiased estimate of 1/p, and its CV could be calculated using standard techniques. The CV could be reduced by increasing the number of transmitters monitored at the time of the survey.

If data from transmitters were available but none could be monitored at the time of the survey, the CV(1/p) would have to be based on observations of the fraction of the walruses with transmitters available to be counted over a period of time. However, the variation in this fraction would be over time and would not be reduced by having more transmitters.

The effect on the total CV(W) of the addition of CV(1/p) is a function of the relative sizes of CV(1/p) and CV(w). If 30 transmitters were monitored at the time of a survey, it is likely that CV(1/p) would be about 0.1–0.2. If one had to rely on information from a series over time, the CV(1/p) would be closer to 0.6–0.8. With the present precision of the surveys, CV(w) = 0.6, the incorporation of a correction factor with a CV(1/p) of 0.2 would add 5.4 percent to the total CV(W), while a CV(1/p) of 0.6 would add 41 percent to the total CV(W). Only when the survey

CV(w) is smaller than the CV(1/p) does the CV for the fraction hauled out dominate the precision of the estimate. The technology exists to deploy 30 transmitter at the time of the survey and we believe effort should be focused on increasing the precision of estimate of the visible population.

Summary

Continuing at the present level of effort is not likely to provide information that can be used to detect the presence *or* the absence of a trend in the walrus population. A large change over a relatively long period of time is required before a trend can be detected. Other methods of population assessment may be more useful in determining status. For example, various population parameters, such as pregnancy rate, proportion of calves to adult females or age at first reproduction, may be related to population status relative to the carrying capacity of the environment (Eberhardt and Siniff 1977, Eberhardt and Simmons 1987, Croxall 1989, Fay et al. 1989, Fowler and Siniff 1991). Russian and U.S. researchers have collected biological data from harvested animals, including body measurements, weights, teeth (used for age determination) and reproductive tracts, from the early 1950s to the present (F. Fay personal communication 1988, G. Fedoseev personal communication 1988). Although available measures of density dependence are relatively imprecise and biased to an unknown degree, the current data may be sufficient to evaluate the utility of various biological parameters. However, the detection of trends in multiple biological parameters used as indices is subject to the same concerns as population estimates, and may be subject to multiple interpretations unless their mechanisms of action are well understood (Garshelis et al. 1990, Estes 1990).

Regardless, some measure of population size still is necessary to calibrate any indices and to ensure that any changes in indices are not due to changes in the carrying capacity of the environment rather than a change in the numbers of walruses (DeMaster 1984). Estimates of population size also are needed to comply with the current requirements of the U.S. Marine Mammal Protection Act. Aerial surveys of Pacific walruses through 1990 give minimum population estimates that cannot be used for indication of trend because of the spatial and temporal aggregation of Pacific walruses and the large area they inhabit. The accuracy of the estimate of population size can be improved by estimating the proportion of the population not seen, with little overall loss in precision of the estimate. In addition, any aerial survey of Pacific walruses with the objective of estimation of total population size should sample enough of the survey area to obtain a number with reasonable precision. Estes and Gilbert (1978) noted more than 56 percent of the sea ice area would have to be sampled to obtain a 95-percent confidence interval that was ± 10 percent of the estimated number.

For now, the best numbers available on the size of the Pacific walrus population are not a sufficient basis for establishing management policy.

Acknowledgments

We thank T. Gerrodette for advice and access to his computer programs for power analysis. Discussions with C. Fowler, R. Harris, R. Merrick, S. Mizroch, L. Pank, E. Rexstad and M. Udevitz were very helpful. As usual, F. H. Fay helped us see the historical perspective of the problem. We appreciated comments from S. Amstrup, F. Fay, G. Garner, L. Holland-Bartels, M. Hunter, W. Krohn, L. Lowry, W. Seitz and M. Udevitz. Funding was by the USFWS, Alaska Fish and Wildlife Research Center to the Maine Cooperative Fish and Wildlife Research Unit and Dr. J. R. Gilbert, University of Maine, Orono.

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Parameters for Monitoring Small Mammal Populations

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Introduction

A variety of environmental problems ranging from loss of biological diversity and habitat to threats from acid rain and global warming have contributed to the call for environmental monitoring. Several important initiatives and programs focused on environmental monitoring have been established over the past decade. The National Acid Precipitation and Assessment Program (NAPAP) is an example of a large, federally funded research effort that lasted over a decade (NAPAP 1991). The U.S. Environmental Protection Agency recently has instituted the Environmental Monitoring and Assessment Program (EMAP) (Messer et al. 1991), an ambitious program that will attempt to monitor the nation's ecological resources.

Assessing biological diversity through inventory and monitoring of an ecosystem is a major component of many of these programs. The Sustainable Biosphere Initiative (SBI) (Lubchenko et al. 1991), the U.S. Army Land Condition-Trend Analysis Program (LCTA) (Diersing et al. 1992) and the IUBS-SCOPE-UNESCO Program on biodiversity (Solbrig 1992) are examples of such programs.

Obviously, the scope of these programs will require a great deal of time, effort and money. As such, it will be desirable to use species which are relatively easy and cost effective to monitor, and small mammals are one such group. The LCTA program currently includes small mammals as well as birds in their surveys of Army lands (Diersing et al. 1992). A Yellowstone National Park program has been monitoring small mammals since 1991 (R. Crabtree and M. Harter personal communication).

There has been a large amount of research devoted to estimating animal abundance and survival, and much of this work has been developed with small mammals in mind (cf. Otis et al. 1978, Seber 1982, 1986, Pollock et al. 1990). In the case of small mammals, most of these methods employ some form of capture-recapture (CR) or removal techniques. The CR data is then used to obtain abundance estimates (in some cases, indices of abundance) and/or survival estimates. Reliability of the methods spends heavily on the samples sizes obtained in CR studies, and sample sizes are a function of the capture probability of the animals (p), the number of traps and the length of the sampling period (Otis et al. 1978, Pollock et al. 1990). Estimators of abundance, either population size (N) or density (D), are based on statistical models which have various assumptions associated with the capture probability of the animals (White et al. 1982: 8). The simplest model (and least realistic) assumes that all animals have an equal probability of capture. A sequence of models which relax the equal catchability assumption has been postulated (Pollock 1974, Otis et al. 1978). This sequence allows for three sources of variation in the capture probabilities: time, behavior and heterogeneity: Model M_t assumes that capture probabilities vary by time or trapping occasion, Model M_b assumes that capture probabilities vary by behavioral responses to capture, and Model M_h assumes that capture probabilities vary by individual animal (i.e., heterogeneity in the capture probabilities exists). A sequence of eight models then is possible, ranging from the simplest, Model M_o (equal catchability), to the most general, Model M_{tbh} (all sources of variation active). All models assume geographic closure on the population, but the models can be further grouped into open CR models which allow demographic processes such as birth, death, immigration and emigration to occur (Pollock et al. 1990), and closed CR models which assume that no demographic processes occur during the study (Otis et al. 1978).

Any monitoring program will have to take into account the importance of landscape heterogeneity (Forman and Godron 1986). This will include issues such as scale and hierarchy, as well as temporal variation (O'Neill et al. 1986, 1989). Sampling protocols obviously will need to be small and efficient in order to include theses factors. Therefore, small mammal trapping may be a potential candidate for use in a monitoring program. Unfortunately, several small mammal studies have indicated that reliable abundance and survival estimation requires large trapping grids, high capture probabilities and at least four trapping occasions (cf. Otis et al. 1978, Pollock et al. 1990). A critical point is that reliable estimation depends on the use of the appropriate model (based on model assumptions about the capture probabilities). This requires that model assumptions be tested, so that the appropriate model can be selected. Unfortunately, the power of these tests can be quite low without sufficiently large sample sizes (Otis et al. 1978:56).

The objective of this study is to examine factors which can affect the results of a small mammal monitoring study. In particular, we will examine population parameters as well as spatial factors that need to be considered in designing a small mammal monitoring study. For example, what can be inferred when using a small number of traps or trapping for a short period, and how does home range size affect the results? This will be accomplished using Monte Carlo computer simulation techniques.

Simulation Methods

A modified version of a computer program simulating small mammal movement and trapping over several time periods was used for this study (Zarnoch 1976, 1979, Wilson and Anderson 1983). The model includes properties of a population of small mammals such as spatial distribution, home range size and shape, and capture probability, and it allows the capture-recapture trapping process to be simulated. For these simulations the following factors were used: (1) population densities of 25 and 100 animals per hectare; (2) average capture probabilities ($\overline{p}$) of 0.1, 0.25 and 0.4; (3) capture probability models M_o (equal capture probabilities for all animals), M_t (capture probabilities vary by occasion) and M_h (all animals have different capture probabilities); (4) study lengths of two and four occasions (trapping days); (5) home range sizes of 0.25 and 0.5 hectare; and (6) trapping grid sizes of 5×5, 10×10, 15×15 at 7 meters spacing. Each simulation proceeded by defining a study area that represented a 200-meter by 200-meter area (4 ha). A trapping grid of either 25, 100 or 225 traps was centered within the 4-hectare study area. Animals were placed within this area according to a random spatial pattern, therefore 100 animals and 400 animals were used for densities of 25/hectare and 100/hectare, respectively. Capture probabilities for each animal were assigned according to the capture probability model with the average capture probability for the entire simulation equal to either 0.1, 0.25 or 0.5. Trapping then was simulated for two or four occasions, and on each occasion animals moved within their home range according to a bivariate normal distribution based on the 95-percent confidence interval (all animals had the same home range size throughout a particular simulation run). Only animals within a certain distance of an unoccupied trap had a chance of capture, and this distance was controlled by the program to ensure that the average capture probability was approximated. Each simulation run consisted of 100 repetitions.

The capture history of all animals caught at least once was recorded and stored for analysis by program CAPTURE (Otis et al. 1978). Program CAPTURE computes population estimates (N) based upon the time, behavior and heterogeneity models. Program CAPTURE can be allowed to choose the "appropriate" model based on the capture history data, but in our case, the appropriate model was known, and this model was used to compute the population estimates.

If the grid sizes, home ranges and capture models were identical for all simulation runs, then population size could be used to compare simulation runs. Density (D) or the number of animals per unit area (N/A) is the obvious measure for comparison. A naive estimate of D would be to divide N by the area of the trapping grid, but because of "edge effect" (capture of animals outside the trapping grid with home ranges that enclose a portion of the trapping grid) the effective trapping grid often is much larger (Dice 1938, Tanaka 1972). In fact, simulations show that the naive density estimates lead to severe overestimation (Wilson and Anderson 1985). Dice (1938) suggested that a boundary strip (W) be added to the effective area of the grid. He suggested using one-half the average home range diameter of the animals as an estimate of W. This procedure has problems when home range is estimated from trapping data, because W can vary depending on trap spacing and the number of recaptures (Stickel 1954, Tanaka 1972). In our case, the true home range sizes were known, and the effective trapping grid was computed as the actual trapping grid plus a distance equal to one-half the diameter of the average home range size, based on the bivariate normal 95-percent confidence interval. Other methods have been suggested for estimating density, but Tanaka (1980) concluded that Dice's method of compensating for edge effect was superior.

The true population size (N_g) of the effective trapping grid was computed by dividing the effective trapping area by the entire study area (4 hectares), and multiplying this proportion by the number of animals in the 4-hectare study area (*see* Table 1). The true population size, N_g was used to compare with estimates from program CAPTURE and with the total number of different animals captured, M_{t+1}, (M_{t+1} was used as an index of population size).

Results and Discussion

Results for Model M_h, 72 simulation runs, are shown in Table 1. The results for Models M_o and M_t were very similar and are available from the first author. The following results and discussion are based on the results for all models.

Population estimates were not possible for many of the simulation runs (primarily with only two trapping occasions), because in these cases there were either no re-

Table 1. Average population estimates, ave(N), and total number of different animals captured, ave(M_{t+1}) for simulations allowing heterogeneity in capture probabilities, Model M_h. The factors are number of trapping occasions, t; average home range size, hr; number of repetitions analyzed per simulation for M_{t+1} and N, Reps(M) and Reps(N), repsectively; the true density, D, the true population size based upon a strip width equal to one-half the average home range size, N_8; and percent relative bias for N and M_{t+1}, PRB(N) and PRB(M_{t+1}) respectively.

t	hr	Rep(M)	Rep(N)	Tr	ave(p)	D	N_8	ave(N)	ave(M_{t+1})	PRB(N)	PRB(M_{t+1})
2	0.5	82	47	25	0.1	400	40	3.6	2.1	–92	–95
2	0.5	99	25	25	0.25	400	46	6.1	4.6	–87	–90
2	0.5	100	100	25	0.4	400	46	8.6	6.7	–81	–86
2	0.5	100	100	100	0.1	400	106	12.4	9.2	–88	–91
2	0.5	100	100	100	0.25	400	106	30.6	21.8	–71	–79
2	0.5	100	100	100	0.4	400	106	43.3	30.8	–59	–71
2	0.5	100	100	225	0.1	400	190	28.9	20.4	–85	–89
2	0.5	100	100	225	0.25	400	192	71.2	50.1	–63	–74
2	0.5	100	100	225	0.4	400	190	98.7	70.2	–48	-63
2	0.5	74	30	25	0.1	100	12	3.2	1.6	–72	–86
2	0.5	80	49	25	0.25	100	12	3.3	2.0	–71	–83
2	0.5	79	41	25	0.4	100	12	3.5	1.9	–70	–83
2	0.5	95	78	100	0.1	100	26	4.6	3.3	–83	–88
2	0.5	100	95	100	0.25	100	26	7.2	5.5	–73	–79
2	0.5	100	100	100	0.4	100	26	9.8	7.5	–63	–72
2	0.5	100	96	225	0.1	100	48	7.5	5.7	–84	–88
2	0.5	100	100	225	0.25	100	48	18.1	13.5	–62	–72
2	0.5	100	100	225	0.4	100	48	23.2	17.3	–51	–64
2	0.25	81	57	25	0.1	400	32	3.8	2.4	–88	–92
2	0.25	99	98	25	0.25	400	32	6.0	4.8	–81	–85
2	0.25	100	99	25	0.4	400	32	8.6	6.6	–73	–79
2	0.25	100	100	100	0.1	400	83	12.8	9.5	–85	–89
2	0.25	100	100	100	0.25	400	83	28.5	20.5	–66	–75
2	0.25	100	100	100	0.4	400	83	41.3	29.7	–50	–64
2	0.25	100	100	225	0.1	400	159	29.9	21.1	–81	–87
2	0.25	100	100	225	0.25	400	159	68.3	48.4	–57	–70
2	0.25	100	100	225	0.4	400	159	94.8	68.8	-40	–57
2	0.25	67	34	25	0.1	100	8	3.3	1.8	–59	–78
2	0.25	78	38	25	0.25	100	8	3.4	1.8	–57	–77
2	0.25	71	38	25	0.4	100	8	3.2	1.8	–59	–78
2	0.25	94	78	100	0.1	100	21	4.5	3.2	–78	–85
2	0.25	100	99	100	0.25	100	21	7.1	5.7	–66	–73
2	0.25	100	99	100	0.4	100	21	9.0	7.2	–57	–66
2	0.25	100	100	225	0.1	100	40	7.3	5.7	–82	–86
2	0.25	100	100	225	0.25	100	40	16.9	12.6	–58	–68
2	0.25	100	100	225	0.4	100	40	23.6	17.7	–41	–56
4	0.5	95	76	25	0.1	400	46	5.0	2.9	–89	–94
4	0.5	100	100	25	0.25	400	46	14.8	8.5	–68	–82
4	0.5	100	100	25	0.4	400	46	23.2	11.6	–50	–75
4	0.5	100	100	100	0.1	400	106	39.5	18.3	–63	–83
4	0.5	100	100	100	0.25	400	106	81.4	38.2	–23	–64
4	0.5	100	100	100	0.4	400	106	100.5	50.9	–5	–52
4	0.5	100	100	225	0.1	400	190	87.9	39.1	–54	–79
4	0.5	100	100	225	0.25	400	190	160.7	82.0	–16	–57
4	0.5	100	100	225	0.4	400	190	198.3	109.2	4	–43

Table 1. *Continued.*

t	hr	Rep(M)	Rep(N)	Tr	ave(p)	D	N_8	ave(N)	ave(M_{t+1})	PRB(N)	PRB(M_{t+1})
4	0.5	93	75	25	0.1	100	12	4.6	2.6	–60	–77
4	0.5	92	76	25	0.25	100	12	4.9	3.0	–58	–74
4	0.5	93	74	25	0.4	100	12	5.0	2.8	–56	–75
4	0.5	100	94	100	0.1	100	26	8.6	5.2	–67	–80
4	0.5	100	100	100	0.25	100	26	17.9	9.5	–32	–64
4	0.5	100	100	100	0.4	100	26	23.5	12.5	–11	–53
4	0.5	100	100	225	0.1	100	48	19.9	10.3	–58	–78
4	0.5	100	100	225	0.25	100	48	39.6	20.8	–17	–56
4	0.5	100	100	225	0.4	100	48	47.9	27.7	1	–42
4	0.25	99	80	25	0.1	400	32	5.2	3.0	–84	–90
4	0.25	100	99	25	0.25	400	32	14.3	8.0	–55	–75
4	0.25	100	100	25	0.4	400	32	21.9	11.3	–31	–64
4	0.25	100	100	100	0.1	400	83	35.3	16.7	–58	–80
4	0.25	100	100	100	0.25	400	83	70.1	35.1	–16	–58
4	0.25	100	100	100	0.4	400	83	85.5	46.9	3	–44
4	0.25	100	100	225	0.1	400	159	85.8	38.7	–46	–76
4	0.25	100	100	225	0.25	400	159	149.8	79.0	– 6	–50
4	0.25	100	100	225	0.4	400	159	176.8	103.4	11	–35
4	0.25	92	62	25	0.1	100	8	4.8	2.6	–39	–67
4	0.25	93	73	25	0.25	100	8	4.4	2.7	–44	–65
4	0.25	94	70	25	0.4	100	8	4.6	2.5	–41	–68
4	0.25	100	97	100	0.1	100	21	9.1	5.5	–56	–74
4	0.25	100	100	100	0.25	100	21	16.7	9.5	–20	–55
4	0.25	100	100	100	0.4	100	21	19.2	11.6	– 8	–44
4	0.25	100	100	225	0.1	100	40	19.9	10.1	–50	–75
4	0.25	100	100	225	0.25	100	40	36.9	20.3	– 7	–49
4	0.25	100	100	225	0.4	100	40	41.9	26.1	5	–34

captures or no captures at all. Six of the 216 simulation runs resulted in only one population estimate out of 100 repetitions, and all of these occurred when the number of trapping occasion was two and the number of traps equaled 25, *see* Reps(N) in the table. The number of repetitions computed for the population index, M_{t+1} (Reps(M) represents the repetitions when at least one animal was captured. For example, there is one case for Model M_h where 33 repetitions resulted in no captures, i.e., Reps(M) = 67. Ideally, percent relative bias (PRB) of an estimator should decrease as samples sizes increase. This occurred for the population estimates from program CAPTURE, PRB(N), but the decline was much less for the PRB of M_{t+1}, PRB(M_{t+1}), which consistently was negative.

Overall, the PRB(N) decreased as the number of trapping occasions increased and PRB neared zero at higher capture probabilities. These results are not surprising, because several previous simulation studies have seen a significant improvement in results as $\bar{p}$ and/or the number of occasions increases (Otis et al. 1978, White et al. 1982, Wilson and Anderson 1985). The decline was more rapid for PRB when the number of trapping occasions was four.

PRB(N) was slightly lower for model M_t than for model M_o. Both models showed less bias than model M_h. PRB for M_{t+1} was fairly consistent for these capture recapture models relative to PRB(N), but it was consistently larger, on an absolute basis, and negative. This is to be expected because M_{t+1}, is essentially the lower bound on the population estimates.

The effect of home range size on the effective trapping grid size was quite large. For example, the size of the naive grid with 25 traps was 0.0784 hectare, but the effective grid size was 0.316 hectare and 0.461 hectare when the average home range size was 0.25 and 0.5 hectare, respectively (400 and 531 percent increases). For the largest grids of 225 traps the naive grid size was 0.9604 hectare, and the effective grid sizes were 1.593 and 1.902 hectares, respectively (166 and 198 percent increases). PRB of N and M_{t+1} were higher when home ranges averaged 0.25 hectare. Edge effect was less pronounced with larger grid sizes, resulting in a decrease in PRB(N); this decrease was much less evident for M_{t+1}.

Many of the essential aspects of designing a CR study are discussed in Skalski and Robson (1992). As with any study, it is important to clearly state the objectives of the small mammal monitoring program. If the objective is to closely track population size over time and space, then these simulation results collaborate what many studies have shown; namely an intensive trapping effort must be undertaken for valid inference (cf., Otis et al. 1978). If the objectives are more concerned with detecting large population changes over space and time, and time and budgets are limited, then some adjustment in the trapping protocol may be possible.

In many cases, the most desirable approach for reducing the time and effort may be a reduction in the number of trapping occasions. Several factors suggest that this may not be a good idea. First, it is impossible to test the assumptions associated with CR methods with only two days of trapping (Otis et al. 1978, Pollock 1990), and as Skalski and Robson (1992: 61) have stated, "Because the validity of a survey model can only be determined *a posteriori*, there is little alternative to the use of model selection to ensure validity of subsequent abundance estimates." Essentially, a study with only a few trapping occasions will not permit an assessment of the reliability of the estimates. Second, the simulation results show that the population estimates can be highly biased when the number of occasions is small, and in some cases population estimates are impossible because of sparse data.

What about using an index of animal abundance, such as the total number of animals captured, M_{t+1}? The idea of an index implies that the index is being calibrated to some known or estimable quantity. In the case of M_{t+1}, it is assumed that this index has been calibrated to the true population or an estimate of the true population size. This calibration step is rarely done for CR studies. If you examine the results in Table 1, you will notice that the PRB for M_{t+1} is somewhat consistent over all factors. And, although there is a very large negative bias in all cases, the argument could be made for using M_{t+1} for detecting relatively large population changes. In fact, estimates of change over time or space are unbiased, provided the bias is constant (Cochran 1977: 380). The results for PRB(M_{t+1}) can be misleading though, because they were based upon comparison of M_{t+1} to the population of the effective trapping grid which had been adjusted for home range size. In an actual study, home range size would have to be estimated, and with smaller grids and fewer occasions, the estimates of home range from sparse data would be subject to significant bias (Tanaka 1972). Therefore, as a researcher, you must assume that capture probabilities, home ranges, etc., do not change from sampling period to sampling period in order to use an index such as M_{t+1}, an unlikely assumption.

Other difficulties with using population estimates, N, or M_{t+1} become evident by examining the ave(N) and ave(M_{t+1}) in Table 1, these are the parameters that would be obtained in an actual field study. In almost all cases, except when sample sizes are low, M_{t+1} and N increase consistently as capture probabilities increase, even though the true population size and the number of traps has remained constant. In an actual study, the conclusion might be that the population had increased when, in fact, only capture probabilities had. On the other hand, with low capture probabilities or with small trapping grids, the conclusion might be that the population hadn't significantly changed when, in fact, the population may have increased by a factor of four (compare D = 100 versus D = 400).

The fact that comparisons of population estimates (with no adjustment for edge effect) also can lead to incorrect conclusions is very disconcerting, because CR population estimates frequently have been used in field studies. The problem stems from the fact the CR theory was developed from ball and urn models that have a clearly defined boundary on the population, and edge effect due to animal movement violates this assumption of geographic closure. Problems with edge effect have been known for a long time (Dice 1938, Stickel 1954, O'Farrell et al. 1977, Otis et al. 1978, Anderson et al. 1983, Wilson and Anderson 1985) and no ideal solution is available. Because of the difficulties outlined above, small mammal monitoring to detect changes over space and time will be difficult unless resources are sufficient for relatively large trapping grids and at least four trapping occasions.

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Detecting Differences in Wildlife Populations Across Time and Space

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Introduction

Understanding of processes in natural populations demands the utmost skill in wildlife biologists. Documenting fluctuations over time within single populations with concomitant auxiliary variables such as climatological data, enables researchers to piece together mechanisms influencing populations. Coupled monitoring of multiple populations over multiple geographic locations further elucidates population processes by detaching time effects from geographic effects. While population monitoring activities do not constitute scientific experiments, in the spirit of manipulation of salient ecological variables, replication of monitoring activities for natural populations over long periods of time and in diverse geographic locations can lead to insights into populations processes (sensu Cook and Campbell 1979). These insights can be translated into hypotheses useful for prediction of changes in population abundance, resulting either from natural perturbations (e.g., wildfire) or anthropogenic modifications (e.g., mineral extraction).

To these ends of biological population monitoring, a number of national programs have been inaugurated. Basic research organizations such as the National Science Foundation have allocated funds for long-term studies in the form of Long-term Ecological Research (LTER) programs. Applied research organizations, the newly formed National Biological Survey (NBS) has as a primary mission the monitoring of the living resources of the United States. As envisioned by Secretary of the Interior Bruce Babbitt, development of information regarding wildlife populations provides tools necessary for avoiding future natural resource management problems. Stewardship agencies, such as the National Park Service (NPS), have initiated their own monitoring activities: "NPS will assemble baseline inventory data describing the natural resources under its stewardship as well as monitor those resources to detect or predict changes. The resulting information will be analyzed to detect changes that may require intervention and to provide reference points for comparison with other, more altered, environments" (Rugh and Peterson 1992).

Application of Appropriate Methodology

The intention of monitoring activities proposed by the various organizations is the detection of changes in ecosystem or population status. To translate this intention into actions, three activities must be performed. First, appropriate population parameters must be targeted for estimation and data collection methods identified. Second, parameter estimation must be carried out in concert with associated measures of precision. Third, inferences must be drawn regarding change or lack thereof from parameter estimates derived from single populations monitored across time, and/or multiple populations monitored across time and geographic locale. Conclusions regarding population change are only valid when arrived at through sound methodology. It is therefore incumbent upon wildlife biologists to conduct investigations of population monitoring in a rigorous manner in all aspects of design, data collection, analysis and interpretation.

Krebs (1994: 152–160) offers advice on various methods for population assessment, including relative and absolute abundance estimates. I will argue that relative abundance estimates are ineffectual for detecting changes in population abundance. Techniques described by Skalski and Robson (1992) offer rigorous methods for deriving inferences regarding differences in population abundance.

Further, sound inferences from which management actions can be formulated, not only require data on population abundance, but also estimates of the processes giving rise to abundance at any point in time. Anticipatory management can be formulated when estimates of mortality, natality, immigration and emigration can be produced.

This paper will focus on a pair of ongoing studies monitoring populations of microtine rodents in sub-Arctic Alaska. Microtines have been subject to numerous investigations in the circumpolar region (Pruitt 1968, Taitt and Krebs 1985, Ims and Steen 1990). They constitute useful indicators of biological systems as a result of their short lifespans, ready accessibility, low handling cost and the ability to acquire adequate samples of these fecund species which may be found at high population densities. They also have been alleged to undergo pronounced fluctuations in abundance (Batzli 1992). Microtine status in the tropic structure also offers the opportunity to anticipate changes in furbearer populations that are dependent upon microtines as a food source (Hanski et al. 1993). Techniques to detect changes in microtine populations at two time scales and three geographic scales will be described.

Methods

Livetrapping experiments for microtine rodents began in the summer of 1992 in Denali National Park and Preserve (DNPP) and the Creamer's field Wildlife Refuge (CFWR) located in Fairbanks, Alaska. Grids of 0.81 hectare, containing 100 23-centimeter Sherman livetraps were situated in muskeg and black spruce habitats in CFWR, and in black spruce and shrub/white spruce habitats in DNPP. Replicate grids were placed within habitats in DNPP, but not in CFWR. Sampling protocol followed the robust design of Pollock (1982) with primary sampling occasions monthly or semi-weekly during snowfree months (June-September), with secondary sampling occasions, in the form of trap checks, taking place two or three times daily over a five-day period. Each captured animal was identified to species, sexed, examined for reproductive condition and weighed to the nearest gram. Each individual was scanned for a unique identification code using a passive integrated transponder (PIT; Biosonics, Inc., Seattle, Washington) tag reader. Previously unmarked individuals were dorsally marked subcutaneously with a PIT tag (Schooley et al. 1993). Although other species of microtines were components of the study, only data for northern red-backed voles (*Clethrionomys rutilus*) will be presented here. Sampling protocol was reviewed and approved by the Animal Care and Use Committee of the University of Alaska—Fairbanks.

Traps regularly spaced on a rectangular grid pose difficulties for density estimation

(Otis et al. 1978: 67–68) because of the difficulty in identifying effective trapping area. Therefore, for the purposes of population monitoring, absolute measures of abundance were estimated using the closed population models of Otis et al. (1978), with additional estimators derived by Chao (1988), Chao et al. (1992) and Burnham (1990). All estimators have been incorporated into a comprehensive computer program (Rexstad and Burnham 1991). Confidence intervals reported here are based on intervals placed around $\hat{f}_0$, then number of animals in the population not captured. These intervals are transformed to confidence intervals for $\hat{N}$ using the following formulas:

$$[M_{t+1}+\frac{f_0}{C}, M_{t+1}+f_0 \cdot C]$$

where M_{t+1} is number of animals caught, f_0 is the number of animals not caught, and

$$C = \exp\left(1.96\sqrt{\ln[1+\frac{var(\hat{N})}{f_0^2}]}\right)$$

These intervals result in lower confidence bounds that do not fall below M_{t+1}, the number of individuals captured during the primary sampling period. Simulation results have shown that the intervals computed in this manner also produce coverage values close to the nominal 95-percent level (Rexstad unpublished data 1992). Abundance comparisons across time and space were conducted using standard two-tailed z-tests.

Results

Variation in population abundance of northern red-backed voles was influenced by a number of factors that must be considered when conducting monitoring programs designed to detect changes in population levels. Partitioning these sources of variation is of fundamental importance in interpreting the results of monitoring programs.

Within-year Variation

Northern red-backed voles breed continuously during the snowfree months in sub-Arctic climates (Banfield 1974). Adults that survive the winter begin reproducing as early as late April, giving rise to some female offspring that are capable of reproducing before the onset of winter (Whitney 1976). This reproductive strategy gives rise to dramatic increases in population abundance during the summer (figures 1 and 2). At the DNPP riparian replicate sampling grid in 1993, abundances in September differed significantly from June levels ($z = 3.78$, $P = 0.0002$). This pattern also is manifested at the CFWR study site in muskeg habitat ($z = 5.22$, $P < 0.0001$).

Between-year Variation

Contrasts of abundance estimates at the DNPP study site between years at comparable times of the year for the upper riparian grid (Figure 1) is representative of patterns found on other grids in the study area. Abundance estimates in July were significantly different between 1992 and 1993 ($z = 4.43$, $P < 0.0001$). Differences were not significant in late July ($z = 2.72$, $P = 0.0065$), but were different again in early September ($z = 7.05$, $P = 0.0001$).

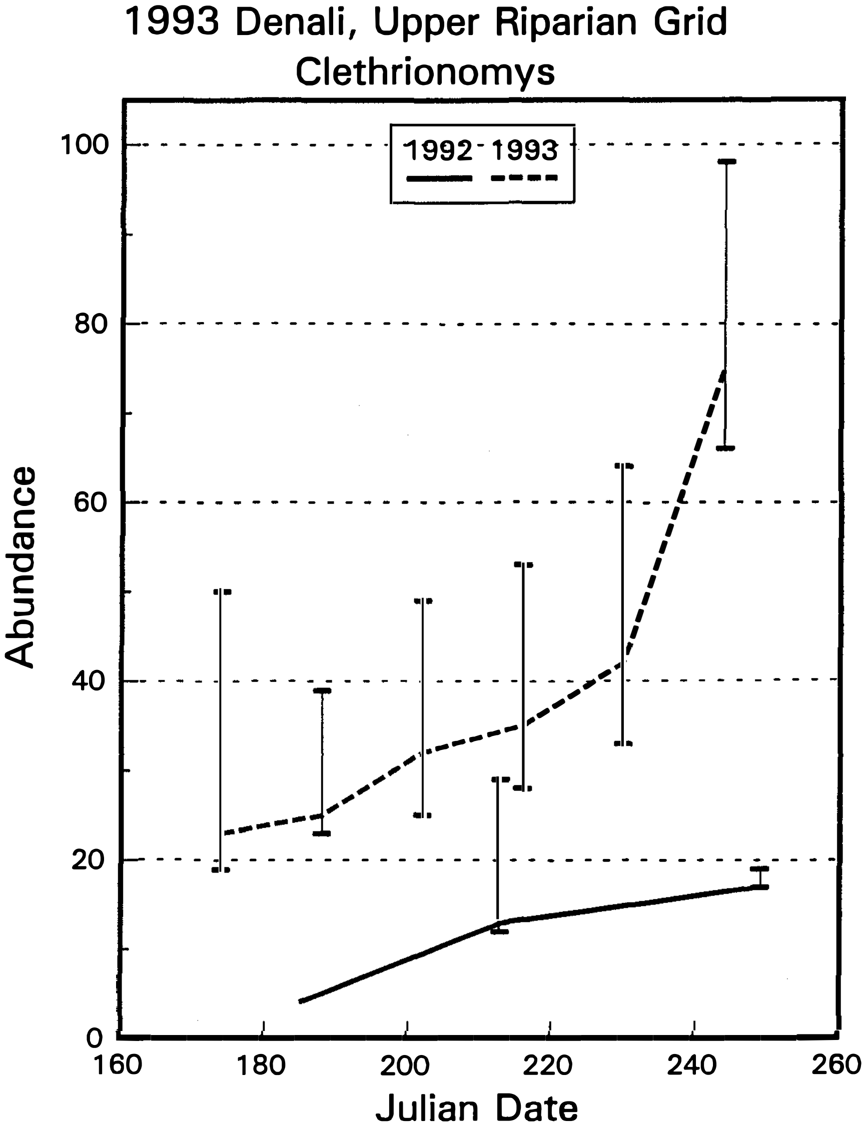

Figure 1. Abundance estimates for 1992 and 1993 field seasons in DNPP study area for upper riparian sampling grid. Note asymetrical confidence intervals.

Spatial Variation

Extrapolating abundance estimates from sample plots to larger regions requires measures of variability at several spatial scales. This includes not only sampling variability associated with the estimation process at the sampling grid, but also ''plot-to-plot'' variability (Skalski and Robson 1992: 27).

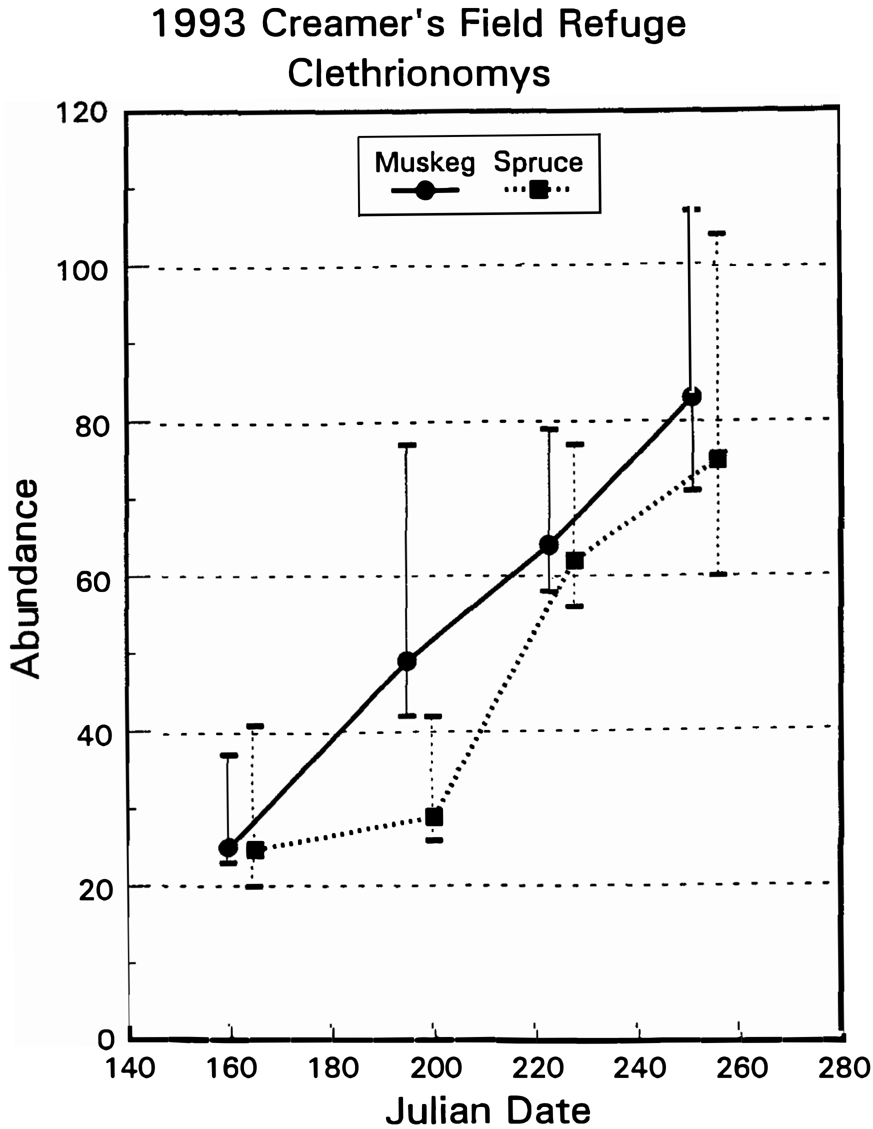

Figure 2. Abundance estimates for 1993 field season in CFWR study area for two habitat types, muskeg and spruce forest. Note asymetrical confidence intervals.

Within habitats. Although sampling grids in the DNPP study site were replicated within areas of homogeneous habitat, northern red-backed voles do not perceive their habitat in the same manner as the investigator. Even with the replicate plots separated by only 100 meters, vole abundance differed between the white spruce replicates in early July during the 1993 field season ($z = 3.57$, $P = 0.0004$). When averaged over the entire season, average abundance did not differ for the riparian habitat ($z = 0.49$, $P = 0.6240$), but did differ in the spruce habitat ($z = 2.14$, $P = 0.0324$).

Between habitats. With differences present to some extent within identified habitat types, it is not surprising this difference is magnified at a larger spatial scale. Seasonal average abundance for the 1993 field season with the DNPP riparian habitat was 41.2 (SE = 3.95), and for the spruce habitat was 36.1 (SE = 2.68). The muskeg habitat at CFWR had an annual seasonal abundance of 55.3 (SE = 3.16), and the spruce habitat had an annual seasonal abundance of 47.8 (SE = 3.23). Temporal variation produced annual estimates with such high variability that habitat differences were not significant at either study site (DNPP: $z = 0.77$, $P = 0.4413$, CFWR: $z = 1.17$, $P = 0.2420$).

Between study sites. Given variation in abundance due to habitat and time during the season, comparisons at larger geographic scales must control for these factors if comparisons are to be unambiguous. Comparing the spruce habitat at the DNPP and CFWR study sites leaves several confounding factors, primarily elevation (DNPP elevation = 900 meters, CFWR elevation = 120 meters). However, abundance did not differ between CFWR and a DNPP replicate grid during early ($z = 0.29$, $P = 0.7748$), middle ($z = 0.77$, $P = 0.4394$) or late ($z = 0$, $P = 0.5000$) in the field season. This brings limited information to bear on geographical synchrony in fluctuations in northern red-backed vole populations within a single year. Comparison of these sites over a number of years would be beneficial in determining if fluctuations in abundance, if present, are influenced by mesoscale environmental events (Henttonen et al. 1985).

Detection of Trends

One goal of long-term population monitoring is the detection of trends in population abundance. Results from two field seasons shed little light on the presence of trends in northern red-backed vole populations. However the magnitude of inter-annual variation (Figure 1) at the DNPP study site suggests it is unlikely populations can sustain these large unidirectional changes over many years, raising the specter of eruptive population dynamics (Lidicker 1988). Analysis of trends (sensu Gerrodette 1987) is meaningless under these circumstances, and other methods of long-term comparison of natural populations are required. If monitoring is being conducted with the intent of detecting environmental change induced by specific anthropogenic activity, the control-treatment pair design of Skalski and Robson (1992: 178–183) may be employed to measure the effect of the activity. However, the activity must be reasonably localized, and the time of the activity must be known such that monitoring can begin prior to imposition of the activity. This eliminates phenomenon such as global climate change from investigation through this design.

Discussion

Monitoring of northern red-backed vole populations in Interior Alaska constitutes a case study of monitoring programs in general. Although the projects have been underway for a period of time, they offer a microcosm in which to view the philosophy of biological monitoring.

Fundamental to design and implementation of monitoring activities is clear determination of their objectives. This subsumes questions regarding species of interest, population attributes of the species to measure and the magnitude of change in those attributes to be detected. Specification of these three characteristics of a monitoring program results in fairly straightforward design of a monitoring program. Unfortunately, it can be difficult to anticipate the resolution of monitoring necessary to achieve the objective of gauging the "health" of a population, and cost considerations weigh heavily upon administrators with the responsibility for overseeing monitoring programs.

Table 1 presents a brief summary of the trade-offs associated with monitoring populations at various levels of intensity. In its simplest form, monitoring wildlife populations merely can ascertain whether any individuals of the species of interest can be found in the area of interest. This constitutes basic inventory activities that are precursors to monitoring activities.

Measures of relative abundance quickly give way to measures of absolute abundance with their associated measures of precision because detection of changes necessitate statistical tests founded upon measures of variation and uncertainty. Process-level monitoring can address the demographic processes operating over time that give rise to population abundance measured at an instant in time. Estimation of reproduction, dispersal and survival can constitute "over the horizon" monitoring able to anticipate changes in population abundance resulting from changes in the underlying population processes. At the most refined level, monitoring through mark-recapture methods where individuals are handled can provide data on changes in the genetic composition of a population or the contaminant loads carried by individuals in the population.

Each level of population monitoring is capable of addressing more specific questions regarding population "health" at the cost of more intensive data gathering. It is the responsibility of resource management agencies to consider the spectrum of questions that can be addressed by monitoring programs in allocating funds to allow the monitoring to proceed. It also must be recognized that monitoring programs are by definition long-term, requiring continual obligation of funds.

Specific recommendations arising from the monitoring studies of northern red-backed voles in Interior Alaska take into account variation resulting from intra- and interannual changes in abundance, as well as variations at the scales of sampling grid, habitat and study area. These specific study recommendations hold insight for investigators planning other long-term monitoring programs.

Temporal Intensity

The timing of trapping events should be considered on three levels: seasonally, monthly and daily. Data gathered early in the summer are fundamental to understand-

Table 1. Levels of resolution possible in biological monitoring activities.

Resolution	Population/individual attribute measured	Relative cost
Coarse	Presence/absence of species (basic inventory)	Low
	Relative abundance (population index)	
	Absolute abundance (estimates and measures of precision)	
	Survival and reproduction (demographic processes)	
Fine	Genetic composition, biotoxin accumulation	High

ing of overwinter survival and onset of reproduction, alleged by Fuller (1969) and Taitt and Krebs (1985) to be a determinant of population eruptions in small mammals. Repeated sampling during the summer months will yield information on dispersal and movements of individuals among habitats, which may be a means by which animals avoid harsh winter environments. Dispersal by young also may be a mechanism by which populations avoid crowding.

Monthly data collection provides information on timing of reproduction and survival. Extreme weather events during the summer are not uncommon in Interior Alaska, and the impact of these events upon measures of population status could be assessed by monthly monitoring. Sampling only in early summer and late autumn still would allow estimation of survival and reproduction, but given the short lifespan of small mammals, recapture rates will be depressed.

Cost Issues

Capture and release of animals is, of course, more labor intensive than removal trapping, because animals must be handled delicately and live traps must be checked more frequently. Time necessary to place traps of either type is roughly equivalent, the added expense of live traps comes in emptying and processing animals. There also is a cost associated with the materials used for marking, which can range from toe-clipping to PIT tags.

In return for the added labor costs of mark-recapture studies, additional information may be derived. Estimation of survival rates is an obvious parameter that can be estimated only with live trapping. Statistically valid estimates also can be obtained repeatedly using live trapping instead of removal trapping. This is because of the "vacuum" created by removing animals from a study area. This has a carry-over effect of depressed abundance that persists until individuals have recolonized the trapping area where animals were removed.

Temporal information, such as seasonal weight dynamics and lifetime reproductive output, indicative of well-being of the small mammal community, only can be evaluated from live individuals.

Statistical Issues

The use of indices for detecting trends in population abundance is an exercise fraught with difficulties. Indices, by definition, have no associated measure of precision and are subject to sources of variation unknown to the investigator. Contrary to the advice of Halvorson (1984), catch indices are inappropriate for population monitoring. Indices make it impossible to place changes in abundance in a statistical framework, which is imperative for scientific investigations or sound management decisions. Without a statistical foundation, it is impossible to state, in a probabilistic fashion, whether populations are changing. All that can be stated is that populations were estimated to be twice as large one year as the next, but it is not possible to state whether the estimates are different (*see* Nichols 1986). Seasonal fluctuations, seen in our data, and reported by other investigators (Taitt and Krebs 1985) also present difficulties for trend analysis. To avoid these fluctuations, an investigator may attempt to "time" trapping occasions consistently from year to year. Unfortunately, there is no way to time such events; due to delayed or advanced spring thaw, or variability in winter severity. As a consequence, monitoring of small mammal populations must be conducted continually (i.e., on a monthly basis during the field season each year).

A more sensitive indicator of population health is productivity (i.e., survival and recruitment). These aspects of population dynamics provide a long-range prognosis for the health of the population. Population abundance in the future is dictated by these mechanisms, such that productivity measures are predictors of future abundance and are available in advance of abundance measures.

Productivity measures are available only through livetrapping studies. A sampling design, with trapping occasions spread throughout the field season, provides the mechanism to measure both survival and recruitment (both in situ and through dispersal) (Nichols and Pollock 1990), resulting in the most refined measure of population-level processes.

As resource managers are asked to produce an ever more up-to-the-minute appraisal of natural resources, population assessment takes on increasing emphasis. The potential for degradation of natural systems, whether from anthropogenic causes or natural perturbation, also is of increasing importance. It therefore is incumbent upon managers to design and implement monitoring programs in a manner that maximizes information gain from limited financial resources.

Acknowledgments

Funds for research projects were provided by the National Park Service and the University of Alaska—Fairbanks. The author benefited from discussions with numerous NPS and NBS personnel, as well as students, technicians and investigators engaged in monitoring programs ongoing in Interior Alaska. Volunteers, too numerous to count, made possible the collection of data at each of the study sites over the past two years.

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Special Session 3. *The Role of Hatcheries and Genetics in Fisheries Management*

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The Texas Fish Hatchery Experience: An Evolutionary Process

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Introduction

Should a fishery manager stock fish with the hope of making fishing better if there is even a minimal chance that stocked fish could further harm a depleted native fish fauna? On the other hand, how could a responsible fish manager, under pressure to improve fishing, justify not introducing a fish reasonably thought to have the potential to provide significant fishing opportunity? How could this same manager argue against introducing a fish likely to provide significant benefits in the face of habitat changes that have left the native fish fauna decimated and incapable of maintaining itself or of providing even minimal fishing opportunity? Similar questions have been asked of fish managers for more than a century.

Stocking has revolutionized fishing in America. Many fisheries have been restored or enhanced by stocking, thus providing tremendous enjoyment, economic opportunity and environmental improvement.

For anglers, a century of hatcheries and stocking has resulted in quality fisheries in waters once barren of fish or once populated by fish having no fishing value. For environmentalists, the issue of stocking non-native species for sport has become another call to action. For fishery administrators, fish stocking has become another in a long list of controversies over management of fishery resources.

At the center of this controversy is the use of hatcheries and hatchery-reared fish in fishery restoration and fish enhancement (Rosen 1986).

The Evolution of Stocking Programs in Texas

In the late 1960s and early 1970s, extensive experimentation with exotic fish species allowed managers to look at an array of potential new fishing opportunities. Single and multiple species fisheries were developed and tested, usually by trial and error.

Even more effort centered around stocking fish native to U.S. waters into areas where the species did not naturally occur. An example of this is the nationwide distribution of walleyes and rainbow trout. Stocking of sport species into newly formed reservoirs created many sport fisheries where none previously had existed. In many western states, stocked fish provide the majority of all fishing, because few native fishes offer sport fishing opportunities.

As the population in western states grew, demand for water supplies increased. In Texas, a frenzy of reservoir building began in the 1920s. Rivers were changed into reservoirs, and riverine fisheries soon were replaced by fisheries more adapted to lakes.

With dependable water supplies came more people, and with increases in population came increased demand for recreational fishing. The vast new man-made lake-like environments created by the impoundments provided great fishing and tremendous opportunity for anglers.

Reservoir construction peaked in the 1950s and 1960s. Texas now has over 2 million surface acres (809,400 ha) of impounded freshwater contained in over 600 public reservoirs and in thousands of smaller private ponds. The state contains nearly 200 reservoirs greater that 500 acres (202 ha) in size.

The newly formed reservoirs were filled with fish produced in hatcheries. Species stocked were selected because they would thrive in the artificial lakes. These stocked fish have provided fisheries where none existed before. Hatcheries and the fish produced in those hatcheries are responsible for creating what is arguably the best all-around freshwater fishing in the nation.

In more recent years, Texas has built on its successes in freshwater to pioneer the use of hatcheries to restore depleted populations of saltwater species. Two case histories that illustrate the success of hatchery and stocking programs in Texas are presented below.

Largemouth bass. Largemouth bass were stocked in Texas even before the era of intensive reservoir building. As long ago as 1893, largemouth bass from federal fish hatcheries in Virginia, Illinois and Missouri were stocked in rivers and private ponds in the state. Those stockings, most likely composed entirely of northern largemouth bass (*Micropterus salmoides salmoides*), continued through the turn of the century and were in full force when reservoir construction began. Consequently, all major and most minor reservoirs in Texas received stockings of largemouth bass from either in-state or out-of-state sources.

The human population of Texas, though growing, was still relatively small during the peak of reservoir construction in the 1950s and 1960s. At that time, the supply of good fishing sites and fish exceeded demand.

This situation changed in the 1970s. Human population rose sharply and the growing popularity of bass fishing placed tremendous pressure on bass populations. Bass fishing, measured in terms of number and size of fish caught, declined in many waters. Eschmeyer's (1955) theory that panfish could not be overharvested in large southern impoundments was being seriously challenged. Examples of overfishing in new impoundments were numerous and examples of growth overfishing in older established impoundments were increasing. The quality of the bass fishery began to decline. Texas biologists began looking to other fishery management tools, including stocking the Florida subspecies of largemouth bass (*Micropterus salmoides floridanus*), to bolster the failing bass fishery.

Florida bass attracted the interest of Texas biologists because of the rapid growth and large size of the subspecies in Florida waters. Originally, their larger size was assumed to be related to Florida's longer growing season and plentiful food supply. The performance of the Florida bass in several California lakes, where they were stocked in the late 1950s and early 1960s (Bottroff and Lembeck) 1978), gave the first indication that the difference in growth rate and maximum size was genetic and not environmentally controlled.

Evaluation in Texas of the two species and their hybrids showed that the Florida subspecies, and their hybrids with the northern subspecies, grew faster over a three-year study period (Inman et al. 1978). Greater growth was concluded to be due to genetic factors. The State of Texas embarked on an extensive Florida bass stocking program in the late 1970s and 1980s. The rationale behind the program was the documented faster growth of the Florida subspecies in Texas, the documented greater maximum size observed in its native range and in California, and the apparent greater fitness of subspecies in lakes.

Texas biologists were seeking to establish a fishery composed of faster growing largemouth bass with greater maximum size potential. With commensurate fishing regulations, this theoretically would increase the number of trophy-sized bass available to anglers and increase standing stock of largemouth bass in Texas impoundments.

Since 1972, when 35,700 Florida bass were introduced experimentally into two reservoirs, over 66 million Florida bass fingerlings have been stocked in over 300 impoundments.

The successful introgression of the Florida subspecies into established northern bass populations has had a profound effect on largemouth bass populations and bass angling in Texas. Evaluations conducted from 1991 through 1993 on 139 reservoirs stocked with Florida bass indicated a mean of 35.6-percent occurrence of Florida largemouth bass alleles in the population. Only 4 of the 139 reservoirs sampled showed 0-percent allele frequency. Five reservoirs had over 90-percent allele frequency.

The effects of the introductions on the production of large bass has been even more dramatic. The state record for largemouth bass, which stood at 13.5 pounds (6.12 kg) from 1947 to 1981, has been broken five times and now stands at 18.18 pounds (8.25 kg). Twenty-seven reservoirs in Texas now have lake records equal to or surpass the previous 13.5 pounds (6.12 kg) record. The number of reservoirs yielding largemouth bass larger that 8 pounds (3.6 kg) increased from only 2 in 1974 to 52 in 1993.

The number of Angler Recognition Awards for catches of largemouth bass greater than 8 pounds (3.6 kg) also increased from less than 15 per year in the early 1970s, when the award program began, to 353 in 1993. The annual mean weight of largemouth bass submitted to the program during that period also increased from about 8.3 pounds (3.8 kg) to nearly 11 pounds (5 kg).

Although much of the increased standing stock of largemouth bass now observed in Texas reservoirs can be credited to more prohibitive length and bag limit restrictions implemented during the period, the maximum size of bass today is the result of the stocking and introgression of Florida bass. In addition, fixation of the Florida bass allele in the population shows that Florida bass are at least as suited to Texas reservoir environments as are northern bass, if not superior in fitness.

Another measure of success of the program has been the strong acceptance by anglers. A mail survey of 1,965 black bass anglers during spring 1992 showed over 83 percent of Texas black bass anglers were moderately to extremely satisfied with black bass fishing in the state (Texas Parks and Wildlife Department unpublished data).

Red drum. During the 1970s and 1980s, red drum (*Sciaenops ocellatus*) along the Gulf of Mexico coast were subjected to heavy fishing by commercial and recreational fishermen. Overharvest led to growth and possibly recruitment overfishing. Texas fishery managers embarked on a long-term program to increase red drum populations to historical levels.

In the early 1970s, a three-part recovery plan for red drum was developed. First, an independent monitoring program to assess relative abundance was implemented. Second, restrictive regulations were enacted to reduce fishing pressure, including a ban on sale of red drum and use of nets, and a bag limit of three fish (20–28 inches long) per day for sport fishermen. And third, an enhancement program was started based on release of hatchery-reared fingerlings and assessment of subsequent survival.

Life history studies showed that red drum was an excellent candidate for stock enhancement. Stocking and rearing methodology was developed by state and university researchers (Arnold et al. 1977, Roberts et al. 1978, Colura et al. 1976, McCarty et al. 1986).

In 1983, a partnership between the Central Power and Light Company, the Gulf Coast Conservation Association, and the Texas Parks and Wildlife Department (TPWD) resulted in the first modern-day, large-scale marine fish hatchery in the nation. Fingerlings produced in early years were used in studies to assess handling methods and survival in the wild. Survival in hauling trailers was 99 percent (Tommasso and Carmichael 1988) and fingerlings stocked in the bay showed an 86-percent, 24-hour survival rate (Hammerschmidt 1986). Fingerlings did well and could be identified up to nine months following stocking (Matlock et al. 1986).

Additional research centered on the wild population of red drum in the bays and on recreational harvest. Stocking was shown to increase the abundance and angler-catch rates of red drum (Matlock 1990). These findings cleared the way for large-scale stocking of red drum in Texas coastal waters.

To date, more than 100 million red drum fingerlings have been stocked. Because mass stocking was and still is controversial, TPWD and outside researchers have conducted extensive studies on genetic makeup of the natural population and fish stocked.

Absence of spatial or temporal allelic heterogeneity among inshore and offshore red drum suggests a randomly mating population in the Northern Gulf of Mexico (Wakeman and Ramsey 1988, Bohlmeyer and Gold 1991, Gold and Richardson 1991, Gold et al. 1993, Gold et al. in press), therefore stocking should have little overall effect on the natural population.

Despite extensive stocking in Texas, genetic variability (e.g., heterozygosity and halotype frequencies) was similar in red drum collected from North Carolina to southern Texas. This lack of reduction in genetic variability suggests no impact on average variability has resulted from the Texas supplemental stocking program. Monitoring for effects of hatchery-raised fish on the natural population continues.

The attention to genetic factors in the stock enhancement program can serve as a blueprint for future enhancement efforts elsewhere. Broodfish are used in spawning and quickly rotated out of the program and replaced with fresh fish from the wild. In addition, male and female pairings are changed after each spawning period to achieve maximum genetic diversity. Genetic makeup of fish stocked is monitored to document any change in genotype through time.

Several tools are being used to assess the effectiveness of stock enhancement. Oxytetracycline is being used to mark otoliths of stocked fish (Bumguardner 1991) in an effort to provide a way to distinguish stocked fish from wild fish. A computer based Optical Pattern Recognition System is being used to compare scales of stocked and wild fish. Genetic research has revealed a unique allele (King et al. 1993) that is not detrimental (Texas Parks and Wildlife Department unpublished data) and can be used to follow marked fish throughout their life.

One recent study showed that 20 percent of the juvenile red drum caught in bag seines in upper Laguna Madre were stocked fish (Texas Parks and Wildlife Department unpublished data). Assuming survival of stocked fish is similar to that of wild fish, up to 20 percent of fish ultimately caught by sport fishermen may be of stocked origin.

While there is still much to learn about artificial enhancement of red drum stocks, the effectiveness of the stock enhancement program to date shows that continued stocking may help reduce or eliminate the wide variations in recruitment from year to year that have characterized the red drum fishery. Recreational fishermen and the recreational fishing industry will benefit from a stable red drum population, but stocking alone will not be enough. Conservative harvest regulations and coastal habitat protection must be essential elements in any comprehensive plan to enhance, stabilize and protect red drum stocks and red drum fishing.

Conclusion

Water bodies and fisheries can be managed in multiple ways to produce multiple benefits. Texas fisheries management reflects this philosophy. Some members of the public disagree with multiple-use and active fishery management, and favor instead a "back-to-nature" approach that seems to discount the century or so of human pollution, dam building and species introductions, and the desire of the nation's anglers for quality fishing opportunities.

While many managers would give almost anything to restore altered and polluted aquatic environments, and managers do work hard to protect native fish species, we simply cannot turn the clock back. There is no cost-effective and technically feasible way to restore many waters and eradicate nonindigenous species. Thus, managing existing altered waters is likely to continue, with the use of hatcheries among prominent and necessary management tools. Hatcheries also can play a prominent role in preserving the genetic integrity of rare and endangered species, as well as provide the means to restore a fish species to the wild.

Too often discounted in the current debate over the use of hatcheries is the tremendous effect that destruction of aquatic habitat has had on native fish. Pollution, dam and reservoir construction, agricultural withdrawals of water and other farming practices, and poor mining, forestry and grazing practices have all taken a significant toll.

Nonetheless, fishery managers are in general agreement that hatcheries are a poor substitute for protection of aquatic habitat. Too often, politicians and construction agency officials have viewed hatcheries as an acceptable substitute for habitat destroyed. Fishery managers rarely proposed such "substitutions," but when faced with the prospect of a reservoir and sure destruction of stream fisheries, fishery managers took the only rational course. In the past, there simply was no stopping a dam or major construction project, and it was hatcheries that provided for the creation of new and often spectacular fishing opportunity.

While some stocking programs have been detrimental effects, these effects generally have been significant because fish stocked were genetically or otherwise incompatible with the fisheries into which they were mixed. On the other hand, important fisheries have been created or maintained due to stocking programs.

Hatcheries can provide fishing opportunity where no other methods will work. Intensive fisheries, such as those being created in urban environments, often require hatchery support. Anglers measure fishery management success by the size and number of fish in their catch. Growth of the fishing-related economy depends on continued improvements in fish populations.

Today's public fishery management programs must be responsive to the needs of the fishing and nonfishing public. This speaks to the need for a variety of management options, applied where needed, to address the biological, political, social and economic needs of a nation hungry for recreational opportunities and committed to improving environmental quality. Hatcheries are not inherently evil or good. Like any tool, used properly and in skilled hands, hatcheries can create wonderful fishing. Misuse of hatcheries can be wasteful of public investment, cause harm and destroy potentially productive fish communities.

Fishery administrators seeking to use hatcheries first must be committed to protecting fishery habitat, controlling harvest and foregoing political expediency when the hatchery "fix" to a fishery problem is demanded wrongly.

The use of hatcheries in Texas to rear red drum, Florida largemouth bass and other species has received world-wide recognition. Efforts to develop a long-term strategy to ensure genetic integrity of the coastal red drum stock and to monitor the genetics of bass, while maintaining a program to upgrade as new information becomes available, is necessary in today's environmentally conscious era. Texas Parks and Wildlife Department biologists believe that stocking, in conjunction with other management tools, has created better fishing. The Texas experience has proven to be one of success, as well as one of long-term commitment to fishermen and the future of fishing.

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The Role of Fish Hatcheries in the Sport Fisheries of the State of Alaska

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Introduction

Alaska contains millions of acres of pristine wilderness, innumerable lakes and thousands of miles of streams. In addition, it contains more miles of coastal shoreline than the rest of the contiguous 48 states combined. Sport fishing in Alaska is an important and growing activity for thousands of visitors and tourists alike. A total of 428,768 sport anglers fished in Alaska in 1992. Alaska residents made up 57 percent of the total; nonresidents comprised 43 percent. For the first time, in 1992, nonresident anglers spent more for licenses than did resident anglers. Sport anglers in 1992 caught an estimated 4.8 million fish, of which 2.0 million were harvested (Mills 1993).

During the early years of statehood, from 1960 to the early 1970s, uncrowded and easily accessible sport fisheries in Alaska were numerous. Today, many sport fisheries are crowded by a growing urban population and a steadily rising number of visitors.

The ability of many aquatic ecosystems to meet the harvest demands of sport anglers has been exceeded in many of the more popular sport fisheries. As a result, numerous popular fisheries have been closed or restricted to maintain or preserve specific fish populations. Harvest restrictions may exacerbate the problem by forcing anglers to fish more remote fish stocks, thereby increasing harvest pressure on less accessible populations of fish. A cycle is created by these effort shifts when increasing numbers of fisheries are restricted or closed.

In the last 15 years, Alaska has developed an extensive enhancement program to benefit recreational anglers. Initially, the sport fish enhancement program was a hodgepodge of individual projects designed to ''make fish for people to catch.'' Some projects were beneficial and well designed, others were not. The only cost consideration was whether there was enough money to produce the fish. A detailed, long-term statewide plan for fish stocking activities has not existed until recently. Likewise, guidelines for evaluating project effectiveness were nebulous and inconsistent. The purpose of this paper is to outline the hatchery role in Alaska sport fisheries, and to discuss procedures which have been developed to govern hatchery operations in Alaska.

The Alaska Sport Fish Hatchery Program

The State of Alaska operates four hatcheries (Elmendorf and Fort Richardson Hatcheries in Anchorage, Clear Hatchery near Fairbanks, and Crystal Lake Hatchery in Petersburg) almost exclusively with Dingell-Johnson (D-J) and Wallop-Breaux (W-B) funds and state fees collected from the sale of sport fishing licenses. These hatcheries produce most of the fish stocked to benefit recreational fisheries and their supporting industries.

In the past, several other state-operated hatcheries raised fish for stocking programs to benefit sport anglers. Some of these hatcheries received D-J and W-B funds as compensation for the sport fish components of production. Recent changes to the State of Alaska's hatchery program resulted in most of these hatcheries being transferred to the private sector or closed. Individual sport fish stocking projects that depend on these hatcheries either have been discontinued, moved to one of the state hatcheries or taken over by the new hatchery operator. Other federal and privately owned hatcheries raise small numbers of fish for sport fisheries enhancement projects. D-J funding is not involved in these projects. Some sport fisheries have developed on private hatchery commercial fisheries stocking projects. These incidental sport fisheries are not considered part of the Alaska sport fish hatchery program. However, sport fish effort and harvest levels on some of the incidental sport fisheries are larger than some of our planned sport fisheries.

Almost 11 million fish were stocked in Alaska waters to benefit sport anglers in 1992 (Table 1). Six different species currently are raised and released: coho salmon (*Oncorhynchus kisutch*), chinook salmon (*Oncorhynchus tshawytscha*), rainbow and steelhead trout (*Oncorhynchus mykiss*), Arctic grayling (*Thymallus arcticus*), Arctic char (*Salvelinus alphinus*), and lake trout (*Salvelinus namaycush*). More than 3.7 million anadromous fish were released, while about 7.3 million fish were released into landlocked lakes. Fort Richardson Hatchery produced 42.5 percent of the total number of fish released. Rainbow trout were the most utilized species and, with grayling, accounted for 51.2 percent of the total fish released and 77.5 percent of the landlocked fish stocked. Chinook salmon comprised 65.4 percent of the anadromous fish stocked.

Types of Sport Fish Stocking Programs in Alaska

Stocking programs are selected to maximize benefits to sport anglers through rehabilitation, enhancement or development. Specific programs are intended to (1) supplement a depressed stock (rehabilitation), (2) increase the number of fish caught beyond historic levels (enhancement), or (3) establish a new fishery (development) (Alaska Department of Fish and Game 1993).

Due to the healthy condition of most Alaska fish stocks, only a few sport fish rehabilitation programs are being carried out in Alaska. An example of rehabilitation is the Chena River Arctic grayling program. The Chena River and its tributaries once supported the largest recreational Arctic grayling fishery in North America (Clark 1991). Estimated annual harvests during the late 1970s through the early 1980s ranged from 20,000 to 40,000 fish. This level of exploitation, combined with poor survival of juvenile fish during the mid-1980s, dramatically reduced this population. Regulatory harvest restrictions failed to protect these fish or provide sustainable harvests. Consequently, a no-harvest policy was imposed in 1991 and stocking using brood stock from the Chena River was initiated to accelerate rebuilding of the population. The stated objective of the program is to rebuild the Chena River Arctic grayling population to a level which, by 1995, will support a sustained annual harvest of 10,000 or more Arctic grayling.

Several enhancement programs are being conducted throughout the state. However, no new enhancement programs are planned. All existing enhancement programs are

Table 1. Numbers of hatchery produced fish stocked in Alaska in 1992 to enhance sport fisheries.

Hatchery	Species	Number Landlocked	Anadromous	Total
Clear	Arctic char	435,670		435,670
	Grayling	2,185,618		2,185,618
Big Lake	Coho salmon	215,000	215,000	430,000
Crooked Creek	Chinook salmon		273,000	273,000
	Coho salmon	246,000	74,000	320,000
	Steelhead		39,700	39,700
Elmendorf	Chinook salmon		1,289,000	1,289,000
	Coho salmon	349,000	283,000	632,000
Fort Richardson	Chinook salmon	200,000	359,000	559,000
	Coho salmon	187,000	515,000	702,000
	Rainbow trout	3,406,971		3,406,971
Deer Mountain	Coho salmon		162,000	162,000
	Rainbow trout	34,200		34,200
	Steelhead		1,030	1,030
Crystal Lake	Chinook salmon		520,000	520,000
Total by species	Arctic char	435,670		
	Grayling	2,185,618		
	Rainbow trout	3,441,171		
	Steelhead		40,730	
	Chinook salmon	200,000	2,441,000	
	Coho salmon	997,000	1,249,000	
Grand total		7,259,459	3,730,730	10,990,189

being evaluated intensively to ensure none of the enhanced populations are being impacted negatively by introduced fish. The Willow Creek chinook salmon program is representative of enhancement. Willow Creek is an easily accessible chinook salmon stream located near Anchorage. The chinook salmon sport fishery was closed during the 1970s due to poor returns. A "weekend only" fishery was initiated in 1979 and 285 fish were harvested. Smolt of Willow Creek brood stock were released beginning in 1983 to increase chinook salmon fishing opportunities by supplementing the stream's natural run with hatchery fish, while maintaining the present quality and quantity of natural chinook salmon production. In 1992, 11 additional days of sport fishing were added to the "weekend only" season. Over 18,000 angler-days of fishing effort were expended to harvest approximately 7,000 chinook salmon. More than half of the fish harvested were of hatchery origin and natural production was at a historically high level (Peltz and Sweet 1993).

Most stocking programs to improve sport fisheries in Alaska create new fisheries where none previously existed. These "development programs" are preferred because there is no or minimal interaction between hatchery and wild stocks of fish. Southcentral Alaska non-anadromous lake stocking is a typical example of a development program. This program was initiated in the 1950s to create new fisheries in lakes where game fish were not present. Rainbow trout, Arctic grayling, Arctic char, lake trout, and landlocked chinook and coho salmon are stocked annually. In 1990, approximately 2.7 million fish were stocked in 173 southcentral Alaska lakes (Alaska Department of Fish and Game 1993). Approximately 128,000 angler-days of sport fishing effort were reported from stocked lakes in 1990 (Mills 1991), resulting in a catch of 299,000 stocked fish of which 109,600 were harvested.

Planning Sport Fish Hatchery Production

Planning should be the first stage in developing a recreational fisheries stocking program. Sport Fish Division recently has standardized a method for planning fish stocking programs. A fishery management plan is prepared during the initial stage of planning. Each fishery management plan lists the following: management objectives to be met by fish stocking, specific measures required to accomplish the objectives and performance criteria that will be used to evaluate whether objectives are achieved. Management objectives recently have been defined in terms of benefits and currently are measured in angler-days (one angler fishing for any portion of a day) of fishing effort. Maintenance or increase in fishing effort due to stocking is a measure of performance and provides an indicator of program success. Specific stocking actions are the numbers of fish and locations for stocking. Performance evaluation criteria require a listing of parameters to be measured (fishing effort, harvest, catch, etc.) and how they will be measured (creel survey, Statewide Harvest Survey, harvest cards, etc.). A single fishery management plan may cover numerous stocking sites over a broad geographical area or a single stocking site.

The second stage in developing a recreational fisheries stocking program is to ensure that fish production in the hatcheries matches fish production demands in the fishery management plans. On a periodic (4–5 years) basis, all sport fisheries management plans which address fish stocking are incorporated into a Statewide Stocking Plan for Recreational Fisheries (SSP). The SSP contains specific information about each stocking location; region of the state, Division of Sport Fish Management Area, reference to a sport fishery management plan which covers the stocking location, release site, species to be released, whether the location is anadromous or landlocked, size of fish to be stocked and number of fish to be stocked each year. If demand for hatchery fish exceeds hatchery capacity, projects are prioritized and fish are allocated to the most important projects. Time is allowed for public viewing of the draft plan. The plan becomes finalized when it is approved by the Commissioner of the Alaska Department of Fish and Game (Department). The SSP finally is submitted to the Regional Director of the U.S. Fish and Wildlife Service for approval, since the major funding source for the projects in the SSP is federal money administered through the U.S. Fish and Wildlife Service (D-J and W-B monies).

The recreational stocking program changes frequently to adjust to success or failure of prior fish plants, angler preferences, acquisition of public lands, human population growth, availability of funding, hatchery limitations and recreational trends. Consequently, changes to the SSP are inevitable and to the extent possible anglers and the general public are alerted to any significant departures from the plan. Most changes appear in an update to the SSP which is made available to the public annually. Due to complexities of long-term rearing of fish in a hatchery, it is unusual to have exactly the planned number of fish for each location available for stocking. It often is necessary for professional staff of the Department to make minor changes in fish numbers, fish species or stock, or exact release location to accommodate variables in fish production.

Regulation of Sport Fish Hatchery Production

The final stage in developing a recreational fisheries stocking program is regulatory review. The State of Alaska strictly regulates transportation, possession or release of live fish in the state. Regulations have existed since the Alaska hatchery program expanded in the 1970s. These regulations are part of the Alaska Administrative Code (Title 5, Chapter 41) and are thus state law. Two specific regulations form the backbone of the fish stocking regulatory process.

The first regulation (5 AAC 41.070.) prohibits importation of any live fish into the state for purposes of stocking or rearing in the waters of the state. Ornamental fish not raised for human consumption or sport fishing purposes may be imported into the state, but may not be reared in or released into the waters of the state. This regulation prohibits introduction of nonindigenous species or stocks of fish into the state.

The second regulation (5 AAC 41.005.) makes it unlawful to transport, possess, export from the state or release into the waters of the state, any live fish without a Fish Transport Permit or FTP. A FTP is issued for a fixed term and authorizes only that operation specified in the permit. Any change of species, brood stock or location requires a new permit. Each applicant for a FTP submits the following information to the Department (5 AAC 41.010.):

(1) species and stock involved;
(2) incubation, rearing and/or release site(a);
(3) number and life history stage involved;
(4) history of previous transport, if any;
(5) disease history of the stock, hatchery or rearing facilities involved, any previous disease treatment or vaccinations, or, if the disease history is incomplete or unavailable, a brood stock inspection and certification;
(6) isolation measures planned to control disease;
(7) description of proposed eggtake methods;
(8) source of water for rearing and proposed effluent discharge location;
(9) identification and status of native stocks involved;
(10) method of transport of release and the expected date of transport or release;
(11) purpose and expected benefits of the project; and
(12) evaluation plans.

Each FTP application is reviewed by the Department. A FTP is issued if it is determined that the proposed transport, possession or release of fish will not adversely affect the continued health and perpetuation of native, wild or hatchery stocks of fish. Terms and conditions may be attached to the FTP if it is determined that terms and conditions are necessary to protect the continued health and perpetuation of native, wild or hatchery stocks of fish. A FTP can be denied if the proposed plans, methods or specifications are not adequate, on the basis of fish disease, genetics, competition, predation or other biological considerations, and to assure the continued health and perpetuation of native, wild or hatchery stocks of fish (5 AAC 41.030).

In addition to regulations, there are Department policies that apply to fish stocking programs in Alaska. The State of Alaska's genetic policy for salmon (Alaska Department of Fish and Game 1985) addresses stock transports, protection of wild stocks and maintenance of genetic variability. The genetic policy is reviewed as part of the FTP application process. The State of Alaska also has adopted a policy relating to fish health and disease control (Alaska Department of Fish and Game 1988). This policy is intended to prevent dissemination of infectious finfish and shellfish diseases within or outside the borders of Alaska without introducing impractical constraints for aquaculture and necessary stock-renewal programs. Again, the FTP process serves as a forum for reviewing fish health and disease control policies as well as regulations. The last policy of note which influences sport fish stocking programs in Alaska exists only in draft form. The Division of Sport Fish wild stock protection policy still is being formalized, but the intent of the policy is clear. Sport Fish Division will not stock hatchery fish in locations where wild stocks of sport fish occur unless: (a) the indigenous wild stock(s) is incapable of supporting a recreational fishery; (b) the indigenous wild stock(s) is important to sport anglers and is found to be depressed; or (c) adequate evaluation can be dedicated to the stocking project to maintain historical levels of natural production, run timing and spawning distribution. As previously mentioned, Sport Fish Division will not initiate any new enhancement stocking programs until evaluation from existing programs has thoroughly documented impacts on indigenous wild stocks of fish. The wild stock protection policy generally is reviewed for compliance as the fishery management plan is being composed.

Review of Sport Fish Hatchery Production

Mechanisms for review of a sport fish hatchery program have been built into the planning and regulation processes. The fishery management plan for each program usually lists a time period for reviewing achievement of program objectives. In addition, program costs during the time period are summarized. Measured objectives are combined with program costs to provide a measure of efficiency (cost per angler-hour of effort generated or fish harvested). Attainment of objectives and measurement of efficiency provide the primary basis for program review. If objectives are achieved and/or the program efficiency is adequate, the program is considered a success and the existing fishery management plan remains in effect with a new time period established for future review. If primary objectives are not achieved and/or the program efficiency is poor, the program is considered a failure and is terminated. If some of the objectives are achieved and/or the program efficiency is marginal, the program is closely scrutinized to determine: (1) if project objectives are realistic, (2) if program changes might increase the possibility of attaining stated objectives and/or improving program efficiency, or (3) if some aspect of program performance is adequate to justify continuing the stocking program. If any changes are made, the fishery management plan is modified accordingly and a new time period is established for further review.

Adherence to stipulations outlined in the issued FPT, and project compliance with Department regulations and policies also are periodically reviewed. The FTP is issued for a fixed term after which a new FTP is required for the program to continue. Renewal of a FTP is reviewed as thoroughly as a new FPT. As previously mentioned, a FTP will be denied if the stocking program doesn't conform to state regulations or Department policies.

Most fish stocking programs have not yet received a thorough review. Fishery management plans for stocking programs were written within the last few years and the time period for review of most programs has not been reached. Total program costs which are used to help measure program efficiency only recently have been monitored. Numerous FTP's have expired and new FTP's have been reviewed for compliance with state regulations and policies. Within the next five years all sport fish stocking programs should receive thorough review. It should be evident after the first round of reviews whether or not the existing review process is adequate to produce good fish stocking programs while protecting wild populations of fish.

Summary

The State of Alaska has an extensive fish stocking program conducted for the benefit of anglers in Alaska. Most programs are easily accessible in highly populated areas where angling pressure on native stocks has exceeded natural production capability. Stocking programs serve two primary purposes. The first is to maintain or increase historic levels of angler participation and harvest. The second has been to protect other accessible wild populations of fish that, in the absence of hatchery fish, would be subjected to unsustainable harvests. Fish produced from the stocking program have satisfied many anglers' desires to catch fish. Consequently, wild populations of fish have been spared from the angler effort directed at stocking programs.

In order to ensure that all sport fish stocking programs provide benefits while protecting existing fish populations, Sport Fish Division of the Alaska Department of Fish and Game recently has assembled the final pieces of a Fish Stocking Program Management Plan (Figure 1). The three main components of the plan are planning, regulation and review. Planning involves preparation of a fishery management plan for each fish stocking program and assembling all plans into a Statewide Stocking Plan. Regulation entails applying for a FTP. The FTP application is reviewed for compliance with all State of Alaska regulations and policies. Issuance of a FTP is the equivalent of granting a license for the program to begin. Review is a periodic visit back to the planning and regulation components. Achievement of the program objectives in the fishery management plans and measurement of program efficiency are the focal points of review. Likewise, compliance with state regulations and policies are reviewed. Completion of the review process will mean the fish stocking program will continue as is, continue with modifications or be discontinued.

Acknowledgments

I would like to thank Mike Haddix, Doug McBride, Al Havens, Craig Whitmore and Tim McDaniel for their comments and suggestions. I am indebted to Bob Lafferty for his assistance with graphics. I am deeply appreciative of the editorial review and moral support provided by Bill Arvey and John Burke.

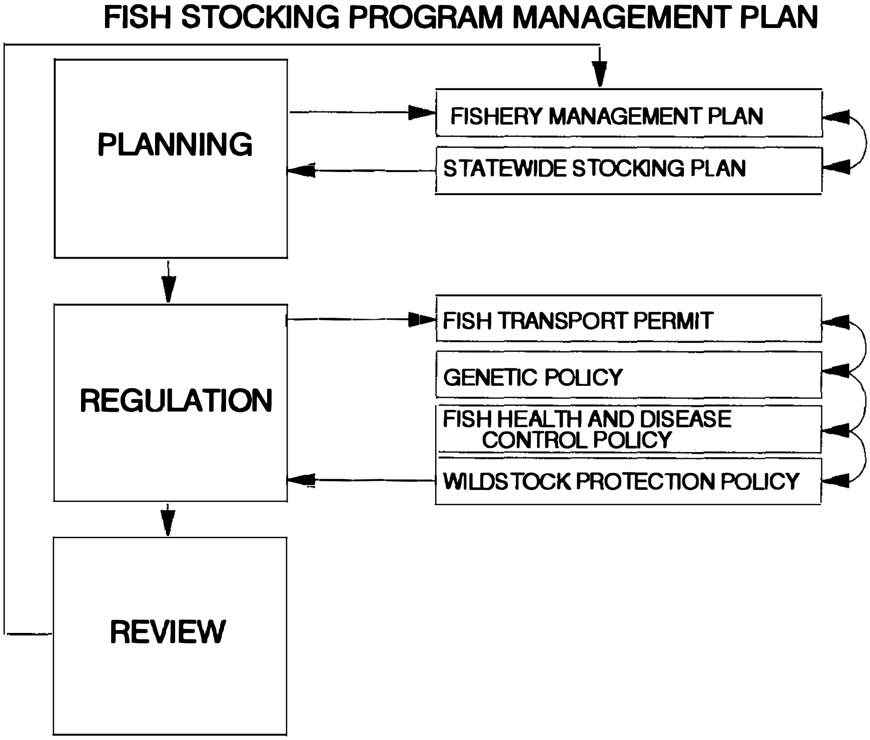

Figure 1. Diagram of the Alaska Department of Fish and Game, Sport Fish Division, fish stocking program management plan.

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Studies of Alaska's Wild Salmon Stocks: Some Insights for Hatchery Supplementation

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Introduction

"To characterize management of wild stocks as controversial would be a considerable understatement... one thing upon which we agree is that these valuable resources have been taken for granted for too long... both managers and scientists have many commitments to make and promises to keep before anyone can feel comfortable with the fate of our wild fish" (Walton and Houston 1984). A decade has passed since these and other similar concerns were voiced at forums such as the Olympic Wild Fish Conference (Walton and Houston 1984) and the Wild Salmon and Trout Conference (Washington Environmental Foundation 1983) where concerns for wild salmonid stocks (*Oncorhynchus* sp.) in the Pacific Northwest were brought to focus. Since then, the body of literature associated with wild stocks has grown exponentially, but we still see serious declines in populations. Konkel and McIntyre (1987) found that 13 percent of Pacific anadromous salmonid stocks declined between 1968 and 1984. Eighty-four percent of declining stocks were located in Washington, Oregon and California. The American Fisheries Society (Nehlsen et al. 1991) lists 214 native stocks of Pacific salmon, steelhead and sea-run cutthroat as depleted, with 101 at high risk of extinction.

Restoration or enhancement of wild stocks through use of hatcheries has a long history in the Pacific Northwest (Kelly et al. 1990). However, this strategy is under an active debate in the fisheries profession (Martin et al. 1992, Hilborn 1992), centered around documented or suspected impacts of hatchery activities on wild stocks. Recommendations have been made to consider genetic diversity of wild stocks and genetic-based approaches to management (Kapuscinski and Philipp 1988, Waples et al. 1990) and, in part, implemented through various state policies as reviewed by Kelly et al. (1990) for the Pacific Northwest.

Salmonid populations of Alaska represent a different picture from much of the Pacific Northwest with respect to status of wild stocks, history of hatchery influence and management agency perspectives, but the wild stock issue still exists (Thomas and Mathisen 1993). Five species of Pacific salmon occur naturally—pink (*O. gorbuscha*), sockeye (*O. nerka*), chum (*O. keta*), chinook (*O. tshawytsha*) and coho (*O. kisutch*). Only 6 percent of the 489 Alaska stocks analyzed by Konkel and McIntyre (1987) showed decreasing escapement trends, although a new effort to define stocks at risk is underway (Tim Baker personal communication: 1994). Alaska has many wild stocks that have had limited hatchery influence. Also, an active state genetics program supports genetics policies established in 1985 (Alaska Department of Fish and Game [ADFG] 1985). As such, characteristics of these stocks may provide valuable insights for efforts to reestablish viable salmon populations in other parts of their range or identify areas of caution in applying hatchery techniques. We summarize the history of hatcheries in Alaska, outline the federal resource management perspective, highlight scientific concerns and present examples where local adaptations of salmonids have and have not been reflected in measured genetic variation.

Historical Perspective

Efforts to ''enhance'' natural production of salmon in Alaska commenced more than 100 years ago (Roppel 1982). However, most early attempts failed because of a poor understanding of the unique life history requirements of salmon. Federal hatcheries operated through the 1920s, but closed in the 1930s, with one experimental hatchery operated through the 1950s (Kelly et al. 1990).

In the 1970s, the State of Alaska initiated an enhancement program and began permitting private nonprofit salmon hatcheries. The state currently leads North America in production of artificially propagated salmon (Holland et al. 1993). As of 1989, Alaska had 41 aquaculture facilities, many of which are located on, adjacent to, or enhancing wild salmon stocks originating from federal lands (Figure 1). Production of salmon by aquaculture facilities has increased steadily since the mid-1970s with releases now approaching 1.4 billion fish annually (Seeb 1993).

Enhancement has taken various forms in Alaska, including habitat rehabilitation and lake fertilization. New runs have been established through introductions using non-indigenous broodstock that can be self-perpetuating (Blackett 1979). In some cases ''terminal'' fisheries are created where salmon are imprinted to a non-natal area for ''complete'' harvest (Clark et al. 1993). Either native or non-native cohorts can be used to supplement production where returns are weak. However, the most common method used in Alaska, and in compliance with the state's genetics policy (ADFG 1985), is the use of native broodstock. Eggs are taken from returning adults, incubated in hatcheries and released as fry to their natal area.

Unlike the rest of the Pacific Northwest, no federal hatchery program exists in Alaska, but federal lands provide critical spawning and nursery areas. For example, almost 70 percent of the sockeye salmon in Cook Inlet originate on U.S. Fish and Wildlife Service or Forest Service lands. These salmon are an international resource with young migrating into the Gulf of Alaska and mingling with fish from British Columbia, Washington and Oregon.

A Federal Perspective in Alaska

The federal perspective on preservation of wild stocks is multifaceted, but in Alaska focuses primarily on a land-management and research role.

The Land Manager

Conservation and management of salmonid resources in Alaska exist in a framework forged by Alaska's unusual land ownership patterns and recent legislative history. Federal holdings of about 245 million acres ($\approx$ 1 million km^2) are managed

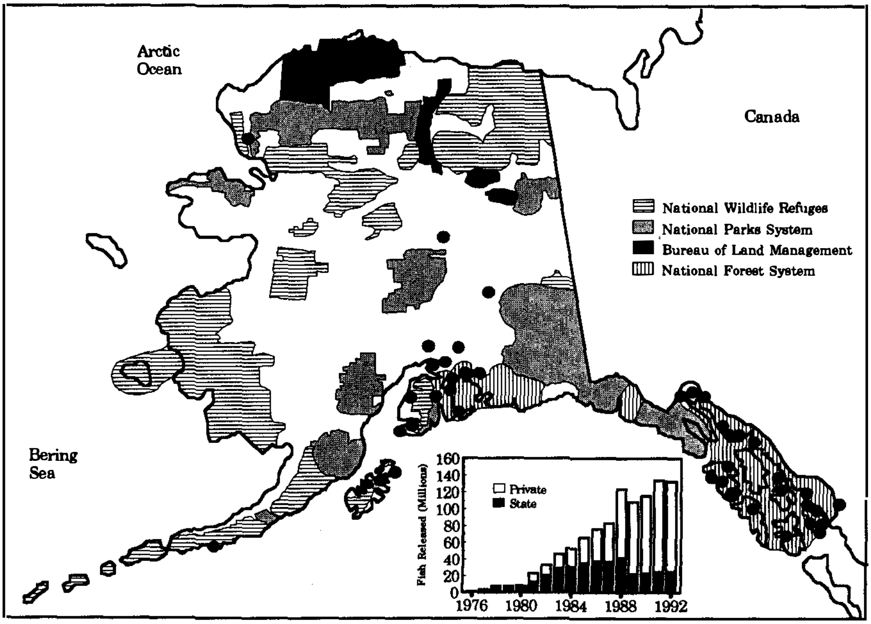

Figure 1. State and private nonprofit aquaculture facilities (circle) in Alaska, in relation to primary federal land holdings (insert illustrates trends in fish released from these facilities, 1976 through 1992).

primarily by the U.S. Fish and Wildlife Service (31 percent), National Park Service (22 percent), Bureau of Land Management (37 percent), and U.S. Forest Service (9 percent). Many of these holdings were created or expanded by the Alaska National Interest Lands Conservation Act of 1980, amended 1988 (Public Law 96–487). The U.S. Fish and Wildlife Service and the National Park Service received guidance significant to the wild stock issue, such as to conserve fish and wildlife populations and habitats in their natural diversity, and to protect populations of fish and wildlife and their habitats. In addition, both agencies have national policies to consider the natural abundance, diversity and ecological integrity of native animals.

National Biological Survey

In 1993, the Secretary of Interior consolidated research components of several agencies and established the National Biological Survey (NBS). With this action, Alaska is included in the "Western Ecoregion," along with Hawaii, California, Oregon, Washington and Idaho. This change encourages the study of Alaska's wild stocks to address restoration issues elsewhere in the Pacific Northwest, as well as for their inherent value in maintaining the integrity of various Alaskan ecosystems.

The Hatchery versus Wild Stock Issue

Potential interactions between propagated and wild salmon are well known (Hindar et al. 1991, Krueger and May 1991, Waples 1991). Genetic alterations, increased

Table 1. Types of salmon enhancement used in Alaska and possible impacts and risks to wild stocks as synthesized from selected literature.

Enhancement type	Possible impact and risk	Source
Introductions	Increased competiton with resident fishes	Krueger and May 1991
	Increased predation on resident fishes	Krueger and May 1991
	Unwanted gene flow (straying) from fry releases	Unwin and Quinn 1993
	Unwanted gene flow (straying) from smolt releases	Unwin and Quinn 1993
	Incidental harvest of other stocks	Wright 1981
Supplementation:		
Non-indigenous stock	Intraspecific genetic change	Waples 1991
	Outbreeding depression	Gharrett and Smoker 1991
	Unwanted gene flow (straying from fry releases)	Unwin and Quinn 1993
	Unwanted gene flow (straying) from smolt releases	Unwin and Quinn 1993
	Decreased fitness from competition, disease	Hemmingsen et al. 1986
	Increased exploitation of native fish	McIntyre and Reisenbichler 1986
Indigenous stock	Intraspecific genetic change	Waples 1991
	Unwanted gene flow (straying) from fry releases	Unwin and Quinn 1993
	Unwanted gene flow (straying) from smolt releases	Unwin and Quinn 1993
	Decreased fitness from compeition, disease	Waples 1991
	Increased exploitation of native fish	McIntyre and Reisenbichler 1986
Habitat modification:		
Stream rehabilitation	Change in stream dynamics	Ryder and Kerr 1989
Lake enrichment	Change in fish community balance	O'Neill and Hyatt 1987

competition and predation, high exploitation of wild salmon in mixed-stock fisheries, and disease introduction are several issues of concern (Table 1). Our emphasis here is on the first three issues.

Genetic Alterations

It is widely accepted that wild salmon have evolved traits over many generations that adapt them to specific environments. Stock transfers (especially those using non-native broodstock) result in intraspecific gene flow that may lead to reduced genetic variability (Waples 1991), lower fitness and survival (Reisenbichler and McIntyre 1977) and outbreeding depression (Gharrett and Smoker 1991). For example, hybrid vigor often is reported in F1 generations of animal matings, but outbreeding depression (poor fitness in F2 and subsequent generations) may be a factor in the decline of some salmonid populations. Even when within-drainage, local broodstocks are used, selection may occur within the hatchery over time or during the egg takes (selection of early returners, large females, etc.) which may result in a once wild gene pool being permanently altered or lost (Waples 1991). Other concerns include ''founder'' effects (when small numbers of parents are used) and lowered disease resistance in wild stocks from reduced genetic diversity (Hindar et al. 1991). Hemmingsen et al. (1986) found that stocks of coho salmon exhibit a genetically based variance in their resistance to pathogens. It is possible that donor stocks can transmit lowered disease resistance to wild fish.

Competition and Predation

Introduction of salmon into streams not previously colonized can cause competition with native fishes, increased predation on resident populations and population instability. Ishida et al. (1993) suggest that density-dependent factors, resulting from intensive enhancement of Japanese chum salmon, may be linked to observed reductions in fish size in the North Pacific Ocean and that wild stocks might be adversely affected. Where stock supplementation is made to revitalize depressed salmon populations, hatchery-incubated brood fry often are fed prior to release, with the larger hatchery fry in a position to outcompete wild cohorts.

Exploitation Rates

Overexploitation of wild stocks in a mixed fishery can occur. For example, Wright (1981) suggests that hatchery stocks of coho salmon can support a catch-to-escapement rate of 19:1, while wild stocks only a 3:1 rate. In addition, when a new fishery is created, other stocks or species in the fishing area may experience high incidental harvest.

Where Ecological Diversity and Genetics Converge

Often, ecologically distinct forms of salmon can be separated with genetic tools (Wilmot and Burger 1985). Stream- and ocean-type chinook salmon in British Columbia spawning in three parts of a drainage could be distinguished by enzyme polymorphisms (Carl and Healey 1984). Variation in body morphology among certain chum salmon stocks (Beacham and Murray 1987) has a genetic component (Beacham et al. 1985). In Alaskan sockeye salmon populations, ecological differences in spawning area, time (Gard et al. 1987) and swimming orientation of emergent fry exist between lake outlet and tributary spawning sites (Raleigh 1967). Such behavioral patterns have a hereditary basis (Raleigh 1967).

Allozyme and mitochondrial DNA patterns of various Alaskan salmonids provide support that certain phenotypic traits have a significant genetic component. For example, Yukon River chum salmon exhibit differences among allozymes between early and late-running stocks (Wilmot et al. in press).

Evidence exists for genetic uniqueness among stocks where formally only one population was expected. Early running fish spawned in tributaries of the Kenai and Kasilof rivers (Figure 2), but late-running fish spawned in main-stem waters (Burger et al. 1985, Faurot and Jones 1990). Both spatial and temporal segregation was supported by genetic analyses: late-running salmon in each of the rivers have an mtDNA haplotype found in only about 8 percent of early running fish (Adams et al. in press). Tustumena Lake sockeye salmon demonstrate similar differences: 50 percent of the late-running salmon sampled from spawning areas in the lake's outlet possessed an mtDNA haplotype not found in early running tributary or lake shoreline spawners (Carl Burger unpublished data). For both chinook and sockeye salmon,

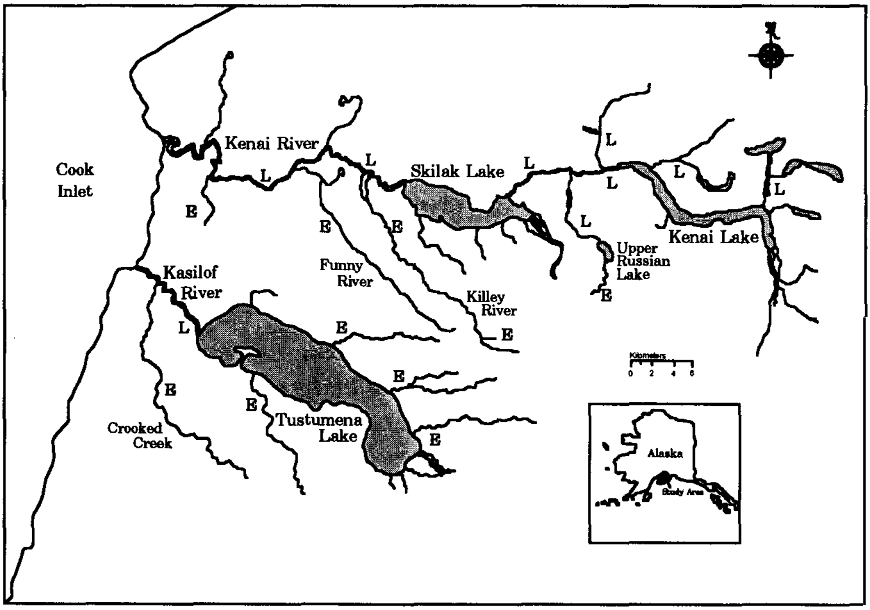

Figure 2. Spawning locations of early and late-running salmon spawning sites in the Kenai Peninsula, Alaska (E: early run salmon; L: late-run salmon).

these differences are highly significant, yet spawning areas of the two runs averaged <30 km apart and, in some cases, were <10km apart (Figure 2). Although fishery managers often consider geographically adjacent populations as good candidates for donor stocks in restoration plans, proximate stocks may differ substantially in phenotypic and genotypic characteristics.

A genetic basis exists for differences in egg development rates found among Alaskan chinook salmon stocks having different run and spawning times (Carl Burger unpublished data). Each population appeared adapted to the unique temperature regime of its home stream. Early running salmon spawned mid-July in tributaries where waters were coldest, while late-running salmon spawned late-August in main-stem rivers warmed by lakes. Eggs also hatched at different times (mid-September versus early November), but fry emerged at similar times the next May. The genetic basis of such differences has a major implication for managers because artificial selection can alter traits if sampling of a donor stock is temporally biased (Gharrett and Smoker 1993).

We See a Difference but What About Genetics?

The literature is replete with examples of ecological differences between populations whose environmental or genotypic basis has yet to be substantiated through genetic tools. Should phenotypic traits be considered during enhancement efforts? Available evidence suggests yes. In many cases (such as in the examples above), genetic techniques improve and subsequent application of these techniques corroborates ecological findings. Therefore, in some cases it may be prudent to conservatively define stocks as discrete based on consistent phenotypic differences until our understanding of the environmental or genetic basis of variability is improved.

For example, Burger and Finn (1993) compared the spawning distribution of sockeye salmon at Tustumena Lake, southcentral Alaska. As previously mentioned, the lake outlet-spawning component was genetically unique. Preliminary mtDNA studies suggested additional genetic differences between the tributary spawners and salmon spawning along the lake's shoreline (beach spawners). Ecological evidence that the beach spawners are a unique subpopulation comes from comparisons of run timing between beach and tributary spawners ($p<0.0001$) and from spawning time ($p<0.02$), yet genetic analyses to date are inconclusive. However, recently diverged populations may not be detectable by molecular genetic procedure (Utter et al. 1993). Based on glaciation patterns (Karlstrom 1964), we believe that beach spawners only could have colonized the lake in the last 2,000 years and that these fish may be differentiating. Conservative management may be appropriate until a body of evidence is compiled. The implications for salmon enhancement in this situation are obvious since all Tustumena sockeye salmon were formerly thought to be a single run of fish.

Other questions exist. Different outmigration timing patterns of juvenile salmon also may be in synch with temperature and aquatic productivity of their rearing areas (Burger and Finn 1993). Such findings are reasonable, but are these characteristics genetically based? We have found that most adult sockeye salmon migrate in a clockwise direction around Tustumena Lake. Why? In the Kenai River, we do not know if offspring from the genetically distinct early and late-running chinook salmon use different rearing habitats. If they do, is this heritable, conferring a selective advantage for survival? The significance to wild salmonids will remain unknown if stock transfers occur before thoughtful analyses are completed.

Conclusion

While it is clear to the engineer that a road culvert will fail if designed for last year's flow regime, that we must build for the future . . . the 100-year event, we as fishery managers have yet to agree on a similar perspective. Alaska is fortunate that it has lagged behind the "lower 48" in anthropogenic impacts and has a diversity of wild salmonid stocks. That is lucky for both Alaskans and citizens of the rest of the Pacific Northwest who have lost much of their salmonid diversity and abundance. One of our best hopes for maintenance or restoration of wild stocks in the Pacific Northwest is development and implementation of clear genetic policy by all resource agencies. Many agencies, including federal, do not have such policies. However, we also must acknowledge that genetics is a rapidly evolving science, with tools of promise, but also limitations. For example, most genetic surveys assess traits which alone may be insufficient to quantify genetic variability in populations (Gharrett and Smoker 1993). Because we lack clear black and white answers, we must manage with patience to ensure the future integrity and continued multiple use of our wild stocks. Our recommendations are not new but warrant restating and are as follows:

1. Establish formal policies among resource agencies to address strategies to maintain identifiable genetic variability in wild stocks. To meet the "diversity" mandates described above, we recommend that federal agency policies conservatively consider stock discreteness based both on genotypic and consistent

phenotypic traits. When artificial propagation is considered, stocks must be monitored and evaluated to ensure that long-term changes do not occur. Threshold characteristics should be identified that would trigger project termination or modification.

2. Develop a partnership and protocol to assess the status and trends of salmonids in a refined enough manner that wild stocks can be adequately monitored. While Konkel and McIntyre (1987) compiled data for 893 Alaskan stocks, 45 percent of these stocks had insufficient data for trend analysis. Eighty-four percent of those stocks (340) were from southcentral Alaska, an area where refuge and national park lands are abundant and sport harvest of sea-run salmon has increased 87 percent between 1982 and 1992 (Mills 1993). Enhancement project-specific information also should be incorporated, such as (a) a tag/recovery program, with recovery efforts in fisheries, spawning areas and proximal streams; (b) enumeration of escapement and outmigrants; (c) genetic sampling and monitoring; and (d) monitoring of fish and dependent wildlife populations within the study area.
3. Identify research needs and establish a partnership mechanism to encourage needed research on wild stocks. Such a cooperative framework could address the issues of stock identification, consequences of local adaptations, and phenotypic and genotypic variation in wild stocks as they relate to federal land and resource management options.

For the federal land manager, wild stocks are a trust resource. Selection of artificial propagation is an option to be approached in an informed and cautious manner to minimize risks to species, populations and ecosystems. In 1994, we still must concur with Walton and Houston (1984) that "both managers and scientists have many commitments to make and promises to keep before anyone can feel comfortable with the fate of our wild fish."

Acknowledgments

We thank T. Gharrett, G. Kyle, J. McIntyre, R. Reisenbichler, S. Rideout, D. Schmidt, G. Sonnevil and R. Wilmot for their constructive comments on this manuscript.

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Stocked Chinook and Coho Salmon Urban Fisheries in Anchorage, Alaska

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The Municipality of Anchorage is Alaska's largest community and home to over 240,000 people, about 40 percent of the state's total population. This 1,952-square mile area is made up of a wide range of land-use settings, from industrial and residential to rural and wilderness areas. Anglers, therefore, have the opportunity to participate in a variety of sport fisheries. As the population of this area has increased, so has the demand for angling opportunities, to the point of maximizing the potential of wild stocks. Urbanization, habitat degradation and limited availability of wild stocks have required that fish abundance and the diversity of angling opportunities be increased through stocking. Initial efforts to increase opportunities in the urban area fisheries were supported primarily through the stocking of rainbow trout (*Oncorhynchus mykiss*) in the area's landlocked lakes. New sport fishing opportunities recently have been developed for anglers to fish for and harvest anadromous chinook salmon (*O. tshawyscha*) and coho salmon (*O. kisutch*). Providing these fisheries allows for the cost-effective increase in angler participation and may help to reduce the pressure on wild stocks. The development of these new salmon fisheries in areas surrounded by industry, public lands and private property required that significant planning be provided in the design of stocking strategies and that the public, land managers and area landowners be involved throughout the process. Present management strategies for these fisheries are directed at providing for an orderly growth in participation through time and area restrictions while maintaining historical levels of natural production.

Introduction

The demand for fishing opportunities in the Anchorage urban area continues to grow. With limitations on the presence and abundance of wild stocks and consistent increases in fishing effort, Anchorage area sport fisheries in lakes an streams have become increasingly reliant on hatchery-produced fish. The stocking of urban lakes with rainbow trout and 'landlocked salmon' began in the '60s and helped to meet the growing demands through the early '80s. Recent advances in establishing successful returns of anadromous chinook and coho salmon in area streams have helped to meet this growing demand in the '80s, and should continue through the 1990s. This paper will discuss the development and current status of each of the three Anchorage urban area salmon stocking programs: the stocked lake fisheries, the urban anadromous chinook salmon fishery and the urban andromous coho fisheries.

All stocking activities related to state-run hatcheries are conducted under the guidelines established in the Statewide Stocking Plan for Recreational Fisheries (Alaska Department of Fish and Game [ADF&G] 1989). The concept of developing a statewide coordination of stocking activities was initiated in 1988 to optimize the use of hatchery facilities, provide consistency and establish priorities in stocking activities. The first plan was completed in 1989 after internal and public review, and provided statewide stocking locations and schedules for 1989 through 1993. A new stocking plan for 1993 through 1996 was completed in May of 1993 (ADF&G 1993).

According to the stocking plan, 54,000 chinook salmon "catchables" are distributed among 13 lakes and 315,000 chinook salmon smolt are stocked into two streams. Three Anchorage streams receive portions of 365,000 coho smolt. A total of 132,500 rainbow trout catchables are distributed among 26 lakes, and two streams divide an additional 12,500 rainbow trout catchables. Catchables are six- to eight-inch fish, large enough to become part of the bag limit at the time of stocking. Approximately 355,000 rainbow trout fry are released in Eklutna Lake. In addition, 15,000 Arctic grayling are distributed among three Anchorage area lakes, and 5,000 Arctic char are stocked in five area lakes.

Recreational Angler Effort

The primary tool used to evaluate angler effort and harvest in Alaska is the Statewide Harvest Survey (SWHS) (Mills 1993). Since its inception in 1977, angler effort in Anchorage has increased continually. In 1991 the effort was more than 350 percent greater than the low in 1978, however, during the past five years this trend has leveled off to an average of about 116,000 angler days. The Anchorage area consistently has represented approximately 5 percent of the total statewide sport fishing effort and about 7 percent of the total effort recorded for southcentral Alaska. The Anchorage stocked lakes program accounts for approximately 60–70 percent of the Anchorage area effort, with the area streams accounting for the majority of the difference. Ship Creek in particular has grown rapidly in recent years. In 1991, Ship Creek accounted for more than 25 percent of the Anchorage area effort, up five fold from 4.5 percent the creek accounted for in 1985 before stocked fish began contributing to the fishery.

In comparison to other major fisheries in southcentral Alaska, the Anchorage area fisheries as a whole, rank second only to the Kenai River in terms of recreational angler effort expended. The Anchorage stocked lakes program alone provides approximately the same quantity of sport fishing effort as the Russian River, and is greater than either the Deshka River or the Little Susitna River.

Recreational Angler Harvest

Harvests of anadromous salmon in the Anchorage area generally have increased in recent years. Chinook salmon harvests have grown substantially, primarily as a result of the Ship Creek stocking program. Harvest of Ship Creek chinook salmon in 1991 exceeded 1,000 fish for the first time on record, and in 1992 the harvest doubled to more than 2,000 fish. Chinook salmon harvest is expected to continue to increase as a fishery on Eagle river develops in response to adult returns from smolt stockings initiated in 1991. Coho and pink salmon provide the largest harvests of the salmon species in the Anchorage area. The coho fishery is dominated by harvests from Ship, Bird and Twentymile creeks, with the Ship Creek fishery being supported primarily by a stocking program. Harvests of coho salmon increased significantly in 1993 when releases of stocked smolt returned to Campbell and Bird creeks.

Harvests of non-anadromous stocked fish have remained relatively stable since

1988 as available hatchery production space is fully allocated and utilized. Rainbow trout are the dominate non-anadromous species harvested from fisheries in the Anchorage area. In 1992, over 33,000 rainbow were harvested, nearly three times the number of any other species of resident fish or salmon. The rainbow harvest is composed almost entirely of lake-stocked fish. Stocked landlocked salmon contributed a harvest of nearly 14,000 fish in 1992. This harvest primarily is attributed to stocked chinook salmon which are harvested primarily as part of a winter ice fishery on the area lakes.

Anchorage Area Stocked Lakes Fisheries

Few of the lakes in the Anchorage area supported resident fish populations of recreational interest prior to the initiation of stocking efforts. Most of the lakes are landlocked and threespine stickleback were the only species present. Beginning in the 1960s, the ADF&G began a stocking program with rainbow trout to increase the area's sport-fishing opportunities.

A total of 26 area lakes and two creeks are stocked each year with approximately 132,500 catchable-size rainbow trout, and approximately 355,000 rainbow trout fry are released into Ekutna Lake. In addition, stocking with other species is conducted to increase the diversity of angling opportunities. Thirteen lakes are stocked during late autumn, with a total of 54,000 chinook salmon (landlocked salmon) catchables to provide winter ice fishing opportunities. These fish are very aggressive and strike readily throughout the winter. Three local lakes also receive a total of 15,000 Arctic grayling fingerlings, and a total of 5,000 Arctic char are stocked in five local lakes. The result of these stocking efforts is the development of significant urban angling opportunities throughout the year in the Anchorage area. This lake stocking program has provided 60 to 70 percent of the Anchorage area's annual sport fishing effort in recent years.

Rainbow trout dominate the harvest in the Anchorage area lakes, comprising 66 percent of the lake harvest in 1992. Landlocked salmon made up most of the remaining 1992 harvest at nearly 28 percent. In spite of the high proportion of the harvest, the 1992 harvest of rainbow trout was the lowest in 10 years due to the increase in harvest of the landlocked salmon. The sport harvest of landlocked chinook salmon has increased from 399 fish in 1986 to nearly 14,000 fish in 1992 as anglers became more aware of the stocking program and the ice fishing opportunities in the area lakes. Arctic char and Arctic grayling contributed 5 percent and 1 percent, respectively in 1992.

A creel survey was conducted during 1986 on four of the Anchorage area lakes to evaluate the stocking program. Results of this program (Roth 1986) indicated that youth and adult males were the primary recreational users of this fishery. The primary purpose of the survey was to determine if current stocking practice of a single annual release of a large number of rainbow trout each spring was suitable for the area lakes. Data indicated that catch rates remained high for two to six weeks after stocking but that catch rates dropped to below one fish per angler-hour after this time. It was recommended, and since has been adopted, that the stocking of rainbow trout be conducted initially after ice-out and then again four to six weeks later. It is believed that the revised practice of multiple stockings has provided more consistent fishing success throughout the season.

Eklutna Lake is the only lake in the Anchorage area stocked with rainbow fry rather than catchable-sized fish. This is because Eklutna Lake is where the excess production is stocked when reduction in numbers at the hatcheries is required to allow adequate growth of remaining fish to catchable size. Therefore, variable numbers of fish have been stocked here, ranging from a low of approximately 50,000 in 1990, to nearly 2.5 million in 1991. Survival of fry and fingerlings is much lower than for catchable-sized fish, and growth is much slower in this glacier-fed lake. However, sampling has shown that these stockings have produced an adequate population of catchable-sized rainbow trout in Ekutna Lake to support a fishery. It is expected that as more people become aware of this opportunity angler effort will increase.

Northern pike have been introduced illegally into at least one Anchorage area lake, Sand Lake. Not only have large adults been caught, but juvenile pike have been brought into the Anchorage office from Sand Lake, indicating that this species is spawning successfully. As the population of pike grows in Sand Lake, the success of the stocking program may diminish. The major concern is that additional illegal introductions do not occur in other area lakes.

In addition to the sport fishing opportunities provided for the general public through the stocking of the area lakes and streams, these stocking efforts also have assisted in the development of youth fishing classes by local sport fishing associations and community schools, the trout pond at the annual fishing fair, and the annual ice fishing jamboree for disabled and disadvantaged anglers.

Anchorage Chinook Salmon Fisheries

Several Anchorage area streams support wild stocks of chinook salmon. However, none of the native populations are large enough to support a sport fishery. As a result, sport fishing for chinook salmon has been closed for the past two decades, with two notable exceptions. Those exceptions are Ship Creek and Eagle River which are open to king salmon fishing as a result of returns from stocking programs. Recreational chinook salmon fishing in the Anchorage area began in 1987 with the opening of Ship Creek to chinook salmon fishing two days per week. The fishery since has expanded to seven days a week with over 2,000 chinook salmon being harvested in 1992. A similar fishery was developed in Eagle River and this fishery was opened for the first time in 1992. Minimal harvest and participation was documented in 1992, however, only a small return from stocked smolt was expected. All other Anchorage area streams remain closed to fishing for chinook salmon.

Ship Creek Chinook Salmon

Prior to World War II, Ship Creek supported a significant wild stock of chinook salmon which supported sport, personal use and subsistence fisheries. However, dams were constructed in the lower 11 miles of the creek during the 1940s and 1950s for power generation and utilization of the creek as a source of water for the Municipality of Anchorage and the military. This development substantially reduced the returns of wild fish to Ship Creek. Attempts to increase the returns to Ship Creek during the period from 1966 through 1980 by the stocking of chinook salmon of Alaska and Oregon origin (Miller 1990) generally were unsuccessful in that consistent numbers of returning adults could not be established. During this period, eggs obtained from these stocks were incubated at the Fire Lake Hatchery and the resultant fry were reared to smolt in the Fort Richardson cooling pond. More consistent returns of chinook salmon to Ship Creek have been established since 1985 due to smolt releases from the ADF&G's Elmendorf Hatchery using Ship Creek brood stock.

Ship Creek was open to sport fishing for chinook salmon from 1957 through 1959, but remained closed from 1960 through 1969. Chinook salmon fishing was allowed during selected periods in Ship Creek downstream of the Chugach Power Plant dam during 1970, 1971 and 1972. From 1973 through 1986, the creek again was closed to chinook salmon fishing, in part due to the concern over the historically low abundance of chinook salmon through Northern Cook Inlet during the early and mid-1970s. Through increased returns provided by annual stocking efforts, the lower portion of Ship Creek downstream of the Chugach dam was reopened to fishing for chinook salmon two days per week for five consecutive weeks during June and July, beginning in 1987.

In recent years, hatchery-produced chinook salmon returns to Ship Creek have provided a unique opportunity for anglers to fish for chinook salmon in an urban setting. The chinook salmon return is a result of the annual release of approximately 105,000 smolt and raised at the Elmendorf Hatchery located adjacent to Ship Creek. As this was an experimental urban chinook salmon fishery, the period open to fishing initially was limited to two days per week to allow fishing opportunity, while at the same time ensuring that sufficient fish were available for upstream viewing opportunities and brood stock needs. The season recently has been expanded to seven days a week from January 1 through July 13. The fishery operates during June and early July in the lower mile of Ship Creek located downstream of the Chugach Power Plant dam. The shoreline of the area open to chinook salmon fishing is owned and managed by the Alaska Railroad.

The sport harvest of chinook salmon in Ship Creek has increased 500 percent, from 437 fish in 1987 to 2,448 fish during 1992. Fishing effort in Ship Creek has increased nearly ten times the average effort levels continue to increase as the popularity of this fishery grows. Returns to Ship Creek are predicted to average approximately 3,000 chinook salmon annually by 1994 as the full compliment of recent smolt releases are realized.

The 1992 Ship Creek chinook salmon escapement was estimated at 789 fish, well over the mean escapement of 479. Approximately 100 fish were taken for brood stock requirements at the Elmendorf Hatchery while the remainder provided viewing opportunities and spawned naturally in the area downstream of the hatchery.

Eagle River Chinook Salmon

The Eagle River drainage originates in the foothills of the Chugach Mountains with most of the flow contributed by Eagle Glacier. The lower portion of the river is on Fort Richardson Army Base and historically has been used as a large weapons test firing range and impact area. All access to the reach from the mouth upstream approximately two miles to the railroad bridge is restricted due to unexploded ordinances in the area. The remaining portion between the railroad bridge to the Glenn Highway bridge is accessed only through Fort Richardson. Upstream of the Glenn Highway, the river meanders through dedicated greenbelt as part of the Chugach State Park. Access to the river is limited to only a few sites, including the campground located immediately upstream of the Glenn Highway and a parking area/boat launch site located at mile 7.4 of Eagle River Road. A new access site near the location of the new Briggs bridge crossing Eagle River from Hiland Road to Eagle River Loop Road is planned for construction in 1993. The current non-angling use pattern for the river during the summer months is primarily as a recreational site for hiking and whitewater float trips.

The Eagle River drainage has been closed to fishing for chinook salmon less than 20 inches in length since 1964. Wild stock chinook salmon return to the Eagle River drainage during June and early July, however, the number of returning adult salmon is too low to support a viable sport fishery. The majority of the chinook salmon spawning has been found to occur in the South Fork of Eagle River in the area downstream of the barrier falls. Surveys of chinook salmon escapement in Eagle River since 1963 have documented from 28 to 513 fish annually.

The king salmon fishery in Eagle River is unique in that it is the only enhanced run of salmon within State Park lands, yet it is in the midst of a heavily populated area. The initial concept for the development of a king salmon fishery in Eagle River was considered in the late 80s during the development of the five-year stocking plan, and initially was scheduled for stocking starting in 1992. This schedule was accelerated due to interest expressed by the residents as indicated through letters and meetings with local politicians and community councils. The first stocking of 105,000 smolt of Ship Creek origin took place on June 1, 1990, subsequent stockings took place in 1991 through 1993.

In 1992, the chinook salmon fishery was opened in Eagle River for the first time since 1964. Approximately 300 wild stock chinook salmon and 1,000 hatchery chinook salmon were available for sport anglers in 1992 based on projected returns. Observations during an informal creel survey, which was part of the cooperative DNR/ADF&G Eagle River access study in 1992, indicated that angler participation was low and documented a harvest of only 16 king salmon. From these observations, it is likely that less than 50 fish were harvested. The majority of angler effort was from the Eagle River campground site, with 64 percent of the total 572 anglers interviewed. All of the 16 harvested fish observed were from the campground site. The majority of the fish were fairly large, indicating that this harvest was primarily from the wild stock, since those expected to return from the stocking efforts would be only one ocean fish. Observations during 1990 and 1992 indicate that a significant illegal fishery takes place in the clear water of the South Fork. The escapement count for chinook salmon in the South Fork in 1992 was 336 which exceeded the escapement goal of 300. Therefore, in spite of the legal and illegal harvests, adequate returns made it to the spawning grounds. Expected returns for 1993 were greater (approximately 300 wild and 1,500–2,000 hatchery fish). Harvest in 1993 was approximately 70 fish. Runs are expected to gradually increase for the next few years as the run reaches full strength of about 3,000 fish in 1996.

In 1992, the river was open for three days a week (Sunday, Tuesday and Thursday) from 6:00 a.m. to 10:00 p.m. beginning on May 26 through July 12. The area open to anglers was limited to three shore sites identified by ADF&G markers and the stretch of river from the north fork site downstream to the Eagle River Loop Road site which was open to boaters. Assessments conducted by Chugach State Parks to evaluate access and impacts indicated that increased activity in the Eagle River Green

Belt resulting from this fishery caused minimal impacts with regard to stream bank degradation or litter.

Changes for the 1993 season included establishing a sanctuary for the chinook salmon in the South Fork by closing the South Fork and the mainstream of Eagle River 100 yards upstream and downstream of the confluence of the South Fork. All fishing in this area was prohibited during the king salmon spawning season to ensure natural escapement levels. In addition, all fishing above mile 9 of Eagle River Road was closed from June 1 through September 15 to avoid conflicts with wildlife viewing (salmon spawning activities) at the Chugach State Parks, Eagle River Visitors Center. Finally, the timing of the fishery was increased to seven days a week from Memorial Day for thirty days. A return of approximately 1,500 to 2,000 chinook salmon was expected for 1993 Observations indicated that approximately 70 fish were harvested in 1993.

Anchorage Coho Salmon Fisheries

Wild stocks of coho salmon are present in several Anchorage area streams, although few of the native populations are large enough to support significant sport fisheries. As a result, sport fishing opportunities for this species in the Anchorage area have been limited. Streams supporting annual returns of coho salmon include Campbell, Rabbit, Bird, Ship, Peters, Glacier, California and Portage creeks, and Eagle, Twentymile and Placer rivers. According to the Statewide Harvest Survey, the most significant sport fisheries for coho salmon in the Anchorage area presently occur in Bird and Ship creeks and Twentymile River. Bird Creek and Twentymile river support wild coho salmon stocks while the Ship Creek coho are primarily a result of hatchery production from the Elmendorf Hatchery. In 1991, an urban coho project was initiated to provide additional recreational fishing opportunities by stocking coho salmon smolt in several urban area streams. This program identified seven streams which will receive all of the stocked anadromous coho in the northern cook inlet area. Three of these streams, Ship, Bird and Campbell creeks, are in the Anchorage area. The other four—Fish, Wasilla and Cottonwood creeks and the Little Susitna River—are the Palmer Wasilla urban areas. Of the streams in the Anchorage area, Ship Creek already received stocked fish, but the numbers were increased to provide additional angling opportunities. Bird Creek, which had a limited coho salmon fishery supported by natural spawning was augmented through stocking to provide additional opportunities. Finally, Campbell Creek was stocked to provide a new fishery which was open for the first time in 1993. Stocking efforts also have been conducted in Ingram Creek to establish a coho salmon sport fishery, however insufficient returns were realized and this program was discontinued. Anchorage area streams currently closed to coho salmon fishing include Potter and Rabbit creeks.

Ship Creek Coho Salmon

Similar to chinook salmon, Ship Creek supported a significant wild return of coho salmon which provided for sport, personal use and subsistence fisheries prior to World War II. The dams constructed in the lower 11 miles of the creek for power generation and as a source of water for the Municipality of Anchorage and the military during the 1940s and 1950s, significantly reduced the returns of wild fish to Ship Creek. To rebuild the runs, the creek was stocked with coho salmon from 1968 through 1977. These efforts proved to be unsuccessful in providing consistent numbers of returning adults to the creek. Nine brood stocks from Ship Creek, Bear Lake (near Seward), Kodiak, Washington and Oregon (Miller (1990) were used in the stocking efforts. During this period, eggs obtained from these stocks were incubated at the Fire Lake Hatchery and the resultant fry were reared to smolt in the Fort Richardson cooling pond. As a result, coho salmon smolt releases were discontinued in Ship Creek from 1978 through 1986. Beginning in 1987, the ADF&G began annual stocking of Ship Creek using smolt reared at Elmendorf Hatchery using Ship Creek brood stock. These efforts have proven to be successful toward providing consistent returns of coho salmon to Ship Creek capable of supporting a recreational fishery.

Ship Creek was open to sport fishing for coho salmon from 1957 through 1959, and again from 1964 through 1992. Presently, only the reach downstream of the Chugach Power Plant dam is open to salmon fishing. Hatchery supported coho salmon returns to Ship Creek in recent years have provided a unique opportunity for anglers to fish for and harvest coho salmon in an urban setting. The coho salmon are primarily the result of the annual release of approximately 65,000 smolt raised at the state's Elmendorf Hatchery located on Ship Creek. The fishery has taken place during August and early September in the lower mile of Ship Creek located below the Chugach Power Plant dam. Much of the area open to salmon fishing is owned and operated by the Alaska Railroad.

Performance of the sport fishery in Ship Creek since 1977 has been estimated from the Statewide Harvest Survey (Mills 1993). Based on these data, the sport harvest of coho salmon in Ship Creek has increased from less than 300 fish for the period from 1977 through 1987, to an average of 1,400 fish during 1988 to 1991. Harvest and effort levels are expected to continue to increase as the popularity of this fishery grows.

Campbell Creek Coho Salmon

Wild coho salmon return each year to Campbell Creek during August and September. The number of returning adults, however, is insufficient to support a viable sport fishery. Most of the return migrates upstream of Lake Otis Parkway, to the North and South Forks, to spawn. Escapement surveys of coho salmon in Campbell Creek from 1986 to 1992 averaged 159 fish, with 157 counted in 1992. Information shows that Campbell Creek historically supported larger annual returns of coho salmon than observed in recent years. Urbanization and development along the creek, loss of wetlands and associated rearing habitat, the input of storm drain runoff and pollutants, and poaching all have led to the reduction for the numbers of coho salmon returning to spawn in this drainage. To increase the returns of coho salmon to Campbell Creek, the annual stocking of 115,000 coho smolt was initiated in 1992. This is being done as part of the urban coho project aimed at increasing angling opportunities for coho salmon in the Anchorage area. The stocking is expected to yield returns of approximately 3,000 fish annually, which were available to anglers in 1993. To utilize these returns, Campbell Creek was opened to coho salmon fishing in 1993 for the first time since 1971. The Campbell Creek greenbelt supports a major segment of the bike trail system in the Anchorage area, which provides excellent public access to the creek from the confluence of the North and South Forks downstream to Campbell Lake.

An assessment program was developed to evaluate the success of these enhancement efforts. This program consists of placing weirs on selected streams to evaluate the returns and to assure adequate escapements, and monitoring of the commercial catch to determine the interception rates of the stocked fish. Commercial catch sampling was conducted in 1992 at two processors in Anchorage and four processors on the Kenai Peninsula. Data collected from the two Anchorage processors indicated that the hatchery-produced coho salmon contributed about 5 to 8 percent of the 1992 commercial harvest in the Northern Cook Inlet district. The 1993 returns to Campbell and Bird creeks were estimated at approximately 6,000 coho salmon to each stream, with about half of that being in the form of harvested fish.

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Culture and Performance of Triploid Rainbow Trout in Alaska

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Introduction

Rainbow trout (*Oncorhynchus mykiss*) are the mainstay of the lake stocking programs in Alaska. Stocked lake fisheries provide significant fishing opportunity, particularly in the Anchorage urban area. In 1992, stocked lake fisheries in the Anchorage area accounted for an estimated 71,194 angler-days of fishing effort (50 percent of the total fishing effort for the area) (Mills 1993). The management objective for these fisheries is to maximize fishing effort at the lowest possible cost without compromising wild stock integrity where present.

The Alaska Department of Fish and Game, Division of Sport Fish, successfully has cultured all-female triploid rainbow trout for stocking applications in Alaska. Potential benefits include sterility, reduced spawning mortality and increased growth. If successful, hatchery production costs could be reduced and stocking could be considered in open systems where inter-breeding with wild stocks is a concern. Also, the potential for bypassing the rigors of spawning to produce larger, older fish could increase the appeal of, and participation in, our current stocking programs.

In this paper we present a summary of: culture practices to produce all-female triploid fish; hatchery performance as measured by frequency of triploidy; a comparison of survival and growth between all-female triploid and mixed-sex diploid fish; and field performance in landlocked lakes as measured by comparative survival and growth between the two treatment groups.

Methods

Culture and Hatchery Performance

All-female triploid rainbow trout were produced at the Alaska Department of Fish and Game, Broodstock Development Center, Fort Richardson, Alaska. The first step in creating these fish was to reverse the sex of genetic females so that they would function reproductively as males. Sex reversal was achieved by feeding genetic females the male hormone testosterone as outlined by Olito and Brock (1991). Because the female rainbow trout is homogametic, an all-female population can be maintained indefinitely by using sex-reversed (XX) males to fertilize eggs. The second step was to use a thermal shock to induce triploidy in eggs that had been fertilized with sperm from sex-reversed (XX) males. The methods used were similar to those described by Chourrout (1980), Thorgaard and Jazwin (1981), Lincoln and Scott (1983) and Bye and Lincoln (1986).

For the hatchery experiment, eggs from a single female were divided equally into two groups. One group of eggs was fertilized with milt from a sex-reversed (XX) male, allowed to sit for 20 minutes in ambient incubation water (10 degrees Celsius), and then heat-shocked in 26 degrees Celsius water for 20 minutes to create triploid zygotes. The remaining group of eggs was fertilized with a normal (XY) male creating mixed-sex diploid zygotes. This procedure was repeated until 20 females had been spawned, creating 20 half-sibling family groups.

On June 16, 1992, equal numbers of fish from each treatment group were removed from incubators and placed in 10 cubic feet (0.011 m^3) circular tanks, five tanks per treatment group. The fish were fed standard rations by hand and monitored through September 17 when they were sampled for length and weight. Individual length and weight measurements were taken on approximately 40 anesthetized fish which had been randomly selected from each tank. In 1993, the experiment was repeated when on June 8, fish from both groups were removed from incubators and placed in 10 tanks (5 tanks per treatment group). The six-week period from initial ponding through July 21 was sufficient to start all fish on artificial feed. After the six-week period, the fish were further split into 20 tanks, (105 fish in each), 10 tanks for each treatment group to maintain approximately equal rearing densities for the duration of the study. From initial ponding through September 22, fish were fed with automatic feeders and tanks were cleaned daily. Dead fish were counted and removed. The mean mortality rate was calculated for each treatment and a t-test was used to detect any difference between the two groups. On September 22, the fish were sampled for length and weight data as previously described. An analysis of variance (ANOVA) with a completely randomized design and a nested treatment arrangement was used to test for significant differences in mean length and weight between the all-female triploid and the mixed-sex diploid rainbow trout.

To identify polyploidy, a sample of blood from each fish was placed into a 1.8-milliliter vial containing 1 milliliter of Alsevers solution. The blood samples were put on ice and sent to the Alaska Department of Fish and Game Fish Genetics Laboratory for flow cytometry analysis. Thorgaard et al. (1982) and Utter et al. (1983) concluded that flow cytometry could be used to rapidly analyze the DNA content of a large number of cells with greater accuracy than that afforded by other accepted techniques.

All-female triploids used for the field performance portion of this study were created by pooling eggs from 10 females, then fertilizing those eggs with sperm from a minimum of three sex-reversed (XX) males. These fertilized eggs were subjected to the same heat-shock procedure described above. This procedure was repeated until enough eggs had been taken to meet production requirements of the Fort Richardson Hatchery.

To determine percent ploidy within the raceway to be used, a random sample of 100 fish was taken in 1991 and 150 fish in 1992 as described above. Mixed-sex diploids used in the field performance studies came from other production raceways at the hatchery.

Field Performance in Landlocked Lakes in Southcentral Alaska

The field experiment was conducted in six landlocked lakes in southcentral Alaska (Figure 1). Stocking took place in July 1991 for Long, Wishbone and "X" lakes and in July 1992 for Dawn, Ravine and Tigger lakes. Long, Wishbone and "X" lakes

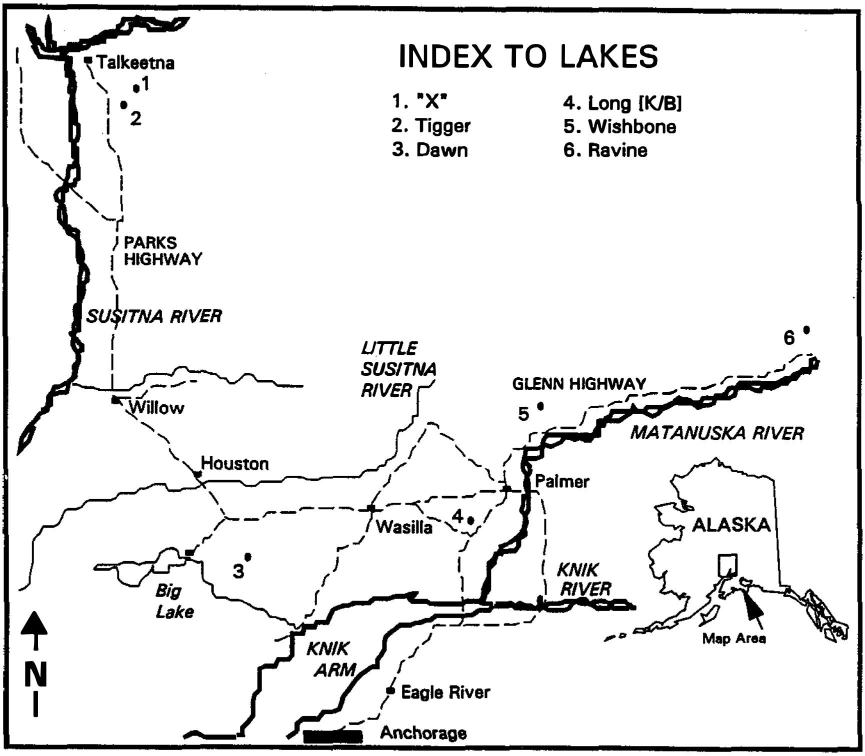

Figure 1. Location of lakes stocked with mixed-sex diploid and all-female triploid rainbow trout.

have been restricted to catch-and-release angling since 1989. Dawn, Ravine and Tigger lakes combined received an estimated total of 523 angler-days fishing effort in 1992. All rainbow trout were 1.5–2 grams age-0 fingerlings at stocking. All fish were marked at the hatchery prior to stocking. Mixed-sex diploid rainbow trout were given a left ventral finclip and all-female triploid fish were marked with a right ventral finclip. All lakes were stocked with diploid and triploid rainbow trout fingerlings at a 50:50 ratio and a density of 200 per surface acre, 100 of each treatment group (see Table 3).

Rainbow trout recruit to the sport fishery at 165 millimeters fork length. This length is reached in autumn, a year after stocking (age 1+); therefore, all sampling took place in late September and early October. Sampling took place in autumn of 1991, 1992 and 1993. Fish were sampled with fyke nets baited with salmon eggs and set parallel to the shoreline at randomly selected sites. Captured rainbow trout were placed in oxygenated water, anesthetized and inspected for the presence of finclips (both a ventral clip and a secondary mark). Lengths were then measured to the nearest millimeter. A secondary mark (adipose clip in 1991 and a partial caudal clip in 1992) was given to all study fish to ensure recognition of a previously handled fish. A second length measurement was taken on all fish handled more than once.

Chi-square tests were used to detect size-selective sampling. Lengths were divided into 100 millimeter categories and the probability of recapture by length group was examined for each lake and year.

A two-factor analysis of variance was used to test the hypothesis that there was no significant difference in mean fork length between the diploid and triploid rainbow trout at stocking, age-0+, age-1+ and age-2+ ($\alpha = 0.05$). Lakes were considered a random effect and ploidy was considered fixed. A Chi-square goodness-of-fit test was used to test for significant differences in catches or survival for each age group.

Results

Culture and Hatchery Performance

Frequency of triploidy. Fish released in study lakes in 1991 were 100 percent triploid (based on a sample of 100 fish). Fish released in 1992 were 99.3 percent triploid (149 out of 150 fish).

Growth. In 1992, the average growth of the mixed-sex diploid rainbow trout was greater than that of the all-female triploid rainbow trout for both length and weight ($P = 0.052$ and $P = 0.056$, respectively, Table 1). In 1993, the average length and weight of the mixed-sex diploid fish was again greater than that of the all-female triploid fish (both P values <0.001).

Survival. There were no significant differences in the mortality rates 1993 between the two treatment groups during either phase of rearing ($P = 0.15$ and $P = 0.47$,

Table 1. Analysis of variance results comparing mean fork length and weight of mixed-sex diploid and all-female triploid rainbow trout taken from the hatchery.

Dependent variable	df	Sum of squares	F	P	Diploid mean	Triploid mean
1992 length						
Treatment	1	3,249	5.21	0.0519		
Tank (treatment)	8	103				
Fish (tank treatment)	390					
Total	399				65.0 mm	59.3 mm
1992 weight						
Treatment	1	64	4.99	0.056		
Tank (treatment)	8	103				
Fish (tank treatment)	390					
Total	399				3.0 g	2.2 g
1993 length						
Treatment	1	5,491	30.94	<0.001		
Tank (treatment)	18	3,194				
Fish (tank treatment)	380					
Total	399				97.1 mm	89.7 mm
1993 weight						
Treatment	1	732	25.22	<0.001		
Tank (treatment)	18	522				
Fish (tank treatment)	380					
Total	399				10.4 g	7.7 g

Table 2. Summary of the hatchery mortality study.

Date	Tank	Treatment	Number stocked	Number died	Morality rate	$\bar{m}$	SE($\bar{m}$)	t Statistic	P value
06/08/93	D3	Diploid	428	4	0.01	0.026	0.000046	−1.08	0.15
	C7	Diploid	423	11	0.03				
	C3	Diploid	594	29	0.05				
	C1	Diploid	369	11	0.03				
	C8	Diploid	389	6	0.02				
	D5	Triploid	403	9	0.02	0.041	0.000156		
	D7	Triploid	222	4	0.02				
	D6	Triploid	249	7	0.03				
	C5	Triploid	376	32	0.09				
	D9	Triploid	264	14	0.05				
07/22/93	C1	Diploid	105	1	0.01	0.024	0.000050	−0.08	0.47
	C6	Diploid	105	4	0.04				
	C10	Diploid	105	0	0.00				
	C7	Diploid	105	6	0.06				
	C3	Diploid	105	5	0.05				
	C8	Diploid	105	0	0.00				
	D1	Diploid	105	0	0.00				
	D3	Diploid	105	3	0.03				
	D4	Diploid	105	5	0.05				
	C9	Diploid	105	1	0.01				
	C5	Triploid	105	4	0.04	0.025	0.000074		
	D5	Triploid	105	5	0.05				
	D6	Triploid	105	4	0.04				
	D2	Triploid	105	0	0.00				
	D7	Triploid	105	1	0.01				
	C2	Triploid	105	9	0.09				
	C4	Triploid	105	1	0.01				
	D10	Triploid	105	0	0.00				
	D8	Triploid	105	1	0.01				
	D9	Triploid	105	1	0.01				

respectively, Table 2). Though not significant, from initial ponding through July 21, the average mortality rate of the all-female triploid fish was 1.6 times greater than that of the mixed-sex diploid fish. From July 21 through the end of the rearing study on September 22, the absolute difference in mean mortality rate between the two treatment groups was only 0.001 percent.

Field Performance in Landlocked Lakes in Southcentral Alaska

In autumn of 1992, age-1+ rainbow trout were sampled from Long, Wishbone and "X" lakes, while age-0+ fish were sampled from Dawn, Ravine and Tigger lakes. In autumn of 1993, age-2+ rainbow trout were sampled from Long, Wishbone and "X" lakes and age-1+ from Dawn, Ravine and Tigger lakes (Table 3).

The Chi-square tests of equal probability of capture regardless of size never were significant (all P-values >0.17). The assumption that the sampling gear was unbiased with respect to size therefore is considered valid.

Table 3. Summary of stocking and sampling of mixed-sex diploid and all-female triploid rainbow trout stocked in six lakes in southcentral Alaska. Standard errors of the means are presented in parenthesis.

	Lake					
	Long	Wishbone	"X"	Dawn	Ravine	Tigger
Surface area (acres)	74.4	52.7	101.4	11.8	12.3	18.9
Number stocked						
Diploid	7,277	5,304	10,152	1,146	1,202	1,881
Triploid	7,451	5,265	10,074	1,147	1,189	1,868
Mean length at stocking						
Diploid	55 (1)	56 (1)	54 (1)	49 (1)	49 (1)	50 (1)
Triploid	54 (1)	53 (1)	54 (1)	47 (1)	47 (1)	48 (1)
Catch age = 0+						
Diploid				189	110	111
Triploid				134	103	33
Mean length age = 0+						
Diploid				97 (1)	100 (1)	86 (1)
Triploid				88 (1)	91 (1)	78 (1)
Catch age = 1+						
Diploid	560	599	689	176	274	47
Triploid	265	284	376	103	389	9
Mean length age = 1+						
Diploid	195 (1)	179 (1)	188 (1)	254 (3)	213 (2)	223 (5)
Triploid	167 (2)	155 (1)	170 (1)	219 (3)	181 (1)	189 (9)
Catch age = 2+						
Diploid	107	234	126			
Triploid	92	127	102			
Mean length age = 2+						
Diploid	305 (5)	268 (3)	280 (3)			
Triploid	275 (4)	229 (2)	252 (2)			

Growth. Mean fork length of the diploid rainbow trout was slightly greater than the mean fork length of the triploid fish at the time of stocking (ANOVA, F = 15.97, df = 1,5, P = 0.01, Tables 3 and 4, Figure 2). Although the trend was the same in Tigger Lake, the small sample size and higher variance did not allow for inclusion of the data in the ANOVA. All-female triploid fish were an average of 4 percent smaller at the time of stocking than the mixed-sex diploid fish. Diploid fish also were significantly larger at age-0+, age-1+ and age-2+ (tables 3 and 4, Figure 2). The average length of age-0+ triploid fish was 7 percent less than that of the diploid rainbow trout and the average length of age-1+ and age-2+ triploid fish was 11 percent less than that of the diploid rainbow trout.

Survival. At age-0+, individual statistical tests showed significantly more diploid rainbow trout than triploid rainbow trout were caught in two of the three lakes (Table 5, Figure 3), catches were relatively equal in the third lake (Ravine). The combined catch of triploid rainbow trout was 34 percent less than the catch of diploid fish. At age-1+, Chi-square tests showed significantly more diploid fish than triploid fish were caught in five of the six lakes (Ravine Lake had the opposite result, Table 5).

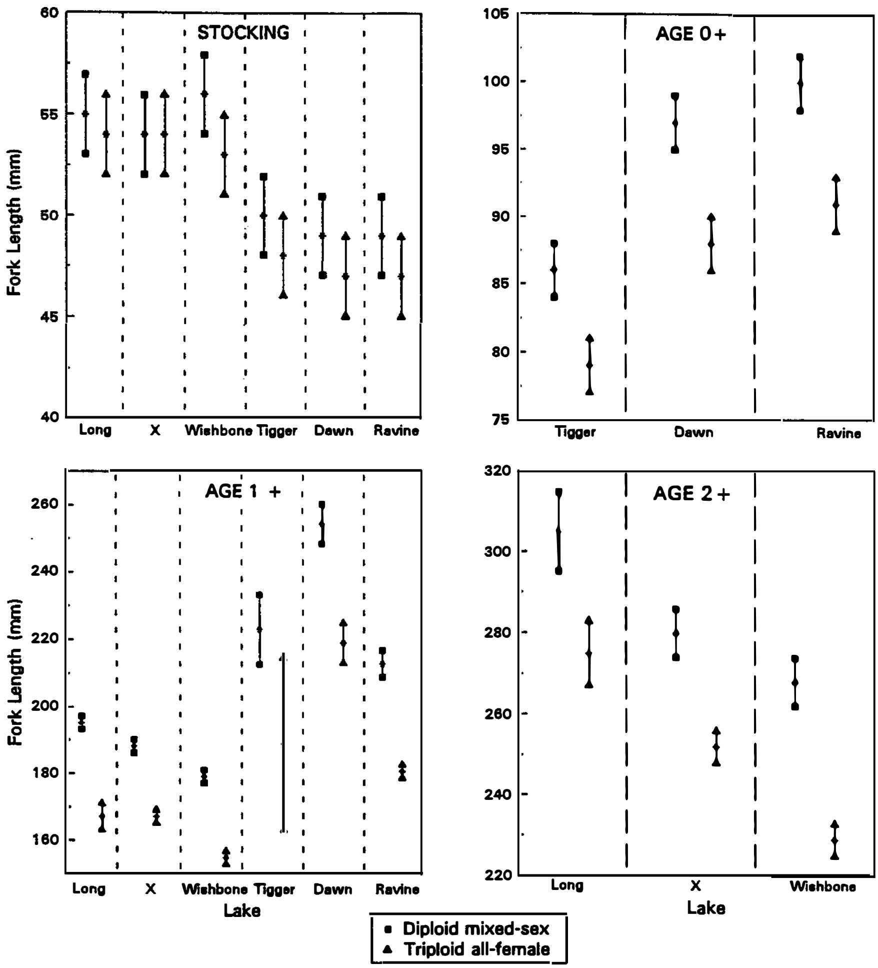

Figure 2. Mean length and 95-percent confidence intervals of mixed-sex diploid and all-female triploid rainbow trout stocked in six lakes in southcentral Alaska.

The combined catch of triploid fish was 39 percent less than that of diploid fish. Catches of the diploid rainbow trout were higher than those of the triploid fish at age-2+ in only one of three lakes (Table 5). Since there was no significant size-selectivity in sampling between the treatment groups, the differences in catch rates are believed to be due to differences in survival.

Discussion

We entered this research looking for an enhanced hatchery product that would translate measurably into more efficient management of stocked lakes. In terms of field performance, we hoped to significantly increase the availability of large fish,

Table 4. Analysis of Variance results comparing mean fork length of mixed-sex diploid and all-female triploid rainbow trout stocked in six lakes in southcentral Alaska.

Source	df	Sum of squares	F	P
Stocking				
Treatment	1	528	15.97	0.010
Lake	5	5,585		
Lake x treatment	5	165		
Fish (lake treatment)	608			
Total	619			
Age = 0+				
Treatment	1	9,176	184.48	0.005
Lake	2	12,629		
Lake x treatment	2	99		
Fish (lake treatment)	674			
Total	679			
Age = 1+				
Treatment	1	119,862	37.30	0.002
Lake	5	1,011,799		
Lake x treatment	5	16,066		
Fish (lake treatment)	3,119			
Total	3,130			
Age = 2+				
Treatment	1	94,207	82.49	0.012
Lake	2	102,869		
Lake x treatment	2	2,284		
Fish (lake treatment)	379			
Total	384			

attracting more angling effort. This did not prove to be the case. While we were successful in creating production sized groups of all-female triploids, the mixed-sex diploid rainbow trout grew better in the hatchery and continued to outperform the all-female triploid rainbow trout throughout the field experiment.

While there were significant differences in growth between diploids and triploids in the 1992 rearing study, the differences observed in 1993 were more pronounced. One reason for the increased divergence in growth observed in 1993 may be a result of the feeding and photoperiod regime followed. In 1992, fish were fed by hand a maximum of eight times during an 8.5-hour daylight period. An automatic feeding system and photoperiod controller were installed in 1993 which allowed feeding every half hour for an extended day length. This not only could account for the gross difference in growth seen between 1992 and 1993, but could, at least in part, account for the more pronounced difference in growth between the diploid and triploid fish in 1993. One could speculate that if capacity for growth were different between two groups of fish, the more one maximized the potential for growth the more dramatic the actual differences would become.

In the field, the all-female triploid rainbow trout were smaller every year of the study, in every lake. The reason for reduced growth of triploid fish in the wild can only be speculated upon. Similar studies found better growth for triploid fish while other studies found diploid fish grew at a higher rate. Simon et al. (1993) suggested the results from growth studies may be strain dependent.

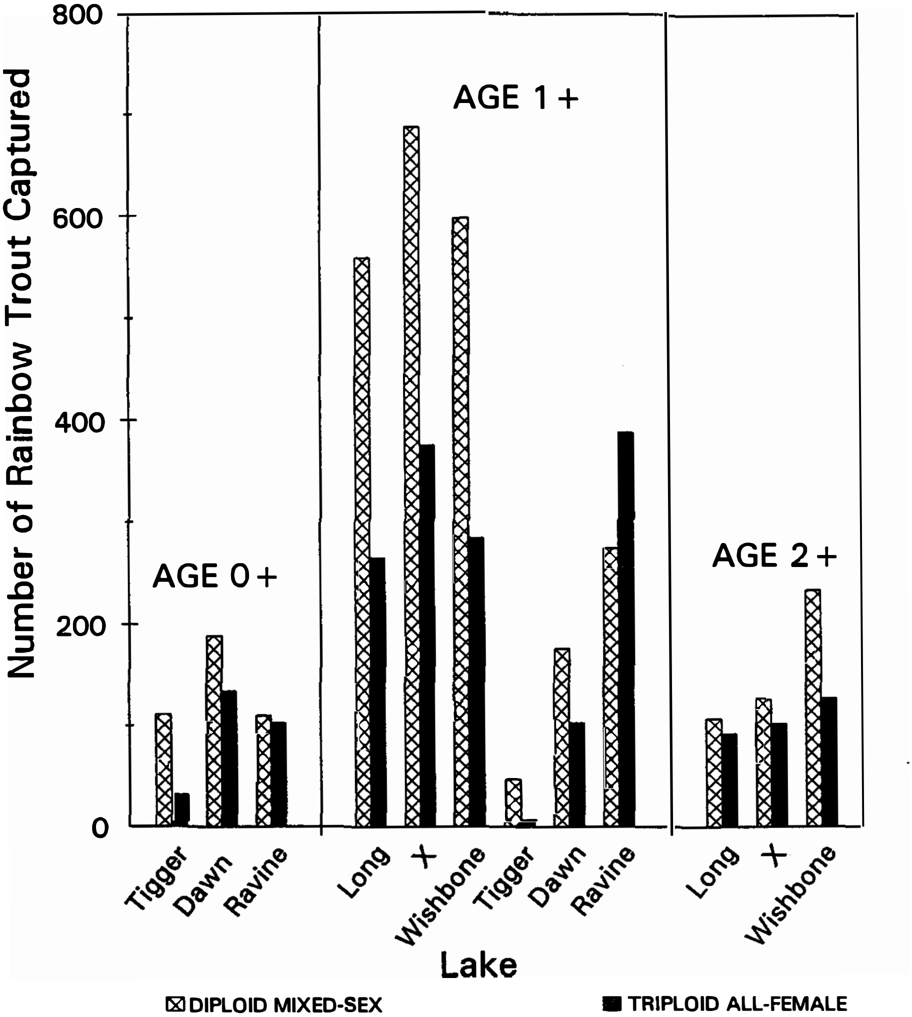

Figure 3. The number of mixed-sex diploid and all-female triploid rainbow trout caught in six lakes in southcentral Alaska.

During the period between initial ponding and July 21, 1993, diploid fish exhibited a greater survival in the hatchery than did their triploid counterparts (though not statistically significant). This difference was not observed thereafter, however. These differences are consistent with what one might expect in a hatchery situation where mortality (induced through stress or lack of proper feeding response) generally is higher in newly ponded fish than in fish that have been feeding successfully for a period of time and are more adapted to their rearing environment.

In the field, the difference in survival between the treatment groups was evident shortly after stocking (triploid fish having a 34 percent lower catch than diploid fish) and remained at the same level through age-1+. The initial reduction in survival of

Table 5. Results of Chi-square tests comparing the catch of mixed-sex diploid and all-female triploid rainbow trout stocked in lakes in southcentral Alaska.

Age	Lake	Catch		Expected catch	χ^2	df	P
		Diploid	Triploid				
Age = 0+	Dawn	189	134	162	4.7	1	0.03
	Ravine	110	103	107	0.1	1	0.75
	Tigger	111	33	72	21.1	1	<0.01
Total		410	270				
Age = 1+	Long	560	265	413	52.8	1	<0.01
	Wishbone	599	284	442	56.2	1	<0.01
	"X"	689	376	533	46.0	1	<0.01
	Dawn	176	103	140	9.6	1	<0.01
	Ravine	274	389	332	10.0	1	<0.01
	Tigger	47	9	28	12.9	1	<0.01
Total		2,345	1,426				
Age = 2+	Long	138	114	126	1.1	1	0.29
	Wishbone	282	176	229	12.3	1	<0.01
	"X"	146	127	137	0.6	1	0.44
Total		566	417				

triploid fish could be attributed to stress. Virtanen et al. (1990) found increased mortality in triploid fish during periods of stress (stocking, high water temperatures, low dissolved oxygen levels). Also, smaller all-female triploid fish could have been out-competed by the larger mixed-sex diploid fish and may have been more vulnerable to predation by older rainbow trout previously stocked in each lake. The reduction in the difference between the survival rates after age-2 could be due to increased mortality in the sexually mature age-2+ male diploid fish. While there was fishing mortality in three of the six lakes, we believe that anglers' preference for larger fish (diploid) would only strengthen our conclusions. However, we are unable to explain fully the results for Ravine Lake. While the results from the growth portion of the study were consistent with those of all other lakes we investigated, the higher survival of the triploid rainbow trout in Ravine Lake was not. One possible explanation could be related to the physical geography of the area. The west side of Ravine Lake is a high mountain ridge with a boulder-strewn slope extending into the lake, whereas the shorelines of the other five lakes are gently sloping and vegetated. The mass of boulders in Ravine Lake could provide protective cover from predation for newly introduced fingerlings.

The results from our study generally are consistent with those from a recent similar experiment with rainbow trout in South Dakota ponds (Simon et al. 1993). In the South Dakota experiment, both the diploid control and triploid treatment were of mixed sex. Like our experiment, survival to age-1+ was significantly lower for triploid fish. In both field experiments, the diploid fish grew better than the triploid rainbow trout. In the South Dakota experiment, the triploid rainbow trout were larger at the time of stocking, however, there was no significant difference in mean length at age-1+, and by age-2+, the diploid fish were significantly larger. In our experiment, the diploid fish were larger at stocking and continued to grow at a greater rate than the triploid rainbow trout. Survival and growth results for older fish in the two experiments were similar. Survival to age-2+ (and older) for triploid fish in the South Dakota experiment was lower than that for diploids in each experiment although most of their differences were not significant.

Recommendations

With diminished survival and slower growth, wide-spread utilization of all-female triploid fish would result in less efficient management. Most stocked lakes in Alaska are landlocked and devoid of wild stocks. However, some candidate lakes either are ''open'' systems which contain indigenous stocks of rainbow trout or are subject to periodic flooding such that stocked fish likely would spawn with wild stocks in other systems. In these applications, only a sterile hatchery product could be considered and the field performance standard for the treatment need not be greater growth or survival. Recruitment from stocking into the fishery must be such that sufficient angling effort is attracted to make the extra expenditure worthwhile. Cost-per-angler-day provides a quantifiable framework for judging the merits of a proposed stocking of this, or any, hatchery product. Results of this study provide estimates of the number and size of fish that could be expected from a stocking of all-female triploid rainbow trout in southcentral Alaska lakes. Fishery survey data (Mills 1993) and stocking records can be used to estimate the resultant expected angling effort from such a program. The total cost of the project (stocking, assessment and management) can be divided by the expected (or realized) number of angler-days to compute a measure of management efficiency. Comparison of this value with those calculated for competing management strategies provides a basis for decision.

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Introduction of Saugeye to Control Overcrowded Crappie Populations[1]

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Introduction

Management of sport fishing in North American reservoirs is limited to a few procedures that address stocking, regulations and habitat manipulations. As reservoir managers are faced with increased demands by an increasing fishing pressure base and/or declining habitat, innovative techniques for enhancing sport fisheries are needed. Panfish populations, because of their high fecundity rates and precocious spawning nature, easily become overpopulated if predator/prey relationships become skewed. Mitzner (1984) identified the "small crappie syndrome" as the most critical crappie management problem facing biologists today.

Prey and predator stockings have been used to elicit desired density and growth responses of target species. On reviewing the literature regarding shad stockings, Devries and Stein (1990) concluded that overall benefits to sportfisheries were inconsistent. However, increasing predator densities appeared to have improved growth rates of targeted prey species (Kempinger and Carline 1978, Gabelhouse 1984).

With the production of the saugeye (*Stizostedion vitreum x S. canadense*), a new, fast-growing predator has become available to fisheries managers. Introduced into Lake Thunderbird, Oklahoma, in 1985, the saugeye became an appealing sportfish, showing rapid growth rates that allowed them to quickly enter the creel (Leeds and Summers 1987). More importantly, however, was a preference in their diet for Thunderbird's small stunted white crappie (*Pomoxis annularis*) (Leeds 1988, Horton and Gilliland 1990). Once these saugeye reached 18 inches (457 mm) TL, crappie became an important prey item.

Investigations into the management implications of saugeye introductions on the stunted crappie population in Thunderbird Reservoir were initiated. Initial findings from this investigation resulted in the conception of a multi-state study sponsored by the Walleye Technical Committee of the Northcentral Division of the American Fisheries Society to evaluate the utility of stocking saugeye into small impoundments to improve fish community balance through increased predation on panfish species.

Methods

Thunderbird Reservoir is a 6,070-acre (2,448 ha) impoundment serving as a municipal water supply for several central Oklahoma communities. The lake has mod-

[1]Contribution No. 221 of the Oklahoma Fishery Research Laboratory.

erate turbidity and dense beds of milfoil (*Myriophyllum* sp.) providing shoreline cover with a shoreline development ratio of 7.9, mean depth of 20 feet (6 m) and a maximum depth of 68 feet (21 m). Major predator species include largemouth bass (*Micropterus salmoides*), white crappie, white bass (*Morone chrysops*) and saugeye. Prey species include inland silversides (*Menidia beryllina*), gizzard shad (*Dorosoma cepedianum*), sunfish (*Lepomis* spp.) and small white crappie.

Saugeye were introduced as fingerlings (1.5 inch: 30 mm TL) in 1985 and stocking continued annually at rates from 12–38/acre (30–93/ha). Food habits of saugeye from Thunderbird Reservoir were obtained from Horton and Gilliland (1990). Adult saugeye were collected in autumn by night electrofishing from 1987 through 1993.

Mean catch rates (number/hour multiplied by 24 and expressed as net-nights), mean length at age and relative weights (W_r; Neumann and Murphy 1991) of white crappie were calculated from autumn trap-net samples collected annually from 1983 through 1993. Catch data were grouped by size (<5.5 inches: 130 mm TL; hereafter referred to as age-0; 5.5–8 inches: 131–199 mm TL, hereafter referred to as intermediate and ≥8 inches, 200 mm TL, hereafter referred to as large). The greatest overlap of lengths at age occurred in the intermediate size group, making this the target length group for density reduction (Boxrucker 1992). The large group was considered to be the minimum size of crappie acceptable for harvest by anglers. Crappie were aged using otoliths.

A non-uniform, random, daylight, roving creel survey was conducted on Thunderbird Reservoir from March through November 1985 through 1993. Twenty 10-hour creel days were surveyed each three-month (season) period. Catch rates were calculated using ratio-of-the-totals method (Summers, 1978).

In spring 1992, fingerling saugeye were stocked at densities of 50 per acre (125/ha) into 21 small impoundments (<1000 acres: 400 ha) in seven midwestern states (Oklahoma, Kansas, Nebraska, Illinois, Iowa, North Dakota and South Dakota). Saugeye populations were sampled in autumn 1992 with night electrofishing and crappie population statistics were calculated using autumn trap-net data.

All statistical tests were performed using Statistical Analysis Systems (SAS) software (1988). The catch, length at age and W_r data were not distributed normally and Log_{10} transformations did not normalize the catch and length data. Therefore, differences in trap-net catch for Thunderbird crappie were compared using a t-test procedure on ranked data. Saugeye stocking success in small impoundments was modeled using simple linear regression estimating relationships between saugeye catch rates and 18 physical and biological factors.

Results

Thunderbird crappie. Catch rates of adult saugeye (≥18 inches: 457 mm TL) in Thunderbird fluctuated between 1987 and 1993, but were not significantly different with the exception of 1991 (Figure 1). Typically, saugeye reach the size (18 inches: 457 mm TL) at which they prey significantly on crappie (Horton and Gilliland 1990) in three growing seasons, and first reached that size in 1987 (Leeds 1988).

Catch rates of intermediate crappie in Thunderbird decreased in years after saugeye reached 18 inches (457 mm TL; $p<0.05$), whereas catches of large crappie increased over the same time period ($p<0.0001$) (Figure 2). Recruitment of age-0 crappie, while high prior to saugeye introduction (1985), declined significantly ($p<0.0001$) and

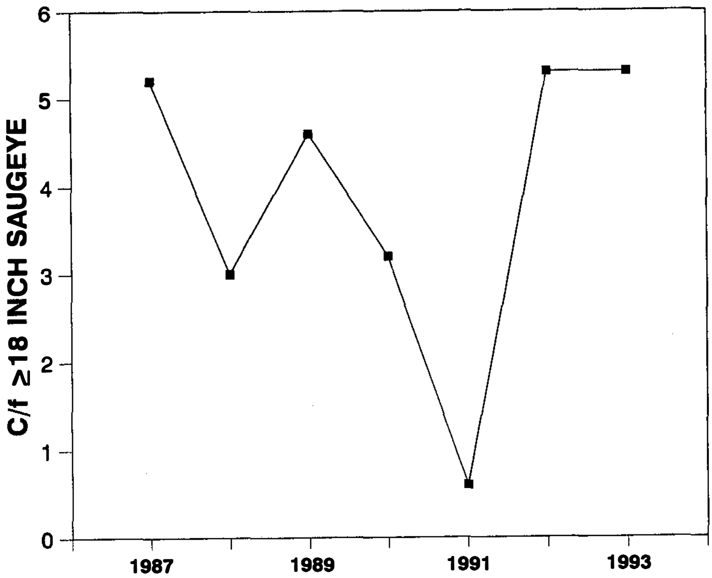

Figure 1. Autumn night electrofishing catch rates (C/f) of saugeye ≥18 inches in Thunderbird Reservoir, Oklahoma, 1987–1993.

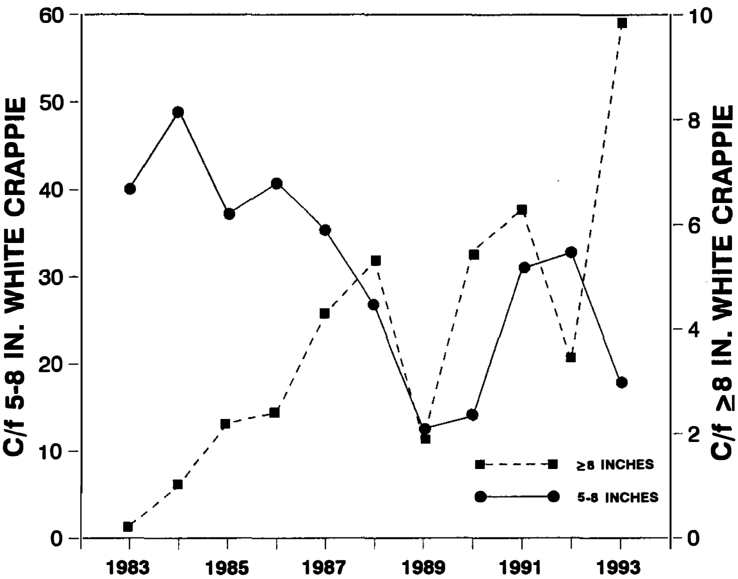

Figure 2. Catch rates (C/f; number/net-night) of two size groups of white crappie from autumn trap-net samples from Thunderbird Reservoir, Oklahoma, prior to saugeye reaching 18 inches TL (1983–1986) and years following saugeye reaching 18 inches TL (1987–1993).

stabilized by 1987 (Figure 3). W_r of intermediate sized crappie improved although W_r of larger crappie declined ($p<0.0001$) (Figure 4). An increase in mean length of crappie ages 1 through 5 was observed after the saugeye population reached 18 inches (457 mm) TL (Figure 5). There also was an increasing trend in crappie anglers' harvest rates. A significant regression model ($p = 0.015$) was found when comparing crappie angler catch rates to trap-net catch rates from the previous autumn ($r^2 = 0.59$).

Multi-state study. Recruitment of stocked saugeye into 21 midwest small impoundments met with varied success. Autumn electrofishing saugeye catch rates (no/hr) ranged from 0 (5 lakes) to 46.5. Although many physical and biological factors were analyzed for their influence on saugeye first-year survival, only a few appeared to have a significant impact. Based on the first two years of introduction, saugeye were more likely to be successful ($r^2 = 0.481$, $p<0.05$), in impoundments with low predator densities (primarily *micropterus* spp. populations). Saugeye also were more likely to be established in lakes that had only moderate densities of crappie (C/f = 20-80/net-night) and poor crappie PSD ($r^2 = 0.478$, $p<0.005$) and finally, gizzard shad presence also positively influenced saugeye recruitment ($t = 2.390$, $p = 0.027$).

Discussion

Trap-net catch rates in Thunderbird Reservoir showed that the size structure of crappie improved after the introduction of saugeye. The decline in intermediate crappie appears to be attributed directly to saugeye and not to fluctuations in year-class strength. Based on growth rates, two strong year-classes (1983 and 1985) should have produced increases in intermediate crappie over the next five years. However the trap-net catch of intermediate crappie continued to decline despite stable recruitment in following years.

Significant increases in large crappie were seen in Thunderbird Reservoir as a consequence of saugeye introduction. These changes are indicative of what occurs when predator densities change relative to their food supply. The W_rs of both size groups of crappie shifted. With the decrease in density of intermediate crappie, the W_rs improved. Correspondingly, the W_rs of large crappie decreased with an increase in density. As the number of larger crappie continues to improve, there will be a need to assure that angler harvest will also increase to maintain a favorable predator/prey balance.

In selecting candidate lakes for saugeye introduction, biological considerations appear to be more critical than physical ones. Several authors have shown that saugeye adapt well to a wide variety of physical habitats (Humphreys 1984, Johnson et al. 1988, Leeds 1989, Lynch et al. 1982). The Walleye Technical Committee study summarized here also points to this fact. However, large predator populations (both largemouth bass and adult crappie) seem to limit the success of saugeye introductions. It was not apparent if this was due to direct predation on young saugeye by these predators or that forage necessary for saugeye survival was limited. While the preliminary results of this study may provide some insight as to lake selection for saugeye, these finds certainly are not conclusive. Health of fish stocked, treatment and method of stocking, and kinds and abundance of zooplankton at time of stocking were not evaluated and all could have contributed to stocking success for failure.

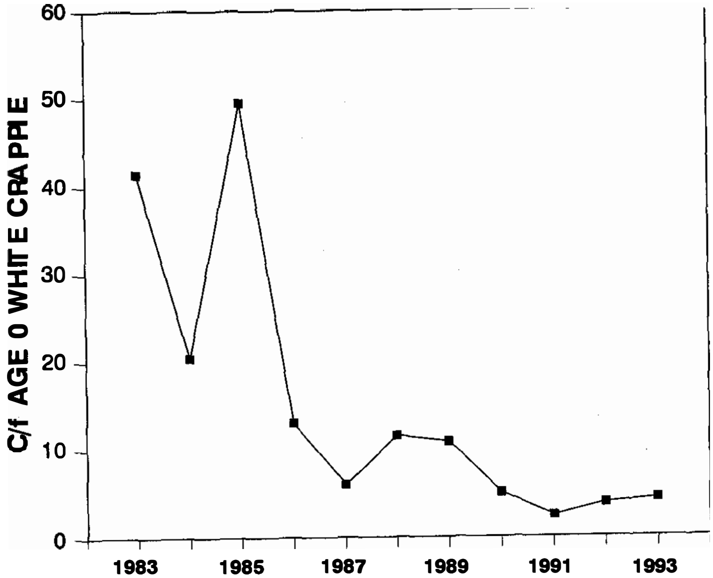

Figure 3. Catch rates (C/f; number/net-night) of white crappie <5 inches TL (age-0) from autumn trap-net samples from Thunderbird Reservoir, Oklahoma.

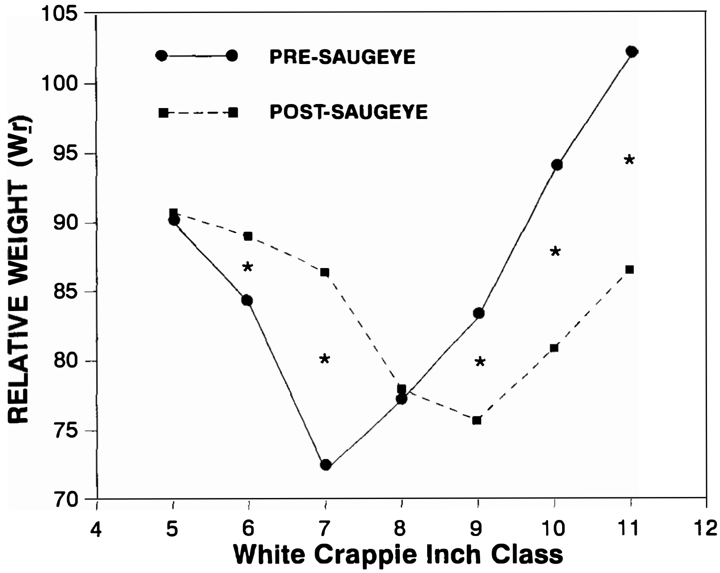

Figure 4. Relative weight (Wr) of white crappie from years prior to saugeye reaching 18 inches TL (PRE-SAUGEYE) and years following saugeye reaching 18 inches TL (POST-SAUGEYE) from Thunderbird Reservoir, Oklahoma. *denotes statistically significant differences.

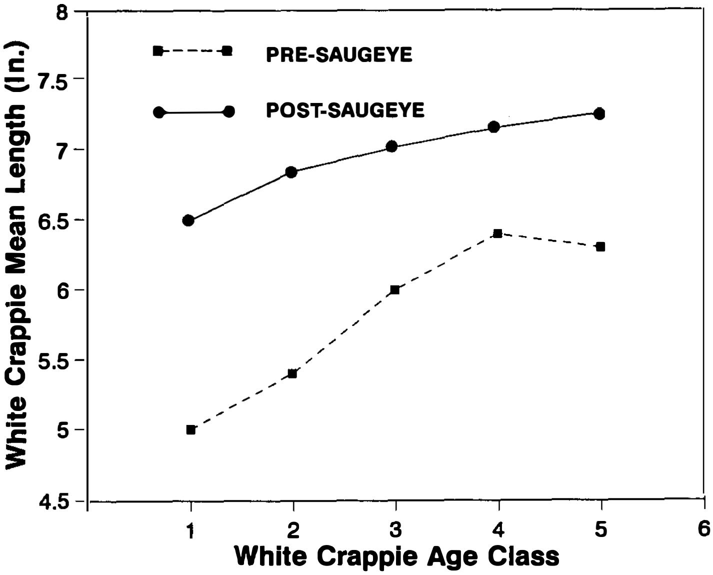

Figure 5. Mean length at age of white crappie from years prior to saugeye reaching 18 inches TL (PRE-SAUGEYE) and years following saugeye reaching 18 inches TL (POST-SAUGEYE) from Thunderbird Reservoir, Oklahoma.

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Reservoirs as Landscapes: Implications for Fish Stocking Programs

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Introduction

Although consideration was given to the applicability of ecological concepts at the landscape level over 50 years ago, the distinct field of landscape ecology emerged only during the 1980s. Studies of spatial dynamics shifted gradually from an initial emphasis on how ecological processes, especially disturbance, affect spatial patterns to contemporary landscape ecology concerned with the mechanisms by which spatial patterns affect processes (Turner 1989). Landscape ecology today includes study of the influences of spatial heterogeneity on biotic and abiotic processes and the management of that spatial heterogeneity (Risser et al. 1984).

As suggested by the root of the term ''landscape,'' concepts of landscape ecology generally have been applied to terrestrial systems. However, it is the trait of spatial heterogeneity, rather than the land component, that is conceptually relevant. Likewise, size (area) has been of much less concern than has heterogeneity in definitions of landscapes. Dimensions tend to be prescribed by the environmental mosaics, and biological responses are studied as they relate to patch characteristics, especially patterns of connectivity of components, spatial and temporal variability, and the impacts of disturbance (Forman and Godron 1981).

Among the processes emphasized in landscape ecology have been the interactions between landscape patterns and animal movements. Current studies of the impacts of habitat fragmentation on neotropical migrant birds are based strongly on such concepts. Likewise, dispersal limitations due to habitat fragmentation may affect population dynamics and genetic integrity, important issues in conservation biology (Hughes and Noss 1992). In addition, predation, productivity and micro-conditions are influenced by local movements and, therefore, landscape patterns.

Applications of landscape ecology to aquatic systems have lagged behind terrestrial systems, and have focused on streams and floodplains (Frissell et al. 1986, Schlosser 1991). Watersheds readily have been characterized as landscapes, but lakes and reservoirs, despite their key roles in watersheds, as well as their large sizes and spatial heterogeneity, seldom have been perceived as landscapes. Man-made reservoirs, because they represent a combination of lake and stream characteristics, exhibit high levels of spatial heterogeneity. In addition, seasonal water level fluctuations in many reservoirs provide disturbance, frequently according to quasi-predictable temporal trends. Our objective in this paper is to draw attention to the fact that reservoirs function as landscapes, with a focus on effects of landscape on dispersal. These effects have profound implications for decision making relative to management, including stocking programs for spatially heterogeneous reservoirs.

Reservoirs as Landscapes

About 1,600 reservoirs have been created throughout North America, generally for multiple uses, including water supply, flood control, navigation and recreation. Although fishing seldom is a primary function, reservoirs support as much as 25 percent of the freshwater fishing in the United States, and fisheries managers have focused on preserving or enhancing recreational fishing opportunities in reservoirs.

A mainstem reservoir is constructed by damming a river, thereby inundating the channel, adjacent floodplain and tributary streams. The morphology of the watershed is assumed by the reservoir, frequently resulting in a highly dendritic surface pattern. In contrast to most natural glaciated lakes, the reservoir is likely to have great spatial complexity.

Also in contrast to natural lakes, the man-made reservoir typically will lack coevolved trophic assemblages and the species present are unlikely to be highly adapted to reservoir habitat conditions (Noble 1986). Despite high total productivity of fishes, generalists dominate many reservoir fish communities, and trophic linkages are weak (Vadas 1990). Consequently, reservoir fisheries offer great potential for improvement through management, but management must be targeted close to the point of anticipated fishery response to avoid the buffering effects of the weak linkages. Consistent with these relationships, stocking programs have been employed widely for introduction or supplementation of predator and prey populations, aquatic vegetation control, and genetic diversification. However, spatial variability has been given little consideration in implementing such programs.

Attention to spatial variability in reservoirs has focused largely on specific longitudinal, vertical and horizontal scales. As one moves from headwaters to outflow, variations in limnology, water quality, morphology, sedimentation, productivity and species composition follow rather predictable trends. Similarly, well-defined patterns in specific characteristics distinguish the water column into hypolimnion and epilimnion, and laterally separate the littoral from the limnetic zones.

Our research with littoral juvenile largemouth bass (*Micropterus salmoides*) suggests that population variability should be addressed from an additional standpoint which accounts for the spatial heterogeneity arising from inundation of river valleys and tributary streams. A hierarchy of spatial components—from microhabitat to cove to embayment to basin—characterizes our system and many similar systems (Figure 1). Littoral fish communities, including species such as juvenile largemouth bass, depend on the more developed littoral areas characteristic of reservoir side-arms, or embayments. Open reservoir basins, with their harsh littoral environments, can act as barriers, restricting movement from one embayment to another. Individual embayments therefore can behave as quasi-independent units in terms of littoral fish population dynamics. As a consequence, management of littoral species such as largemouth bass must be directed to the embayment level of the hierarchy, or lower, and impacts may be only local.

Landscape concepts are equally applicable for limnetic species. Primarily pelagic species, such as shad (*Dorosoma* spp.), crappie (*Pomoxis* spp.) and the *Morone* species, range more widely than littoral species, and decisions concerning stocking and management may be more appropriate at the basin or reservoir-wide level of the landscape hierarchy. Stocking of highly mobile species which may emigrate from

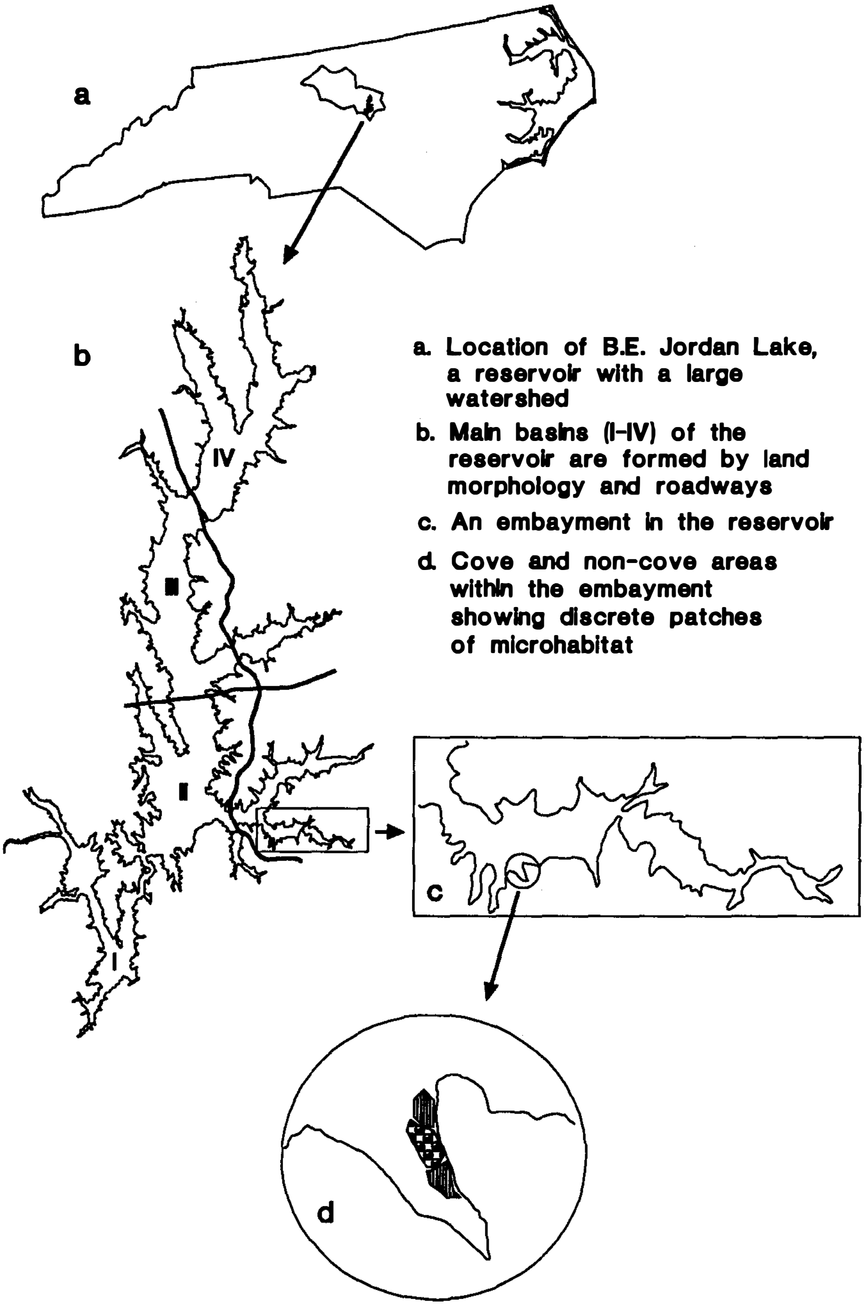

Figure 1. Landscape hierarchy of B. Everett Jordan Lake, North Carolina.

reservoirs should be considered at watershed or broader landscape scales, including connections with other systems.

Implications for Stocking

Largemouth bass, as a principal sport fish in U.S. reservoirs, have been the subject of extensive management efforts. Bass management in reservoirs has included introductions of prey species; addition of new subspecies to provide genetic diversification; habitat manipulation, especially the addition or removal of cover; and, primarily, harvest regulations. Supplemental stocking of bass in reservoirs where natural reproduction and recruitment are limiting typically has been unsuccessful (Keith 1986). Among the explanations for this lack of success have been intra-cohort competition and inferior performance of hatchery-produced fish in natural environments. Nevertheless, hatchery fish have had adequate survival to significantly alter genetic characteristics of largemouth bass in many southern reservoirs where Florida largemouth bass have been introduced into established populations of northern largemouth bass (*e.g.*, Kulzer et al. 1985). These results suggest that traditional methods of evaluating stocking programs, such as lakewide censuses or creel surveys, may underestimate stocking success.

Stocking programs, when employed, typically have been conducted with the intent of supplementing natural reproduction on a lake-wide basis. Nevertheless, stocking sites usually are chosen on the basis of convenience of access for hatchery trucks. Consequently, locations of boat launches, causeways and bridges are more likely to influence the location of stocking sites than are habitat characteristics or local fish population levels. Proper connectivity of the stocking areas to suitable habitats may be essential to dispersal, and perhaps survival of stocked fish. Fortunately, fish stocked into new environments frequently exhibit immediate exploratory behavior which may at least help them encounter a nearby patch of satisfactory habitat. But, without further dispersal to facilitate distribution among available spatial resources, the stocking hardly can be expected to succeed.

Spatial Heterogeneity and Fish Distribution

Since 1987 in B.E. Jordan Lake, North Carolina, and since 1992 in Lago Lucchetti, Puerto Rico, we have been studying the abundance and distribution of young largemouth bass. One objective is to understand better the behavior of young bass and their habitat requirements, with the goal of determining whether fingerling stocking programs can impact recruitment.

During a previous long-term study on Lake Conroe, a 21,000-acre (8,500 ha) reservoir in Texas, densities of young bass in cove rotenone samples appeared to vary consistently among six coves over most years (Klussman et al. 1988). Initial investigations of four bays using electrofishing in 14,000-acre (5,700 ha) Jordan Lake also showed that bass abundance varied rather consistently among bays (Phillips 1994). As much as three-fold differences in juvenile bass abundances were observed among embayments in any given year. These differences were equal to the year-to-year variations in abundance observed in individual embayments. Despite systemwide events such as pronounced annual differences in water level regimes, embayments seemed to function as discrete patches of littoral habitats.

Since adult largemouth bass are known to be quite sedentary, we hypothesized that young bass also exhibit limited movements. We initiated a two-year study to determine whether young, naturally reproduced bass were dispersing (Copeland and Noble 1994). Fingerling bass were collected from two widely separated coves of a 160-acre (65 ha) bay as soon as they were large enough to handle. Fish were tagged with individually unique binary-coded microtags and immediately released at the collection site. Recaptures over the next 15 months came primarily from within the marking coves and adjacent areas, with most fish found within 200 meters (220 yd) of their release site. In the second year, fish were tagged from both cove and non-cove areas, and recaptures again occurred primarily at or near the marking sites. None were found outside the bay, despite intensive sampling efforts. Young bass appear to remain within a small home range and stay in it for their first two growing seasons.

But what do hatchery bass do? To find out if hatchery bass would exhibit similar limited movements, we conducted another two-year study in the same bay, during which we compared movements of wild and hatchery bass tagged as fingerlings (Jackson et al. 1993). Although hatchery bass dispersed slightly more than wild bass, most movement appeared to be within the first few days after stocking. Thereafter, they also tended to stay within a limited range and most were recaptured within 640 meters (700 yd) of their release sites.

Recapture efforts in this bay of Jordan Lake involved multiple electrofishing samples around the entire bay. During this repeated sampling, it appeared that, much like the variation among bays found earlier, bass catch-per-effort varied consistently among areas within the bay, over sampling periods which spanned most of the annual cycle. Subsequent quantification of spatial variation in catch-per-effort has borne out this observation. Irwin (1994) has determined that density differences are consistent with micro-habitat characteristics, which may be extensive, sometimes encompassing an entire cove or stretch of shoreline, or may be localized to areas of shoreline of as little as a few meters.

One cove of about 10 acres (4 ha) which exhibited consistently low densities of young bass was selected for an adaptive fish management experiment to evaluate potential for supplemental stocking of low-density areas. Based on movement distances found in the previous studies, the cove was large enough to retain the stocked fish if movements were comparable to our previous studies. However, virtually all recaptures came from higher quality habitat outside, but near the cove where stocked. Not only did the fish leave the low-density cove with low-quality habitat, those which moved appeared to stop at nearby high-quality habitats rather than dispersing throughout the bay.

A parallel study was conducted lake-wide in Lago Lucchetti, Puerto Rico, a 250-acre (100 ha) reservoir. Gross habitat characteristics, assessed visually according to experience in North Carolina, and young bass densities as estimated from electrofishing catch-per-effort, varied consistently from one side of the reservoir to the other. When micro-tagged bass were supplementally stocked on the high-density, high-quality habitat side of the reservoir, recaptures primarily were from the stocking area. In contrast, when fish were stocked into the low-density area, they distributed widely throughout the reservoir within a few weeks, and were more likely to be recaptured in the high-quality habitat than near the stocking sites.

Unfortunately, micro-tag recapture data do not provide the continual data needed to map the route taken from stocking site to recapture site. Refinement of habitat characterization has not yet been conducted for Lago Lucchetti, so even if bass followed the shoreline, as most bass appear to do when they move in Jordan Lake, it is uncertain whether bass in Lago Lucchetti passed over high-quality habitats as they moved.

From Concepts to Practice

Clearly, spatial heterogeneity is a characteristic of reservoir systems and affects the dispersal and distribution of young largemouth bass. Spatial characters, therefore, need to be considered in management decisions (Table 1). However, before such a dimension is brought into decision making, information beyond effects on dispersal is needed. It is not only important that dispersal does or does not occur, but also what effect movement or non-movement ultimately has on population dynamics and cohort productivity (recruitment) for a particular area. Additionally, both juvenile and adult population levels must be determined in light of specific habitat carrying capacities in order to assess the efficacy of supplemental stocking.

Since stocked fish may move from low-density, low-quality habitats to high-density, high-quality habitats, it may be impossible to impact densities in low-quality target areas. In such cases, habitat enhancement of low-quality areas would appear

Table 1. Landscape hierarchy for reservoirs, with characteristics of habitat and/or fishery management.

Landscape hierarchical level	Management or decision level
Watershed	Water/nutrient inflow Management of emigrant species
Reservoir	Boating access Stocking/harvest of pelagic species Water level management Vegetation control
Basin	Shoreline access Recreational development Stocking of pelagic species
Embayment	Stocking/harvest of littoral species Genetic diversification Spawning refuges Access restrictions
Cove/non-cove Shoreline	Habitat enhancement Shoreline stabilization Habitat enhancement Disturbance minimization
Microhabitat	Disturbance minimization

to be a better management approach, followed by supplemental stocking if natural colonization of improved habitats does not occur. Although stocking into high-density areas rather than low-density areas is counter-intuitive, there appears to be merit to such an approach. In our intensively studied embayment, we observed as much as a three-fold variation in year-class strength over seven years, without detectable effects on survival. This suggests that carrying capacity for juvenile bass exceeds density in most years, and that even high-quality habitats could support up to three times as many bass in some years.

At our current level of understanding, we are unable to determine precisely at what level of structural hierarchy management should be directed. Because of the differences in densities and growth that we have observed among embayments, largemouth bass management strategies such as stocking should be feasible at the embayment level. If open basins impede movements of older fish among bays, embayment management could become more comprehensive, to the point of managing different embayments independently. The effectiveness of management at smaller scales, such as coves or shorelines, probably depends on the interactions between bay size and habitat spatial pattern. Hypothetically, under quality habitat conditions, stocked fish movements would be limited enough to facilitate management at the cove level.

Adequate habitat characterization could become the limitation to selecting the scale at which management should be conducted and to decision making on stocking sites and stocking rates. We have been able to calculate average habitat characteristics in embayments ranging from 160 to 425 acres (65–172 ha) by surveying approximately 5 percent of the shoreline. At this scale, habitat quality correlates with mean densities of young bass calculated from shoreline electrofishing samples. We typically are able to conduct habitat analyses of an embayment in less than two days. However, this sampling intensity provides little indication of microhabitat patchiness or connectivity, and finer resolution may be required to guide management decisions.

Reservoirs which serve flood control functions also exhibit disturbance regimes that are addressable through landscape ecology. Viewed on a whole-lake basis, water level fluctuations simply result in surface area changes, and may be of little importance to pelagic fish population dynamics. However, within individual embayments, water level fluctuations can have dramatic impacts on the types, quantities and connectivity of habitats available for littoral fish species. In Jordan Lake, high-quality habitat tends to become more limited and more widely dispersed as summer water levels decline. Such relationships can have important implications for management decisions, including stocking rates, when water level regimes are somewhat predictable.

Given the complexities of making multi-criteria management decisions at different scales, we have turned to current technologies for more efficient assessment of the components of our reservoir landscapes. Geographic information systems (GIS) are powerful tools designed to incorporate data at any spatial scale, and can be used extensively in fisheries management (Giles and Nielsen 1992). After a GIS database is created for a reservoir and its surrounding watershed, many management questions can be addressed and management scenarios developed with consideration for spatial heterogeneity and hierarchy. GIS can provide visual identification of suitable areas for localized stocking programs based on criteria such as access, habitat availability and patch connectivity for dispersal of fish. The role of other factors, such as disturbance (water level fluctuations) and carrying capacity, also can be examined for specific areas.

Defining the relationships between success of stocking strategies and reservoir landscape heterogeneity requires precise analysis of the interactions of fish population parameters and physical habitats at the appropriate scale. The combination of these factors ultimately defines the management unit in the landscape. What is to be expected if a landscape approach to assessment of stocking programs is employed by fisheries managers? Effectiveness and efficiency of stocking programs will increase, and greater fishery potential will be realized.

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Special Session 4. *Integrating Game and Nongame Management*

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Introduction

The purpose of this session is to highlight past and present successes in order to enhance future cooperation and combat any deviciveness that could be caused by increased competition for limited funds.

People from both game and nongame programs often make some valid points: (1) it's about time nongame received additional emphasis; (2) emphasis should be on game because sportsmen foot the bill; (3) efforts directed at game also have benefited nongame; and (4) incidental nongame benefits were not on purpose, and were not well documented or evaluated. In some cases, benefits to game also have not been documented or evaluated well enough. One of our papers will address the issue of evaluating past actions.

Definitions

What is nongame? It's not in the dictionary! But, there are several definitions for game, including: (1) wild animals taken in hunting for sport or food; (2) the flesh of game animals; (3) an object of ridicule or attack (i.e., fair game); and (4) having a resolute unyielding spirit. Well, whether a game biologist is one with unyielding spirit or an object of ridicule may depend on your viewpoint!

My definition of ''game'' is any wild animal utilized for sport hunting, subsistence or trapping. There are many other definitions—some for specific purposes. For instance, under the Migratory Bird Treaty Act, a trumpeter swan (*Cygnus buccinator*) is a gamebird—even though there are no open seasons—but a crow (*Corvus brachyrhynchos*) is not a gamebird.

Historically, the term "game" was used in reference to wildlife, including game and nongame. All wildlife was considered "game" by early explorers, such as Lewis and Clark. Further, many primitive societies, people in underdeveloped countries and even people living in remote portions of North America still utilize what some of us call nongame for food and clothing.

Some people think that, in the future, we may no longer have sport hunting, I hope that they are wrong because I support hunting as a management tool and a legitimate form of recreation. But, if they are right, will future students of wildlife conservation laugh as they look back at the 20th Century as that time period when we differentiated between game and nongame? Another viewpoint is that, to those people who do not hunt or trap, all wildlife is nongame.

The term "integrate" means to incorporate into a larger unit; "cooperate" means to work with another for mutual benefit. So, cooperation is when I am flying a deer (*Odocoileus* spp.) survey and I count cranes (*Grus* spp.) for somebody else's program. But, integration is when both efforts are combined into a single program. However, the difference is less distinct when both programs are supervised by the same agency. It's really one program, so cooperation actually can be, at a higher level, a form of integration.

From the title of this session you would believe it focuses on management, as opposed to research. We all know the basic differences, although management often includes data collection and monitoring, and research can involve management decisions and actions. This session focuses on integrating management—but we just as easily could have focused on integrating research efforts. Considerable research has gone into the topic of integrative management and that will be reflected in today's papers.

Concepts

The most important point I could make today is that we all share certain common goals and approaches. But, there are buzzwords for certain concepts which some people normally associate with one program or another. Really, these concepts reflect shared approaches.

Biological diversity. Much discussion has occurred about the exact definition of biological diversity, but most of us share the same gut feelings about diversity. We believe it is desirable to achieve the fullest compliment of native populations that would have been sustained under a natural functioning ecosystem. Inherently, we have problems with introductions of exotic species (from European starlings (*Sturnus vulgaris*) to Barbary sheep (*Ammotragus lervia*)), especially where they negatively impact native populations. When we strive for diversity, I believe most of us focus on long-term maintenance of diversity, as opposed to short-term artificial actions.

Natural processes. Most agree that under ideal situations, we prefer to allow natural processes to function and create diversity. Where natural processes no longer can function, we try to emulate them. For example, moist-soil management and green-tree reservoirs are used to mimic natural hydrological regimes. Where it is necessary, we pursue providing artificial situations to compensate for missing habitats, food sources or other factors. Here, too, there are similarities between the programs; providing wood duck (*Aix sponsa*) nest boxes is comparable to creating artificial cavities in pines for red-cockaded woodpeckers (*Picoides borealis*). But, again, most of us prefer natural stands which can provide suitable cavity trees on a regular basis.

Ecosystem management. Lately, there has been a lot of talk about taking a broader look at ecosystems (ecoregions). When I refer to ecosystems, I also think of a deeper approach. In wetland management, for instance, it's not only important to consider how a given wetland might relate to surrounding wetlands within or outside an artificial boundary, but to consider the relationships between adjacent uplands, water quality, sediments, invertebrates—ecosystems. Ideally, we prefer to conserve and manage large enough portions of the landscape so that we include all the parts integral to a healthy, functioning ecosystem and so that we have the luxury of minimizing management actions. But, instead—because this often is impossible—we try to do the best we can.

Partnerships. We recognize that government agencies can only do so much. Increasingly, we have worked with private individuals and organizations to incorporate their activities and support through innovative new partnerships. Important programs include agricultural programs like the Conservation Reserve Program, as well as wildlife programs like Partners In Flight, North American Waterfowl Management Plan, etc. Private-lands programs are incorporating broader wildlife concerns as well.

New constituencies. Really, there are no ''non-traditional'' uses of wildlife; but, recently, our efforts have been somewhat redistributed and, some would argue, more evenly distributed among all uses. *Montana Outdoors* magazine recently featured the northern pintail (*Anas acuta*) in their section on watchable wildlife; I thought this was great because many game species are very watchable. Several displays on watchable wildlife at the poster session also highlight game species. In many cases, the same species can be considered game or nongame, depending on the orientation or intent of the user . This will be highlighted in our last paper.

Closing

I think we need to focus on common sense and efficiency in combining activities. Many benefits can be gained through cooperative pursuit of funding, and reductions of conflicting and overlapping efforts. The opportunities are great. For instance, Missouri has been initiating interdisciplinary, long-term research projects. Such integrated research will allow them to explore relationships between wildlife species and their environments that otherwise would not have been feasible.

The importance of habitat quality and quantity is our strongest common bond. We don't really manage wildlife much, we manage habitat. What we most often do is manage people or encourage them to manage habitat. That's why outreach, education and basic communication are so important, as are providing incentives for proper management.

I firmly believe sportsmen also support nongame efforts, and that their experiences afield are richer when they observe or interact with nongame wildlife—like duck hunters in a blind watching an osprey (*Pandion haliaetus*) catch a fish and then shake the water from its wings as it rises.

There are many ongoing efforts common to both programs, and many efforts directed at cooperation and integration. We share common challenges and direction. To highlight this, we have some exciting papers for you this morning.

Influences of Waterfowl Management on Nongame Birds: The North Dakota Experience

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Introduction

Wildlife managers traditionally have been expected to manage for game. More recently, they have been asked also to manage for nongame species. This added responsibility posed a problem; although a general belief was that "what is good for game animals is good for nongame," little objective evidence supported the claim. Nor was there evidence that management for game species was detrimental to nongame. Further, managers had little guidance for practices that benefit nongame.

This quandary led to the effort described herein. We attempted to determine the effects, both positive and negative, that current management activities for game species have on nongame species in North Dakota. We focused on waterfowl management activities and their influence on all species of birds. The authors, convened by R. L. Kreil, possessed a range of experience and expertise with birds and their habitat needs in North Dakota. Some of us are birders, whose professions are unrelated to wildlife, whereas others of us have careers in wildlife science. Although some authors are employed by natural resource agencies, we did not represent these agencies during our discussions.

The Process

Twice we met for two days each; we also did much work outside the meetings. U.S. Fish and Wildlife Service (Service) officials with responsibility for waterfowl management in North Dakota provided a list and description of common management practices in the state. One of the managers participated in the second meeting to clarify the extent of application, criteria used and the responses of waterfowl to the practices under discussion. We discussed the effects of 26 practices (Table 1) on each bird species that regularly occurs in the state (Faanes and Stewart 1982). We paid particular attention to 22 species of special concern that either have a limited geographical range with a substantial share of the population breeding in North Dakota, have declined significantly at the state or continental level, or are indicators of rare, unique or threatened habitats.

We tried to reach a consensus about the effects of management practices. We often found that too little was known about these effects and the habitat needs of certain species to reach a decision comfortably. A thorough review of the literature would have been helpful but was precluded by time constraints. Therefore, we based our conclusions on personal knowledge and experience.

We categorized the effect of a particular practice on each species as *very beneficial*, *beneficial*, *negative*, *very negative* or *unknown*. We did not list species for which we judged effects as neutral or insignificant. For example, when we evaluated wetland creation in a central North Dakota grassland, we concluded that creating wetlands has no effect on rock wrens (*Salpinctes obsoletus*), because they do not occur there. The same no-effect determination was made for yellow warblers (*Dendroica petechia*), even though they occur in the area, because the habitats that they use are not affected by this practice. We sometimes made additional comments to provide some

Table 1. Waterfowl management practices in North Dakota that were evaluated by the review team.

Grazing—short term
Grazing systems—rotation
Wetland restoration
Wetland creation
Wetland creation (in a wet meadow, type II area)
Wetland creation (west of Missouri River)
Re-seeding uplands to dense nesting cover
Re-seeding uplands to native grasslands
Cattail control by glyphosate
Cattail control by burning
Wetland manipulation/management on hayland or pastures
Wetland enhancement
Delayed haying
No till/minimum till
Predator trapping on islands
Island creation/peninsula cutoffs
Predator fence exclosures
Prescribed burning
Haying on wildlife areas
Cropping—to rejuvenate nesting cover or establish lure crops
Tree planting—multi-row shelterbelts
Weed control by chemicals, mowing, grazing, etc.
Gravel shoreline
Nest structures/boxes
Bird feeders
Chemical fallow

explanations of potentially difficult and unclear points that were considered during our evaluation.

Our product was a written report, "A review of wildlife management practices in North Dakota: Effects on nongame bird populations and habitats," provided to Refuges and Wildlife, Region 6, of the U.S. Fish and Wildlife Service. In this paper we describe our procedures, illustrate results of our analyses and provide a perspective on management of North Dakota habitats, specifically grasslands.

Example: Short-term Grazing

An example dealing with the management practice of short-term grazing will provide some insight into the process and illustrate the results of our evaluation (Figure 1). A brief description of the practice, as provided by managers, first is given. In this instance, the intended objectives are removing litter, favoring warm-season grasses and grazing cool-season grasses. Some measure of the scope or extent of the practice also is given, which, in this example, is 20,000 to 25,000 acres of Service land grazed annually.

Qualifiers expand on the description and give guidelines about the practice and situations to which it should be applied. For short-term grazing, we mention that impacts will vary by location and habitat conditions. Qualifiers also allude to differing responses by birds in the short term versus long term.

Results provide our assessment of the (proximate) effects on various species. Short-term grazing was not deemed very beneficial or very negative for any species. We judged it as beneficial for 11 species, particularly those that favor short, grassy vegetation for breeding habitat. We assessed the practice as negative for 14 species and for dabbling ducks as a group; these species tend to favor more luxuriant grassy cover, which grazing reduces. We listed the effects on six species as unknown; some of these uncertainties were due to differences between nesting and foraging habitats.

The Comments section indicates that the frequency of treatment mentioned under Qualifiers (once every one to five years) was not adhered to consistently. The degree to which these guidelines are followed probably depends on various constraints and interests of individual mangers. We also suggested that the intended purpose of the treatment—to reduce invading cool-season grasses such as Kentucky bluegrass (*Poa pratensis*)—was unlikely to be achieved.

Example: Wetland Manipulation on Hayland or Pasture

This practice is targeted at privately owned land, where new wetlands are created or drained wetlands are restored temporarily (Figure 2). A treated wetland is dewatered after May 15 but before June 15 to permit grazing or haying for the rest of the season.

We concluded that the practice is very beneficial to spring-migrating ducks and shorebirds, and to several species of swallows. We thought it would be negative for American coots (*Fulica americana*), pied-billed grebes (*Podilymbus podiceps*), soras (*Porzana carolina*), virginia rails (*Rallus limicola*) and Wilson's phalaropes (*Phalaropus tricolor*), which nest over water. Any of these birds may begin nesting

MANAGEMENT PRACTICE: Grazing--short term

Mostly done to stimulate grass production through litter removal or removing competing species of grass, done for short-term (2- 4-week) periods, aimed at grazing cool-season exotics (also can affect cool-season natives) and enhancing warm-season natives. The FWS typically grazes 20,000 to 25,000 acres annually.

QUALIFIERS:

1. 2-4 weeks duration in May.
2. Purpose is to promote taller native grasses and reduce cool-season invaders such as Poa pratensis.
3. Some of the species mentioned will be affected only if wetlands are present.
4. Impacts will vary depending upon geographic location and excess vegetation, e.g., east to west changes in amount of prairie, growing potential, and litter build-up.
5. The practice may result in an immediate short-term decrease for some species; however, in the long term the practice may be beneficial to all the negatively impacted species and detrimental to the positively affected species.
6. Frequency of use is once every 1 to 5 years (but variable; may be annual for a few years, then cease for 10-15 years).

Very beneficial: [++] No species indicated.

Beneficial: [+] ferruginous hawk, killdeer, willet, marbled godwit, common nighthawk, horned lark, Sprague's pipit, Baird's sparrow, chestnut-collared longspur, Brewer's blackbird, brown-headed cowbird

Negative: [-] dabbling ducks, American bittern, northern harrier, ring-necked pheasant, prairie chicken (nesting habitat), Virginia rail, sora, upland sandpiper, short-eared owl, mourning dove, dickcissel, grasshopper sparrow, Le Conte's sparrow, sharp-tailed sparrow, bobolink

Very negative: [--] No species indicated.

Unknown: [?] gray partridge, Wilson's phalarope, clay-colored sparrow, western meadowlark, lark bunting, savannah sparrow

COMMENTS: Pattern of use is inconsistent with regard to frequency. Practice also reduces cool-season natives. Doubtful that it reduces Poa pratensis (according to A. D. Kruse, among others).

in the flooded area, which is drained soon thereafter. Effects on ducks, American bitterns (*Botaurus lentiginosus*) and other shorebirds were listed as unknown, but we recognized that responses depend on the availability of alternative nesting cover and brood-rearing water in the vicinity of the treatment area. The practice may harm breeding ducks, for example, if it attracted birds to an area because of the flooded wetland and then left them or their water-dependent young stranded after drawdown.

MANAGEMENT PRACTICE: Wetland manipulation on hayland or pastures

Drained, partially drained, or created wetlands are enhanced or partially restored with water control structures in active hayland or pasture. These projects are normally designed to provide temporary water for pair habitat and use by spring migrants, while increasing soil moisture for forage production. The landowner is allowed to draw down the wetland for haying or grazing purposes normally between May 15 and June 15 with the structure being closed after harvest to catch fall rains and next spring's runoff.

This type of practice is not as beneficial as complete restoration. However, the success of these projects is important in demonstrating that wetlands can be an important component of a successful agricultural operation. This practice fills a niche that may provide wildlife benefits on thousands of wetland acres and is designed to show that wildlife and agriculture can co-exist and mutually benefit. Since 1987, 30 wetlands totalling 429 acres have been manipulated in North Dakota.

QUALIFIERS:

1. Typically involves drained wetlands that private landowners do <u>not</u> want restored, but wish to use for forage production (hay or pasture).
2. Temporarily flooded until May 15 - June 15. Uses water control structures. The water is drawn down rapidly after an agreed-upon date between May 15 and June 15.
3. Done in areas with sufficient brood water.

Very beneficial: [++] Migrating waterfowl (ducks), migrating shorebirds

Beneficial: [+] swallows

Negative: [-] American coot, pied-billed grebe

Very negative: [--] sora, Virginia rail, Wilson's phalarope

Unknown: [?] ducks, nesting American bitterns, nesting shorebirds (these unknowns are dependent upon available nesting cover and brood water)

COMMENTS: In order for this management practice to be beneficial, nesting habitat and brood water must be available in adjacent areas. Later drawdowns would provide greater benefits and lessen negative impacts on nesting species. The later the drawdown the greater the benefits.

The Alternative to Management

Any management practice has feasible alternatives. One alternative is to do nothing, which can be done either after a conscious decision—that leaving alone is the best management—or by default—through failing to take any other action. Doing nothing should be considered as objectively as any other practice; it may be the most appropriate strategy for a given place and time. Some individuals believe that purchasing land provides all the protection necessary and that leaving the land idle generally is the preferred management alternative. At the other extreme are some managers who feel that they must actively manage all their lands.

We discussed the consequences of the no-action alternative in terms of long-term effects on the habitat and bird communities in North Dakota. We focused on grassland, the most extensive natural ecosystem in the state. We could have examined wetlands, the other major natural habitat in the state, in a similar vein. If others repeat our exercise for another area, they may wish to consider different ecosystems.

Historically, disturbance played an important role in the formation and maintenance of North Dakota's grasslands. The prairie was grazed heavily but intermittently by huge herds of bison (*Bison bison*), which left the landscape in a mosaic of habitats ranging from severely grazed to ungrazed. Grasslands also were subjected to fires, some set naturally by lightning, others set intentionally by Native Americans for a variety of purposes. Furthermore, varying climatic regimes, geological formations and topographic features added diversity to the landscape. It is with this perspective that management of prairies should be viewed.

Settlement by Europeans altered the majority of natural grasslands in North Dakota. Cultivation was the most direct and immediate agent of change, and a large part of the state has had its prairie turned upside down. Other effects were less direct, but equally destructive. Among these were intentional or accidental introductions of Eurasian plant species, such as Kentucky bluegrass and leafy spurge (*Euphorbia esula*), which have invaded native grasslands and disrupted the original plant communities. Efforts to reduce weedy plants through use of herbicides have had further detrimental effects on native vegetation, especially forbs. Grazing by free-ranging bison has been replaced by grazing by domestic livestock, often confined in small pastures for the entire growing season at stocking rates that lead to severe overgrazing, with attendant soil erosion and changes in plant composition. Fire suppression by settlers also facilitated increases of woody vegetation, especially in moister parts of the state.

Most lands managed by the Service and other agencies that manage public natural resources are small islands in a mosaic of privately owned land. The same land-use practices have been applied on these public lands as on private lands but in different proportions. Much less publicly owned wildlife land is cultivated annually and much more is left idle for extended periods of time either as part of a management plan or due to lack of resources, local public concerns or characteristics of the tract.

The consequences of idling grassland and suppressing fire for long periods may be summarized in three scenarios of succession, which depend on the prevailing precipitation regime. In the more mesic areas, especially in eastern North Dakota, the grassland ultimately is transformed to woodland, dominated by small trees and large shrubs such as green ash (*Fraxinus pennsylvanica*), Russian olive (*Elaeagnus angustifolia*) and chokecherry (*Prunus virginiana*), with an understory of smaller shrubs and introduced grasses. The second scenario, applicable to somewhat drier areas, has succession proceed to a shrub community dominated by wolfberry (*Symphoricarpos occidentalis*), silverberry (*Elaeagnus argentea*) and Woods rose (*Rosa woodsii*). The third scenario, anticipated in the more arid parts of the state, does not have a woody community arise; instead, the grassland becomes choked with an accumulation of litter.

Breeding bird communities change drastically under these vegetation successions from grassland. The first scenario (for mesic areas) leads to increases in numbers of many shrubland and woodland-edge species, such as willow flycatcher (*Empidonax traillii*), eastern and western kingbirds (*Tyrannus tyrannus* and *T. verticalis*), house wren (*Troglodytes aedon*), American robin (*Turdus migratorius*), gray catbird (*Dumetella carolinensis*) and brown thrasher (*Toxostoma rufum*). The second scenario (in drier areas) favors species such as clay-colored sparrow (*Spizella pallida*) and common yellowthroat (*Geothlypis trichas*). Few species likely benefit to any degree from the third scenario (in more arid areas). In contrast, any of these successional changes reduce populations of almost all true grassland bird species, such as ferruginous hawk (*Buteo regalis*), willet (*Catoptrophorus semipalmatus*), marbled godwit (*Limosa fedoa*), burrowing owl (*Athene cunicularia*), Sprague's pipit (*Anthus spragueii*), Baird's sparrow (*Ammodramus bairdii*) and chestnut-collared longspur (*Calcarius ornatus*).

The establishment of tall, woody vegetation in prairie landscapes affects the bird community in several ways. Most obvious is the direct loss of prairie plant species, through competition for light, water or nutrients. Insects that use those plants and serve as a food base for many birds then disappear. Also, certain grassland birds avoid areas with woody vegetation, even where appreciable grasses and prairie forbs remain. Woody vegetation can fragment a grassland, dividing it into noncontiguous parts that are too small individually to be used by area-sensitive prairie birds. The ecological influence of woody plants extends well beyond their canopies. Trees and tall shrubs provide nesting sites and hunting perches for raptors, travel lanes and denning sites for mammalian predators, and vantage points from which brown-headed cowbirds (*Molothrus ater*) can survey the surrounding area and locate nests to parasitize. Thus, the intrusion of woody vegetation has far-reaching consequences to grassland bird communities.

Overall, succession to woody vegetation, as anticipated under the first two scenarios, leads to an *increased* total number of species in an area. This local species diversity usually is enhanced by having a large number of different habitats and habitat edges in close proximity. Local species diversity should be distinguished from the concept of biodiversity and the goal of preserving as many species and population as possible.

How can publicly owned wildlife lands in North Dakota contribute best to biodiversity? Although these lands could be managed to increase local numbers of shrubland, woodland and woodland-edge species, the areas probably will not make important contributions to maintaining continental populations of those species. Most such species have widespread distributions and are much more common elsewhere. Most have large populations that are not in jeopardy. Many grassland species, however, especially those of the mixed-grass prairie, have little alternative habitat outside the northern plains. The distributions of some of these species center in or near North Dakota; no major populations are elsewhere. Further, many grassland species have suffered population declines at least as severe as birds of eastern forests, which have received greater popular and scientific attention. The lark bunting (*Calamospiza melanocorys*) and grasshopper sparrow (*Ammodramus savannarum*), as examples, each declined 60 percent during the past quarter-century (Johnson and Schwartz 1993).

One mission of the Service and other wildlife management agencies is to protect and manage wildlife populations. Their goal is not to pack as many species as possible onto the parcels of land it manages, as might befit a zoo. Accordingly, the primary interest is in maintaining natural ecosystems and biodiversity, not enhancing local species diversity. Both game and nongame prairie species need protection in grassland states such as North Dakota, which is in the heart of their breeding range. Management should be directed at grassland (and wetland) species, especially endemic ones, in preference to those of other habitat affinities and distributions.

Exceptions exist, but most species are maintained best by sustaining, in as natural a condition as is feasible, the ecosystems on which they rely. For that reason, we believe that management of publicly owned wildlife lands in North Dakota should be oriented toward protecting and restoring large tracts of the most natural ecosystems extant. As a consequence, those actions will protect biodiversity, although local species diversity will not be maximized. This approach may not be optimum for those who enjoy the natural values of unnatural habitats, such as bird watching in shelterbelts on a national wildlife refuge dominated by grassland, but it will favor the long-term protection of the widest array of game and nongame bird species. Compromises, such as restricting woody vegetation primarily to riparian areas, would increase local diversity and allow associated public uses but still would permit restoration of native plant and animal communities in most of the area.

Managers often are responsible for large areas of degraded grassland. Restoration of those habitats to a more natural condition may result in a local reduction in the number of species using those areas. The public should consider those losses as an acceptable trade-off made by managers in favor of preserving natural biodiversity, including game and nongame species, of the northern Great Plains.

Recommendations and Research Needs

During our deliberations and our interactions with managers, it became obvious that many consequences of management were not well understood. In certain instances, managers were conducting activities to favor particular species, but at least some experts thought that the actions could be detrimental to those species. In other cases, consequences for target species, as well as nontarget species, simply were unknown. For some actions, effects on target species were understood, but influences on other species were not. And for some practices, immediate effects were known, but long-term ones were not. Effects of some practices differ markedly by geographic region or time of application (usually mediated by climatic influences); such spatial and temporal variations need to be appreciated. Because the costs of various management actions differ, the expected results should be quantified so that alternative courses can be objectively considered.

We view with concern the uncertainty about the effects of management practices. We recognize that decisions sometimes must be based on incomplete knowledge. Nonetheless, we strongly recommend two general courses of action. First, research should be conducted on proposed management activities. Particularly in need of study are actions that meet one or more of the following conditions: (1) they are expected to influence large areas of land, (2) they have drastic effects, (3) they are applied where sensitive plant or animal species occur, or (4) they have little previous history on which to base conclusions. For some management practices, research findings are available; these should be reviewed and evaluated.

The second recommendation is that responses be monitored after management is implemented. Previous research should lead to some expectation of the results managers anticipate. Follow-up monitoring will assess whether or not the results meet those expectations. If not, further evaluation of the management action is warranted. Careful monitoring also helps understanding the geographical and temporal influences on the results of management.

It might be argued that research and monitoring are too expensive, that problems are immediate and that action must be taken without delay. We believe that the issues—and the resources—are too important *not* to evaluate carefully. Moreover, conducting management practices that have not been evaluated and may not have the desired effects can be a serious waste of funds.

We made specific research recommendations, including: (1) gather basic life-history information on species of special concern, which would allow for a better evaluation of the effects of current or proposed management practices; (2) initiate and (importantly) continue broad-scale review of landscape ecology and land-use changes, which combined with breeding bird surveys and other population studies would allow for a better evaluation of actual changes; and (3) determine actual effects of practices such as grazing and fire on cool-season grasses such as Kentucky bluegrass and on native species, to resolve apparent inconsistencies among research findings and expected and actual results.

Conclusions

We emphasize that the report we provided is not the final word, but only a beginning. Further research and careful monitoring of the results will lead to a clearer understanding of the values of management actions to both game and nongame species. Since the report was completed and distributed, we have received comments about our process and the results of the review. We encourage further scrutiny of our product, for that will improve our recommendations and ultimately enhance management and the natural resources themselves. More recently, we learned that our report is being used as a template for evaluation of the effects of the North American Waterfowl Management Plan on non-waterfowl migratory bird populations.

If the evaluation were to be repeated elsewhere, we offer two suggestions. First, involve managers throughout the process. They not only provide essential information about the practices, but also gain a better appreciation of the process and the resulting product. Second, try to agree on the objectives of the management practice. This is important for both the managers and the review group and keeps everyone focused on the same target.

Acknowledgments

We thank several U.S. Fish and Wildlife Service personnel, notably Stephanie L. Jones, who provided a grant for the participants' travel and for printing the report; Michael McEnroe and Kevin Willis, who graciously gave the group detailed information about the management practices; and Arnold D. Kruse, who offered insight into certain practices. Biologists at the Northern Prairie Wildlife Research Center provided a preliminary evaluation of our review and technical perspectives on the biology and habitat needs of some species. Mark Dryer and Karen Kreil, U.S. Fish and Wildlife Service, and Craig Bihrle, North Dakota Game and Fish Department, offered many constructive comments on various drafts of the report. L. D. Igl, H. A. Kantrud, R. E. Kirby, A. D. Kruse, M. R. McEnroe and E. Rockwell offered comments on this manuscript.

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Reclaiming Minelands to Benefit Wildlife

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Introduction

In 1977, Congress passed the Surface Mining Control and Reclamation Act (SMCRA) designed to regulate surface mining and the reclamation of mines in the United States. The act covers many different subjects, including reclamation objectives to benefit fish and wildlife populations. Since the enactment of SMCRA most reclamation plans have included components used to enhance fish and wildlife habitat. The plans have been based largely on field experience and biological opinion derived from broad ecological theory. Unfortunately, mine operators and regulatory agencies have not had adequate scientific data describing how species of fish and wildlife respond to reclamation efforts, particularly in the high plains grassland of the central United States.

This research effort was initiated in 1990 in Wyoming's northcentral high plains shrub/grassland. The overall purpose was to determine how different forms of reclamation affect wildlife. The specific objectives were: (1) compare wildlife communities associated with reclaimed and native surfaces within and between mineland areas; (2) evaluate the effects of vegetation, topographic diversity and rock piles on birds and small mammals in reclaimed areas; and (3) determine what factors of reclaimed impoundments influence their use by waterfowl and other water birds.

Study Area

The study sites were selected in northeastern Wyoming near the cities of Gillette and Upton. These areas were selected because of the high concentrations of coal and bentonite mines that are in close proximity. The region is a relatively high elevation (greater than 1,500 meters), sagebrush (*Artemisia spp.*)/grassland community with a rolling topography. Snow can occur in any month of the year but primarily occurs between November and April. Late spring snowstorms in May and June sometimes have major impacts on wildlife in the areas. In the central portion of the study area there are approximately 20 operating surface coal mines. Approximately five of these mines have extensive reclamation efforts. In the northeastern portion of the study area, bentonite mining has created over 866 ponds. Some of these ponds have been created for mitigation of wetland losses elsewhere. Some of them have been completely reclaimed and some have been left unclaimed. The reclaimed ponds are surrounded by grassland sage communities, interspersed with fragments of ponderosa pine. The wetland resources outside of these impoundments are minimal and restricted to seasonal playas and small rivers and streams. Stock ponds in the area also can offer some habitat for migrating and breeding waterfowl (Rumble 1989).

Methods

Terrestrial Wildlife

To evaluate vegetation, birds and mammals, plots on 22 reclaimed and 14 native terrestrial sites were selected. A total of 40 reclaimed and 34 native 100-by-300 meter (3 ha) belt transects were established to sample birds and habitat. Small mammals were sampled on subplots 100 by 100 meters (1 ha). Big game observations also were made on these plots.

Vegetation sampling began in late June and was completed by mid-August in 1990 and 1991. Line intercept sampling (Eberhardt 1978) was used to estimate shrub density, percent canopy cover and the relative abundance of shrub species. A total of four transects, each 50 meters long, was placed on each hectare. Vertical canopy cover and ground cover were estimated by growth forms using a Robel pole (Robel et al. 1970) and a Daubenmire quadrat (Daubenmire 1959). Growth forms included Wyoming big sagebrush (*Artemisia tridentata*), other shrubs, perennial grasses, annual grasses, forbes, succulents and sedges. Basal cover was quantified as percent bareground, rock, litter, lichen and stems. Topographic diversity was quantified using the Land Surface Ruggedness Index (LSRI). The LSRI as described by Beasom (1983) estimates the topographic diversity of approximately 8.3 hectares. At reclaimed sites, rock piles were quantified by height, volume and number present on the mammal and bird transect. Volume was calculated by extending a 50-meter tape over the rock pile along the longest axis and the shortest axis. The product of length and width was then multiplied by height to estimate the volume. Conversions were made to determine rock piles per unit area.

Breeding bird sampling began 15 minutes prior to sunrise from May to mid-June in 1990 and 1991 on the terrestrial sites. All birds observed within the transect boundaries were identified to species and sex by an observer walking along the mid-point of the transect. The perpendicular distance from the transect midline to the bird was estimated. These data allowed for density estimates of bird populations, or number of birds per 3 hectares. Bird species diversity (Shannon and Weaver 1949) and species richness (number of species), along with density, were used to evaluate bird use of the habitat.

Collapsible Sherman traps were baited with oats and placed 10 meters apart on a 10-by-10 or 100-station grid on all of the study sites. Traps were set for five consecutive nights each year. Sex, species, age and weight were recorded for each individual caught. Individuals were permanently marked to document trap history and estimate population numbers.

Waterbirds

Waterbird sampling was conducted at 92 impoundments that were selected based on size and accessibility. Pond size ranged from 0.25 to 10 hectares. Physical measurements taken at impoundments included water quality (pH, chloride, hardness and turbidity), surface area, shoreline length, slope of adjacent banks, average depth, maximum depth, percent of drawdown, distances between wetlands, number of wetlands within 1 kilometer of the sample wetland and distances to disturbances. A shoreline development index also was computed for each wetland (Belanger and Couture 1988). Aquatic vegetative measurements taken were percent coverage and species composition of both emergent and submersed vegetation. Estimates were averaged to estimate coverage for the entire pond. Nesting habitat was estimated by quantifying visual obstruction, percent coverage and type of terrestrial vegetation within 50 meters of a sample wetland (Robel et al. 1977, Daubenmire 1959).

Between May 10 and August 15, 1991 and 1992, biweekly waterfowl counts were made on the impoundments. Monthly counts were made in March, April, September and October. Migrating and breeding waterfowl were surveyed from a half hour before sunrise until a half hour after sunset. All birds counted before May 15 and after August 15 were considered migrants. Pairs and lone males were counted to estimate the breeding population on each pond prior to June 23. Blood counts were conducted form June 10 to August 12. All wetlands were surveyed by walking the shoreline and checking the emergent cover.

Analysis

Two tailed paired t–tests were used to test for differences in bird communities and vegetation between years and between mines. Analysis of variance (ANOVA) was used to test for treatment effect on bird density, diversity and richness. Linear regression techniques were used to study habitat association of birds. Likewise, two tailed pair t–tests were used to test for differences in small mammal communities between years and between mines. An ANOVA was used to test for treatment effects on small mammal communities in reclaimed habitat as measured by their density, diversity and richness.

Mann-Whitney rank-sum tests (Day and Quinn 1989) were used to test univariate differences in variables among used verses unused wetlands. A pooled variance t–test and separate variance t–test were used for variables that had homogenous and heterogeneous variances, respectively (Dixon 1988). Stepwise logistic regression was used to identify habitat variables that could explain the use of wetlands by waterfowl.

Results

Terrestrial Wildlife

The most abundant bird species on reclaimed terrestrial sites was the lark bunting (*Calamospiza melonocorys*), while on native sites, the western meadowlark (*Sturnella neglecta*) was the most abundant. Most species were approximately equally distributed among reclaimed and native sites except the Brewer's sparrow (*Spizella breweri*) and grasshopper sparrow (*Ammodramus savannarum*). Brewer's sparrows were far more abundant on native grassland sites, especially where big sagebrush was abundant, while grasshopper sparrows were more abundant on reclaimed surfaces.

Overall, bird species richness on reclaimed surface areas was relatively similar between study sites. However, bird density and diversity did differ. The mean density, diversity and richness of bird communities were greatest on treatments that included rock piles. Rock piles appeared to have a stronger influence on bird richness, density and diversity than the topographic diversity. Topographic diversity did influence species diversity. Birds had very strong, non-linear associations with topographic diversity on reclaimed and undisturbed surfaces. The optimum LSRI to maximize bird density was approximately 15 in reclaimed and 9 in native habitat.

In reclaimed habitats, canopy cover from perennial forbs seemed to influence the number and diversity of bird species that were present. Regression analysis showed that percent canopy cover and frequency of shrubs were correlated with an increase in bird density and bird species diversity. Bird richness was increased by vegetation height and number of succulents, both of which contribute to vegetation diversity. There were large differences observed among the vegetation communities of reclaimed sites and native sites. Native sites were structurally more variable because of the dominance of bunchgrasses and Wyoming big sagebrush. Although dominated by rhizomatous and annual grasses, reclaimed sites had much more vegetative cover (Mean = 44 percent) than native sites (mean = 23 percent). Bunchgrasses and shrubs were present on reclaimed surfaces, but rarely in a composition similar to native surfaces. Because of these differences, the vegetation seral stage of the reclamation considered in this study offers a different, perhaps even new habitat ''type'' to bird communities.

Bird communities in native habitat had strong associations with the total canopy cover and frequency of full shrubs. The vegetation height and frequency of succulents (cacti) enhanced only species richness in bird communities. Of these four vegetation community variables, the most important to bird communities in the native habitat was total canopy cover.

A trapping effort of 25,000 trap nights resulted in a capture of 1,221 individuals representing 14 species. The deer mouse (*Peromyscus maniculatus*) was the most commonly caught small mammal (78 percent of all captures). In reclaimed habitat, the western harvest mouse (*Reithrodontomys megalotis*) was found 19 percent of the time while only 5 percent of the time in native habitat. Rock piles and topographic diversity increased small mammal density and richness in reclaimed areas. In native habitat the deer mouse dominated the captures, however, *microtus spp.*, western harvest mouse and olive-backed pocket mouse (*Perognathus fasciatus*) were fairly evenly distributed. Reduced plant litter was highly associated with an increase in small mammal density. A high abundance of shrub cover was associated with an increase in density and diversity of small mammals.

Slope angle was the best topographic predictor of small mammal density in reclaimed habitat. However, the LSRI and slope angle were both effective in predicting small mammal density in native habitat (LSRI slightly better). Reclaimed sites that had maximum slope angles around 10 degrees tended to have lower small mammal densities. Small mammal density increased with slope angles up to 20 degrees in native habitat, and may increase beyond 20 degrees, but an insufficient number of sites with greater than 20-degree slopes were found.

Throughout the area, numerous predators such as red fox (*Vulpes fulva*) were observed denning in and around rock piles and foraging on reclaimed surface. Red fox appeared to be attracted to reclaimed areas. Avian predators such as short-eared owls (*Asio flammeus*), golden eagles (*Aquila chrysaetos*), Swainson's hawk (*Buteo swainsoni*) and ferruginous hawks (*Buteo regalis*) were commonly observed foraging at reclaimed areas, undoubtedly due to the abundance of small mammals.

Large mammals such as antelope (*Antilocapra americana*), mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus* and elk (*Cervus canadensis*) foraged on the reclaimed surface areas once the grasses reached the stage that provided adequate forage throughout spring and summer. In addition, many of these big game species were observed in the winter when heavy snows were blowing. Rock piles and topographic features provided protected areas for these animals to use.

Waterbirds

Nineteen species of migrating waterfowl and eleven other waterbirds were found at 59 reclaimed ponds in the study area. During migration, Canada geese (*Branta canadensis*) used 29 ponds, mallards (*Anas platyrhynchos*) 36, blue-winged teal (*Anas discors*) 16, green-winged teal (*Anas carolinensis*) 20, redheads (*Aythya americana*) 12, and ring-necked ducks (*Aythya collaris*) 14. Breeding waterfowl were divided into geese, puddle ducks and diving ducks to increase sample sizes for used wetlands and to develop broad preferences or trends. Only Canada geese, mallards and blue-winged teal were documented to breed at the study area. Broods of mallards and blue-winged teal were found at 15 ponds, while Canada geese broods were found at 9 ponds. Waterfowl used wetlands extensively during migration in late April and May, as well as September and October. Mallards were the most common migrating and breeding bird, as well as the most common waterfowl brood.

Canada geese used ponds that were significantly larger and subject to less draw-down than unused ponds. The number of ponds within 1 kilometer and the distance to nearest wetland were highly correlated with geese use. Geese used ponds containing a higher percentage of forbs than unused ponds.

Diving ducks were found on ponds that had a higher percent slope around the pond than unused. This variable undoubtedly reflects the steepness of the pond bank and therefore the pond depth. Puddle ducks were associated with wetlands that had greater amounts of submersed vegetation and were larger (Mean = 1.4 ha) and deeper (mean = 1.5 m) than nonused wetlands.

Discussion

The enactment of SMCRA in 1977 indicated the intent to consider wildlife in reclamation of disturbed lands. Many forms of reclamation occur depending on the community type. Our results indicate that habitat features can be constructed in reclaimed grassland/sage communities that allow the displaced animals to return and also enhance the area for other wildlife populations. Therefore, it is very important that reclamation plans contain specific goals for species and communities of wildlife desired. The results we found should help reclamation planners develop habitat features to establish wildlife communities. It is possible to plan for a wildlife community by considering species diversity, species richness and or specific species that might be desired.

In arid grasslands, species diversity was enhanced by contouring the land, including structures like rock piles, and reclaiming both shrub and grassland habitats. Species richness or species-specific responses were dependent upon features of the habitat. While rock piles enhanced small mammal populations, they did not provide places for raptors to nest or perch.

Rock outcrops in the native habitat typically are small (Height ≤ 1 m), very numerous (20–38 per 3 ha), very close in proximity (5–25 m) and usually located on or near ridgetops. Data collected in native habitat suggest that sites with greater numbers of rock outcrops per hectare, placed close together (clustered) and having a consistent height attracted birds in greater abundance. Our data show that most mines are constructing rock piles too large. Constructing smaller rock piles during reclamation will provide for more material to construct more rock piles. However, rock piles on reclaimed surfaces need to be slightly taller (height $\approx$ 1.2 m) than rock outcrops in native habitat, because the vegetation on the reclamation is taller. In addition, rock piles always should be associated with ridges.

Ground and above-ground nesting birds need some form of protective cover, thus plots of shrubs add to the diversity of the bird community. Perennial forbs appeared to have an influence on the ground-nesting bird community. Native sites with denser shrub cover had more ground-nesting birds than reclaimed areas.

When big game species are desired in the area, adequate forage must be available on a year-round basis. This means consideration must be given to the types of grass and shrub species planted and contouring the land so plots free from snow cover are available in winter and early spring. Cattle grazing and fencing must be carefully planned if big game use is a goal.

Water impoundments add to total wildlife diversity. These areas provide sources of food, water and shelter. The size, configuration, number and slope of water impoundments influence the type of waterbirds that use them. Vegetation (emergent and submersed) also should be considered in planning for waterfowl. Wetland complexes of three to four impoundments appear to be important to waterbirds. Complexes provide alternative sites when birds are disturbed by humans or predators. A series of ponds also means a greater availability of water throughout brood rearing and molting. Likewise, a varied habitat and dependable food supply can help meet the needs of birds during various life stages. Wetland complexes also provide a mosaic of habitat for fish, amphibians, reptiles and mammals.

The key to reclaiming mined land for wildlife is to first examine the community of wildlife. Second, consider the needs of the species including different life stages. Third, develop a mosaic of habitat that will attract the community desired. And finally, place specific features that may be required by a desired species. Reclaimed minelands are an opportunity to enhance communities of both game and nongame wildlife.

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Integrated Grassland Bird Habitat Restoration in Wisconsin Using GIS Habitat Modeling

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The Problem

Populations of many wildlife species dependent on grassland or wetland habitat have undergone severe declines in the Midwest during the last several decades (Edwards 1985, Robbins et al. 1986, Dahlgren 1988, Caithamer et al. 1993). In Wisconsin, annual surveys by the Wisconsin Department of Natural Resources (WDNR) document that blue-winged teal (*Anas discors*) populations declined 53 percent from 1973–93 (Gatti 1988, Andryk et al. 1993), while ring-necked pheasants (*Phasianus colchicus*) declined 67 percent from 1944–93 (WDNR 1989, Rolley 1993). Populations of several nongame grassland bird species, such as the grasshopper sparrow (*Ammodramus savannarum*), western meadowlark (*Sturnella neglecta*) and dickcissel (*Spiza americana*) have declined over 80 percent in Wisconsin from 1966–91 (U.S. Fish and Wildlife Service unpublished data: 1991), and are on the state's list of special concern species (Sample 1989). Habitat changes concurrent with these wildlife declines are equally dramatic: Wisconsin has lost over 50 percent of its original wetlands and 99 percent of its original prairies and oak savannas (WDNR unpublished data: 1993). Urbanization expanded over 50 percent from 1960–1985 (Wisconsin Chapter Soil Conservation Society of America 1987), planted corn acreage increased 50 percent from 1950–91 (Wisconsin Agricultural Statistics Service 1992) and pasture acreage decreased 57 percent from 1950–87 (U.S. Bureau of Census 1989). The continued loss of important grassland and wetland habitats threatens the future existence of the entire grassland/wetland wildlife community, game and nongame alike.

Our Solution

The Habitat Restoration Area Program

With these problems in mind, in 1990 the Wisconsin legislature established the Habitat Restoration Area Program, whose goal is to restore critical wildlife habitat on a landscape scale and thereby reverse the decline of many grassland and wetland wildlife species. The Habitat Restoration Area Program provides the WDNR with a stable, long-term commitment of state-bonded funds for land acquisition, easement and management that will total $15,000,000 over 10 years. Most of this program ($12,000,000) will be focused in a single area in southern Wisconsin, the Glacial Habitat Restoration Area (GHRA), where program costs and benefits are being evaluated.

Pilot Study Area: The GHRA

The GHRA is the largest concentration of wildlife funding in Wisconsin's history, covering 838 square miles (217,000 ha) and including parts of four counties: Winnebago, Fond du Lac, Dodge and Columbia. The management objective is to strategically restore 11,000 acres (4,450 ha) of wetlands in historical basins and establish 38,600 acres (15,620 ha) of idle grass nest cover through a variety of programs, including perpetual easements and fee title acquisition. The GHRA management plan calls for changing 10 percent of the landscape within an active agricultural setting. Management has focused on 21 target grassland bird species (Table 1), which collectively require a diversity of nesting habitat ranging from dry to wet, dense to sparse, and small acreages to large blocks. Restoring such a large habitat base, scattered across the landscape in a pattern that optimally will benefit the different target species, is a difficult task at best.

A Geographic Information System (GIS), using ARC/INFO software, was assembled to develop an integrated management plan for the GHRA landscape. This pro-

Table 1. Grassland bird species[1] targeted for habitat restoration.

Common name	Scientific name
Henslow's sparrow	*Ammodramus henslowii*
LeConte's sparrow	*A. leconteii*
Grasshopper sparrow	*A. savannarum*
Blue-winged teal	*Anas discors*
Mallard	*A. platyrhynchos*
Short-eared owl	*Asio flammeus*
Upland sandpiper	*Bartramia longicauda*
Northern harrier	*Circus cyaneus*
Sedge wren	*Cistothorus platensis*
Bobolink	*Dolichonyx oryzivorus*
Brewer's blackbird	*Euphagus cyanocephalus*
Loggerhead shrike	*Lanius ludovicianus*
Savannah sparrow	*Passerculus sandwichensis*
Wilson's phalarope	*Phalaropus tricolor*
Ring-necked pheasant	*Phasianus colchicus*
Vesper sparrow	*Pooecetes gramineus*
Dickcissel	*Spiza americana*
Clay-colored sparrow	*Spizella pallida*
Field sparrow	*S. pusilla*
Eastern meadowlark	*Sturnella magna*
Western meadowlark	*S. neglecta*

[1]Names after A.O.U. (1983).

gram is the first operational use of a GIS by the WDNR Bureau of Wildlife Management and, as such, is being used as both a demonstration and testing ground for broader application of this powerful management tool. For each of the 21 target bird species, habitat guidelines were drafted and are being translated into spatial models using GIS data layers of the GHRA.

GIS Data Layers

The data layers we entered in the GIS fall into three general groups: habitat modeling, management siting and mapping (Table 2). All layers were transformed into the Wisconsin Transverse Mercator (WTM) projection, an adaptation of the Universal Transverse Mercator projection centered on 90 degrees longitude to place the entire state in a single projection zone. The WTM coordinates of the section corners in the public land survey system in the GHRA were used to reference all data layers geographically.

Three layers were used for wildlife habitat modeling: landcover from LANDSAT satellite imagery, and wetlands from inventories of the WDNR and the U.S. Soil Conservation Service (SCS). We purchased two quarter scenes of LANDSAT Thematic Mapper data for May 7 and June 24 1990, to cover the entire GHRA. The landscape was initially classified into 24 cover types, and later pooled into 16 cover types (Table 3) with a resolution size of 0.2 acres (812 m^2) and an average landscape accuracy of 90 percent (Polzer 1992).

Table 2. Data layers for the GIS of the Glacial Habitat Restoration Area in southern Wisconsin.

Wildlife habitat modeling	
Data layer	Data source
Public land survey system	1:24,000 U.S. Geological Survey (USGS) quadrangles
WDNR wetlands	1:24,000 orthophotoquads
SCS wetlands	1:12,400 National High Altitude Photography (NHAP)
1990 landcover	LANDSAT Thematic Mapper
Management siting	
Data layer	Data source
SCS soils	1:15,800–20,000 NHAP photos
Archaeological sites and historic buildings	1:24,000 USGS quadrangles from Wisconsin State Historical Society (WSHS)
Land ownership	1:4,800 drawings from county tax listing offices
ASCS land management	1:7,900–15,800 NHAP photos
WDNR natural heritage	1:24,000 USGS quadrangles
Grassland bird abundance	1:40,000 county plat maps from WDNR bird surveys
Mapping	
Data layer	Data source
Lakes, rivers, streams	1:24,000 USGS quadrangles
State and local roads	1:100,000 USGS digital line graphs
1830s land cover	1:20,300 drawings of original General Land Office Survey records from Wis. Geol. Nat. Hist. Surv.
1930s land cover	1:14,100 drawings of Wisconsin Land Economic Inventory from WSHS

Table 3. Landcover types used to classify the Glacial Habitat Restoration Area landscape from LANDSAT Thematic Mapper data.

Original cover types	Final cover types	Percentage class accuracy	Percentage of landscape
Corn	Row cash crops	93	31.1
Beans			
Peas			
Oats	Small grains	82	6.0
Wheat			
Hay	Hay	82	13.5
Orchard	Orchard	25	0.0
Upland pasture	Pasture	71	3.2
Wetland pasture			
Gravel pit	Gravel pit	60	0.0
Urban	Urban	100	6.5
Upland deciduous trees	Upland deciduous trees	100	9.0
Upland coniferous trees	Upland coniferous trees	86	0.5
Idle cool season grass	Idle grass	70	7.1
Idle warm season grass			
Idle oldfield			
Idle forbs	Idle forbs	93	3.3
Wetland trees	Wetland trees	75	0.7
Wetland shrubs	Wetland shrubs	64	1.7
Reed canary grass	Wetland shallow herbaceous	93	6.6
Sedges			
Cattail	Wetland deep herbaceous	100	5.1
Bullrush			
Open water	Open water	100	5.8

Two different sources of wetland data were used because of the importance of wetlands to target wildlife species. The Wisconsin Wetlands Inventory (WWI), a statewide inventory patterned after the National Wetland Inventory (Cowardin et al. 1979), already existed in digital form. The WWI classified wetland types (WDNR 1992a, WDNR 1992b) but had not been updated in this area since its 1978 creation and did not include wetlands smaller than 5 acres (2 ha). We also digitized the SCS wetland inventory, which is dated 1990 and includes wetlands as small as 0.5 acres (0.2 ha); however, this inventory does not classify wetland types. The two wetland layers were overlaid and compared with the LANDSAT landcover to classify wetlands existing in 1990.

Six data layers were used to refine sites that satisfy the habitat models: soils, archaeological sites, land ownership, agricultural lands retired by the U.S. Agricultural Stabilization and Conservation Service (ASCS), important natural features within the WDNR Natural Heritage Inventory (NHI), and current grassland bird abundance. We created a layer of soils in the GHRA by digitizing or scanning the original drafted compilations of the published soil surveys in the four counties (Link 1973, Mitchell 1978, Fox and Lee 1980, Mitchell 1980) and merging them with attribute data from the state soil survey data base. The soil layer was used to located hydric soils for wetland restoration and xeric soils for short-grass management. We created a layer of landowners in the GHRA by digitizing all tax parcel sheets maintained by the tax listing offices of the four county governments and merging them with county tax parcel data bases of names and addresses. Land ownership data are used by wildlife managers to contact landowners during land negotiations. We are digitizing all farm tracts and fields enrolled in the Conservation Reserve, Wetland Reserve and Waterbank Programs of the ASCS; this data layer will be used by wildlife managers to identify where federal habitat restoration can mesh with WDNR efforts.

We are creating a layer of known locations of rare plants, animals and communities by digitizing the WDNR NHI, which contains occurrences of 12 prairie/savanna relics and 14 wetlands of significant quality within the GHRA. The NHI also contains 63 plant and animal locations within the GHRA, but does not include any target grassland bird species. Current patterns of abundance of the latter were digitized and plotted from extensive annual survey data. Grassland birds were surveyed each spring within the GHRA along 20 roadside routes patterned after the federal Breeding Bird Survey (Robbins et al. 1986). These two data layers will be used to focus habitat restoration around existing native grassland/wetland communities and areas of grassland bird abundance.

We created a layer of all historic buildings and archaeological sites in the GHRA from data provided by the Wisconsin State Historical Society. This data layer along with the NHI species occurrences was used to avoid conflicts in our restoration management. Other data layers were created to produce graphics that help orient data layers (hydrography and roads) or help convince the public of the need for restoration (historic landcover from the 1830s and 1930s).

Habitat models

We developed a spatial model for wetland restoration and four models for grassland restoration (dabbling ducks, pheasant, nongame birds—scattered approach; and nongame birds—large block approach) that address the needs of target species. The nest-cover models were overlaid to determine where habitat needs coincide and where integrated management efforts will be most efficient.

Wetland restoration. The wetland restoration model seeks to restore temporary and seasonal wetland types (i.e., feeding areas for dabbling ducks and phalaropes) within 1 miles (1.6 km) of permanent and semipermanent wetlands (duck brood-rearing areas). Wetlands of the latter type over 10 acres (4.0 ha) in size were selected from an overlay of WDNR and SCS wetlands and LANDSAT landcover. We delineated a buffer area including all land within 1 mile of the selected brood wetlands, in which temporary wetlands will be restored. An overlay of soils and existing wetlands on the buffer area identified drained wetlands as hydric soils without existing wetlands. Finally, the land ownership and archaeological layers were overlaid on the drained wetlands to identify landowners for easement/acquisition contacts.

Nest cover—dabbling ducks. The duck nest-cover model seeks to provide 5 percent of the GHRA uplands in idle grass/forb nest cover located within one-half mile (0.8 km) of breeding pair wetlands, which are within 1 mile of brood wetlands (*see* above). Existing permanent and temporary wetlands within the 1-mile buffer of brood wetlands were selected from the composite wetland data. A one-half mile buffer radius was delineated around the selected wetlands, within which nest cover could be restored. Areas impractical for nest-cover establishment (e.g., trees, urban, wetlands) were selected from the landcover layer and subtracted from the potential restoration area.

Nest cover—pheasant. The pheasant nest-cover model seeks to provide 10 percent of the GHRA uplands as idle grass/forb nest cover within 1 mile (1.6 km) of dense shrub or cattail wetlands, where pheasants concentrate in winter. Dense shrub and cattail wetlands over 20 acres (8.1 ha) in size were selected from the composite wetland layers. A 1-mile radius buffer area was delineated around the selected wetlands, within which nest cover could be restored. Areas impractical for nest cover establishment were again subtracted from the buffer area (as above).

Nest cover—nongame bird scattered approach. The scattered model for nongame grassland bird nest cover seeks to provide 10 percent of the GHRA uplands as idle grass/forb nest cover, but in larger fields and a greater variety of cover types than for ducks or pheasants. This model also excludes a 164-foot (50 m) zone around wooded and urban habitats, to minimize nest predation and parasitism. Relatively sparse, short nest cover is required for several of these bird species (Sample 1989) and this is most easily established and maintained on infertile and xeric soils. Soils that meet these criteria were selected from the soils layer for potential nest-cover restoration. Buffer analysis of the impractical nest-cover areas (woody and deepwater wetlands, and the 164-foot buffer around wooded and urban landcover) delineated areas where a nest cover block 20 acres (8.1 ha) or larger could fit. This buffer area was overlaid with a 1,053-foot (321 m) buffer around existing nest cover, to yield a potential restoration area that further aggregates nest cover.

Nest cover—nongame bird large block approach. The large block model for nongame grassland bird nest cover seeks to provide a block of idle grass/forb nest cover 240 acres (97 ha) or larger in each of 12 survey townships to address needs of area-sensitive target species. Another buffer analysis of the impractical nest-cover areas delineated places where a nest cover block 240 acres or larger could fit. Existing nest cover was overlaid onto the potential restoration blocks to identify blocks where restoration will require the least work.

Nest cover—integrated. A simple, unweighted overlay of the nest-cover models produced a fifth (integrated) nest-cover model that was used for the management plan. Restoration areas identified by two or more of the previous nest-cover models were selected as the priority areas for restoration management. Finally, the land ownership and archaeological layers were overlaid on this model to identify appropriate landowners for easement/acquisition contacts.

Conclusion

The GIS approach to integrated restoration management took considerable time (four years) and money ($270,000) to assemble. However, its advantages over a manual approach are its ability to handle large, complex data sets and its flexibility to incorporate (i.e., integrate) new data, species, species relationships or weighting factors into simple or complex formulas that allow for inexpensive and rapid revisions of the management plan. We have added six new data layers, seven new target bird species and new cooperators not envisioned when the project began less than three years ago. A manual approach to these same tasks would have been cost prohibitive. Additionally, the GIS provides a more-defensible management plan because of its biological basis, and objective and consistent treatment of the broad landscape. This already has proved useful in the political reality of a state agency under public review at many levels (state, county, local township and individual landowner). Our quantitative models, along with annual survey data of target species, also allow us to simulate wildlife responses to habitat manipulations and test model assumptions.

The assembly of our GIS was expensive because we were alone and in the lead in this new area (i.e., bleeding on the cutting edge of technology). Offices in the four GHRA counties (ASCS, SCS, tax listing) currently are all working toward automated land records while databases of statewide landcover and natural resources also are under development. We have shared our data with eight agencies/groups and expect to be receiving data from others by the end of our 10-year program, substantially reducing costs for updates and further application of this integrated management tool.

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Removal of Introduced Foxes: A Case Study in Restoration of Native Birds

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Introduction

The widespread introduction of mammals for "sport, fur, or food" (Fish and Wildlife Conservation Act 1980) has reduced the natural biodiversity of island ecosystems around the world (e.g., Laycock 1966, Coblentz 1978, Moors and Atkinson 1984, Scott et al. 1984, King 1985, Coblentz 1990, Bailey and Kaiser 1993). In Alaska, as in other parts of the world, wildlife management frequently has included introductions of game animals for hunting and release of furbearers for trapping (Murie 1940, Burris and McKnight 1973).

One such management program, the leasing of Alaskan islands for fox farming (Ashbrook and Walker 1925), proved to be disastrous for native birds, particularly along the Alaska Peninsula and Aleutian Islands (Murie 1959, Jones and Byrd 1979, Bailey 1993). For the past 30 years, periodic efforts have been made by the U.S. Fish and Wildlife Service to remove introduced arctic (*Alopex lagopus*) and red foxes (*Vulpes vulpes*) from selected islands (Bailey 1993) because most of the area is within the National Wildlife Refuge system and the Aleutians are an International Biosphere Reserve. The purpose of fox removal has been restoration of native bird populations, particularly the threatened Aleutian Canada goose (*Branta canadensis leucopareia*) (Byrd in press).

Fox Introductions and Impacts on Native Birds

During the two centuries between Vitus Bering's discovery of Alaska in 1741 and WWII, foxes were introduced to over 450 Alaskan islands (Bailey 1993). The majority of releases took place between 1900 and 1930 (Bailey 1993). The normal technique was to liberate foxes and return later to trap. In some cases, supplemental feeding, often with marine mammals or birds, was also practiced. Although the impacts of predation by these introduced canids were not quantified carefully, the pattern was clear. Foxes eliminated or drastically reduced most species of surface-nesting birds, including seabirds in earthen burrows (Dall 1874, Murie 1959, Bailey 1993, Litvinenko 1993). Fortunately, foxes eventually died out on most islands in southeastern and southcentral Alaska after bird populations were depleted, but conditions along the Alaska Peninsula, and particularly the Aleutian Islands, allowed foxes to persist (Bailey 1993).

The Aleutian Islands form an 1,800-kilometer long chain which separates the Bering Sea from the North Pacific Ocean. Approximately 33 percent (21 of 64) of the avian taxa nesting on the islands are endemics. In addition, millions of seabirds, representing most North Pacific species, nested on the islands prior to fox introductions (Murie 1959). The magnitude of loss of native birds was not documented, but comparisons of islands where foxes were introduced with islands where foxes never occurred suggest the toll on diversity and particularly biomass was enormous (Byrd and Day 1986, Bailey 1993).

Frequently, it is impossible to mitigate ecosystem modifications by removing introduced exotics because of public opposition, unavailability of appropriate tools, cost, or other factors (Soule 1990). Nevertheless, in the case of introduced foxes, it has been possible to remove these aliens from at least 21 islands in Alaska (Bailey 1993).

In the mid-1970s, we recognized the need to document the recovery of native bird populations after foxes were removed. Nizki and Alaid islands were chosen for that purpose because they contained habitat for most of the species thought to have been most severely affected by foxes, and they were small enough to allow documentation of population changes for a number of species.

The Nizki/Alaid Story

Study Area

Nizki and Alaid islands are centered at about 52 degrees 45' N, 173 degrees 55' E, and, along with Shemya Island, comprise the Semichi group in the western Aleutian Islands. Nizki and Alaid frequently are one island, being joined by a sand bar that washes out periodically, with a combined area of about 1,200 hectares. The highest hill is 190 meters high, but rolling hills under 60 meters are more typical of the islands' topography. The vegetation communities on these treeless, windswept islands are typical of the Aleutians (Byrd 1984). Both islands contain scattered small ponds (most <0.05 ha). The coastlines are irregular with numerous offshore islets and rocks.

In the late 1800s and early 1900s, Nizki and Alaid were considered excellent waterfowl breeding islands (Turner 1885, 1886, Clark 1910). In addition, other native birds—including at least seven endemics—likely were common (Murie 1937). Arctic foxes were introduced to Nizki and Alaid in 1911 (Gray 1939), and nesting birds had been reduced drastically or extirpated by 1937 (Murie 1937). Nevertheless, remnant populations of some species survived on offshore islets or islands in lakes.

Methods

Introduced arctic foxes were eradicated from the islands in 1975 and 1976, with 140 animals killed by shooting, trapping and M-44 cyanide devices. A survey of the islands in 1977 revealed that no foxes remained.

Counts were made of 12 species of birds representing different nesting guilds (cliff nesters, surface nesters, crevice nesters and burrow nesters) before (1975–1976) and after (1984 and 1990) foxes were removed. Additional counts were made of some of the species of birds in 1979, 1983 and 1992.

Each summer, all birds were counted: (1) on every lake, (2) within 100 meters of the coastline during small boat surveys, (3) along all beaches, and (4) within breeding colonies (for colonial-nesting seabirds). Terrestrial routes were mapped so that comparisons could be made among years. All counts were timed to coincide with periods when breeding birds were most conspicuous (e.g., prelaying period for eiders in nearshore waters, incubation period for gulls and puffins).

Results

Following fox removal in the mid-1970s, populations of all surveyed species increased (Figure 1). The Aleutian Canada goose was reintroduced by translocating birds from a distant island, but all other species reoccupied Nizki and Alaid through natural pioneering or still were present at reduced levels.

Of the 12 species monitored, the smallest increases occurred in populations of red-faced cormorant (*Phalacrocorax urile*), which nested mostly on inaccessible ledges, and common loon (*Gavia immer*), which nested on islands in large lakes. Dispersed, inconspicuous nesting species like dabbling ducks (*Anas* spp.) and rock sandpipers (*Calidris ptilocnemis*) demonstrated only two- to three-fold increases.

The most substantial increases occurred in species that were more conspicuous, either because of their size or colonial breeding habits, and which nested in locations accessible to foxes. Aleutian Canada geese, which were nearly extinct as a result of fox predation, were reintroduced to Nizki and Alaid in 1981, and by 1992 at least 34 pairs were nesting. Numbers of common eiders (*Somateria mellissima*) tripled between the mid-1970s and 1984. Although birds were not counted in a comparable way after 1984, the number of nests on Nizki increased from none in 1975 to more than 200 in 1992. Red-throated loons (*Gavia stellata*), pelagic cormorants (*P. palegicus*) and tufted puffins (*Fratercula cirrhata*) all increased four- to five-fold following fox removal, and glaucous-winged gulls (*Larus glaucescens*) and pigeon guillemots (*Cepphus columba*) had even larger proportional increases. Guillemots formerly must have nested primarily among boulders with crevices large enough to admit foxes.

Overall, about 5,000 individuals of 12 monitored species were counted at Nizki and Alaid islands in the mid-1970s. Following the removal of foxes, numbers of these species increased to about 14,000 individuals by 1990. For most of the species, increases likely are continuing. Furthermore, additional species of birds (e.g., storm-petrels [*Oceanodroma* spp.]) may reoccupy Nizki and Alaid in the future.

Management Implications

The response of native birds to the removal of introduced foxes at Nizki and Alaid islands illustrates the benefits of this type of management action to restore native species. Although not so carefully documented, similar increases have been noted on other Alaskan islands following removal of foxes (Bailey 1993). Admittedly, many island ecosystems (e.g., Hawaii) have been much more drastically modified by introductions than have most Alaskan islands, nevertheless, selective removal of exotics even in more complicated situations can have major benefits for native species.

The damage caused by exotics often is far greater than more highly publicized purturbations such as the T/V *Exxon Valdez* oil spill, yet it often is difficult to secure adequate funding and permission to use the most effective tools (e.g., toxins) for removal of introduced predators. In spite of these difficulties, land managers, partic-

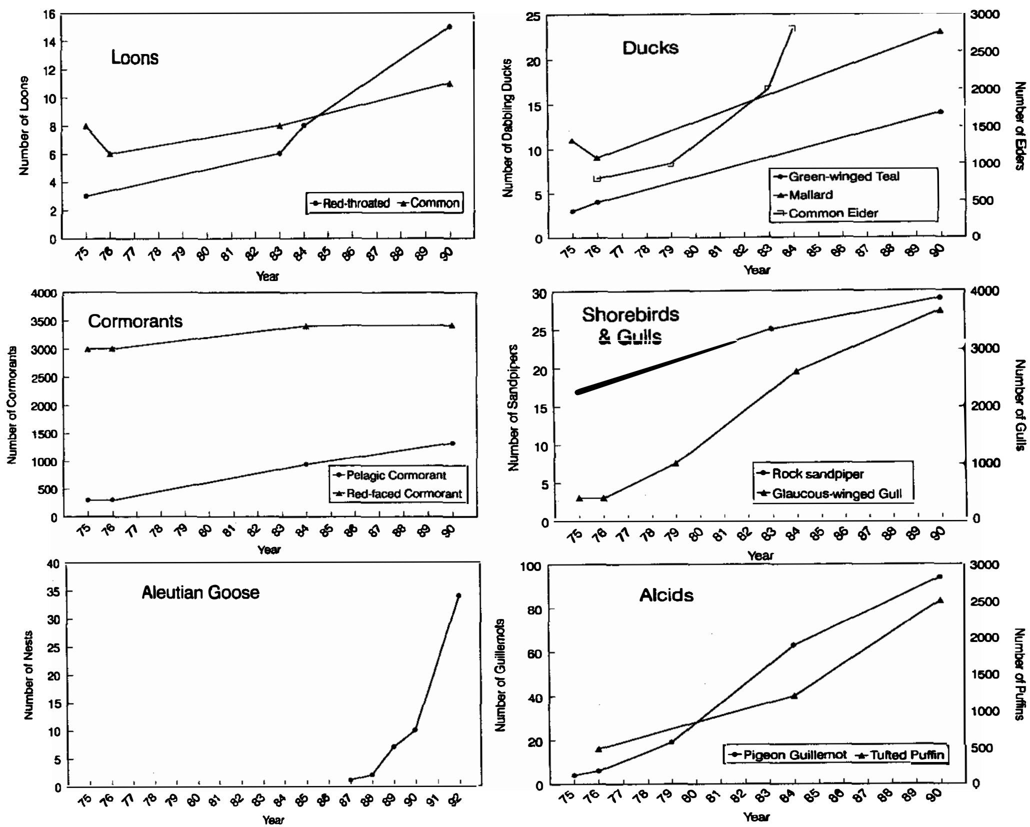

Figure 1. Trends in populations of 12 species of birds following removal of introduced arctic foxes at Nizki and Alaid islands in 1976.

ularly those responsible for island ecosystems, should consider removal of exotics as one of the most productive actions that can be taken to restore native biota.

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Passerine Abundance and Productivity Indices in Grasslands Managed for Waterfowl Nesting Cover

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Introduction

The North American Waterfowl Management Plan (NAWMP) is one of the most comprehensive wildlife management projects ever attempted. The Prairie Habitat Joint Venture (PHJV) plans to spend nearly one billion dollars by the year 2000 (PHJV undated). Fifty-five percent of this budget is allocated to intensive management programs that improve upland-nesting habitat, especially through development of dense nesting cover (DNC). Plans to develop 54,000 acres (21,862 ha) of DNC in Saskatchewan (PHJV undated) are well underway, with approximately 35,000 acres (14,000 ha) developed to date (D. Duncan personal communication: 1993).

Evaluating the impact of the NAWMP on non-waterfowl species is a priority of PHJV partners (NAWMP 1986). Declines in some migratory passerine populations (Robbins et al. 1986, Peterjohn and Sauer 1993) make this group particularly important. From 1966–1991, grassland-nesting birds had a higher proportion of declining species than did any other avian guild in North America (Peterjohn and Sauer 1993). During this period, the central region had significantly higher proportions of declining species than did any other region (Peterjohn and Sauer 1993).

NAWMP partners designed their habitat management programs to conserve or increase populations of waterfowl and nongame species across extensive areas. Habitat quality is a function of population density, survival rates and reproductive success (Van Horne 1983). My objective was to assess the impact of DNC management on passerines by comparing passerine abundance and productivity in DNC, cultivated cropfields and native grasslands. Cultivated cropfields represent the habitat that managers "improve" when planting DNC. Idling native pastures is another PHJV management tool, so this habitat also was compared to DNC and cropfields.

Study Area

My research was conducted on the western edge of the aspen (*Populus tremuloides*) parkland region of eastcentral Saskatchewan (Smith et al. 1964). Eighty-three percent of the uplands in this region are under intensive crop cultivation (Sugden and Beyersbergen 1984). The study area extended from Yorkton to the west side of Big Quill Lake, encompassing approximately 3,250 square miles (8,420 km^2). I selected this area because it has been managed longer and more intensively than any other NAWMP project area in Saskatchewan.

I sampled all DNC sites (n = 16) available in the study area. Most sites consisted of mature grasslands in their third to fifth growing season. Four of 16 DNC sites that I sampled were planted with native grasses, consisting of western wheatgrass (*Agropyron smithii*), northern wheatgrass (*A. dasystachyum*) and green needlegrass (*Stipa viridula*). Six sites were planted with introduced grasses, consisting of intermediate wheatgrass (*A. intermedium*), tall wheatgrass (*A. elongatum*), slender wheatgrass (*A. trachycaulum*), bromegrass (*Bromus* spp.), alfalfa (*Medicago sativa*) and sweetclover (*Melilotus* spp.) Four sites were planted with a mixture of native and introduced grasses, and two sites were idle hayfields of bromegrass and other introduced grasses (Duebbert et al. 1981).

Idle native grasslands were uncommon in this region, so I sampled all of those known to be available. Native grasslands I sampled (n = 18) had never been tilled, and had not been grazed, burned or hayed in at least three years. Only cropfields seeded with wheat (*Triticum aestivum*) were sampled (n = 19), because this is the most common crop cultivated in Saskatchewan. Many wheat fields contained some odd habitat features within the cultivated landscape. These features consisted of isolated shrub clumps, wetland basins or rock piles—usually less than 200 square feet (18.6 m^2) in size. Birds observed around these habitat features were included in surveys, because these habitat features were common within wheat fields.

Wheat fields could not be sampled randomly because many landowners would not grant access to planted cropfields. I selected wheat fields based on their geographic distribution and proximity to DNC and native sites, so that study sites for all three habitats were evenly distributed throughout the study area. Wheat field selection depended mainly on landowner approval of research activities. Though this lack of randomness may prevent strong inferences to all wheat fields in this region, cultivated habitats in this region are quite homogenous due to similarities in landscapes and agricultural practices.

Most study sites comprised one-quarter section of land (160 acres: 64.8 ha), but a few sites in each habitat type were considerably larger or smaller. Average sizes of DNC sites, native grasslands and wheat fields were 106.9, 162.3 and 153.0 acres (43.3, 65.7 and 62.1 ha), respectively. Most sites were separated by at least 1 mile (1.6 km), although adjacent fields were used in three cases.

Methods

Two circular survey plots with 328-feet (100 m) radius (Hutto et al. 1986) were randomly placed at least 164 feet (50 m) from the edge of each study site. Plots usually were separated by 160–330 feet (50–100 m), and each plot was devoid of wetland basins or aspen clumps. For sites that were within large, continuous habitat tracts, distances between the two plots were similar to those of other sites.

Abundance data were collected at all 53 sites by counting birds from the center of both plots for 10 minutes. Each plot was surveyed three times during May 24–July 12 1993, between 0445–0830 CST. Surveys were not conducted on days with winds in excess of 12 miles per hour (20 km/h) or precipitation (Mikol 1980, Robbins 1981). Numbers of breeding birds were estimated by counting territorial males (Mikol 1980). Birds detected outside of marked plots also were recorded separately if observed within the target habitat (Hutto et al. 1986).

Productivity data were collected at eleven sites from each habitat type. For convenience, I chose the 33 sites closest to our field station. Both plots on each site were visited weekly from June 14–August 3 1993. An observer walked slowly through each plot for 30 minutes, recording all observations indicating productivity, to calculate behavioral productivity indices (Dale 1992). Observations were classified as: category one—behaviors indicating the presence of a nest (e.g., distraction displays, alarm calls); category two—fledglings of any species (Hartley 1994).

Statistics

Species abundances were calculated as mean numbers of individuals (pooling all species), per plot, per visit. Species richness for each site is the number of species observed within one or both survey plots. I calculated Shannon diversity indices (Magurran 1988) for each site, using abundances and richness values described above. I calculated category one and two behavioral productivity indices by averaging observations from each plot over the six-week sampling period. Means are reported ±1 standard deviation.

To meet the assumption of equal variances in the ANOVA model, abundances, species richness values, Shannon diversity indices, and category one and two behavioral productivity data were square-root transformed. Excepting category one productivity data, all parameters were compared among habitats by a one-way ANOVA in Proc GLM of SAS (SAS Institute Inc. 1990). Category one productivity data were compared among habitats by a nested ANOVA model in Proc GLM of SAS. I used orthogonal contrasts to compare specific differences between habitat means.

Results

Territorial males of 14, 15 and 6 species were counted within census plots in DNC, native grasslands and wheat fields, respectively (Table 1). Species richness averaged 4.6 ± 1.4, 5.3 ± 1.6, and 1.7 ± 0.9 species per site, in respective habitats. Species richness did not differ significantly between DNC and native grasslands ($p = 0.195$), but wheat fields had significantly fewer species ($p = 0.0001$) than did other habitats. Average numbers of individuals in DNC, native grasslands and wheat fields were 4.4 ± 1.5, 3.8 ± 1.0 and 0.7 ± 0.5 individuals per plot, per visit, respectively. Abundances in DNC and native grasslands were not significantly different ($p = 0.222$), but these habitats had significantly more individuals than did wheat fields ($p = 0.0001$).

Behavioral productivity indices of nests of all species (category one) averaged 0.791 ± 0.507, 0.977 ± 0.413 and 0.099 ± 0.098 observations per plot, per visit in DNC, native grasslands and wheat fields, respectively. Numbers of category one observations in DNC and native grasslands did not differ ($p = 0.281$), but both habitats had significantly more observations than did wheat fields ($p = 0.0001$). Fledglings of all species (category two) averaged 0.158 ± 0.127, 0.144 ± 0.124 and 0.015 ± 0.034 per plot, per visit, in respective habitats. Numbers of fledglings observed in DNC and native grasslands did not differ ($p = 0.793$), but were substantially greater than the number of fledglings in wheat fields ($p = 0.0002$).

Shannon diversity indices for DNC, native grasslands and wheat fields averaged 1.28 ± 0.25, 1.27 ± 0.30, and 0.44 ± 0.42, respectively. Shannon indices for DNC and native grasslands did not differ ($p = 0.91$), but were greater than those for wheat fields ($p = 0.0001$).

Discussion

Fields planted with DNC are used by several passerine species. Compared to idle native grasslands, DNC sites did not have significantly different numbers of species or individuals per site. Renken and Dinsmore (1987), however, reported that idle native grasslands in North Dakota had more species and more individuals than did DNC sites. There is a great deal of overlap in species composition at DNC and native sites, but each habitat seems to be preferred by some species (Table 1). Sedge wrens (*Cistothorus platensis*) were observed only in DNC habitats. Bobolink (*Dolichonyx oryzivorus*) and common yellowthroat (*Geothlypis trichas*) were much more common in DNC. Sprague's pipit (*Anthus spragueii*), a scarce but regular breeder in native grasslands, was absent from other habitats. This species is of concern to managers because of its limited geographic distribution (Dale 1991, Peterjohn and Sauer 1993). Both DNC and native grasslands are used by some declining species. Clay-colored sparrow (*Spizella pallida*), song sparrow (*Melospiza melodia*), bobolink, red-winged blackbird (*Agelaius phoeniceus*) and American goldfinch (*Carduelis tristis*) have suffered significant continental population declines since 1966 (Peterjohn and Sauer 1993).

My results are similar to those of Blankespoor (1980), who reported 13 passerine species using two DNC sites in South Dakota. Higgins et al. (1984) reported eight species breeding at three North Dakota DNC sites. In an evaluation of four DNC sites in North Dakota, Renken and Dinsmore (1987) found 15 species using DNC during a two-year period. They reported bobolink, sedge wren, clay-colored sparrow

Table 1. Passerine surveys from DNC sites (n = 16), native grasslands (n = 18), and wheat fields (n = 19) in eastcentral Saskatchewan, 1993. Mean (±1 s.d.) is number of individuals per plot, per visit. Occurrence (Occur.) is percentage of sites with species observed within a survey plot. Abundance/plot is average number of individuals of all species, per plot, per visit. See American Ornithologists' Union (1983) for scientific names of bird species.

	DNC		Native		Wheat	
Species	Mean	Occur.	Mean	Occur.	Mean	Occur.
LeConte's sparrow	1.34 (1.03)	100.0	0.46 (0.60)	94.4	0	0
Savannah sparrow	1.14 (1.14)	93.7	1.82 (1.11)	100.0	0.21 (0.58)	57.9
Clay-colored sparrow	1.08 (1.21)	93.7	0.65 (0.83)	88.9	0.11 (0.31)	10.5
Sedge wren	0.35 (0.81)	37.5	0	0	0	0
Bobolink	0.23 (0.61)	43.7	0.01 (0.10)	5.6	0	0
Common yellowthroat	0.08 (0.31)	25.0	0.03 (0.17)	11.1	0	0
Western meadowlark	0.03 (0.23)	6.3	0.28 (0.56)	66.7	0.02 (0.15)	5.3
Vesper sparrow	0.02 (0.14)	12.5	0.03 (0.17)	16.7	0.02 (0.15)	5.3
Sharp-tailed sparrow	0.02 (0.14)	12.5	0.02 (0.19)	5.6	0	0
Brown-headed cowbird	0.01 (0.10)	6.3	0.18 (0.60)	44.4	0.02 (0.15)	5.3
Eastern kingbird	0.01 (0.10)	6.3	0.10 (0.39)	22.2	0	0
Red-winged blackbird	0.01 (0.10)	6.3	0.03 (0.17)	16.7	0	0
Song sparrow	0.01 (0.10)	6.3	0.02 (0.14)	11.1	0	0
American goldfinch	0.01 (0.10)	6.3	0	0	0	0
Sprague's pipit	0	0	0.07 (0.26)	33.3	0	0
Western kingbird	0	0	0.05 (0.40)	5.6	0	0
Horned lark	0	0	0.03 (0.17)	5.6	0.39 (0.71)	84.2
Abundance/plot	**4.4 (1.5)**		**3.8 (1.0)**		**0.7 (0.4)**	

and savannah sparrow (*Passerculus sandwichensis*) as the most common species in DNC. In contrast to my results, they reported LeConte's sparrow (*Ammodramus leconteii*) and sharp-tailed sparrow (*A. caudacutus*) in DNC, but not in native grasslands (Renken and Dinsmore 1987).

My survey results from wheat fields are similar to those of Owens and Myres (1973), who observed only horned lark (*Eremophila alpestris*) and vesper sparrow (*Pooecetes gramineus*) breeding in an Alberta cropfield. Although Higgins (1975) reported planted cropfields to be nearly as productive for shorebirds as were untilled uplands, the 19 wheat fields that I sampled had significantly lower productivity than did DNC or native grasslands. Planted in place of cultivated wheat fields, DNC clearly is an improvement for passerine birds, with the exception of horned larks.

Inclusion of individuals observed outside of survey plots (but within the target habitat patch) changes the apparent habitat use of less abundant species. Considering these additional observations, sedge wrens, common yellowthroats and sharp-tailed sparrows were observed at 63, 69 and 31 percent of DNC sites, respectively. Considering these additional data, Sprague's pipits, brown-headed cowbirds (*Molothrus ater*) and common yellowthroats were observed at 61, 78 and 33 percent of native grasslands, respectively. Including observations of birds outside of plots had the most impact on wheat fields. Using these data, clay-colored sparrows were observed at 63 percent of wheat fields, and the overall species list jumps from 6 to 14 species breeding in wheat fields.

Behavioral productivity indices are not meant to estimate productivity, but only to make comparisons between habitat types. If this technique is biased toward certain habitats, however, behavioral productivity indices may overestimate the indications of nests or fledglings in that habitat. Native grasslands in this region have shorter and less dense cover than do DNC sites (Hartley 1994). If birds are easier to observe in native grasslands, productivity indices may be biased (towards underestimation) against DNC. I do not know if such a bias exists. Future research should combine behavioral productivity indices with intensive productivity studies in grassland habitats, to better understand the relationship between nesting success and behavioral productivity indices.

DNC management is of great benefit to many grassland passerines. However, if waterfowl production is similar in native and DNC habitats, managers also should place a high priority on acquiring native grasslands in addition to cultivated lands seeded to DNC. Native grasslands, though grazed, are likely to maintain a component of the original prairie vegetative community, so this habitat is especially important to conservation biologists. Due to cultivation practices, many native grasslands probably are shrubbier than DNC sites. Species richness and density are significantly higher on sites with greater shrub coverage (Arnold and Higgins 1986). Future research should compare DNC and native habitats with respect to other taxa such as invertebrates or mammals.

Acknowledgments

This research was supported by Louisiana State University Agricultural Center, Saskatchewan Wetlands Conservation Corporation, and Delta Waterfowl and Wetlands Research Station. L. Sansone, P. Gronomeyer, B. Piazza and B. Hershey provided valuable field assistance. This project was made possible by the cooperation and assistance of Ducks Unlimited Canada, and many landowners who allowed us to walk through their property and crops regularly. F. Rohwer, D. Duncan and A. Afton provided useful comments on drafts of this manuscript.

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Conservation Reserve Program: Benefit for Grassland Birds in the Northern Plains

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Introduction

During the past few decades numbers of some species of upland-nesting birds in North America have declined. Duck species such as mallard (*Anas platyrhynchos*), northern pintail (*A. acuta*) and blue-winged teal (*A. discors*) have declined since the early 1970s and have remained low since 1985 (Caithamer et al. 1993). Some grassland-dependent nonwaterfowl species also have declined since 1966, as indicated by the North American Breeding Bird Survey (BBS) (Robbins et al. 1986). For prairie-nesting ducks, population declines can be attributed mostly to low recruitment, partially as a result of low nest success. Klett et al. (1988) concluded that nest success (probability of ≥1 egg of clutch hatches) in much of the U.S. Prairie Pothole Region was inadequate to maintain populations of the five most common upland-nesting duck species studied, and that predators were the most important cause of nest failure. Over the years, as grassland areas have been converted to cropland, ducks have concentrated their nesting in the remaining areas of available habitat, where predators such as red fox (*Vulpes vulpes*), striped skunk (*Mephitis mephitis*) and badger (*Taxidea taxus*) forage (Cowardin et al. 1983).

The reasons for declining populations of grassland nonwaterfowl birds are not clear but the loss of suitable grassland-nesting habitat probably is an important factor. Currently, approximately 95 percent of the land in North Dakota is used for agricultural purposes, of which over 60 percent is used for annual crop production (Haugse 1990). Of the grassland that remains, 95 percent is used for livestock production. This probably had a severe impact on grassland bird species that seek idle grass cover for nesting.

The 1985 and 1990 U.S. Farm Bills include provisions under the Food Security Act to fund a cropland-idling program called the Conservation Reserve Program (CRP). Over 36 million acres have been enrolled nationwide in the CRP since 1985 (Osborn 1993), and up to 25 percent of cropland in some counties has been converted primarily to grass. In North Dakota, nearly 3 million acres have been enrolled. Over 90 percent of the CRP plantings in North Dakota are grass and grass-legume mix composed primarily of wheatgrass (*Agropyron* spp.), smooth brome (*Bromus inermis*), alfalfa (*Medicago sativa*) and sweetclover (*Melilotus* spp.). Mixes of these species have been reported to attract high densities of nesting ducks (Duebbert and Kantrud 1974). According to the CRP provisions, the land must remain idle for the 10-year contract period, with the exception of emergency provisions for haying or grazing. CRP appears to have great potential for benefiting many species of grassland-nesting birds.

There have been efforts to document the importance of the CRP to migratory birds in the Upper Great Plains of the U.S. Kantrud (1993) studied duck nest success in CRP cover and concluded that nest success was higher than in planted cover on U.S. Fish and Wildlife Service (FWS) Waterfowl Production Areas (WPAs). Johnson and Schwartz (1993a) measured the use of CRP fields by nonwaterfowl birds and reported that several species have responded positively by colonizing CRP fields. They concluded that CRP has the potential to help reverse the population declines of several species.

We investigated the importance of CRP to upland-nesting ducks and certain other grassland-nesting birds. For ducks, we compared nest success in CRP cover with nest success in planted cover on WPAs in the same period (1992–93) and with that of an earlier period (1980–84). For nonwaterfowl, we used BBS data to compare the trends in populations of certain species found in CRP, for the periods 1966–86 (pre-CRP cover establishment) and 1987–92 (post-CRP cover establishment) in North Dakota.

Study Area and Methods

For duck nest success, our study area included that portion of North and South Dakota east or north of the Missouri River (Figure 1). This area corresponds roughly to the Prairie Pothole Region of North and South Dakota and also was studied by Klett et al. (1988). For nonwaterfowl bird population trends, our study area included all of North Dakota.

Duck Nest Success

In the spring/summer 1992 and 1993, we located duck nests (scrape or bowl containing ≥1 egg) in CRP cover and planted cover on WPAs using methods described by Klett et al. (1986). For our study area, we obtained a sample of 2-mile by 2-mile (3.2 km by 3.2 km) blocks from another study (*see* Cowardin et al. 1988). From this sample, we selected blocks that met two criteria: (1) a minimum of 40 acres (16.2 ha) of CRP cover; and (2) sufficient ponds to attract ≥20 breeding pairs of mallards estimated from a pair-pond regression model similar to that described by Cowardin et al. (1988).

For each block, we selected the nearest WPA (farthest WPA was 9 miles [14.5 km]) that had ≥40 acres (16.2 ha) of planted cover. Planted cover on WPAs consisted of a vegetation mix similar to CRP cover. Each block and associated WPA constituted a study site (Figure 1). For each study site, fields to be searched by treatment (CRP or planted cover) were selected randomly from all fields available until the last field selected resulted in ≥200 acres (80.9 ha) of that treatment for the study site. For study sites with ≤200 acres of a treatment, all fields of that treatment were searched. Each field was searched three times.

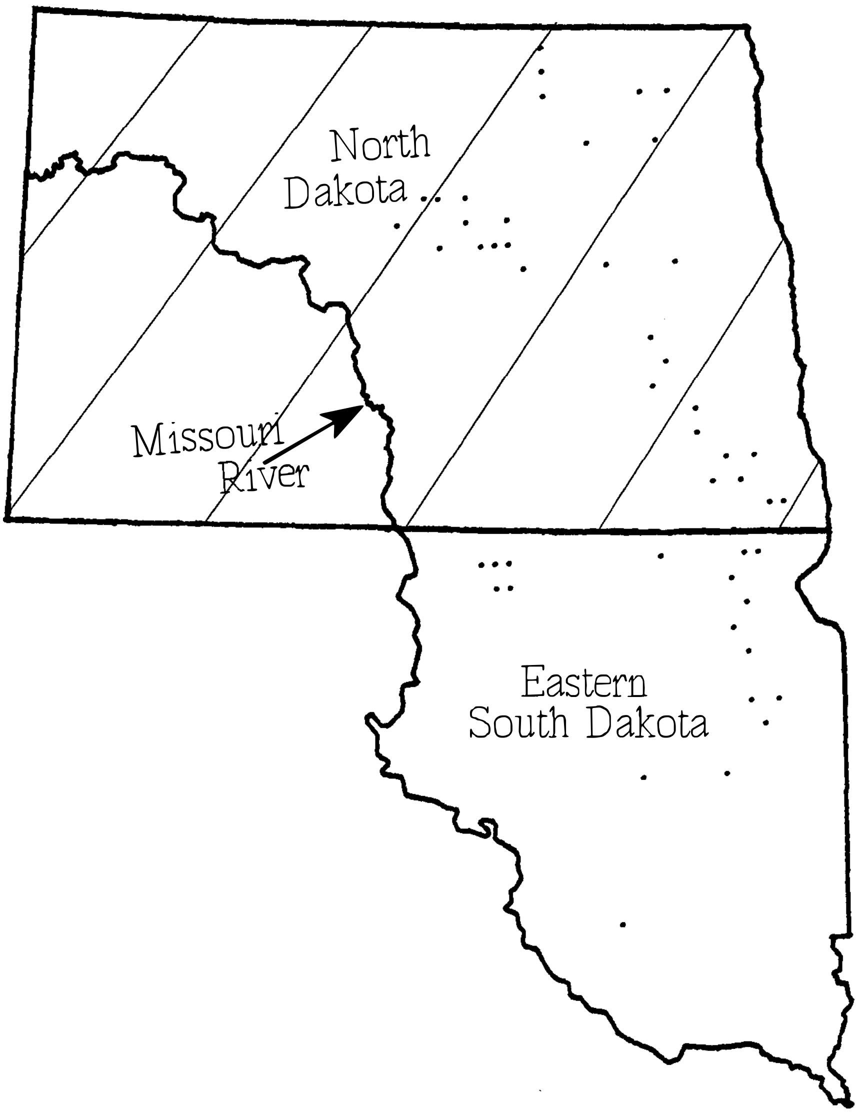

Figure 1. Locations of study sites (dots) in eastern South Dakota and North Dakota used to estimate duck nest success. Study area used for nonwaterfowl bird population trends analyses is indicated by hatching.

We calculated daily nest survival rates (DSR) for each treatment by study site using the modified Mayfield method of Johnson (1979). We modeled DSR as a function of treatment, year and location of each study site. For spatial effects, we considered linear and quadratic terms involving universal transverse mercator (UTM) coordinates of Easting and Northing values and their cross products. Four regression models, corresponding to the four combinations of treatment and year, were developed and statistically compared for each species. The method of weighted least squares, with weights equal to the number of exposure days for each DSR estimate, was used to fit the model.

To compare our results with those of Klett et al. (1988), we required overall nest success estimates for the North Dakota portion of our study area (Klett et al. 1988 did not report estimates for the remainder of our study area). We used our model of DSR to estimate the average DSR for each of the nine Wetland Management Districts (Districts) in North Dakota, using the centroid UTM coordinates for each District as an explanatory variable. Some of the nine Districts (four in 1992 and two in 1993) had UTM coordinates that were farther west than our westernmost study sites (Easting = 482296 and 369094, respectively). To avoid extrapolating beyond the geographic range of our data, we truncated our models at the westernmost study site and used the estimated DSR for that study site as a constant for Districts farther west. Nest success by District was estimated by raising the District DSR to the power equal to the mean laying plus incubation period for successful clutches (Klett et al. 1986). Nest success by year for North Dakota then was estimated by weighting each District's nest success by the proportion of breeding pairs that occurred in that District, as estimated from surveys conducted by the FWS, and averaging. Overall nest success was estimated by averaging the individual year values weighted by population estimates in each year.

Nonwaterfowl Bird Populations

We estimated population trends from the BBS (Peterjohn and Sauer 1993) during the pre- (1966–86) and post-CRP (1987–92) periods for grassland-nesting nonwaterfowl birds regularly observed in CRP fields in North Dakota. We used all species reported in Table 2 of Johnson and Schwartz (1993b). We estimated trends (a percentage change per year estimated as a weighted average of slopes of linear regression on each route [Geissler and Sauer 1990]) for each species in each period.

To evaluate the effects of CRP on these species, we classified them into two groups: species primarily nesting within grassland habitats whose trends may be associated directly with the availability of CRP grasslands (CRP-influenced species), and species nesting in various habitats in addition to grasslands whose trends may not be associated necessarily with the availability of CRP grasslands (CRP-neutral species, Table 1). We then used *t*-tests to determine whether the difference in mean trends ($\text{trend}_{\text{post-CRP}} - \text{trend}_{\text{pre-CRP}}$) for CRP-influenced species was similar to those of CRP-neutral species. Rejection of this null hypothesis in favor of a one-sided alternative hypothesis would indicate that CRP-influenced species were more likely to have positive population changes than CRP-neutral species between the pre- and post-CRP periods.

Results

Duck Nest Success

In 1992 and 1993 we searched 9,567 acres (3,872 ha) in 137 CRP fields and 5,745 acres (2,325 ha) in 95 planted cover fields on 52 study sites (Figure 1). We found 1,197 duck nests in CRP cover and 624 duck nests in planted cover, of which 1,121 and 578, respectively, could be used for estimating DSR. Principal species in CRP cover were blue-winged teal (32.1 percent), gadwall (*A. strepera*) (28.3 percent),

Table 1. Trends in abundance of grassland-nesting nonwaterfowl birds for the periods 1966–86 and 1987–92 in North Dakota with associated significance levels and sample sizes (*n* of routes).

Species	1966-86		1987-92	
	Trend[a]	n	Trend[a]	n
CRP-influenced species				
Bobolink (*Dolichonyx oryzivorus*)	-0.15	37	-1.46	42
Western meadowlark (*Sturnella neglecta*)	-0.94	37	1.98↑	43
Chestnut-collared longspur (*Calcarius ornatus*)	-0.86	29	8.26↑↑↑	32
Savannah sparrow (*Passerculus sandwichensis*)	-3.14	35	-6.03	41
Baird's sparrow (*Ammodramus bairdii*)	-1.19	25	-15.29↓↓↓	24
Grasshopper sparrow (*Ammodramus savannarum*)	-7.86↓↓↓	37	10.06↑↑	42
Dickcissel (*Spiza americana*)	-6.51↓↓↓	25	-9.86	24
Lark bunting (*Callamospiza melanocorys*)	-7.70↓↓↓	29	21.73↑↑↑	35
CPR-Neutral species				
Killdeer (*Charadrius vociferus*)	1.92↑	37	-13.02↓↓↓	43
Mourning dove (*Zenaida macroura*)	4.17↑↑↑	37	6.51↑↑↑	43
Eastern kingbird (*Tyrannus tyrannus*)	5.13↑↑↑	37	7.79↑↑↑	43
Western kingbird (*Tyrannus verticalis*)	5.85↑↑↑	36	12.61↑↑↑	43
Horned lark (*Eremophila alpestris*)	1.09	37	0.74	43
Brown-headed cowbird (*Molothrus ater*)	4.15↑↑↑	37	-2.98	43
Red-winged blackbird (*Agelaius phoeniceus*)	-1.38↓	37	-10.89↓↓↓	43
Vesper sparrow (*Pooecetes gramineus*)	2.59↑↑	37	3.22	42
Clay-colored sparrow (*Spizella pallida*)	-3.88↓↓↓	36	-1.46	42
Barn swallow (*Hirundo rustica*)	5.09↑↑↑	37	-4.68↓↓	43
Common yellowthroat (*Geothlypis trichas*)	-0.36	37	-2.86	43
Sedge wren (*Cistothorus platensis*)	-1.15	22	-10.07	23

[a]Percentage change per year, ↑(↓) P<0.10; ↑↑(↓↓) P<0.05; ↓↓↓(↑↑↑) P<0.01.

mallard (21.6 percent), northern shoveler (*A. clypeata*) (8.6 percent) and northern pintail (8.4 percent). The species composition in planted cover was blue-winged teal (51.4 percent), gadwall (22.9 percent), mallard (12.2 percent), northern shoveler (10.3 percent) and northern pintail (3.0 percent). Sufficient information to develop models of DSR was obtained for mallard, gadwall and blue-winged teal.

Mallard. The four regression lines for DSR of mallard did not differ ($F_{3,66} = 0.09$, $P = 0.966$), indicating no treatment or year effects. All data were combined into a single model that indicated DSR increased from east to west ($F_{1,66} = 8.34$, $P = 0.005$).

Gadwall. Daily survival rates of gadwall nests increased from east to west ($F_{1,82} = 4.49$, $P = 0.037$) and the rate of increase did not depend on treatment or year ($F_{3,79} = 0.50$, $P = 0.681$). The regression line for CRP in 1992, however, was lower ($P<0.05$) than the other three lines. The other three did not differ from each other ($P>0.05$).

Blue-winged teal. Daily survival rates of blue-winged teal nests varied from east to west, but not linearly or consistently from year to year ($P<0.05$). The shapes of the curves for CRP and for planted cover were the same within each year, but differed

Table 2. Comparison of estimated nest success in Conservation Reserve Program cover and planted cover in 1992–93 with estimates for planted cover in 1980–84[a] for mallard, gadwall and blue-winged teal.

	Nest success (percentage)					
	Mallard		Gadwall		Blue-winged teal	
Habitat	1980–84	1992–93	1980–84	1992–93	1980–84	1992–93
CPR	[b]	24	[b]	22	[b]	25
Planted cover	9	24	12	30	12	18

[a]From Klett et al. 1988.
[b]Habitat not available in 1980–84.

between years. Tests indicated that DSR was higher in CRP than in planted cover in 1992 but the two did not differ in 1993.

Our overall estimates of nest success for CRP and planted cover in North Dakota were higher than estimates of nest success in planted cover reported by Klett et al. (1988) for 1980–84 (Table 2).

Nonwaterfowl Bird Populations

Prior to the CRP, all eight of the CRP-influenced species had negative estimates of trends, but post-CRP, four of the species had positive trend estimates (Table 1). In contrast, four of the twelve CRP-neutral species had negative point estimates of trends pre-CRP, but seven had negative trends post-CRP. Evaluation of differences in trend ($trend_{post\text{-}CRP} - trend_{pre\text{-}CRP}$) indicates that the CRP-influenced species were more likely to be increasing during the later period (mean differences: 4.72 [CRP-influenced], –3.19 [CRP-neutral], $t = 1.73$, df = 18, $P = 0.052$).

Discussion and Conclusions

The results of our investigations suggest that CRP cover is providing benefits for some grassland-nesting birds. For ducks, we found nest success in CRP cover and planted cover in 1992–93 to be 6–18 percent higher than that reported for planted cover in 1980–84 by Klett et al. (1988) (Table 2). Nest success in CRP for three principal species was 2–9 percent higher than that believed necessary to maintain populations (i.e., 15 percent for mallard, 20 percent for gadwall and blue-winged teal) (Cowardin et al. 1985, Klett et al. 1988). Our estimates of nest success in CRP cover were comparable to that reported for CRP cover by Kantrud (1993) for North Dakota and Minnesota combined in 1989–91. For seven combined species of ducks, he reported higher nest success in CRP (23 percent) than in planted cover (8 percent). His study was conducted during drought years, and CRP fields were farther from ponds than were planted cover fields. Kantrud speculated this may have been a cause of the difference in nest success because predator activity probably is greater near wetlands. We did not find evidence of a difference in nest success between CRP and planted cover in 1992–93. We purposefully selected study sites that were not affected strongly by drought. Although we did not measure distances from our fields to the nearest pond, it was common to have ponds within and adjacent to both our CRP and planted cover fields. If what Kantrud (1993) speculated is correct, it may partially explain why our nest success estimates were similar in CRP and planted cover.

We can only speculate as to why nest success in planted cover was higher in our study than that reported by Klett et al. (1988) for the 1980–84 period. It is possible that the increased amount of grass cover provided by the CRP had a positive effect on nest success in planted cover and other cover types by dispersing nests or providing a larger prey base (Lysne 1991). If this is true, then benefits of CRP to grassland-nesting ducks extend beyond the CRP cover itself.

Another explanation is that the expanding coyote (*Canis latrans*) populations and declining red fox populations in our study area during the pre- and post-CRP periods (Sovada 1993, Sargeant et al. 1987) may have increased nest success in all cover types. Sovada (1993) found that nest success of upland-nesting ducks was about 15 percent higher in her study areas where coyotes were active and there was little or no activity by red fox compared with study areas where the reverse was true. On study areas where foxes were active, however, average nest success (17 percent) was higher than that reported by Klett et al. (1988) for planted cover in 1980–84. Sovada's study was conducted during 1990–92 after most CRP cover had been established. She speculated that CRP may have elevated nest success in all cover types. Thus, increased CRP cover and changing canid populations may be working in concert to yield increased nest success for ducks. This high nest success combined with the attractiveness and availability of CRP cover suggests that this Program has great potential for increasing duck production in the Prairie Pothole Region of the U.S.

Breeding Bird Survey data indicate that some grassland-nesting species show population increases coincident with the post-CRP period, reversing the negative trends of the pre-CRP period. We believe that CRP provides substantial benefits for certain species, especially those restricted to grassland habitats during the breeding season. For example, the population status of grasshopper sparrow (*Ammodramus savannarum*) and lark bunting (*Calamospiza melanocorys*) improved markedly during the post-CRP period. These two species were reported by Johnson and Schwartz (1993b) to be the most abundant breeding birds in CRP fields in four states including North Dakota, and their increasing populations reflect the increased availability of nesting habitats offered by the CRP in this region.

Not all grassland-obligate species increased during the post-CRP period, indicating that factors other than breeding habitat availability may be strongly influencing their population trends. For example, bobolink (*Dolichonyx oryzivorus*) and dickcissel (*Spiza americana*) are neotropical migrants, and their declines during the post-CRP period may be the result of conditions encountered during their migrations or on their wintering grounds in South America. Additionally, species with specialized habitat requirements, such as Baird's sparrow (*Ammodramus bairdii*), may be able to occupy only a small proportion of the CRP habitats that have been created. However, their presence in CRP cover, relative to the alternative (cropland) (Johnson and Schwartz 1993b), indicates that these species still benefit from the CRP.

Three considerations should be made when interpreting these results. First, any grouping of birds is subject to criticism because of the unique life-history characteristics associated with each species (e.g., Mannan et al. 1984). We believe these results are robust to minor changes in the groupings, and encourage interested readers to evaluate the patterns demonstrated in Table 1 for alternative groupings. Second, regional patterns of population changes are influenced by many factors, including drought. Between 1987 and 1992, the number of ponds declined in most of North Dakota as a result of drought (Hunnicutt and Reynolds 1993). We believe this extended drought could negatively affect some species irrespective of other changes such as increased habitat availability provided by the CRP. Finally, our analyses considered only North Dakota, and the species investigated also occur elsewhere. Therefore, it is possible that the population changes observed were at least partly a result of redistribution of birds from other areas. In conclusion, we feel that CRP is benefiting both upland-nesting ducks and some species of grassland-nesting nonwaterfowl birds.

Most of the vegetative cover associated with the CRP became progressively available from plantings during the period 1987 to 1992. Unfortunately, this same period coincided generally with drought conditions across much of the northern plains. Low soil moisture resulted in less than optimal conditions for vegetative growth and runoff was insufficient to fill most prairie wetlands. These conditions prevented upland-nesting ducks and likely some nonwaterfowl birds from taking full advantage of the increased availability of grass cover provided by the CRP. Additionally, emergency haying has been allowed on a significant portion of the CRP acres in recent years which has reduced vegetation used by birds arriving in early spring. Since summer 1993, precipitation has increased and the outlook is favorable for improved wetland conditions and increased vegetation growth in 1994. Our evaluation of CRP cover and nesting ducks is scheduled to continue for two more years. We look forward to providing additional insight into the importance of CRP to grassland-nesting birds.

Acknowledgments

We thank the many individuals who have participated in the BBS for North Dakota. We also thank members of the nest-searching crews who participated in the duck nest success study. We especially thank R. Renner of Ducks Unlimited, Inc. and R. Greenwood of the National Biological Survey, for their assistance in coordinating the duck nest success study. We acknowledge the financial support provided by the Central and Mississippi Flyway Councils, Ducks Unlimited's Institute for Wetland and Waterfowl Research, and the U.S. Bureau of Reclamation. B. Batt and R. Meeks of Ducks Unlimited, Inc., provided administrative assistance for the duck nest success study. We appreciate the logistical assistance provided by H. Hoistad, R. Hollevoet, F. Giese, R. Gilbert, J. Koerner, S. Kresl, D. Potter, D. Walls, P. VanNingen and their staffs. S. Vaa and M. Johnson provided field assistance. A. Schaff typed the manuscript. We thank W. Newton for statistical advice and D. Johnson, R. Koford, G. Pendleton and A. Sargeant for reviewing the manuscript.

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Conservation of Nongame Birds and Waterfowl: Conflict or Complement?

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Introduction

Nongame and gamebirds share habitat over much of North America. This commonality could promote either cooperation or conflict among diverse conservation interests. Waterfowl have been emphasized in North American conservation programs and management-oriented research and monitoring efforts for more than 50 years, versus about 25 years for most common nongame species. Declines in breeding bird populations have been documented over the past several decades in some forest songbirds in the eastern and central United States (Robbins et al. 1989b, Terborgh 1989, Böhning-Gaese et al. 1993), in grassland nongame species (Knopf 1994, Johnson and Schwartz 1993) and in several species of waterfowl (Johnson and Shaffer 1987, Bethke and Nudds 1994). Here we explore the view that remarkable similarities occur in the probable causes of these declines.

Our perception is that many researchers or managers who specialize in waterfowl or in nongame birds often are unaware of ecological information (and resulting conservation/management implications) obtained by the complementary group. Consequently, ecological patterns and processes common to waterfowl and nongame birds may be overlooked, thus retarding progress of both science and conservation. Moreover, failing to fully integrate ecological knowledge and conservation activities for waterfowl and nongame birds could result in management conflicts, promote the perception of conflicts where none exist, obscure the potential for cooperative conservation activities, and unnecessarily limit the breadth of public support for conservation programs to sustain bird populations in particular and biological diversity in general. Here we review the general patterns and probable causes of population declines in nongame birds and waterfowl, particularly issues specific to habitats and events on the breeding grounds.

Declining Bird Populations

Songbirds in Eastern Hardwood Forests

Patterns of persistent population decline in songbirds first were detected in long-term intensive surveys of breeding birds conducted on relatively small tracts (<350 ha) of eastern hardwood forest (reviewed by Terborgh 1989, Askins et al. 1990). Although the forest tracts themselves seemed to have changed relatively little between the 1940s and 1980s, populations of certain species had declined markedly or disappeared entirely. Patterns of population decline were most consistent in migratory species associated with forest interior conditions. Declining populations in the absence of apparent local change in breeding habitat, and the migratory status of many of the declining species, logically led to early suggestions that habitat changes on migration or wintering areas were involved (Briggs and Criswell 1979, Ambuel and Temple 1982). Considerable evidence has accumulated over the past decade suggesting that deforestation in the Neotropics has had, and will continue to have, negative consequences for some species of migratory, forest-breeding songbirds (Terborgh 1989, Robbins et al. 1989b). Nevertheless, it also has become increasingly clear that changes on the breeding grounds have created conditions that are sufficient to account for population declines in many cases. We emphasize these changes here because they provide the most striking parallels between forest songbirds, grassland songbirds and waterfowl.

If population declines in migratory birds primarily were associated with deteriorating conditions on the breeding grounds, then breeding populations in large, intact forest tracts should show little change. Surveys in virgin forest stands of the Great Smoky Mountains National Park (209,000 ha) conducted in 1947–1948 (Kendeigh and Fawver 1981) and 1982–1983 (Wilcove 1988) showed remarkably similar populations of migrants between the two sampling periods. Other long-term studies in relatively large tracts of contiguous forest in the eastern U.S. (reviewed by Askins et al. 1990) generally showed migrant populations to be stable or increasing; the declines noted were moderate and compatible with explanations based on local successional change.

Numerous studies have documented the presence or absence of breeding bird species in woodlot or forest tracts of various sizes (Forman et al. 1976, Galli et al. 1976, Whitcomb et al. 1977, Robbins 1980, Martin 1981, Lynch and Whigham 1984, Blake and Karr 1987, Robbins et al. 1989a). Area sensitivity (the tendency for a species to be absent from relatively small tracts) is more pronounced for long-distance (neotropical) migrants than for short-distance migrants or resident species (Blake and Karr 1987). Males of some species are relatively unsuccessful at attracting mates in small forest tracts (Gibbs and Faaborg 1990). Area-sensitive species may decline or be absent from small forest tracts because reproductive success there is inadequate to replace mortality. Neotropical migrants might be relatively susceptible to any decrease in productivity because they tend to have relatively small clutches and low propensity to renest. Nest success of open-nesting migrant passerine species commonly is 30–40 percent, with predation accounting for 70–80 percent of all losses, and parasitism by brown-headed cowbirds (*Molothrus ater*) for about 10 percent (Martin 1992). Hence, any factors resulting in increased rates of nest predation or parasitism have high potential for eliminating species from small tracts or causing population declines.

High diversity of bird species and high population density of some bird species commonly occur along habitat discontinuities or edges. Gates and Gysel (1978) studied open-nesting passerines along a forest/field ecotone, and found over half of the nests within 15 meters of the edge. However, nest predation rates (61 percent of total nesting mortality) were 3–5 times higher within 50 meters of the edge than beyond. Nest parasitism by brown-headed cowbirds (11 percent of total nesting mortality) also appeared to be higher within 5 meters of the edge than beyond. Overall, fledgling success increased steadily with increasing distance from edge. The relationship was attributed primarily to density-dependent predation and to the tendency for predators to move along edges. Thus, the increasing amounts of edge associated with fragmentation of forest tracts can have major ramifications for the vulnerability of breeding birds to nest predators and parasites (Whitcomb et al. 1981, Yahner 1988). Wilcove (1985) used artificial nests to determine whether nest predation rates differed between suburban forest fragments (<14 ha), rural forest fragments, large forest tracts (<1,000 ha) and the 209,000-hectare Great Smoky Mountains National Park. After seven days of exposure, 71 percent of the nests in suburban fragments had been destroyed by predation versus 48 percent in rural fragments, 20–50 percent in large forest tracts and 2 percent in the Great Smokies. Predation rates on artificial nests may differ substantially from those on real nests (Martin 1987, Storaas 1988, Willebrand and Marcström 1988); nevertheless, the results strongly suggest that birds nesting in small forest fragments face predation pressures many times higher than existed historically over much of the eastern United States. Furthermore, a relatively high predation rate (roughly 25 times baseline) occurred in a 283-hectare tract, demonstrating that the problem is not confined to the smallest fragments. Wilcove suggested that avian and mammalian nest predators existed at unnaturally high levels in small forest tracts because of a variety of human influences, including absence of large predators, increased edge and other landscape-level changes. Terborgh (1989) stressed the probable benefits to nest predators and parasites of supplemental winter food provided by agriculture and, ironically, recreational bird feeding.

Earlier conclusions that population declines in eastern neotropical migrant birds are associated with tropical deforestation have been questioned because geographic scope of data from the breeding grounds was limited and because the species considered varied in many traits besides migratory status (Martin 1992, Böhning-Gaese et al. 1993). Using the same data source used by Robbins et al. (1989b) to show a median decline of 0.97 percent per year among neotropical migrant species from 1978–1987, Böhning-Gaese et al. (1993) broadened geographic coverage on the breeding grounds and restricted their analysis to an ecologically homogeneous group of insectivorous passerines. They concluded that migratory status was associated with a nonsignificant downward trend (–0.08 percent per year) during 1978–87, but that species most vulnerable to predation declined significantly (–0.8 percent per year). Nest parasitism by cowbirds appeared to play a detectable but secondary role.

Pulliam (1988) provided a theoretical framework for addressing the prospect that habitats may function as "sinks" where reproduction fails to replace losses to mortality. Conversely, conditions in "source" habitats are such that annual production exceeds mortality, and surplus individuals are available through immigration to support populations in sink habitats. Population sink conditions have been identified for several passerine species in eastern hardwood forests (Temple and Carey 1988, Robinson 1992). Most such evidence derives from relatively small (<100 ha) forest fragments, but tracts up to 2,000 hectares in the Shawnee National Forest of southern Illinois exhibit discouragingly high rates of nest predation and parasitism (Robinson 1993). Sherry and Holmes' (1992) study of American Redstarts (*Setophaga ruticella*) in the largely unfragmented Hubbard Brook Experimental Forest in New Hampshire demonstrated that even under presumably natural conditions, sites may shift from source to sink status and back over time. In this case, the shift apparently was associated with changes in weather and in populations of nest predators; neither habitat fragmentation nor significant amounts of nest parasitism were involved.

In summary, considerable evidence suggests that anthropogenic changes in eastern hardwood forest landscapes and communities of nest predators and parasites promote high rates of nest predation and parasitism on some songbirds. These changes appear to be correlated with the loss of some bird species from small forest tracts and general population declines in the most vulnerable species. The situation is severe enough to create population sinks, often in small forest fragments but sometimes in reasonably large forest tracts as well.

Grassland Birds

Although they have received considerably less attention than the forest-breeding neotropical migrants, grassland birds show much clearer and stronger evidence of general population decline. Only 17 percent of grassland-nesting species exhibited increasing population trends between 1966 and 1991 compared to 59 percent of neotropical migrants, and all grassland sparrows are in decline continentally, 5 of the 10 at >3.0 percent annually (Knopf 1994). Some of these declines presumably are related to agricultural conversion of grasslands; population densities in 16 of 20 species were seven times higher on grassland fields in northcentral United States than on croplands (Johnson and Schwartz 1993).

As in eastern hardwood forest systems, small patch size and proximity to edges are associated with reduced productivity in grassland nesting songbirds. Johnson and Temple (1986, 1990) studied nesting ecology of five ground-nesting passerine species in tallgrass prairie tracts (16–486 ha) of western Minnesota. Productivity was highest for nests in recently burned, large (>130 ha) prairie tracts in sites far (>45 m) from a wooded edge. Reductions in productivity were attributable primarily to predation (>57 percent of 322 nests). Species of nest predators were not identified, but anthropogenic changes in populations of raccoons (*Procyon lotor*) and red fox (*Vulpes vulpes*) were mentioned. Approximately 18 percent of all nests were parasitized by cowbirds, reducing productivity by about 1.5 fledglings in each parasitized nest that escaped predation ($n \approx 23$). Hence, parasitism reduced productivity by approximately 35 potential recruits, while nest predation accounted for approximately 550. Although some of the prairie tracts studied were among the largest remaining in Minnesota, and were intensively managed by burning to maintain high-quality prairie vegetation, population sink conditions prevailed even in the most productive sites. Vickery et al. (1992) noted that nest predation rates on threatened grassland birds in Maine were high (58 percent–unadjusted for exposure), and attributed most losses to incidental nesting predation by striped skunks (*Mephitis mephitis*). Determining the species involved in predation on songbird nests is exceedingly difficult because virtually no evidence is left at the site in most cases. Although nest predation and nest parasitism commonly receive equal emphasis, the relative demographic importance of parasitism is easily overestimated (Martin 1992).

In summary, many species of grassland birds exhibit clear trends of serious, widespread population decline. High rates of nest predation, and secondarily of nest parasitism, create population sink conditions that are most severe in small fragments but extend to some of the largest remaining prairie tracts.

Dabbling Ducks

Annual surveys of ducks across most of their breeding range began in 1955, but periodic droughts that greatly reduce wetland abundance and alter the distribution of waterfowl complicate efforts to document long-term population trends. Breeding population indices of the 10 most common duck species declined 33 percent from 1955–59 to 1989–93.

Johnson and Schaffer (1987) noted that populations of mallards (*Anas platyrhynchos*) in surveyed portions of North America peaked in the late 1950s, declined through 1965, increased to a slightly lower peak in 1970 and declined through 1985. Correlation between pond numbers and numbers declined from $r = 0.47$ in 1955–1970 to $r = 0.27$ in 1971–1985, indicating that mallards no longer filled the wetland component of their breeding habitat to the extent they had previously. The decrease in mallard/pond correlation showed some association with increases in cropland (i.e., decreases in nesting habitat), although variation was substantial. Bethke and Nudds (1994) followed a similar approach, using duck/wetland correlations established during 1955–1974 to demonstrate deficits in duck abundance occurring during 1975–1980. These deficits, expressed as a percentage of the expected long-term mean abundances, were 45 percent for northern pintails (*Anas acuta*), 29 percent for mallards and ≤18 percent for other species. Substantial proportions of the deficits (80 percent in pintails and 65 percent in mallards) were associated with agricultural conversion of upland habitats.

Compelling evidence has surfaced over the past several decades that remarkably low nest success among dabbling ducks is common across much of the breeding range. Klett et al. (1988) used nest records for more than 15,000 dabbling duck nests found in the United States portion of the prairie pothole region to calculate nest success for 50 combinations of species, years and subregions; in 40 (80 percent) of these, estimated nest success was below levels necessary to sustain populations. In mallards, for example, nest success was 5 percent in western Minnesota and eastern North Dakota, 8–11 percent in central North Dakota, 9–10 percent in eastern South Dakota, and 19 percent in central South Dakota. Predation, primarily by mammals, accounted for 82 percent of nest failures for mallards and 77 percent for all species. Greenwood et al. (1987) studied nest success of mallards at 17 sites broadly distributed across the prairie/parkland region of southern Manitoba, Alberta and Saskatchewan. Nest success averaged 12 percent and was below population maintenance levels in 24 of 31 (77 percent) area-year estimates. Predation was involved in 88 percent of nest failures.

The relationship between nest success and dabbling ducks and patch size of nesting habitat is unclear (Clark and Nudds 1991). Higgins et al. (1992) showed that dabbling duck nest densities were highest in small (<16 ha) fields, but that nest success did not vary predictably among field sizes. In this study, and in fact most studies conducted in the prairie pothole region, relatively little is known about nest success in reasonably large fields (say, >260 ha). This is largely because few fields this large *exist* in the prairie pothole region, and many of those that do remain are degraded by intensive, season-long livestock grazing. Furthermore, prairie grasslands are highly heterogenous both in vegetation (does a shrub-dominated segment of a grass field represent a separate patch?) and land use (does haying, grazing or burning part of a grassland tract reduce patch size?).

In spite of these limitations, the relationship between fragmentation of prairie and parkland habitats by tillage agriculture and nest success is reasonably clear. Greenwood et al. (1987) found that mallard nest success was strongly correlated ($r = 0.67–0.80$) with percentage of grassland habitats remaining on 26-square kilometer study areas. Furthermore, nests tended to be concentrated in large (130–580 ha) native grass pastures; most of the viable nest success rates recorded during the study occurred on study areas containing these large pastures. Klett et al. (1988) did not explicitly address the relationship between grassland habitat remaining in regions and nest success, but their data clearly reinforce this pattern. Finally, in northcentral Montana where roughly 80 percent of grasslands remain (although they are heavily grazed by livestock), dabbling duck nest success commonly is ≥50 percent (Holm 1984, Ball et al. 1988).

Long-term changes in predator populations in the prairie pothole region, the causes of those changes and their ramifications for breeding ducks were addressed by Cowardin et al. (1983), Johnson and Sargeant (1977), Johnson et al. (1989), Sargeant (1972, 1983), Sargeant and Allen (1989) and Sargeant et al. (1987, 1993). Gray wolves (*Canis lupus*) were the dominant carnivore over much of the northern prairies and plains prior to the late 1800s, and they were heavily dependent on large, highly mobile herds of bison (*Bison bison*). Many of today's most important species of nest predators, such as red fox, raccoon and American crow (*Corvus brachyrhynchos*), were absent, rare or narrowly distributed. Anthropogenic changes that drastically altered the predator community structure and population densities included: elimination of bison; extirpation of top carnivores (particularly canids); provision of den and nest sites (abandoned buildings, foundations, culverts, shelterbelts, tree encroachment through fire suppression, etc.); and supplemental winter food supplies (agricultural crop residues, roadkills, domestic carrion, etc.). Large predators (wolves) that were well adapted to a large, mobile prey resource and smaller predators that were specialized to exploit locally abundant prey resources were replaced by medium to small generalist predators all strongly subsidized by humans and existing at relatively high densities. Thus, wolves were replaced by coyotes (*Canis latrans*), and eventually by red fox. Wolves generally appear to control coyote abundance, and a similar interaction between coyotes and red fox is particularly well documented. Historic information provides numerous examples of inverse correlations in coyote and fox distribution and abundance. This pattern also is evident in current distribution patterns across the prairie pothole region (proportion of coyotes increasing and foxes decreasing, from southeast to northwest). Where both species occur, foxes often persist along the edges of coyote territories, commonly near roads or other human developments. Exclusion of foxes by coyotes seems to occur mainly through avoidance, although numerous instances of coyotes killing foxes have been recorded. Coyote and raccoon populations also exhibit significant negative correlations. Although both coyote and fox densities vary substantially, densities of foxes often are 3–4 times higher than those of coyotes. Furthermore, foxes commonly cache ducks and duck eggs. In combination with their much higher overall density, caching limits the possibility of predator "swamping" by ducks, and probably minimizes buffering by other prey items of fox predation on ducks and their nests. Predation rates on duck nests in the prairie pothole region are strongly correlated with abundance of red fox both for early and late-season nests. Finally, in parts of North and South Dakota, duck nest success averaged 32 percent on 17 study areas where coyotes were common, and 17 percent on 13 areas where foxes were the principal canid (Sovada 1993). Habitat conditions were similar among study areas; hence study areas differed between source and sink status based largely on differences in predator community.

Among dabbling ducks, and particularly among early nesting species such as mallards and pintails, most recent studies indicate that vast areas of the prairie pothole region operate as sinks in most years. The few exceptions to this pattern include exceptionally large grassland tracts (Holm 1984, Greenwood et al. 1987, Ball et al. 1988), areas dominated by coyotes (Sovada 1993), islands (Duebbert 1966, Duebbert et al. 1983), overwater nest sites (Arnold et al. 1993) and sites where mammalian nest predators are reduced or excluded (Duebbert and Lokemoen 1980, Greenwood et al. 1990). One particularly troubling aspect of this situation is that sink status appears to be influenced most strongly by large-scale patterns, while settling patterns in many duck species are strongly influenced by small-scale patterns, particularly the availability of water. Consequently, efforts to restore drained wetlands in regions where existing upland habitats and predator communities cause low nest success may merely increase the attractiveness of a sink and have a demographic effect that is opposite to what is desired and expected.

Declining populations of several dabbling duck species, like those of some songbirds in eastern hardwood forest and in prairie fragments, appear to be associated with anthropogenic changes in both habitat integrity and predator communities.

Are Nest Success and Population Status Linked?

Population change in birds depends on the balance between fecundity (probability of breeding x number of nesting attempts x clutch size) and survival (eggs, nestlings, fledglings, juveniles, adults). Changes to any one component may be demographically irrelevant if accompanied by compensatory changes in another (say, if nesting/fledgling success increased but post-fledgling survival decreased). Understanding the demographic consequences of nest predation and parasitism requires demonstration of a functional link between fledgling success in one year and recruitment to the breeding population the next. Among passerines, such links have been shown on a local scale for prairie warblers (*Dendroica discolor*) (Nolan 1978) and American redstarts (Sherry and Holmes 1992). In American redstarts, nesting success in one year accounted for 57 percent of variation in yearling recruitment to the population the next. Similar evidence exists for the mallard on a continental scale (Reynolds and Sauer 1991) where production of young in one year accounted for 25 percent of the variation in populations the next year.

Experimental reduction in populations of nest predators (Balser et al. 1968, Duebbert and Kantrud 1974, Duebbert and Lokemoen 1980, Greenwood 1986) provides strong evidence that nest predation is the primary source of nesting mortality in dabbling ducks and that decreases in nest mortality from predation are not offset by increases in some other form of mortality during nesting (Martin 1991). Furthermore, dabbling duck populations experiencing relatively high nest success commonly reach remarkably high densities.

Common Ecological Patterns and Common Conservation Issues

The ecological parallels stressed in this paper suggest that both research efforts and conservation practices for nongame birds and ducks should be integrated. Although we have argued that similar processes have important impacts on reproductive success in both groups, we are unaware of a single publication where productivity of both was assessed on the same area. We suggest that standardized monitoring of avian productivity on northern grasslands should include both nongame birds and ducks. A statistically valid sampling approach based on 10.4-square kilometer plots for which habitat conditions are known (Cowardin et al. 1988) currently is operational and could form the foundation for such efforts. Similarly, existing Waterfowl Production Areas on northern prairies might form core areas for efforts to improve habitat for grassland birds in general, whether through additional land acquisition or working with private landowners to improve conditions on surrounding grasslands.

The population problems addressed here arise from pervasive, large-scale modification of landscapes and predator communities. Hence, the solutions that will be most effective and will influence the broadest segment of species will be those that can be implemented on a large scale. Effecting large-scale change will be difficult in all situations, but particularly so in grassland systems, where few large tracts of public land exist (at least in central and eastern regions) and strong financial pressures appear to favor maintenance of existing land uses. The dictum, developed largely in eastern hardwood forest systems, that conservation areas should be as large as possible, fit equally well for grassland songbirds and prairie ducks. However, the possibility that substantial amounts of grassland in large tracts can be returned to public ownership for avian conservation purposes seems slight. Financial constraints, "downsizing" of government functions and substantial public resistance to increases in land ownership by government entities all will limit progress on this front. The approach should be pursued, of course, and integration of game and nongame interests is essential. Nevertheless, most existing and potential grasslands will remain in private ownership. Hence, restoring and maintaining grassland ecosystem functions, including sustainable bird communities and populations, depends largely on developing effective conservation programs on private lands.

One promising approach to improving conditions for birds on private lands involves agricultural programs implemented primarily to control erosion and reduce crop surpluses. The Conservation Reserve Program of the U.S. Department of Agriculture has resulted in reestablishment of perennial grassland on millions of hectares of highly erodible cropland for a 10-year period, literally dwarfing the amount of grassland protected for other conservation purposes. Substantial benefits accrue to songbirds (Johnson and Schwartz 1993), waterfowl (Reynolds et al. 1994) and resident game species. Concerted and coordinated efforts to maintain the Conservation Reserve Program when current contracts begin to expire in 1996 and to improve benefits of the program for birds represent an exceedingly important task. Broad conservation projects such as the U.S. Prairie Pothole Joint Venture and the Canadian Prairie Habitat Joint Venture, both under the North American Waterfowl Management Plan, offer substantial benefits for a wide variety of bird species and exemplify the ecosystem-wide approach that will be necessary to achieve significant progress. Establishment of rotational grazing systems, and other efforts to return agriculture on the prairie and plains to a sustainable basis, must form a major part of conservation efforts for grassland birds.

The protection or restoration of very large tracts of forest or grassland habitat with inherently low rates of nest predation and parasitism represents an ideal that should be pursued relentlessly. Nevertheless, many managers face the dilemma of being responsible for overseeing areas that are reasonably large (and hence, exceedingly expensive), yet that function as population sinks for ducks and/or songbirds because of landscape-level problems. For these situations, we desperately need methods for reducing levels of nest predation and parasitism. Lethal control of cowbirds (Robinson 1993) or generalist nest predators (Lokemoen 1984) may be effective, but often faces opposition from some segments of society (Garrott et al. 1993). Nonlethal alternatives such as reducing anthropogenic subsidies and favoring predator species that are compatible with viable reproduction by birds should be explored. Ideal methods would be ecologically sound (even if second best to landscape approaches), socially acceptable and of benefit to a broad spectrum of the impacted species. High rates of nest predation and parasitism represent critical problems for many species of birds in most small tracts of habitat and some large tracts. Understanding the ultimate and proximate causes, the diversity of species impacted and the pervasive nature of the problem represents an important step toward finding viable solutions.

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Managing Habitats to Avoid Game and Nongame Conflicts

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Introduction

Many natural resource management agencies are embroiled in funding and policy conflicts, attempting to manage both game and nongame wildlife populations on stagnant or decreasing budgets. This often results in failure to meet objectives for either game or nongame populations and in dissatisfied user-groups (customers). The game and nongame factions often have differed in the past; our contention is that this divisiveness needs to be forgotten, and new, coordinated efforts at managing "wildlife" habitat, not game or nongame wildlife, should become the focus of our management efforts. In this paper, we discuss several examples of how this philosophy currently is working in the hopes that new, broader partnerships can be developed in the future.

A Perspective of the Past

Prior to the 1980s, wildlife management in North America tended to be synonymous with game management. Even today, many groups do not see much difference between the terms "game" and "wildlife." For the most part, state wildlife management agencies have been funded from hunter license revenues. Thus, before the 1970s these agencies primarily were concerned with managing game populations to provide opportunities for hunting. The Migratory Bird Treaty Act of 1916 placed responsibility for migratory bird management on the U.S. Fish and Wildlife Service, however, for the most part, federal management of migratory bird populations was oriented toward migratory gamebirds: waterfowl, woodcock and doves. During these "early years" of wildlife management, nongame animals were not on the agenda of most wildlife management agencies.

Because both state and federal natural resource management agencies were concerned mostly with game species, and because population information on game animals was necessary to establish harvest regulations, methods of inventorying wildlife were developed primarily for game animals. It simply made sense to have population information on those species which were subject to harvest pressures. This caused a proliferation of information on inventory and management of game species. A review of all *Journal of Wildlife Management*, *Wildlife Monographs* and *Wildlife Society Bulletin* papers indicates that of 40 papers on survey methods or actual surveys of birds, 29 (73 percent) are devoted to game species, while only 11 (27 percent) detail techniques for inventorying a nongame species.

This is not meant to imply that there was no interest in nongame wildlife. Many biologists were conducting studies of nongame migratory birds, as evidenced by the numerous papers published in ornithological society journals such as *The Auk, The Condor* and *The Wilson Bulletin*, just to name the most popular North American journals. Additionally, many states started nongame programs during the 1970s and 1980s and began developing state lists of nongame species.

Also in the 1970s, concern about populations of nongame species began to mount. The Breeding Bird Survey (Robbins et al. 1986), the first national census to provide indices of nongame bird populations, indicated many populations of nongame species were declining (Peterjohn and Sauers 1993). Organizations and individuals interested in nongame species began to place pressure on natural resource management agencies to address concerns about declining populations of nongame species.

The problem was that very few new revenues came along with the pressure to conduct additional studies and surveys, and to manage nongame species. In fact, because of decreasing hunting license sales, state resource agencies' budgets were decreasing. Some efforts at raising new revenues, such as nongame ''checkoffs'' on state income tax returns, initially provided a source of income that appeared promising. The funds derived from income tax returns, however, have tended to decrease over time in most states. Natural resource agencies were in the difficult position of trying to add new nongame programs while maintaining their traditional game programs with few if any increases in budget.

What often has been ignored in the game versus nongame conflict is the fact that wildlife agencies, both state and federal, had been protecting and managing *habitat* for years. While many management activities conducted on these lands were geared toward game species, the lands benefitted a wide spectrum of species. Were it not for dollars spent by hunters, much public land used by numerous species of wildlife would be lost to development and other forms of degradation.

This brings us to the heart of our argument. Both hunters and nonhunters need to recognize that successful management of wildlife into the 21st century will not result from competition between game and nongame programs. Success will come with concentrating on *managing habitat* and selecting management techniques that will benefit both game and nongame. Land managers must recognize that with slight modifications to existing habitat management practices, there need not be a conflict between nongame or game benefits; both can come out winners.

One of the major drawbacks of this, of course, is the lack of quantifiable information needed to develop nongame management plans. Most resource managers graduated from wildlife curriculums and need additional training on nongame species biology. Basic information in the management literature on life history, food habits and habitat requirements often is buried in the quagmire of ornithological literature or may not even exist. Additionally, we must continue to recognize that humans more easily identify with animals rather than with their habitat needs. In general, people will be more willing to write a check to save a critter than to protect a field. There is nothing wrong with this, but to successfully manage wildlife into the 21st century, we must put our bowling shirts away and begin to work together.

Evidence that this tactic can work exists. We detail some examples of this philosophy below.

Examples

Critics could say that the North American Waterfowl Management Plan (NAWMP), the Partners in Flight Program (PiF) and the Western Hemisphere Shorebird Reserve Network (WHSRN) are narrowly focused initiatives, each conceived to benefit only a limited range of species, i.e., waterfowl, neotropical migrant songbirds and shorebirds, respectively. But if we look at how the three programs can work together in habitats of mutual interest and how they benefit a variety of species, we can see that a wide range of wildlife, especially those not necessarily under the guise of these groups, can benefit from their collective efforts.

Of the three programs, NAWMP has affected the greatest land base through habitat management actions. As of the end of 1992, over 1.6 million acres have been protected, 570,000 acres restored and 2.1 million acres enhanced in the United States and Canada. Yet, not one of these acres of accomplishment is managed in such a way that groups of wildlife other than waterfowl are excluded. Many signs surround the projects initiated under the NAWMP, yet not one of these signs states "for the exclusive use of waterfowl." For example, a single moist-soil unit developed in the midwest can be used by more than 150 species of birds, including 25 species of waterfowl, 25 shorebirds and more than 50 species of passerines (Fredrickson and Reid 1986).

In fact, NAWMP partners worked with WHSRN to publish a handbook which discusses techniques that can be used by wetland managers to modify wetland-management practices that benefit primarily waterfowl so that they also benefit migrating, wintering or breeding shorebirds (Helmers 1992). In a cooperative effort between WHSRN and the U.S. Fish and Wildlife Service (USFWS), a series of workshops have been held across the country which provide specific regional details on managing wetlands for both waterfowl and shorebirds. These workshops have been highly regarded and serve as an excellent example of how groups with interests in different species that use the same habitats can work out habitat-management methods that benefit both species groups.

The cooperative efforts do not end here. Streeter et al. (1993) detailed numerous examples of how NAWMP partners are working on wetland projects that benefit not only waterfowl, but numerous species of shorebirds. For example, NAWMP programs which promote the winter flooding of agricultural fields in Louisiana for waterfowl are used heavily by 33 species of shorebirds. Likewise, in papers presented at this symposium you have heard how waterfowl management projects are highly beneficial to grassland-nesting species (Johnson et al. 1994, Ball et al. 1994) and about an agriculture program that is highly beneficial to ducks and other grassland-nesting species (Reynolds et al. 1994).

Other areas for cooperation exist. The PiF Midwest Working Group has indicated a need to concentrate management efforts on grassland-nesting migratory bird species (S. Jones, USFWS, personal communication). Likewise, the 1994 update of the NAWMP has reaffirmed that the partners involved with this group need to increase efforts at habitat protection and restoration in grasslands which are important for waterfowl recruitment. Not surprisingly, grasslands of high importance to breeding waterfowl also are important to grassland-nesting neotropical migrants.

Numerous grassland-nesting species benefit where NAWMP partners implementing habitat restoration projects in the prairies of Alberta have moved away from seeding traditional mixes of "dense nesting cover" across all upland areas surrounding prairie pothole wetlands and are using a "landscaping approach" by seeding a "dry zone" mix of grasses on the higher elevations, a "moderate zone" mix of grass and forbes at mid elevations, and a "moist zone" mix of vegetation adjacent to the prairie pothole wetlands. This results in fields more characteristic of the native vegetation zones that existed before the grasslands were plowed under for cropland. This "landscape approach" to vegetation management allows a wider diversity of avian species to utilize this cover for nesting and is an example of how managers can develop management plans which emulate habitat that existed before the land was plowed.

Bottomland hardwood forest in the Mississippi alluvial valley is another declining habitat type that provides benefits to both waterfowl and neotropical migrant birds. The PiF Southeastern Working Group has identified bottomland hardwood forest as being a priority habitat for neotropical migrants (Anonymous 1993). Likewise, the NAWMP partners in the southeast have restored over 11,000 acres of forested wetlands. Will there be signs keeping the neotropical migrants out of the tree tops? Will there be fences keeping the ducks out of the flooded timber? Both groups will benefit.

Summary

These are general examples of how barriers between constituencies with species interests can be overcome if each group can identify mutually beneficial projects related to the species' habitat needs. It is not likely, nor necessarily beneficial, for groups with special-species interests to suddenly become "biodiversity" organizations. It is human nature to have specific interests. Those interested in ducks should not feel guilty for their devotion to waterfowl. Those interested in songbirds should not feel isolated and resentful that the habitat needs of these species have not received the funding to date that has been directed toward gamebirds. Each of these groups must recognize that funding bases may continue to erode and limit the programs of natural resource management agencies. Game and nongame factions will do best in this fiscal climate if they seek common ground and co-manage habitat for species guilds, rather than attempting to promote one species group at the expense of another.

As the famous "Pogo" comic strip once stated: "We have met the enemy and they are us." Wildlife professionals do not need to breed dissention and unhealthy competition between "game" and "nongame" organizations. We should promote managing habitats using established management practices that benefit a range of species. Let "united we stand" be our motto to work against factions that would let North American wildlife "take care of themselves" and those groups that would pit one species against another for management resources. Let's work together at managing habitat for the benefit of wildlife.

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Managing Brown Bears as Both Game and Nongame: Past Experience and Future Prospects

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Introduction

Growing demand for bear viewing as an organized recreational activity in Alaska is a relatively new challenge for resource management agencies. Wildlife-related recreational activities continue to be popular for millions of Americans (U.S. Department of the Interior and U.S. Department of Commerce) and nature tourism (Whelan 1991) continues to expand in Alaska. Alaska supports 25,000–39,000 brown/grizzly (*Ursus arctos*) bears in various populations remaining generally stable in recent decades (Miller 1993). Some of these bear populations occur at high densities and at concentration areas, providing the unprecedented opportunity for bear viewing to a public having a variety of perceptions and desires. Bear hunting also is popular in Alaska and the combined interest in bears as a hunted and nonhunted species creates an increased demand on this wildlife resource.

Knowledge of the characteristics of successful bear viewing programs is limited compared with ecological and population dynamics studies of brown bears (e.g., Miller 1990, Schoen and Beier 1990, Reynolds and Bourdreau 1990, Sellers 1993). Recently, studies have benefitted from the opportunities provided at bear viewing areas where behavior, long-term reproduction and human impacts can be evaluated (e.g., Warner 1987, Ablert and Bowyer 1991, Aumiller and Matt in press, Fagen and Fagen in press, Sellers and Aumiller in press) through close observation. These studies are helpful in evaluating the success of many aspects of bear viewing programs.

This paper discusses increasing demand for bear viewing in Alaska and identifies well-known viewing programs that are managed by state or federal agencies. We then focus on two areas where viewing occurs in sanctuaries closed to brown bear hunting. Our objectives are to examine briefly the histories of the two areas and identify common elements of success. Hunting and brown bear viewing uses, including public perceptions are examined in more detail.

Bear Viewing in Alaska

At least six bear viewing locations are managed by resource agencies in Alaska (Figure 1, Table 1). The areas differ in geographic region, the number and species of bears, regulations, number of visitors, access, and types of visitors. Denali National Park and Preserve is one well-known viewing area accessible by road. Brooks River falls and camp in Katmai National Park and Preserve attracts both bear viewers and sport anglers. Total visitation increased from 3,332 in 1976 to 13,920 in 1992 (National Park Service 1993). O'Malley River is a newly developed bear viewing program in the Kodiak National Wildlife Refuge. Anan Creek near Wrangell in southeast Alaska primarily is a black bear (*Ursus americanus*) viewing area that has been closed to black bear hunting since 1937. Two additional bear viewing areas, McNeil River State Game Sanctuary on the Alaska Peninsula and Stan Price Wildlife Sanctuary within the Pack Creek Cooperative Management Area on Admiralty Island, are our focus. Other bear viewing areas occur across Alaska, most of which have no agency presence.

McNeil State Game Sanctuary

McNeil River is located about 211 miles (340 km) southwest of Anchorage in Kamishak Bay at the base of the Alaska Peninsula. It seasonally attracts the largest concentration of brown bears in the world. As many as 67 bears have been observed at one time and 126 individuals were identified during a single summer season (Walker and Aumiller 1993). Some bears are attracted to sockeye salmon (*Oncorhynchus nerka*) in Mikfik Creek in June, but most seek chum salmon (*Oncorynchus keta*) at McNeil falls in July.

The McNeil drainage was closed to bear hunting in 1955. The Alaska Legislature created the McNeil State Game Sanctuary in 1967 with the intent to "provide permanent protection to brown bears and other fish and wildlife and their habitats, so that their resources may be preserved for scientific, aesthetic and educational purposes." Originally consisting of 130 square miles (337 km^2), the sanctuary was expanded to its present size of 178 square miles (461 km^2) in 1991. An adjacent refuge of 207 square miles (536 km^2) was added at the same time. Additional closures to hunting outside the sanctuary and expansion of what is now known as Katmai National Park cumulatively placed the sanctuary on the northern edge of about 6,032 square miles (15,623 km^2) closed to bear hunting (Sellers and Aumiller in press).

Regulation of bear viewing activity began in 1973 with a lottery permit system (Faro and Eide 1974). A maximum of 10 bear viewers were escorted to the falls or other viewing areas each day (Table 1). Many modifications have been made to the McNeil viewing program, but a priority to minimize impact on brown bears remains central to the program.

Sanctuary visitors increased from 48 in 1973 to 225 in 1993. Number of permit applications, a measure of demand, increased from 669 in 1979 to 2,150 in 1993 (Figure 2). Although people and bears sometimes are as close as 6 feet (2 m), no human injuries have occurred during the 20 years of the permit program and no bears have had to be destroyed. Sanctuary staff rely heavily on brown bear habituation, defined as "reduction in the frequency or strength of response following repeated exposure to an inconsequential stimulus" (Aumiller and Matt in press). Food conditioning of bears is strictly avoided. History and operations are detailed elsewhere (Walker and Aumiller 1993, Aumiller and Matt in press).

Table 1. Characteristics of selected bear viewing areas in Alaska.

Name/location	Permit required	Daily use/seasonal use	Agency oversight	Salmon species	Other nearby uses	Fees
McNeil River State Game Sanctuary—Alaska Peninsula	Yes/pre-issued lottery permits	10 daily to viewing area/295 season	AFD&G[1]	Chum, socheye	Limited commercial and sport fishing	Lottery permit: $100–250—fees vary
Stan Price State Wildlife Sanctuary—Pack Creek, Admiralty Island National Monument/Kootznoowoo Wilderness	Yes/reservations and daily lottery	24 daily permits/~1,000 season	USFS[2] & ADFG	Chum, pink	Recreational boating, camping, commercial fishing, deer hunting	$10 administrative fee charged by USFS
Denali National Park and Preserve—interior Alaska	No, limited bus access	Estimate 200,000 1992 season	NPS[3]	None	Hiking, camping	$3.00 park-use fee; $4.50 bus reservation
Anan Creek—near Wrangell, southeast Alaska	No, limited cabin space and marginal boat anchorage	~30 daily/1,830 in 1992, and 1,512 in 1993 season	USFS	Pink	Sportfishing, brown bear hunting, commercial fishing	None
O'Malley River—Karluk Lake, Kodiak Island	Reservations with private operator	6 to viewing areas, 114/season	USFWS[4]	Sockeye	Same location sportfishing and hunting at other times	$1,400
Brooks River—Katmai National Park and Preserve—Alaska Peninsula	Unlimited day use, limited overnight space	120 overnight, 13,920 for 1992 season	NPS	Sockeye	Sportfishing	None

[1]Alaska Department of Fish and Game.
[2]USDA Forest Service.
[3]National Park Service.
[4]U. S. Fish and Wildlife Service.

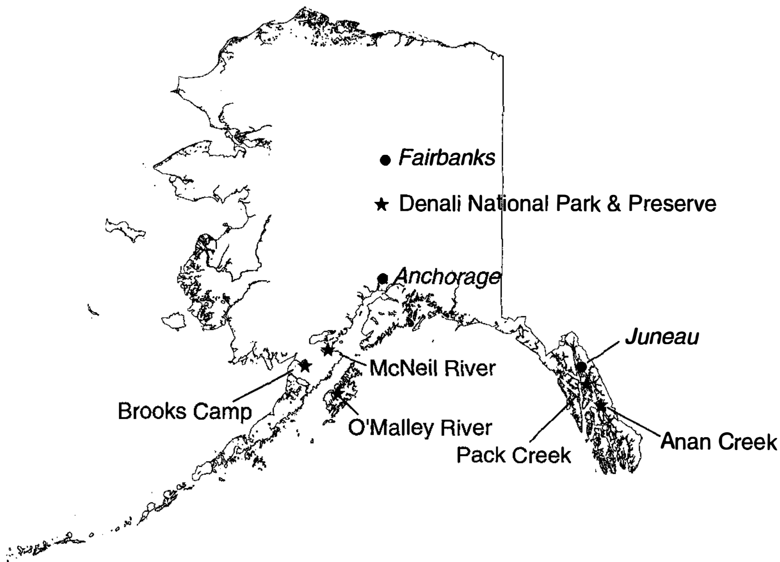

Figure 1. Selected bear viewing areas managed by resource agencies in Alaska.

Stan Price State Wildlife Sanctuary—Pack Creek

The Stan Price State Wildlife Sanctuary was established in 1990 to protect fish and wildlife populations and public use values. The sanctuary covers 613 acres (248 ha) of state tide and submerged lands. This area is managed jointly with the USDA Forest Service (Forest Service). Uplands are within the Admiralty Island National Monument and Kootznoowoo Wilderness that was established in 1980. The area is co-managed within the larger Pack Creek Cooperative Management Area. Summer access was authorized by advance permit beginning in 1988 (Table 1), however, permits first were limited in 1992.

Pack Creek and adjacent areas have a long history of closure to brown bear hunting. Closures began in 1935 when 20 square miles (52 km^2) were closed at the same time as an additional 60 square miles (155 km^2) on interior portions of Admiralty Island. The Pack Creek drainage was not known originally as a special brown bear concentration area. Bear habituation began in 1955 when Stanton Price and his wife moved to the tidal meadow at Pack Creek. In 1984, about 75 square miles (194 km^2) were added to the closed area, bringing the size of the closure to 95 square miles (246 km^2), about 6 percent of Admiralty Island. Up until the mid-1980s, there was little on-site agency presence. During the 1980s, viewing interest increased greatly (Figure 2) and some bears obtained food from visitors. On-site agency supervision began in 1985. A permit system now is administered by the Forest Service.

A minimum of 15–22 individually recognized brown bears used the Pack Creek estuary and stream mouth from 1987–1991 (R. Fagen and J. Fagen unpublished data). Observing five or six bears at once is considered a good viewing day; most visitors view at least one bear. Pack Creek is only 25 miles from Juneau and easily visited by boat or plan for day use.

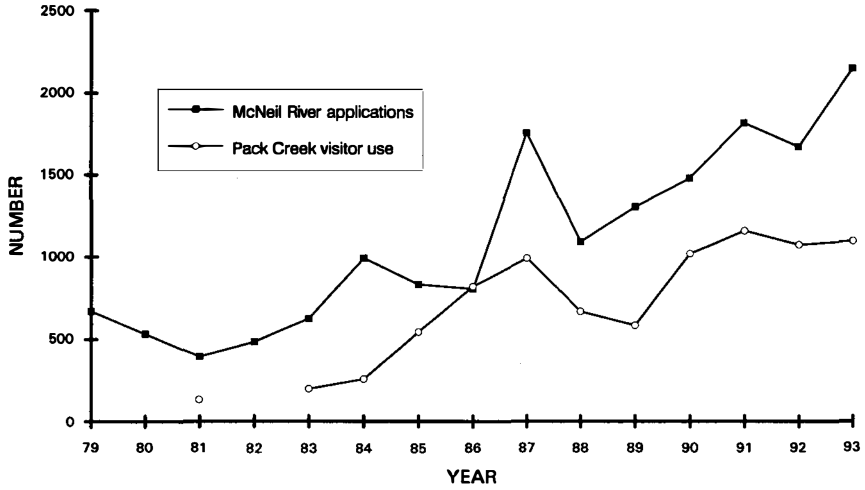

Figure 2. Demand for brown bear viewing at McNeil State Game Sanctuary and Pack Creek Cooperative Management Area, Alaska. The number of visitors is restricted at McNeil River, so demand is best expressed by permit applications. Number of visitors were restricted at Pack Creek in 1992, but all available permits have not been issued annually, so the number of visitors still indicates demand.

Competing Interests

Human activities, regulations and the physical settings have affected program administration at McNeil River and Pack Creek. Sportfishing is restricted at both locations, avoiding a conflict that often exists at other viewing areas. Wild salmon populations originating in both areas are commercially fished. There have been tensions between commercial fishing activities and bear viewers at McNeil River. Construction of a salmon fish pass on the adjacent Paint River stimulated extensive public debate and resulted in the most recent sanctuary extensions and creation of the adjacent McNeil State Game Refuge. Debate about hunting has focused on the limited hunting allowed within the McNeil refuge. Various proposals to expand or reduce the size of a hunting closure surrounding Pack Creek have focused attention on the size of the closure. Through the process, resource professionals have learned of the importance of wildlife viewing from segments of the public and adequate public policy review.

Public Perceptions

To further understand public perception in the form of allocation arguments, we examined more than 300 letters to the Board of Game regarding recent hunting closure proposals near Pack Creek. Most letter writers favored either existing boundaries (45 percent) or expanding the area of closure (40 percent).

The reasons most often cited by all user groups for maintaining or expanding the Pack Creek area closure were related to equity (88 percent), economics (71 percent) and ethics (63 percent). The equity issue revolved around the belief that viewers should be allocated their "fair share." This often was expressed as the percentage of the area on Admiralty Island closed to hunting and an emphasis of the unique experience of viewing habituated brown bears in a natural or wilderness setting. Alaska residents often mentioned economic arguments, particularly the comparison of the economic value to the tourist industry from repeated viewing of bears versus the single economic return of a guided bear hunt. The ethics issue was related to the belief that hunting habituated bears was unethical. Often cited reasons for supporting a reduction of the area closure were general support for all proposals to open closed areas (53 percent), a belief that hunting and viewing were compatible in the same area (42 percent), and that there was no biological basis for the closure (32 percent).

The group of perceptions suggest differences in beliefs about the appropriate use of the Admiralty Island brown bear resource. These arguments, as well as the bear viewers' perception that the viewing experience is unique and inviolable, are common to other bear viewing areas like McNeil sanctuary. For example, informal sampling of McNeil sanctuary visitors revealed support for the closing of nearby areas to bear hunting and protection of highly habituated individual bears.

Comparing McNeil River and Pack Creek

Human use is higher at Pack Creek, with about 1,000 summer visitors, while the visitation at McNeil is now about 250 people. The ease of access to Pack Creek leads to a variety of users, most of whom have no experience with brown bears in a wild setting. McNeil River access is limited by a lottery permit system. Additionally, the high commitment and cost to reach McNeil River might limit it to individuals with strong personal desire.

Visitors to Pack Creek are not constantly accompanied by agency staff as occurs at McNeil River. Permits are administered by the Forest Service and are allocated between guided and non-guided trips. Guided wilderness charter businesses depend on Pack Creek permits. Experience at Pack Creek suggests that it is not necessary to have an extraordinary concentration of bears for a successful viewing program.

Both areas share an important similarity that probably is common to all successful bear viewing areas. The management priority at McNeil River and Pack Creek favors undisturbed bears. Controlling land use with cooperative agency efforts and public support have been used to achieve this goal and both are considered essential by managers.

Viewing and Hunting Relationships

Bear Hunting Patterns

Brown bear hunting has occurred for many decades on Admiralty Island with about 50 bears being taken annually in recent years. Admiralty Island has a high brown bear density (~1 bear/mi^2; 400–460 bears/1,000 km^2) based on three multiple sample density estimates (Schoen and Beier 1990, K. Titus and L. Beier unpublished data). This equates to an island population estimate of 1,700 bears making for high hunter interest. Brown bear hunting has remained constant over a four-year period with about 350 hunters going afield annually in Game Management Unit 4. Permits are not restricted, so hunter use is an indicator of demand.

Brown bears are a big game management priority on the Alaska Peninsula and the area attracts hunters from around the world. Two Alaska Peninsula brown bear density estimates were 0.49 bears per square mile (190 bears/1,000 km^2) at Black Lake (Miller and Sellers 1992) and 1.42 bears per square mile (550 bears/1,000 km^2) in Katmai National Park (S. Miller and R. Sellers unpublished data). In areas open to hunting on the Alaska Peninsula, Sellers (1993) estimates an overall brown bear density of 0.24 bears per square mile (93 bears/1,000 km^2) and a population of 5,679. Annual hunting season harvests averaged just over 273 bears for the Alaska Peninsula. In the game management subunit closest to McNeil River, harvests over the last five years varied from 12 to 35 bears. Hunting regulations are adjusted to harvest no more than 5 percent of the population in this area.

Bear Hunting and Viewing

The influence of brown bear hunting on brown bear viewing is a combination of biological, public policy and perceptual issues. Managers attribute maintaining habitat quality and control of visitors as the most important factors contributing to the success of the McNeil sanctuary program (R. Sellers personal communication). Other factors, including closure of hunting in adjacent areas that include home ranges of bears using the sanctuary and increasing or maintaining salmon runs, have contributed to program success.

The McNeil sanctuary includes some bears that are exposed to light hunting pressure outside the sanctuary associated with the alternate-year hunting season on the Alaska Peninsula. Limited hunting is allowed in the McNeil River State Game Refuge directly north of the sanctuary and sanctuary bears can travel to areas open to hunting. Counts of adult bears in the sanctuary between 1979–1989 suggest hunting effects. Although overall counts continued to increase during this 11-year period, the rate of increase slowed in the alternative years following hunting seasons outside the sanctuary (P = 0.0022, one-sided Wilcoxon rank test, [Statxact software, Cytel Software Corp 1991], Figure 3). The dampening effect on the number of bears using the sanctuary is described by Sellers and Aumiller (in press).

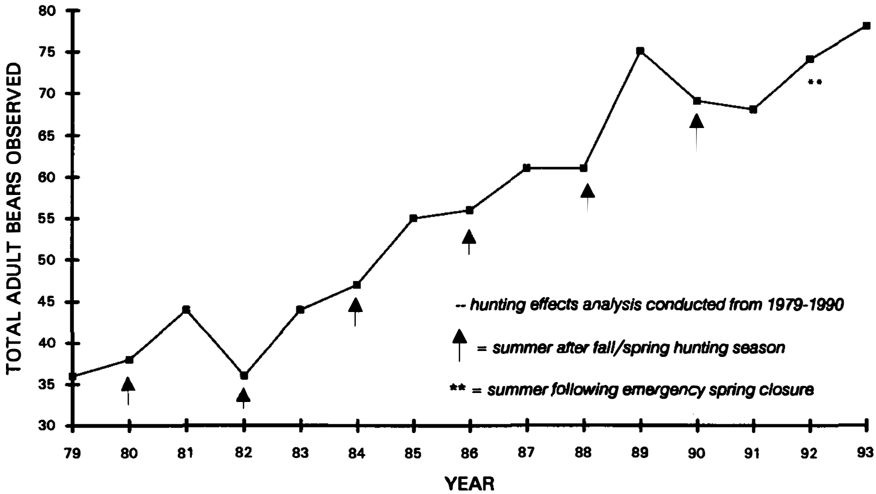

Figure 3. Total number of adult brown bears observed by year at the falls at McNeil River State Game Sanctuary, Alaska.

The evolution of sanctuaries and organized viewing activity adjacent to areas where bears are hunted has raised new management issues. Interest in bear viewing emphasizes the characteristics and welfare of individual bears, while bear management objectives traditionally focus on broad-based management goals. We have little quantitative information on the importance of individual bears to the maintenance of viewing opportunities, but we suspect that certain individual bears are important for program success and maintenance.

The effects of hunting activity on the behavior of bears visiting recreational viewing areas is not fully understood, nor is it clear whether all bears that become habituated to people in viewing areas are more susceptible to hunting mortality elsewhere. Experience at both McNeil River and Pack Creek suggest that it is feasible to operate successful bear viewing programs adjacent to larger hunted bear populations. Pack Creek is surrounded by a larger hunted population. We characterize the hunting pressure near McNeil River as very light and near Pack Creek as light to moderate. It seems reasonable to assume that hunting will have some influence on the numbers and behavior or bears at viewing areas. Yet, this influence may be acceptable if management objectives do not stress maximizing viewing opportunities.

At both McNeil River and Pack Creek, this issue translates to the policy question of where to draw the boundary for hunting. The Alaska Board of Game decision to continue hunting in McNeil Refuge immediately north of the sanctuary was based on an estimate of sustainable yield that coincided with recent harvest. A permit hunt during each spring and autumn semiannual season has been authorized with the intent of not exceeding historic and sustained harvest levels of three bears per calendar year.

These decisional criteria are oriented around traditional bear management objectives that manage for populations. However, management of bear viewing areas such as McNeil River and Pack Creek is behaviorally oriented and management objectives should account for additional objectives that provide for the maintenance of individual bears. Human behavior is constrained and brown bears are habituated at viewing areas. These criteria beg the question of how to design bear management programs and measure success. If natural resource managers are to do this, they must devise measures that are sensitive to at least four dimensions: bear population status, bear behavior and habituation, differing human perceptions of the bear viewing experience, and conflicts between the two uses.

Additional Information Needs

Better information continues to be desirable about bear ecology and human dimensions of bear viewing. The home ranges of bears at McNeil are not completely understood. Home range and movement data are available for brown bears from Admiralty Island that provide some information about the adequacy of the closure to hunting. Of five bears radio-collared at Pack Creek, one male was killed outside the area closed to hunting, while nearly all of the females' home ranges were completely within the present closed are of 95 square miles (J. Schoen unpublished data). Interpretation of Schoen's data indicated that 67 percent of the Pack Creek radio-collared bears would be at risk to hunting if the closed area was decreased. This might indicate that the current closure size is adequate for protecting female bears with established home ranges centered on Pack Creek. Additional information on bear movements associated with viewing areas may be useful, but radio-collaring bears at viewing areas may not be desirable.

Resource valuation studies such as that proposed for McNeil River (Swanson et al. 1992) would help compare the dollar value of bear viewing with other activities that might influence bear populations. Clayton and Mendelsohn (1993) report the results of contingent valuation questions asked of McNeil Sanctuary visitors. Results indicated that visitors are willing to pay an average \$228–\$277 per person for a permit to visit the sanctuary. This information was used to adjust McNeil permit fees.

Knowledge about the motivations for bear viewing as a social-psychological experience also is needed. What combination of wilderness experience, number of bears seen and proximity to bears are important to visitors? Resource managers are only beginning to acquire this type of non-biological information.

Summary

Bear management that provides for viewing and other opportunities depends on a number of factors. These include biophysical settings that concentrate bears, localized management priorities including spatial and temporal zoning, health bear populations, and a public willingness to support effective regulations. The emergence of popular bear viewing areas near locales where hunting, sportfishing, commercial fishing and guiding activities already have been established has created the need for additional management information.

Our current state of knowledge indicates that successful bear viewing programs and bear hunting can coexist under constraints in place at the two viewing areas we examined. However, hunting, viewing and other activity may have other interactions to a natural system that may or may not be a desirable management goal. We urge caution in oversimplifying these relationships because long-term and subtle interactions of these management approaches are not understood. Experience gained at McNeil sanctuary since 1973 and at Pack Creek since the mid-1980s guided our discussion. The most important lessons from these experiences are:

- Successful bear viewing programs require locations that will have unusual bear concentrations or a reasonable certainty of viewing bears. Large numbers of bears are not required for successful viewing programs.
- Successful bear viewing programs may occur best in locations with adequate and non-declining populations and where there is a high probability of protecting certain individual bears.
- High-quality viewing programs require control of land uses at and adjacent to the viewing areas and a strong commitment to reduce or eliminate any human uses that interfere with natural brown bear behavior and habituation.
- The development and maintenance of a viewing program must include a public policy-making process. Sensitivity to viewer values and changing public desires is important for measuring program success.

Acknowledgments

We thank S. D. Miller for data preparation and E. F. Becker for statistical testing of hunting effects. L. Beier, J. Fagen and J. Neary provided historic and current insights about Admiralty Island history and Pack Creek bears. USDA Forest Service staff share in the collection of data and management at Pack Creek and we appreciate their information. S. Mills (National Park Service) and V. Barnes (U.S. Fish and Wildlife Service) provided information on other Alaska bear viewing areas. The detailed reviews of S. Miller and R. Sellers are especially appreciated.

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Closing Remarks: The Whole is Greater than the Sum of the Parts

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In closing, I first want to thank all of the speakers and authors for the considerable effort that went into their presentations. They have given us an excellent overview of many of the varied aspects of what we call integrated management.

At one end of the spectrum, integrated management simply can be viewed as keeping track of all the parts, which is not to say that we should become so enamored of the individual parts that we overlook the functioning of the whole. Carried to its logical conclusion, integrated management becomes nothing less than a holistic approach to resource management, or what is being widely referred to today as ecosystems management. Far from being merely an organizational effort to blur the distinctions between game and nongame, I think that integrated management requires a major philosophical change in the way that we manage wildlife.

Several of the previous speakers alluded to real or perceived conflicts between game and nongame management. I would like to focus on some of the common bonds.

The most common denominator is habitat. All species use it, and game and nongame species seem to live quite harmoniously together in their chosen natural communities. Only where habitats have been altered by man are major conflicts likely to appear. In our manipulations of habitat, we must recognize that not all habitat patches are equally valuable to wildlife. In particular, we must avoid creating population sinks in the name of habitat restoration or improvement.

We need to consider ecological processes and do a better job of documenting the effects of our management actions on all other organisms in the affected community. It no longer is acceptable, for example, to manage a tract of habitat exclusively for grouse or deer without first considering the costs and benefits of those management actions to other species.

Traditionally, we have tended to practice single-species management, with emphasis on game and endangered species. With rare exceptions, we can no longer afford that luxury. We must manage wildlife on the basis of communities and landscapes. Integrated management simply makes good economic and ecological sense.

Public outreach and education are essential for the successful implementation of integrated management. We must make an effort to identify our customers and constituents, and strive to educate all segments of our society about our wildlife heritage. Only in that way can we hope to build broad support for wildlife conservation activities. Just as we have come to value biodiversity in nature, we must promote cultural diversity in the workplace. This is essential if we hope to relate effectively to the multitude of audiences that look to us to conserve wildlife and their habitats for use and enjoyment by people.

To a greater degree than ever before, we need to incorporate economic and social considerations into our decision-making process. If we are to be successful in conserving a natural diversity of wildlife and preserving opportunities for the American public to enjoy a quality wildlife experience, we must be innovative in creating interstate and international alliances. We need to be proactive in forging nontraditional partnerships with corporate America to maintain and improve the quality of the environment that we all share with wildlife.

Recently, when I remarked to a colleague that I thought the term "wildlife management" was in danger of becoming obsolete, and that we needed to reach out and embrace some of the tenants of conservation biology, I was told that such an attitude was demeaning to wildlife professionals. I was surprised by this reaction because in my mind game biologists were in the forefront of the American conservation movement and remain so today. I'm not advocating that we turn our backs on the achievements of our game management past. Rather, we should take the best traditions of wildlife management and incorporate them into the new paradigms of conservation biology, biological diversity and ecosystems management. Our ultimate goal should be to achieve balance and harmony in our wildlife programs.

Our challenge today is to maintain all the interlocking parts so that our successors someday can put together all the pieces and make sense of that mysterious puzzle called ecosystems.

Special Session 5. *Biological Diversity and Sustainable Ecological Systems*

Chair
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Cochair
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Biological Diversity and Sustainable Ecological Systems: Bringing Diverse Goals Together for Conservation

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Introduction

As natural resource managers and scientists struggle to examine, define and apply concepts of sustainable ecosystem management, divisiveness and fragmentation within the conservation and natural resource community is intense. Today, many of us find ourselves encouraged by strong public interest and support for proper resource stewardship, but immersed in increasingly competitive, fragmented and specialized niches represented by narrowly defined organizations and institutions. We now save the seals, enlist for elk or negotiate for Neotropicals.

In this realm, are there common areas where we can agree on natural resource management and direction? Do we have common messages to communicate to diverse publics? How can professionals adequately represent the diverse interests of a society intensely interested in our common natural resources?

Ecosystem management can embrace many common values necessary to effectively integrate biological information into systems of organization and management. Is ecosystem management new? Principles and methods important to ecosystem management have been successfully applied in the past. Restoration success stories abound. Forests of the eastern United States are much more plentiful than they were eight decades ago. Techniques and concepts used in game restoration have proven effective. Can we take our experience and apply it to restoration needs to meet ecosystem management objectives? I believe we can.

Today, under the banner of ecosystem management and biodiversity, we see major efforts in agencies and institutions to broaden or define missions, to organize, investigate and carry out many principles needed for ecosystem management. Proposals abound to change organizational structures, coordinate and integrate information across administrative boundaries, and even start new agencies focusing on ecosystem management and conservation of biodiversity. The proposed National Biological Survey within the U.S. Department of Interior and the emphasis on ecosystem management for the USDA Forest Service are two examples of efforts to strengthen information and management for ecosystems at the federal level.

Ecosystem management, with its broad definition, is an effective label in bringing together different interests. The banner can serve as a common base from which to articulate specific diverse objectives. Conserving biological diversity and sustainable ecosystems can be an underlying goal of narrow and sometimes fragmented interests within the conservation community.

About This and Earlier Sessions

In 1992, a Special Session of the 57th North American Wildlife and Natural Resources Conference addressed "Biological Diversity in Wildlife Management." The session provided a framework for addressing biodiversity on various ecological and biological scales.

Presentation in this Session will add principles under the heading of sustainable ecosystem management for biodiversity conservation. Aspects of human values, the use of species to address community and ecosystem conservation, and botanical and aquatic concerns are presented. The Session includes a conceptual framework for defining and integrating ecosystem goals, and, through diverse examples, demonstrates key principles of ecosystem management on a variety of spatial, administrative and biological scales.

A Framework to Conserve Biological Diversity through Sustainable Land Management

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Introduction

Sustainable land management has no "book" to summarize extensive trial and error experience into a meaningful model or set of guidelines for implementation (Dumanski 1994). As a result, we sense dissolution among biologists and resource managers in implementing sustainable land management to conserve biological diversity. Part of the dissolution is due to the complexity of ecosytems (Odum 1992), the minor role science has played in the conservation of biological diversity (Weston 1992), and lack of coordination and planning among natural resource agencies (Knopf 1992). Given the lack of consensus about methods, it often is hard to say how decisions are made and priorities established to achieve either the conservation of biological diversity (Pimm and Gittleman 1993) or sustainable land management (Samson and Knopf 1994a).

The conservation of biological diversity and ecosystem management are part of the more general problem of sustainable land management (Samson 1992). Biological diversity is the variety of life and the ecological processes native to a particular landscape (Wilcove and Samson 1987). Ecosystem management—"people trying to accomplish something bounded in space" (Gordon 1993: 240)—to conserve diversity differs from traditional resource management in three ways. First, substantive connections among people in government and non-government organizations are key to implementation (Honadle 1993). Second, resource management must emphasize how species are distributed across space (Samson and Knopf 1982, 1993, Pimm and Gittlemann 1992). Conserving areas with high species richness is of little value if they share the same set of species (Magurran 1988). Third, the health of both landscape- and ecosystem-level processes that support and sustain species, communities and ecosystems across space and over time is a priority (Odum 1992) and, perhaps, the management priority (Norton and Ulanowicz 1992). In this paper, we outline a framework to conserve biological diversity through ecosystem management and consider the more important components—the Ecosystem Joint Venture, diversity, ecological processes and monitoring—in some detail.

The Framework

The Ecosystem Joint Venture (EJV)

Broad conservation goals are identified in the 29 federal laws that relate to the conservation of biological diversity in the United States. While these may facilitate visionary statements by agencies and broad policy formulation, a more explicit framework is needed to conserve the two components of biological diversity—the variety of life and the ecological processes characteristic to a particular ecosystem—to be of value in planning and management. The EJV is the first step in our framework (Figure 1). The recommendations to manage natural resources by ecosystem boundaries is not new (Caldwell 1970). Defining regional boundaries of national agencies along ecosystem boundaries (*e.g.*, the major vegetative provinces of North America) would better align resource management decisions to conserve biological diversity (Knopf 1992). Current, overlapping jurisdictions among resource agencies complicate the management picture. An example, the North Platte River is managed by 120 different state and federal agencies, each with some form of jurisdiction. Unclear division of responsibility and authority only serves to slow the strengthening of resource management by agencies (Honadle 1993).

A core set of organizational relationships and social and economic data are part of the proposed framework to ecosystem management (Figure 1). Ecosystem management and conservation of biological diversity are intimately tied to human values and institutions (Salwasser et al. 1993). Similar to issues in ecology, it must be recognized that the social decision process and human behavior are complex (Romm 1993). Tradeoffs for specific ecosystems or the environment as a whole should include social data and understanding of the decision-making process (Hass 1991). Second, economists should consider the function and value of ecosystems as a whole (Toman 1993). This will require a fundamental shift in economic theory—ranging from the application of less damaging technology in agriculture to incorporating costs that appear distant, i.e., global warming, depletion of the ozone layer, etc., in local and regional habitat conversions (Dailey and Ehrlich 1992).

Understanding connections among social values and economic resources may uncover new resource management strategies (Honadle 1993). The loss and degradation of wetland habitats are the major waterfowl management problem in North America. The North American Waterfowl Plan (U.S. Fish and Wildlife Service 1986) seeks to reverse downward trends in waterfowl populations and restore wetland habitats. An innovative strategy to implement goals of the North American Waterfowl Plan is the Joint Venture—restoration of the prairie pothole ecosystem, inventory and management of the Playa Lake ecosystem, and so on. The Joint Venture is agreed to by all that participate and is set out in a plan that details the contributions of non-government organizations, individuals, agencies (federal and state), provinces, territories and governments. This innovative effort serves well as the template for the EJV (Knopf 1992). A committee with representatives from the National Academy of Sciences, professional societies, non-government organizations and universities would approve and coordinate EJVs.

Quite apart from the difficulties in aligning administrative and ecosystem borders are a number of inherent difficulties in selecting an ecosystem classification scheme. Nothing more than approximations exist for the distribution of most taxa that can be used to establish a biogeographic base needed to delineate an ecosystem. One must only compare biogeographical provinces proposed for reptiles and amphibians, birds, fish, or vegetation in North America to gain an appreciation of the current lack of congruence (Brown and Gipson 1983). This confusion among classification schemes should not come as a surprise (Rodda 1993). Each taxon has different biological attributes and, as a consequence, patterns in species distribution reflect different

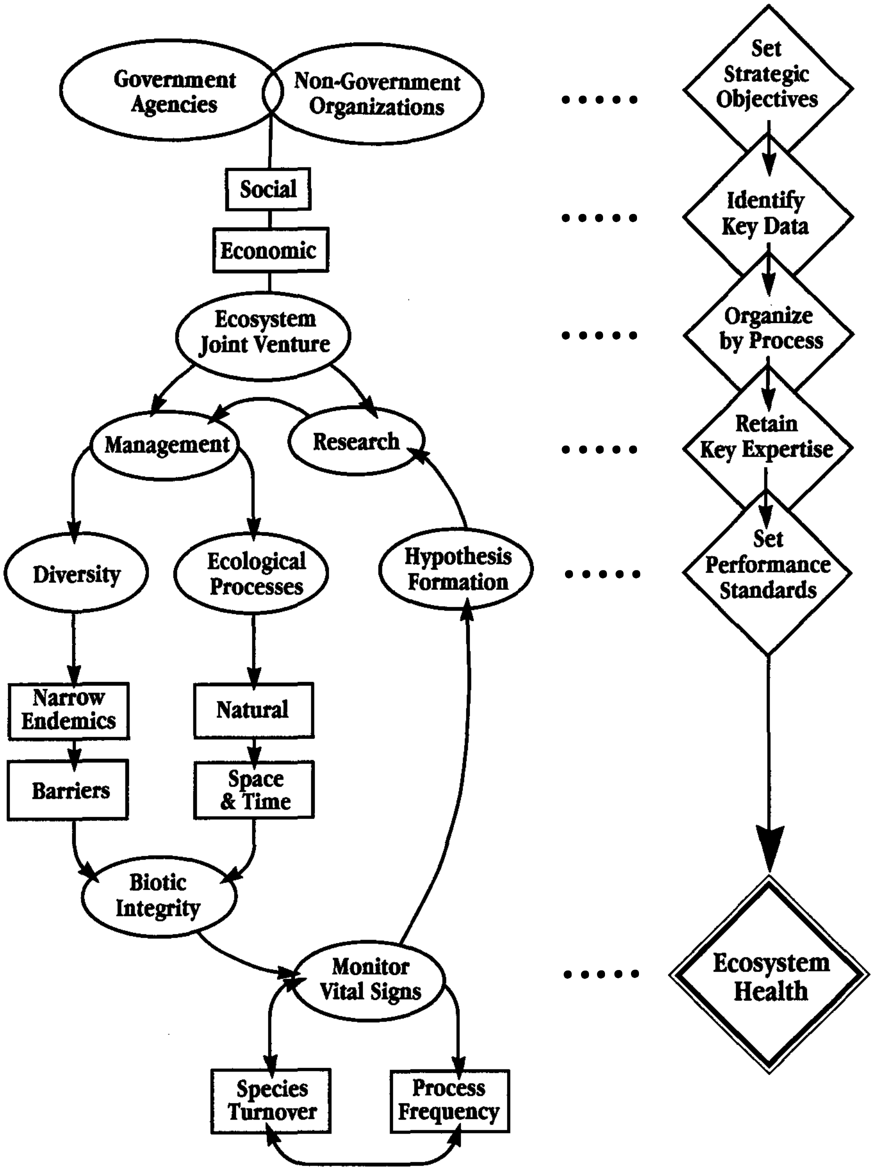

Figure 1. A framework to conserve biological diversity through ecosystem management. Circles enclose framework elements, rectangles enclose key data requirements and diamonds enclose management tasks.

ecological tolerances and requirements. An agreed-upon ecosystem classification system, however, is essential to region-wide planning through ecosystem conservation (Orians 1992).

A more basic and philosophical question is whether the conservation of biological diversity of widely distributed species be accorded the same priority as an uncommon species with a restricted geographic range. Species are neither distributed at random nor in a uniform manner across a continent. Examining species' ranges reveals that certain taxa exhibit distinctive patterns in distribution. Endemic species are those restricted to a large geographic area such as a continent. A narrow endemic is restricted to a particular geographic area or habitat (Kruckeburg and Rabinowitz 1985), and it is the narrow endemic that is most in risk of dropping out when ecosystem functions are deteriorating. Cosmopolitan species are those widely distributed utilizing numerous habitats. The emergence of classification schemes in the early 1800s was based on the distribution of taxa, particularly those with restricted distributions, and how those taxa were organized in a hierarchical fashion. Since then, historic and evolutionary arguments provide a strong basis for the classification schemes based on taxonomic hierarchies. Two centuries of support for the Linnaean hierarchical taxonomic system is clear evidence of the utility of such a species-oriented approach in conservation (Master 1991).

One recent and popular approach to delineate ecoregions that describe patterns in vegetation is to utilize climate (Barbour and Billings 1993). The comparison of the vegetation provinces described by Barbour and Billings to the ranges of bird species north of the Texas/Mexico border shows marginal congruence (Samson and Knopf 1994b). Over 59 percent of the ranges of bird species extend across 3 or more of the 16 vegetation provinces described by Barbour and Billings (1993). Similarly, over 58 percent of the ranges of mammal species north of the Texas/Mexico border extend across 3 or more vegetation provinces. A further impediment in use of climate-based provinces is the inability to predict events at a scale less than an ecosystem (Neilson 1991). An inability to predict local- versus broad-scale events is a major ecosystem observation (Odum 1992). The climate-climax approach also may predict an incorrect vegetation community given the role of ecological processes. An example, the mixed hardwood on the southeastern coastal plain (Kuckler 1964) was in fact historically a long leaf pine community maintained by fire.

There are several solutions to the dilemma of selecting and implementing an ecosystem classification scheme. One is to use the Linnaen taxonomic hierarchy with each major scheme a layer within a geographic information system (Scott et al. 1993). A second is to implement a systematic sampling system, such as the Environmental Protection Agency's Environmental Mapping and Assessment Program (Gallant et al. 1989), and subsequently examine the issue of bioregionalism. A third is to focus on the narrow endemic. Advantages to the latter approach include the fact that narrow endemic-based ecosytems have been shaped by both ecological processes and historical events, they describe general distribution patterns for a variety of life—angiosperms to butterflies to small mammals (Brown and Gipson 1983)—and priorities among ecosystems for conservation can be established (ICBP 1992, Samson and Knopf 1994b).

Diversity

Historically, formulating principles of wildlife conservation has rested on the concept of diversity, specifically alpha or point diversity (Samson and Knopf 1982).

Equally if not more important to the conservation of biological diversity is beta diversity (Pimm and Gittleman 1992). While alpha diversity may describe site-specific species richness, beta diversity describes how many sites are needed to conserve a full variety of species along an ecocline, within an ecoregion or continentally. Gamma diversity is important, for it describes species assemblages unique to a continent or large, distinct ecosystem. A hypothetical example illustrates the point. Two local sites, A and B, hold five species each while a third site supports a single species. If sites A and B share an identical set of species, the appropriate strategy to conserve diversity is to manage for site A or B and site C, assuming the species on site C is unique. The sole priority may be to manage for site C, if species associated with sites A and B are widely distributed generalist species.

An example in the application of diversity is management of the avifauna of the Great Plains of North America (Knopf 1986), an area of native prairie that once extended south from Canada to the Mexico border and west from Indiana to the Rocky Mountains. No other major North American ecosystem has experienced the extent of fragmentation, consistent and regular declines in population numbers of endemic bird species, and loss of essential resources such as soil and water (Knopf 1994). Three hundred and thirty of the 435 bird species that breed in the United States occur on the Great Plains. Nevertheless, only 5 percent (12 species) of the North American avifauna are believed to have evolved in this region, with another 20 species thought to have originated on the Great Plains but ranging more widely today. Importantly, only within the Great Plains can the 32 endemic species be conserved, whereas the remaining 290 species are ecological generalists that occur in a variety of habitats across the North American continent. Furthermore, it is precisely the majority of these endemic species, either because of their limited distribution and/or narrow ecological requirements, that are endangered, threatened or declining (Knopf and Samson 1994).

Securing the future of the endemic species characteristic to the Great Plains requires the conservation of native habitat and the ecological processes that have shaped endemic plant and animal species on the Great Plains over time. A third and growing management obstacle is the non-indigenous species (NIS) (U.S. Congress Office of Technology Assessment 1993). An early goal of the development of biogeographic regions was to identify barriers that block the interchange of organisms between adjacent regions. The goal remains useful in implementing ecosystem management to conserve biological diversity (Figure 1). For example, over 40 percent of the species on native grasslands and 70 percent on agricultural lands are NIS. The ever-growing dissolution of both political and ecological barriers to NIS now is viewed as a growing and perhaps the major threat to the conservation of biological diversity (Cullota 1991, Samson 1992, U.S. Congress Office of Technology Assessment 1993). An example on the Great Plains is the fragmentation of this once major evolutionary barrier as a consequence of non-historic forested corridors that now border major east/west riverways. This corridor-dependent fragmentation has lead to hybridization of certain eastern and western birds—a loss in the variety of bird life beyond that associated with fragmentation of the eastern deciduous forest. An emphasis on ecological barriers—the most significant factor in the evolution of endemics and in the development of regional assemblages of plants and animals—is essential in implementing ecosystem management to conserve biological diversity.

Ecological Processes

No strategy for the conservation of biological diversity will succeed without an emphasis on ecological processes (Ulstrand 1992) (Figure 1). Habitat management, as part of ecosystem management to conserve biological diversity, should emulate both the scale and natural patch dynamics of an ecosystem (Urban et al. 1987: 25). The pattern in disturbance is useful in explaining patterns in plant communities. On a within-site scale, the intermediate disturbance hypothesis predicts that the "intermediate" disturbance frequency coincides with the life histories of the greatest number of species (Clark 1991: 1,102). On a larger or landscape scale, the disturbance heterogeneity hypothesis predicts the number of plant species is related to the extent and timing of disturbance across the landscape. Using the disturbance heterogeneity hypothesis, the disturbance effects can be modeled to reflect disturbance-mediated changes in the structure of forest (Wolfe personal communication) or grassland landscapes (Collins 1992). By extension, one can describe changes in bird species distribution as a consequence of disturbance.

As predicted for plant species by the disturbance heterogeneity models, the number of bird species varies with the extent of disturbance in the northern Rocky Mountains and plains (Figure 2). Few grassland bird species—Sprague's pipit (*Anthus spragueii*) and mountain plover (*Claradrius montanus*)—prefer recently disturbed sites and, at the other end of the disturbance spectrum, only a few species—the Baird's sparrow (*Ammodramus bairdii*) and other late seral species—find preferred habitats. More dramatic is the relation of the number of bird species and extent of disturbance in the coniferous forest. Bird species richness increases with disturbance because the landscape includes more species typical of the natural mosaic characteristic to the western fire-dependent coniferous forests. A dip in habitat variety and bird species richness accompanies the loss (or control) of the natural pattern in disturbance. Historically, about 400,000 to 500,000 acres of forest and upwards to one million acres of grasslands burned annually. Of interest is the difficulty early naturalists and surveyors had describing these ecosystems in the mid- to late 1800s because of smoke-impaired vision.

Monitoring

The implementation of an approach to ecosystem management places a premium on relevant and accurate information to monitor both the resource base and management activities. Several federal agencies and an increasing number of state agencies either have in place or are calling for biological monitoring of water quality (Karr 1991). Historically, implementation of such programs has been hampered by dominance of a reductionism view—the tendency to dissect a subject into smaller and smaller isolated parts as an effort to identify the problem's essential elements. A recent and more successful direction is to develop biological indices that concentrate on the biological integrity of a particular community or ecosystem (Figure 1). Such approaches are appropriate when they characterize the biotic integrity of a system, specifically the comparison of the integrity of a community (aquatic and terrestrial) native to a particular system with that currently present.

The comparison of current with historic is an important temporal component often neglected in monitoring (Samson 1992). More difficult is the integration of monitoring within a spatial scale suitable to ecosystem management to conserve biological di-

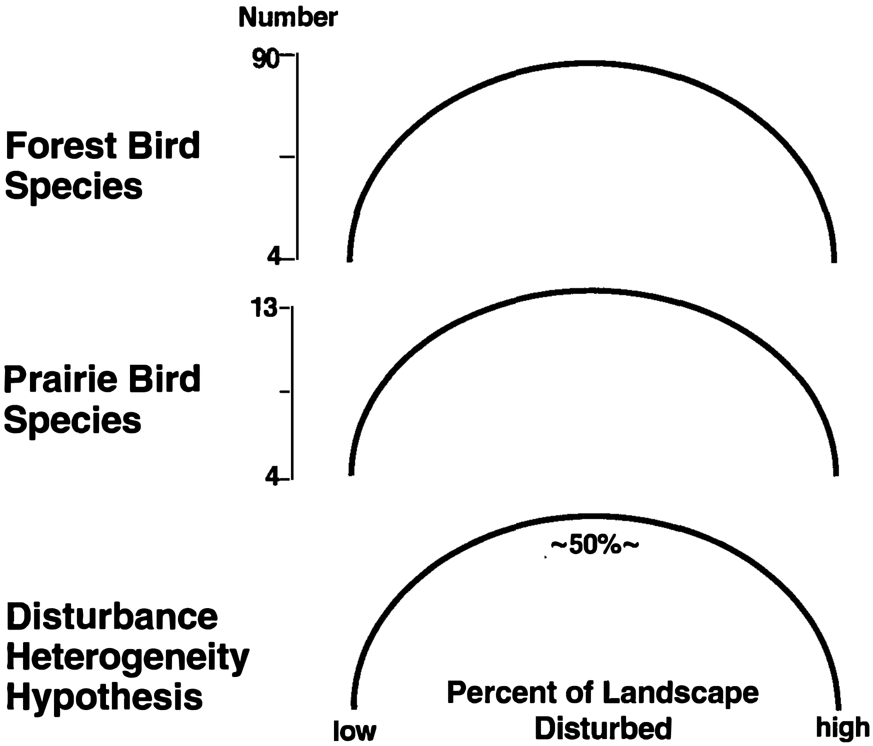

Figure 2. Estimated relation of the extent of disturbance and number of breeding bird species in northern montane coniferous forests and grasslands.

versity—both the variety of life and the ecological processes native to an ecosystem, landscape or site. Ecological processes are argued to be the ''autopoetic'' (self renewal) feature of ecosystems that support and sustain the components across space and time (Norton and Ulanowicz 1992). Health and protection of the health of landscape-level processes should, therefore, be a central goal of any policy or strategy to implement ecosystem management to conserve biological diversity.

In Table 1, we propose a framework to ensure that biological diversity and, therefore, ecosystems remain intact. This ecosystem/biological diversity monitoring framework is based on the concept of turnover. Turnover in geological time is the rate at which species evolve and subsequently become extinct. In ecological time, one calculates a turnover rate where turnover (t) is equal to 1 minus the number of species present at time t divided by the sum of the number of species present at time t + 1, the number of colonizing species, and the number of extinctions. An intact, sustainable ecosystem would have a value near 0, a value near 1 indicates a high level of colonizations and/or extinctions. The time interval to calculate t + 1 will vary, some ecosystems are old, others are recent.

In terms of an ecosystem, we suggest the turnover (specifically, composition or the variety of life) as a monitoring tool to ecosystem management/biological diversity be low and approximate that expected under natural conditions (Table 1). At the landscape level, one must recognize the dynamic nature and expect a higher rate of

Table 1. Monitoring to sustain ecosystems to conserve biological diversity. Monitoring at the ecosystem scale should ensure a high probability of retaining native species and natural processes characteristic to the system. Requirements at the landscape and site levels are less restrictive given the inverse relationship of predictability and ecosystem level (Odum 1992).

	Composition	Function
Ecosystem	<.01%	±.01%
Landscape	<.10%	±.10%
Site	<.15%	±.15%

colonization and local extinction given the mosaic-process dependent character of landscapes. Whether monitoring at the site is realistic is open to question, given the dynamic nature of local events (Odum 1992). Nevertheless, the variety of life at a site should retain a relatively high proportion of those species expected to be present. A number of ecologists (O'Neil et al. 1986) argue for process/functional as opposed to population/community types of ecosystems. The second column in Table 1 focuses on the frequency of dominant ecological processes—fire, wind hydrologic events—thought necessary to sustain an ecosystem and its components. The percentages listed in Table 1 surround the most common frequency of the dominant ecological processes at each spatial scale. Analytical tools needed to refine these estimates are available (Samson and Knopf 1994b), as are indices that capture the array of processes characteristic to a particular ecosystem (Norton and Ulanomicz 1992).

Research

So few suitable examples of the scientific method in management are in use (Knopf and Scott 1990, Murphy and Noon 1992, Hansen et al. 1993) that a comparative evaluation of their scope in terms of needed spatial and temporal scale required to manage ecosystems rarely is available (Murphy 1989). Nevertheless, recognizing that natural systems are hierarchically organized and they react to change is adequate to raise testable ecosystem management questions that in turn become the basis for land and resource management (Figure 1). There are major opportunities for scientists to become involved in the design and conduct of land management for the mutual benefit of science, resource managers and the resource (Hansen et al. 1993). A critical antecedent to the use of adaptive management is to invest in and develop a modest number of credible hypotheses that relate to management. We suggest diversity and ecological processes. They provide a basis to construct meaningful hypotheses, as well as provide answers to ecosystem management that resource managers can trust.

Management

In our view (and after Johansson 1993), as indicated in Figure 1, management in ecosystem management to conserve biological diversity requires 6 tasks:

- Identify strategic objectives. Narrow the focus in identifying objectives to the key biological and ecological elements of the ecosystems, i.e., narrow endemics and dominant ecological processes.
- Focus on what is essential ecological, social and economic data to accomplish strategic objectives.

- Organize around management processes—such as concept or product development, not functional disciplines.
- Preserve key expertise. Appoint a team of experts as the ''managers'' to achieve key strategic objectives and limit supervisory roles.
- Set specific perfonnance objectives and schedules to accomplish each process.
- Connect ideas of ecosystem health and an ''organism'' to the technology and history of the human organism.

Summary

More than 60 years ago, Aldo Leopold introduced his pioneering book *Game Management* (1933) as the art of producing sustainable populations of game. The art was to control those environmental factors which impact wildlife populations, with success dependent on the selection of the right factors, rather than on heavy investments of labor and materials. The design was to rely on the control of a few factors so that only an expert could distinguish the unmanaged from the managed landscape. Sustainability was the purposeful and continuing alignment of those few factors. We suggest those key elements—the selection of the right factors (narrow endemics and ecological processes), control of those within a narrow window similar to the natural landscape and and emphasis of sustainability—remain key to resolving today's issues in the conservation of natural resources.

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Proactive Strategies for Conserving Biological Diversity: An Unprecedented Opportunity in Alaska

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Introduction

Wildlife management in North America has broadened from its early focus on the conservation and management of a few game species. Today, our concerns and responsibilities encompass population genetics, many plant and animal species, community ecology, and ecosystem processes—the elements of biological diversity. As human populations continue to grow and place greater demands on the earth's finite resources, most ecologists foresee significant declines in global biological diversity (Wilson and Peter 1988, National Science Board 1989). The Environmental Protection Agency's Science Advisory Board identified the loss of biological diversity as one of the most significant risks to ecological systems and human welfare (U.S. Environmental Protection Agency [EPA] 1990). The primary causes of species extinctions include habitat loss, introduction of exotic species, over-exploitation by humans and environmental pollution (Ehrlich 1988, Reid and Miller 1989).

Much of the emphasis on conservation of biological diversity has focussed on the tropics (*see* Wilson and Peter 1988); however, conservation biologists and wildlife managers also are recognizing the necessity of addressing these issues in North America (Trauger and Hall 1992, Franklin 1993). The Endangered Species Act (ESA) of 1973, as amended, is the most significant law in the United States for protecting biological diversity. ESA, however, basically is reactive, its focus is on single-species, it always is expensive and it is commonly inefficient (Noss 1991, LaRoe 1993). The most cost-effective method for conserving biological diversity is an ecosystem-oriented approach (Office of Technology Assessment 1987, Franklin 1993). Although the National Environmental Policy Act and the National Forest Management Act are more ecosystem-oriented, their implementation requires baseline ecological data which often are lacking. Establishing habitat reserves sufficient to maintain all species within representative ecosystems, before any become threatened or endangered, can effectively reduce the often prohibitive costs of recovery management.

Due to a unique combination of environmental attributes and human history, Alaska provides an unprecendented opportunity for developing a proactive approach for conserving biological diversity. Our objectives in this paper are to identify some of the key attributes that endow Alaska with unique and formative conservation opportunities and to outline a framework for an Alaskan conservation strategy.

Alaska Conservation Opportunities and Challenges

The Setting

Alaska encompasses vast and varied landscapes extending from the world's largest old-growth temperate rainforest in south-coastal Alaska, through numerous mountain ranges, permanent ice fields and interior boreal forests, to the expansive tundra of the North Slope, the only Arctic ecosystem in the United States. Alaska is bordered by 39,000 miles (62,750 km) of marine shoreline, more than the rest of the United States combined. While species diversity is relatively low in Alaska compared to more southerly latitudes, Alaska's ecosystems contribute significantly to global biological diversity. For example, Alaska coastal waters are inhabited by 100 million seabirds representing 66 species, 32 species of marine mammals and some of the most productive marine fish stocks in the world. Alaska's diverse and abundant wetlands provide nesting habitat for over 10 million ducks, geese and swans representing 37 species. Nowhere else in the United States do large carnivores, including wolves (*Canis lupis*), grizzly bears (*Ursus arctos*), lynx (*Lynx canadensis*) and wolverines (*Gulo luscus*), remain integral and secure components of the ecosystem. These examples are but the tip of Alaska's biodiversity iceberg.

Unique Circumstances for Conservation

Habitat loss and fragmentation are considered the most significant threats to biological diversity (Wilcox and Murphy 1985). Today, rapidly growing human populations are the primary cause of habitat degradation. One fifth the size of the contiguous United States, Alaska's human density is one person per square mile (2.6/km^2). This is the lowest density in the nation, which has an average of 70 people per square mile (181/km^2). As a consequence of Alaska's remoteness, low-density population and recent development history, less than 5 percent of the landscape has been altered by agricultural, industrial or urban development.

Wetlands are at risk throughout the world. The contiguous United States, for example, has lost 53 percent of its original wetlands, including losses of 91 percent in California (Dahl 1990). In contrast, of the 170 million acres (68.8 million ha) of wetlands in Alaska, less than 1 percent have been lost (Dahl 1990). In the contiguous United States, only 20 percent of riparian vegetation remains in natural or semi-natural condition (Hunt 1988). We estimate conservatively that more than 95 percent of Alaska's riparian vegetation remains in a natural condition.

In general, public lands provide the greatest opportunities for conservation. Thirty-seven percent of the nation's lands are managed under federal jurisdiction (Crumpacker et al. 1988). Alaska, in comparison, is 88-percent public land (60 percent federal, 28 percent state). Conservation of biological diversity, however, is not a management priority for most public lands in the United States. For example, approximately 3 percent of lands in the contiguous United States are legally protected for their natural values (Scott et al. 1993). In contrast, approximately 40 percent of Alaska's lands are protected instate and federal habitat reserves (e.g., parks, preserves, refuges, wilderness, critical habitats, etc.).

Establishment of reserves, however, does not guarantee ecosystem integrity. Reserve size and connectivity are critical elements for maintaining viable populations and for sustaining ecosystem integrity (Wilcove et al. 1986). Most reserves in the contiguous states essentially are habitat ''islands'' in a ''sea'' of altered and fragmented habitat. The size of most habitat reserves in the contiguous U.S. may be inadequate for maintaining viable populations of large carnivores (Grumbine 1990). For example, Yellowstone National Park, at 2.2 million acres (0.9 million ha), is the largest national park in the lower 48 states. The State of Alaska, in comparison, encompasses 17 habitat reserves larger than Yellowstone. These vary in size from 2.3 to 19.6 million acres (0.9–7.9 million ha), and much of the surrounding lands have had little or no habitat alteration.

Unlike most states, Alaska still maintains viable populations of nearly all of its post-Pleistocene plants and animals. To our knowledge, only the Steller sea cow (*Hydrodamalis gigas*), spectacled cormorant (*Phalacrocorax perspicillatus*), wood bison (*Bison bison athabasacae*), musk ox (*Ovibos moschatus*) (now reintroduced) and possibly the Eskimo curlew (*Numenius borealis*) (last sighted in Alaska in 1886) have become extinct or extirpated from the state. One indicator of the ecological risk within a region is its number of threatened and endangered species. The U.S. Fish and Wildlife Service lists only seven threatened or endangered species in Alaska, the fewest of any state.

We suggest that Alaska's ecological integrity exceeds that of any state in the nation and is, in some respects, biologically comparable to the American West a century ago. Thus, the management options available to Alaska's resource managers are unparalleled in the nation. Armed with ecological theory (e.g., island biogeography, metapopulation dynamics, population viability) and high-tech management tools (e.g., molecular genetics, population modeling, remote sensing, geographic information systems), managers in Alaska are virtually starting with a ''clean slate'' and the tools necessary for maintaining the integrity of intact ecosystems.

Significant Challenges for Conservation

Like the western frontier and many developing countries, Alaska's economy is based largely on the extraction of natural resources. Unlike the western frontier, however, the technology for resource development is much more advanced, and the demands on natural resources are many-fold greater and significantly increasing. Thus, the accelerating pressures on Alaska's ecosystems are cause for concern. Specific conservation concerns include logging high-volume old-growth forests in southeast Alaska, logging riparian white spruce communities in the interior boreal forests, oil and gas exploration and development in the Arctic, mining near riparian and aquatic systems, overexploitation of the Bering Sea fishery, interactions between hatchery and wild-stock salmon, and marine contamination (e.g., the *Exxon Valdez* oil spill). The cumulative effects of these and other resource developments may have significant and long-term impacts on Alaska's ecosystems.

To address these concerns, we must develop a better understanding of how Alaska's ecosystems function. Since before statehood in 1959, most research in Alaska has focused on game species and commercial fisheries. For most, if not all, of Alaska's ecosystems, basic inventories are incomplete, and compared to vertebrates, significantly less is known about the diversity and ecology of Alaska's invertebrates. For example, University of Alaska entomologist, Mark Oswood (personal communication: 1994), estimated that less than 10 percent of Alaska's aquatic insect fauna has been inventoried.

Although Alaska's habitat reserves are among the most extensive in the world, it has not yet been determined if these areas adequately represent the state's biological diversity. On the Tongass Forest, for example, existing reserves protect ample rock, ice and scrub forest communities, but the highly productive lowland forest types are proportionately under-represented in reserves (USDA Forest Service 1991). A comprehensive evaluation of all existing reserves and the surrounding land matrix is of fundamental importance for designing a statewide conservation plan.

Framework for a Proactive Conservation Strategy

Conservation in Alaska is at a crossroad, confronted by both threats and opportunities. We have not yet developed a comprehensive statewide conservation strategy, however. Our objective here is to outline the key elements of a proactive strategy for conserving Alaska's biological diversity. The first step is to clearly recognize Alaska's potential for following in the footsteps of the lower 48 states. Much of the groundwork necessary for implementing this strategy already has been laid, and we acknowledge the important foundation that many scientists and managers have built.

Cooperation

Because of Alaska's immense size and the many resource agencies involved in land management, coordination among agencies often is difficult and fragmented. Ecosystems are not confined by jurisdictional boundaries, however. The key for building a proactive conservation program in Alaska is interagency cooperation and interdisciplinary coordination.

Identifying potential cooperators is an initial step in designing a conservation strategy. Potential cooperators include, but are not limited to: the Alaska Departments of Fish and Game, Natural Resources and Environmental Conservation, the U.S. Fish and Wildlife Service, Geological Survey, National Park Service, Bureau of Land Management, National Biological Survey, Forest Service, National Marine Fisheries Service, Soil Conservation Service, U.S. Army, U.S. Air Force, University of Alaska, and representatives of conservation organizations, Alaskan natives and Alaska's resource development industries. The Alaska Natural Heritage Program (AKNHP), at the University of Alaska Anchorage, serves an important function as a repository for Alaska's biological inventory and also could facilitate inter-agency coordination of inventory and monitoring procedures to enhance compatibility among data bases.

To further enhance cooperation in Alaska, we recommend developing an interagency memorandum of understanding (MOU) outlining a coordinated strategy. This should include defining terminology and setting identifiable conservation goals and objectives. Establishing a statewide technical task force to facilitate this effort would be beneficial. The Conservation of Arctic Flora and Fauna Initiative and Alaska Partners in Flight Program represent good prototypes for cooperation.

Conservation Management

Classification systems. A fundamental requirement for establishing a statewide biological inventory program is the development of ecosystem, vegetation and habitat classification systems. For planning and management at the landscape scale, we need classification systems which can be applied across agency boundaries.

Examples of classification systems include the National Hierarchical Framework of Ecological Units (ECOMAP 1993), National Wetlands Inventory (Cowardin et al. 1979), Ecoregions of Alaska (Gallant et al. in review) and Alaska Vegetation Classification (Viereck et al. 1992). Kessel (1979) developed a habitat classification system for Alaska birds which could be modified for other animal species.

Biological inventory and monitoring. Comprehensive regional inventories are necessary for developing a proactive conservation program. The USGS/EROS Alaska Field Office recently has digitized several Alaska inventories, including the ecoregions of Alaska (Gallant et al. in review), physiographic divisions, forest types, geology, soils and land-management jurisdictions. The vegetation classification of Viereck et al. (1992) could provide additional inventory layers as each of the five levels is mapped and digitized in Alaska. Distribution maps of selected vertebrate species are available from the Alaska Department of Fish and Game, the U.S. Fish and Wildlife Service, and the Alaska Natural Heritage Program. Additional inventory layers (e.g., anadromous fish streams, invertebrates, rare species, etc.) also will be useful when they become available.

To be useful for management, inventory data must be accessible for analysis. Geographic information systems (GIS) provide managers with the tools to solve complex spatial problems that are difficult or impossible to resolve with traditional techniques. Alaska's GIS capability is considerable. For example, at least seven state and federal agencies plus the University of Alaska have GIS capability, and interagency communication and data transfer is enhanced because most agencies use ARC/INFO software.

Regional monitoring objectives should be identified and prioritized at several levels (e.g., populations, habitats). Interagency and interdisciplinary coordination on biological monitoring can save limited agency resources and avoid unnecessary redundancy. This could be a coordinating role for an interagency technical committee.

Gap Analysis. Gap Analysis is a landscape-scale inventory of state, regional and national biological diversity (Scott et al. 1993). In Alaska, we have the opportunity for incorporating a landscape perspective into land management before significant fragmentation occurs. A comprehensive biological inventory and Gap Analysis are fundamental steps for assessing Alaska's reserve system and setting state conservation strategies. Using GIS technology and biological inventories, Gap Analysis identifies ecologically significant lands, determines where they are and what proportion are protected, and helps natural resource managers identify high-priority conservation needs. This comprehensive landscape approach, spearheaded by the national Biological Survey, has been completed in several states and initiated in 20 others. A Gap Analysis will be initiated in Alaska in 1995.

Now is the time for resource managers to assess Alaska's current reserve network, determine how adequately it represents regional biological diversity and evaluate connectivity between reserves. A Gap Analysis will provide a comprehensive evaluation of the efficacy of regional reserves for conserving biological diversity. Gap Analysis also offers a proactive tool to help guide management decisions affecting habitat on non-reserve lands. By using GIS technology and overlays of biological inventories, managers can evaluate different resource management alternatives and select those that have the least environmental costs. This would have both economic and environmental benefits.

Conservation Research

Research is our best long-term investment for expanding our understanding of ecosystem processes. Important research opportunities in Alaska include interdisciplinary baseline studies of unimpaired ecosystems; predator/prey relationships involving large carnivores; effects of insularity and habitat fragmentation on metapopulation dynamics and population viability; and the influence of fisheries on marine ecosystems.

Adaptive management has significant potential in Alaska for increasing the link between research and management, and incorporating an experimental design in to many of our land management actions. "Adaptive Resource Management" (Walters 1986) is a coordinated approach for expanding our understanding of ecological systems while conducting routine management programs. For example, the Alexander Archipelago in southeastern Alaska has thousands of islands, some of which are pristine, and others of which have been impacted by varying levels of clearcut logging. This setting provides a natural laboratory in which the effects of habitat loss and fragmentation can be evaluated in terms of their influence on biological diversity and ecosystem functions on individual islands.

Education

Public education is fundamental for building public and political support for conservation. In concert with traditional classroom education (e.g., Project WILD), Watchable Wildlife programs provide experiences that inspire awareness and understanding of wildlife and ecological relationships, and spark a personal commitment to conservation. Unless people understand what biological diversity is and how it is beneficial to their lives, our best conservation efforts surely will fail.

Conclusions

Although Alaska faces significant conservation challenges during the decades ahead, we have one of the best chances on earth for conserving our biological diversity. In Alaska, we still have the possibility of focusing on habitats and ecosystems *before* we face a conservation crisis. Alaska also offers unique opportunities for expanding our understanding of natural systems and for designing and testing new conservation strategies useful beyond our borders.

Key elements needed for formulating a proactive conservation strategy for Alaska include: (1) compatible ecological classification systems, (2) comprehensive biological inventories, (3) regional Gap Analyses to identify areas at risk and guide land management planning, (4) a coordinated research program that focuses on ecosystem processes, (5) public education, (6) an MOU encouraging interagency cooperation and coordination in conserving biological diversity, and (7) establishment of a technical task force to begin designing a proactive conservation strategy that evaluates the size, spacing and biological composition of existing reserves, the linkages among reserves, and management prescriptions in the matrix outside reserves.

An ecosystem perspective, long-term planning on a landscape scale and interagency cooperation are fundamental to proactive conservation. Alaska resource managers, political decision makers and the public must recognize the conservation of biological diversity, not as an economic impediment but as a community and economic asset. This will require balancing long-term economic and ecological sustainability with short-term economic gain.

The time has come to roll up our sleeves, join forces, and begin crafting a conservation strategy. Our opportunity for proactive conservation will never be greater than during this decade in Alaska.

Acknowledgments

D. Anderson, K. Boggs, D. Duffy, J. Hughes, R. Lipkin, S. Miller, R. Sinnottt, J. Seeb, N. Tankersley, K. Titus, J. VerHoef and J. Wright provided helpful comments on earlier drafts of the manuscript. M. Kirchhoff and J. Sherburne shared their ideas for an Alaska conservation strategy.

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Red-cockaded Woodpecker Recovery: An Ecological Approach to Managing Biological Diversity

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Introduction

Historically, the red-cockaded woodpecker (*Picoides borealis*) occupied a wide range throughout the pine belt of the southeastern United States. Its range extended from Missouri, Kentucky and Maryland southward to Florida and westward to eastern Texas. Due to losses of foraging and nesting habitat, the red-cockaded woodpecker (RCW) was listed as an endangered species in 1970. Since that time, suitable habitat continually has decreased and now the birds' range has been reduced primarily to public lands (mainly national forests) in the southern United States.

The RCW is a cooperative breeder and helper birds aid the breeding pair in rearing their offspring (Lennartz and Harlow 1979, Walters et al. 1988, 1992). The species is nonmigratory and individual groups (family units of one or more birds) maintain year-round territories around their cavity tree cluster and foraging habitat. Preferred RCW habitat consists of open parklike stands of mature pine with little or no midstory vegetation. The RCW is unique in that it is the only woodpecker which excavates cavities almost exclusively in living pines. There is a definite preference for older trees and trees infected with redheart fungus (*Phellinus pini*) for cavity excavation (Conner and Locke 1982, Hooper et al. 1991a, Rudolph and Conner 1991). Redheart fungus usually is not abundant in southern pines until the trees are 80 to 100 years old, but the fungus may infect pines as young as 40 years of age.

Developing the Recovery Strategy

When developing the recovery strategy, we analyzed five main topics including causes of RCW population declines, past RCW management strategies, other threatened, endangered or sensitive species occurring in similar habitats, natural disturbance processes of the southern pine ecosystems, and ecosystem function.

Primary Causes of RCW Population Declines

Although RCWs will use many pine forest types, they are most closely associated with the longleaf pine (*Pinus palustris*) forest. Historically, longleaf pine dominated between 60 and 80 million acres (24.3 and 32.4 million ha) of the coastal plain region. Today, less than 4 million acres (1.6 million ha) of the original longleaf pine type remain as second growth stands (Landers et al. 1989). Habitat loss, fragmentation and the associated demographic isolation are primary causes of RCW population declines (Lennartz et al. 1983, Conner and Rudolph 1991). Another major cause of population decline is the lack of suitable and potential cavity trees, and a high rate of cavity tree mortality (Ligon 1970, Lennartz et al. 1983, Conner et al. 1991).

The last major cause of RCW population decline is hardwood midstory development. The development of dense hardwood midstory is a major cause of cluster abandonment caused either by adverse changes in habitat conditions or increased cavity competition (Hovis and Labisky 1985, Conner and Rudolph 1989, Loeb 1993). The lack of hardwood midstory control in foraging habitat also can have impacts. In stands with tall, dense midstories, a significant portion of the female's preferred foraging substrate may be unavailable and in some cases avoided (R. N. Conner and D. C. Rudolph personal communication: 1993). The development of pine midstory may have similar affects.

Past RCW Management

With the passage of the Endangered Species Act in 1973, the USDA Forest Service (USFS) began implementing RCW management in an attempt to recover the species. As new knowledge of RCW ecology became available, USFS management direction was refined and intensified. Even with more intensive and protective management, most national forest RCW population continued to decline (Costa and Escano 1989). The only population to have a documented increase was on the Francis Marion National Forest (Hooper et al. 1991b).

Most of the past RCW management direction included prescribed burning during the winter months (dormant season), timber rotations of less than 80 years and extensive use of clearcutting for forest regeneration. Past RCW management direction failed to use a landscape-scale approach to management. Even the most protective management direction could lead to fragmentation and demographic isolation, with the most severe effects on the smaller, widely dispersed populations (Conner and Rudolph 1991).

Other Threatened, Endangered or Sensitive Species

The species included in this analysis are either known to occur or likely to occur in RCW habitats or microhabitats within RCW habitat, or species known not to occur but for which suitable habitat exists. Examples of the latter category include Florida panther (*Felis concolor coryi*) and red wolf (*Canus rufus*). There are 172 other species associated with RCW habitat for which there are viability concerns, including 18 endangered, 10 threatened and 144 sensitive species, of which 68 are candidates for listing under the Endangered Species Act. These species include 120 plants, 17 mammals, 7 birds, 13 reptiles, 6 amphibians, 9 insects and 1 arachnid. These species evolved with the red-cockaded woodpecker and their population declines likely were caused by the same factors: habitat loss, fragmentation, alteration of natural disturbance regimes and fire control.

Natural Disturbance Processes of Southern Pine Ecosystems

Disturbance affects structural and habitat diversity as well as overall species diversity (Hobbs and Huenneke 1992, Landres 1992). A "natural" community does not occur after a long period without large-scale disturbance; rather, several communities are "natural" for any site at any point in time (Sprugel 1991). Large- and small-scale disturbances resulting in even-aged regeneration are natural. Some recent examples of landscape-scale disturbances include the 1980 explosion and eruption of Mount St. Helen's, the 1988 Greater Yellowstone fires and the landfall of hurricane Hugo in 1989. Hurricanes have been a major disturbance force in the southeastern coastal plain forests (Hooper and McAdie 1994) and have had major impacts on red-cockaded woodpeckers (Engstrom and Evans 1990, Hooper et al. 1990).

Hooper and McAdie (1994) discuss the potential impacts of hurricanes on long-term RCW management and recovery. They state 11 of the 15 RCW recovery populations are vulnerable to hurricane damage. Repeated hits to the same areas could extirpate RCWs from these areas. Currently, at least two recovery areas are in the process of restoration from hurricane damage. Silvicultural practices can be used to reduce the damage caused by hurricanes to RCW habitat.

Of all the natural disturbance forces exerting change on the landscape, only fire has been readily controlled by man. The total extent of fire in the southeastern United States has decreased almost 95 percent in the past 50 years. Control of fire and other alterations of natural processes have influenced the structure, function and composition of most ecosystems (Samson 1992). Hobbs and Huenneke (1992) state suppression of fire in ecosystems dominated by fire-adapted species can severely disrupt ecosystem processes, which may have implication for the conservation of native, fire-tolerant species. The alteration of natural fire disturbance regimes in the Southeast, has affected plant communities as evidenced by more than 100 threatened, endangered or sensitive plant species occurring in RCW habitat that should benefit from prescribed burning. Recurring fires have been a long-standing evolutionary agent of habitat change to which native species are adapted (Christensen 1977). In the longleaf pine ecosystem, Landers et al. (1989) stated "Under natural conditions, frequent fires probably kept pine-grasslands and mesic hardwoods so widely separated that competition between the respective wildlife groups was minimal."

Frequency of disturbance has been and will continue to be a major influence on biological diversity. Again, fire is the only natural disturbance where man can significantly influence frequency of occurrence. Natural disturbance regimes help maintain stable communities. Human interference with natural disturbance regimes has affected biological diversity. To conserve biological diversity and achieve a biodiversity more similar to that which existed historically, active management, including fire, ecosystem restoration and disturbance regimes that closely resemble natural frequencies, will be required. Before active management is initiated, an understanding of ecosystem function and species interaction is essential (Walker 1992, Samson and Knopf 1982).

Ecosystem Function

Walker (1992) states that the best way to minimize species loss is to maintain the integrity of ecosystem function. Understanding the role of biodiversity in natural processes and how ecological processes shape patterns of diversity is central to sustainability of all ecological systems. Understanding natural processes is important to maintain and/or restore natural ecological systems and viability of species and communities associated with these systems (Samson 1992). Saunders et al. (1991) affirmed the importance of ecosystem function to individual species when they stated, "The question of whether management should be for individual species or whole ecosystems is largely irrelevant because individual species require functioning ecosystems to survive."

One of the objectives in maintaining ecosystem function should be to retain the complexity within the system. Terborgh (1988) suggests addition or deletion of species can have effects throughout the community. Loss of certain species can affect closely associated species sometimes leading to secondary extinction events (Wilcox and Murphy 1985). An example of this could be occurring with the gopher tortoise (*Gopherus polyphemus*) and the associated herptiles that utilize tortoise burrows. Habitat alteration, changes in natural fire regimes, predation and habitat fragmentation have led to the gopher tortoise being listed as threatened in parts of its range. As tortoise populations declined, local extirpations occurred and species associated with gopher tortoise burrows also have declined. Associated with gopher tortoise burrows are 11 animal species that currently are listed or are candidates for listing. None of these 11 species, or the gopher tortoise, are known to have increasing populations (U.S. Fish and Wildlife Service 1991). Declines in these species are directly related to habitat alteration and the disruption of species associations which affect ecosystem function.

Habitat fragmentation also has serious effects on ecosystem functions. Fragmentation was listed as one of the causes of population decline of gopher tortoise. Conner and Rudolph (1991) have disclosed the effect of fragmentation on the RCW. Jennersten (1988) analyzed the effects of fragmentation on plants and discovered significantly more seeds produced per flower in unfragmented habitat. He concluded that the difference in natural seed set between fragmented and unfragmented habitat can be explained by differences in pollinator services.

Some critical points to remember about functioning ecosystems are that ecosystems are constantly in a state of flux, disturbance is natural and necessary for survival of certain species and for maintaining environmental heterogeneity necessary for a wide variety of species, and species composition changes over time with succession (Landres 1992). Managing for functional ecosystems is essential for managing biological diversity (Hansen et al. 1991). Human influence on ecosystems has resulted in fragmented habitats, altered natural disturbance regimes, affected ecosystem function and changed species diversity. Maintaining population viability and conserving biodiversity will require active management and, in many cases, ecosystem restoration.

Key Elements of the Red-cockaded Woodpecker Recovery Strategy

Based on the analysis of the major areas of concern, seven keys to RCW recovery were identified. We believe five of these represent an ecological approach to recovery. The other two elements represent intensive management to reverse downward RCW population trends and alleviate problems associated with demographic isolation.

Ecological Elements of Recovery

Habitat management area designation. Margules et al. (1988) and Sanders et al. (1991), when discussing managing for biodiversity in an already fragmented system state, the first step in management must be the determination of the minimum subset of existing remnants required to represent the diversity of a given area. The habitat management area (HMA) delineation process is this first step. Habitat management area designation involves the delineation of an area that represents the desired future demographic configuration of an RCW population. It is a strategy for management at a landscape scale. The intent is to manage an area large enough to avoid or overcome the adverse effects of fragmentation and to reduce the risks involved with small populations and stochastic events. The minimum size HMA identified is 16,000 acres (6,477 ha). In many cases, entire national forests are identified as HMAs. Total acres involved in HMAs, including suitable and unsuitable RCW habitat, may exceed 3 million acres (1.2 million ha).

Management intensity levels. Four management intensity levels (MILs) were identified based on RCW population size. It is well established that small, widely dispersed populations are more susceptible to extirpation (Gilpin and Soule 1986, Goodman 1987). Conner and Rudolph (1991) have shown small RCW populations are more susceptible to habitat changes than larger populations. Based on this, RCW populations with less than 25 annually breeding pairs receive the most intensive RCW management, while being most restrictive in regard to the production of forest products. Populations with more than 250 annually breeding pairs are considered recovered, and would receive the least intensive RCW management and have the fewest restrictions on other resource management.

Midstory control. The adverse effects of midstory development have been previously discussed. The existing midstory conditions have developed because of changes in the natural fire regime. The USFS has controlled most wild fires occurring on national forest lands. Prescribed burning for RCW habitat improvement has been completed primarily in the vegetative dormant season and has had little effect on controlling hardwood midstory development. The dense hardwood shrub and midstory vegetation has impacted RCW and the fire-adapted plant communities of the Southeast, as evidenced by more than 100 plant species occurring in RCW habitats for which the USFS has viability concerns.

The RCW recovery strategy emphasizes prescribed burning for midstory control, with much of the burning occurring during the growing season, and implements a three- to five-year burning cycle. This closely mimics the natural five regime, and should result in improved habitat conditions and positive effects on the biological diversity of southern pine forests. Prescribed burning will not be effective initially because of the large size of much of the existing midstory vegetation. Initial treatments could include cutting with chainsaws, individual stem treatment with herbicides and mechanical equipment. After initial treatments to control vegetation, prescribed burning will be used to maintain the desired habitat conditions.

Longer timber rotations. Past RCW management relied on an 80-year rotation for longleaf pine and a 70-year rotation for other pine species. The RCW recovery strategy implements a 120-year rotation for longleaf and shortleaf pine (*Pinus echinata*) and a 100-year rotation for loblolly (*P. taeda*) and slash (*P. elliottii*) pines. These extended rotations are based on the RCW's preference for older trees and the rate of heartwood and heart rot development. Clark (1992) determined that on an average site it would take 70 years for loblolly pine and 90 years for longleaf pine to develop an adequate core of heartwood for RCW cavity excavation. Past timber rotations would allow the harvest of forest stands before they become suitable for RCW cavity excavation, forcing the woodpecker to rely on a recruitment stand strategy.

It is essential that these longer rotations be implemented and a balanced age class distribution achieved. The balanced age class distribution will allow a sustained flow of RCW habitat through time. This is critical because only a remnant of the original habitat exists today to recover the species.

Full range of vegetative management options. To maintain the open stand conditions RCW prefer and ensure a sustained flow of habitat through time, forest management must occur. All silvicultural methods must be available to properly manage RCW habitat. Habitat management will range from thinning and prescribed burning to forest regeneration, to perpetuate RCW habitat. Regeneration methods will range from clearcutting to single tree selection. Clearcutting will be used primarily for ecosystem restoration to restore the naturally occurring pine types in areas that have undergone forest type conversions. The most commonly used regeneration method will be the irregular shelterwood. With this method the residual trees are left in perpetuity. This will provide old trees scattered across the landscape. The amount of basal area retained on site varies by MIL, with the greatest number of residuals left in the smaller, more vulnerable populations. The purpose of this technique is to minimize the effects of fragmentation.

Intensive Management Elements of Recovery

The two intensive management elements of recovery are artificial cavities and translocations of young RCWs. These strategies will be used to reverse downward RCW population trends and to overcome the effects of past fragmentation that led to demographic isolation.

Artificial cavities. Artificial cavities will be used to increase the supply of cavities in active clusters and to stimulate colonization of unoccupied habitat (Copeyon 1990, Copeyon et al. 1991). They also have proven effective in stabilizing populations following cavity loss from natural causes (Watson et al. 1994, Conner and Rudolph 1994). Three types of artificial cavities will be used, including drill cavities, drilled cavity start-holes and cavity inserts.

Translocation of young RCWs. Translocation involves the moving of juvenile RCW from one location to another to create a potential breeding pair. In most cases, the appropriate sex juvenile RCW is moved to a single-bird group creating a potential breeding pair. A second type of translocation results in the establishment of new RCW groups by releasing a nonrelated juvenile male an female together in unoccupied habitat (Rudolph et al. 1992). Both methods of translocation have been successful, but they must be used in conjunction with artificial cavities and midstory control to be effective.

Summary

The USFS red-cockaded woodpecker recovery strategy is based on conservation biology principles and implements an ecological approach to recovery. It implements landscape-scale management by identifying habitat management areas. These HMAs represent the desired future RCW population configuration. Within HMAs, longer timber rotations will be established. Management intensity levels are established that restrict timber harvest levels and methods. These MILs are based on population size, with the smallest populations having the most restrictions and the most intensive direct habitat improvements for RCW. The combination of habitat management areas, longer timber rotation and management intensity levels should allow RCW populations to overcome the effects of past habitat fragmentation, and should preclude future fragmentation and demographic isolation.

The recovery strategy implements a prescribed burning regime on a three-to five-year cycle and includes growing season burning. This burning regime should closely mimic the natural fire regimes of southern pine ecosystems. This burning regime will not only maintain the open habitat conditions the RCW prefers, but should benefit numerous other species. Of 172 other threatened, endangered or sensitive species that could be affected by this strategy, 165 should benefit in some way from the burning regimes. The other seven species should not be adversely affected. These species include three dragonflies and four plants that grow in rock outcrops.

A full range of vegetative management techniques will be used, ranging from thinning, prescribed burning and single tree selection, to clearcutting. Clearcutting will be used primarily for restoring the naturally occurring pine type to areas that have undergone previous forest type conversions. The most commonly applied harvest technique will be the irregular shelterwood method. This method will provide for old trees scattered across the landscape. Managing ecosystems for a sustained flow of RCW habitat will result in the sustained yield of forest products. Managing the forest to maintain open conditions and regenerating habitat to meet future needs should result in the production of approximately 800 million board feet of timber annually. This sustained flow should allow for local economic stability.

Artificial cavities and translocations are intensive management techniques that will be used together. They will reverse downward population trends and help overcome problems of demographic isolation. These technologies are of critical importance because if we do not reverse the downward population trends in the smaller, more vulnerable populations, they will be extirpated.

Implementation of the RCW recovery strategy should benefit threatened, endangered or sensitive species, as well as overall biological diversity. Over time, habitat conditions should develop that would allow for a biological diversity more similar to that which existed historically.

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Establishing Riparian Habitat Linkages in the Channeled Scablands of Eastern Washington

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Our problems are man-made, therefore, they can be solved by man.
John F. Kennedy

"Channeled Scablands" is the name given to an area of eastern Washington where there have been geologic phenomena produced by two separate types of flooding. The first floods, occurring some 16 million years ago, were flows of molten lava. These flows created a vast region of 'flood basalts," burying eastern Washington and adjacent portions of Idaho and Oregon under thousands of feet of black basalt. This area now covers over 50,000 square miles (83,000 km^2) (Alwin 1984).

During the latter Pleistocene, sometime between 13,000 and 20,000 years ago, another series of floods scoured this landscape ... only this time it was water, not lava. This late Pleistocene period coincides with the Ice Age, a time of cool, wet climate when winter snowfalls did not melt completely as seasons changed. Eventually, large continental ice sheets covered much of Canada, spilling southward into the United States. In Washington, the Cordilleran Ice Sheet spread about as far south as the Columbia River. As the Ice Age ended, a large glacial lobe which extended into what is now northern Idaho and western Montana formed a dam across the canyon of the Clark Fork River, creating a giant lake geologists call Lake Missoula. This 2,000-foot (615 m) barrier held back more than 2,900 square miles (4,833 km^2) of surface water totalling over 500 cubic miles (833 km^3) in water volume (Allen et al. 1983, Bretz 1959).

As the Ice Age waned, this ice dam periodically burst, sending forth a giant wall of water calculated to exceed the combined flows of all the world's rivers by 20 times (Allen et al. 1983). The total number of floods is unknown, but is thought to be in excess of 20. The effect of these repeated raging torrents, many hundreds of feet deep, is certainly better known. These sculpted basalt landscapes are the "Channeled Scablands," and provide the basis for our work in eastern Washington.

Today these rock-rimmed floodways intertwine among thousands of acres of both dryland and irrigated cropland in the Columbia Basin, forming islands and corridors of the only remaining "natural" landscape to be found in this region. These channels contain both ephemeral and perennial stream courses, springs, meadows, pothole ponds, and small wetlands.

The ecosystem represented in this portion of the Columbia Basin is termed "shrub-steppe," after work by Daubenmire (1970) and others (Franklin et al. 1988). This ecosystem is characterized by a number of different sagebrush/bunchgrass plant communities as well as a variety of other smaller, though locally important, habitat types. For example, the riparian and wetland habitats have ecological significance in this arid landscape which far exceeds the percentage of their representation. Ponds, wetlands and stream courses make up only 1 percent or less of the area. Nearly every species of native wildlife depends on these habitats for some element of their life history. In excess of 80 percent of the vertebrate species in the channeled scablands rely on these areas (F. C. Dobler personal communication: 1992). The disproportionate importance of riparian habitats in an arid landscape also is reflected in similar sagebrush habitats in Oregon and other portions of the intermountain west (Maser et al. 1979).

The deep soil areas of the Columbia Basin were long ago recognized as having great potential and importance for agriculture. Massive changes to the steppe vegetation of the Northwest have been the result of years of cultivation, grazing and introduced exotic plants (Franklin et al. 1973). It is estimated that nearly 65 percent of the original 15,000,000 acre (6,000,000 ha) shrub-steppe landscape has disappeared since westward migration and settlement in the 1840s (Dobler 1990, Washington Department of Wildlife 1993). Much of the remaining natural habitat exists only in small fragments or narrow corridors. Many of these remaining fragments and habitat corridors actually are the basalt flood channels left from the Ice Age floods. They have little value for cultivated agriculture due to nonexistent or shallow, rocky soils. In the Columbia Basin shrub-steppe ecosystem, 26 vertebrate species have been identified by state or federal authorities as being of special concern due to habitat loss or fragmentation, and general population decline (Washington Department of Wildlife 1991).

Recognizing the value of these remaining habitats and the desirability of maintaining or reestablishing viable and sustainable plant and animal communities within the channeled scablands area, the Bureau of Land Management (BLM), in cooperation with other state and federal agencies, began a program of land exchange and acquisition in the late 1980s. These efforts were supported by the BLM's emphasis on riparian habitat management and a strategy developed through a habitat management plan developed in 1986 (BLM 1986 et rev.).

Originally, these efforts targeted private lands that contained valuable or potentially valuable riparian habitats which were adjacent to existing parcels of public lands, already administered by BLM. As the program gathered momentum, significant habitats were identified in areas outside the original planning zone. It became apparent that riparian, wetland and even upland habitats could be linked using land exchange as a tool to facilitate these landscape linkages and provide habitats of viable size and location within this fragmented region. At least, the concept seemed appropriate for the establishment of a habitat base for management and protection of the ecological integrity of riparian systems, and maintaining or enhancing these systems to provide for viable wildlife populations and plant communities. Small parcels of forested BLM lands in northeastern Washington were identified as surplus to public needs through a systematic inventory and evaluation process. Most of these parcels were "land-locked" by surrounding private ownership and had no public access. These scattered parcels were pooled to provide a "trading stock" supply of properties that could support acquisition of important riparian habitats in the channeled scablands. To date, over 25,000 acres have been acquired using this process. The program was further enhanced by using a third-party facilitator who specializes in multi-owner land exchanges. This addition to the process assisted by shortening time frames and simplifying the laborious agency process of conducting land exchanges, completing environmental analyses and complex land appraisals.

Not all of the acquired lands are in good ecological condition. Many of the parcels had been used for livestock grazing for over 100 years and are considered to be in poor ecological condition. Other authors (Evans 1989, Franklin et al. 1988) have noted the effects of long-term grazing on Columbia Basin riparian and shrub-steppe plant communities. Though in poor ecological condition from grazing and the effects of a prolonged six-year drought, most are considered to have some restoration potential that can be realized through proper management and enhancement.

Riparian, wetland and upland habitat enhancement and restoration efforts are underway, and will attempt to recreate the natural productivity of these habitats through seeding, planting, water control and intensive grazing management (including exclusion and rest). Thanks to the work of local conservation groups and the Washington congressional contingent, the Spokane District of BLM was allocated a special four-year wildlife budget element for this purpose (M. Weland personal communication: 1992). This ongoing project is identified as the Upper Crab Creek Management Area, after the main drainage system in the central channeled scablands. The goals of this project are to enhance, restore and manage both upland and riparian habitats to maintain and improve conditions for resident and migratory wildlife, long-term protection of plant and animal community integrity, and dispersed primitive recreation opportunities. Other uses may be allowed under intensive management.

Conduct of biological and physical investigations, identification of ecological problems and development of management objectives based on proposed solutions to these problems were really the easy parts of this project. Acquiring private lands in a rural county (population less than 10,000) with strong roots in agricultural development presented the BLM with some difficult social issues that surfaced almost immediately (Devitt personal communication: 1993) Initial attitudes toward the BLM plans to acquire native ranges and stream corridors with precious arid land water sources were driven by a fear of government control, loss of tax base and the lingering question of some hidden agenda (BLM 1990). From the onset, it was clear that a careful strategy needed to be developed to blend the needs of the community with the riparian and upland wildlife habitat enhancement goals of the BLM.

Taking these concepts to the field required a "ground-up" approach. BLM found that it needed to involve staff members in the fabric of the community in order to build the partnerships needed to make the project successful over time. The willingness of the District staff to spend countless hours with individuals, community leaders and committees began to reveal several opportunities. Local community leaders had several goals for economic diversification (A. Herdrick personal communication: 1993). Coupled with landowners who were willing to explore ideas for federal ownership and management of natural resources, this linkage began to identify a strategy to develop community pride in the area's natural history and heritage, and to use these values to develop locally initiated programs to use public land resources to diversify not only the economic base, but the quality of life and educational opportunities.

As the first successful land exchanges were completed, challenges and misconceptions continued to surface. Questions such as "Which government are we dealing with?" often arose, as missions of various state and federal agencies working in the county were confusing to local residents and county officials. A simultaneous effort was made to work closely with local government and community leaders to bridge the bureaucratic morass and garner local and regional political support. This community support, often given grudgingly, was extremely important as the BLM strived for program support in the Congressional appropriation arena.

Involvement of media was a necessary element of the program if the project goals were to be realized. As a tool for outreach and information, connections with area newspapers and publication provided a format for two-way communication that soon became invaluable. The energy spent on informing local journalists through every step of project development helped alleviate concerns of a ''hidden agenda''(Chrisman 1993). Through regular media contact, these information exchanges alerted BLM to potential community concerns before they developed into suspicions and rumors which could potentially cause irreversible damage to project implementation.

Planning agency strategy to adopt biological objectives for public land management in the face of skeptical and sometimes adversarial elements of a small community is a challenge that is becoming commonplace in the West . . . and will certainly continue to be so. Development of sound social, economic and political foundations which compliment biological resource objectives will be the normal mode of business as government activities are carefully scrutinized by a concerned public. A fully documented, ''no secrets' approach seemed to be the best practice in this case. Although differences of opinion will continue to plague these relationships, these differences now can be discussed in open forum, with mutual respect. Partnerships and personal trust, at least to some degree, have been established and maintained within the community between BLM staff and local residents (Walter 1993). Instead of being labeled as that ''blankety-blank government,'' the agency now is seen as a neighbor and a functioning part of the community.

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Insect Biological Weed Control: An Important and Underutilized Management Tool for Maintaining Native Plant Communities Threatened by Exotic Plant Introductions

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Introduction

The biota of North America is changing at an unprecedented rate due to human disruptions of natural systems coupled with deliberate and accidental introductions of foreign species (Mooney and Drake 1986). Over the last two centuries, at least 4,500 species of foreign origin have established free-living populations in the United States (U. S. Congress 1993). These include several thousand plant and insect species and several hundred vertebrate, mollusk, fish and plant pathogens. Although some non-indigenous species are beneficial, many have proven harmful, creating a growing economic and environmental burden for the country.

Invasions by high-profile species, such as zebra mussel (*Dreissena polymorpha*) in the Great Lakes, gypsy moth (*Lymantria dispar*) in northeastern and northwestern forests, and fire ants (*Solenopsis invicta*) in southeastern soils, have peaked regional and national interest. The success of these and other exotic species is attributed to their broad adaptability, high rates or reproduction, growth and dispersal, and the absence of natural control mechanisms, such as predators, parasites and diseases typically found in their native range. Many are capable of prolific growth and range expansion, which subsequently allows them to replace native species and alter the ecological integrity and biological diversity of entire ecosystems.

As we proceed into the 21st century, the potential for massive alterations of native flora and fauna resulting from introduced non-indigenous species is great. Unheralded, yet already prevalent, is the subtle invasion of foreign plant species into the natural areas of our country. As recently as the past two to three decades, land managers regarded invasive weeds as a relatively minor problem. Today, the problem has become so widespread that it can no longer be ignored.

Classic Examples

The Everglades National Park in southern Florida is an integral part of one of the largest contiguous complexes of preserved ecosystems in the eastern U. S. The park contains about 840 plant species, 26 percent (217) of which are not native to the area (Whiteaker and Doren 1989). Many of these aliens were brought in by early settlers. One such species, *Melaleuca quinquenervia,* is an ornamental tree introduced from Australia. Released from its complex of evolved natural enemies, *Melaleuca* is displacing sawgrass marshes, sloughs, forests and other natural habitats. The plant now occurs on over 450,000 acres in the region and is a serious threat to the integrity of all southern Florida's natural systems (*see* LaRosa et al. 1992).

Leafy spurge (*Euphorbia esula*), a deep rooted perennial from Europe, is found on rangelands and pasture and federal/state parklands throughout the northern Great Plains. In North Dakota alone, this species has spread from 220,000 acres in 1962 to 1.1 million acres in 1990 (*see* Bangsund and Leistritz 1991). Unpalatable to livestock and responsible for declines in native mixed prairie grasses and their associated fauna (Belcher and Wilson 1989), leafy spurge has proven difficult to control using conventional techniques, including herbicides. Projected economic losses in Montana, Wyoming, North Dakota and South Dakota are estimated to approach $144 million annually by 1995 (Bangsund and Leistritz 1991).

The Central Valley of California and the Intermountain West region of the northwestern U. S. are host to a number of exotic plant species. Continual disturbance of grasslands has occurred since appearance of the first European settlers. Much of the regional vegetation has been irreparably altered by alien plant species introduced and spread through agricultural practices and intensive grazing by cattle and sheep. The present outlook is for continual change in the vegetation as new immigrants displace the old by the same mechanisms that fostered earlier invasions: disturbance and transport (Mack 1988).

Purple loosestrife (*Lythrum salicaria*), an exotic wetland perennial, was introduced from Europe in the early 1800s. The plant is responsible for the degradation of many prime wetland habitats throughout the temperate regions of the U. S. and Canada (Thompson et al. 1987). Large, monotypic stands reduce the biotic diversity of wetlands by replacing native plants, thereby eliminating the natural foods and cover essential to many wetland wildlife species, including waterfowl. No effective method is available to control *L. saliacaria,* except where it occurs in small localized stands and can be intensively managed.

Conventional Control Techniques

Prevention of new introductions of non-indigenous species is the first line of defense. The need for a more restrictive national policy to accomplish this is widely acknowledged and will require future federal and state legislation and regulation (U. S. Congress 1993). However, future unintentional introductions are inevitable, as are illegal ones. Perfect screening, detection and control are technically impossible and will remain so for the foreseeable future (U. S. Congress 1993).

At the regional and local levels, prevention and eradication of newly arriving alien plant species can be accomplished by heightening public awareness, reducing the transport of seed or vegetative propagules into areas, decreasing disturbances that promote the spread of weeds, and selectively removing plants. In rangelands and wilderness areas this can involve requiring weed-free livestock feed, washing vehicles prior to entering sensitive areas, limiting human access and hand pulling or using herbicides to eradicate individual plants (Kummerow 1992). In agricultural systems, efforts to control invasive plant species usually are directed at simplifying the system and limiting the number of plant propagules to a tolerable level (Groves 1989). Standard practices of mowing, tilling and herbicide application often can accomplish the desired effect.

Maintaining or enhancing long-term biological diversity in natural systems infested with alien plant species creates a more complex situation. Here, an integrated approach is needed, using one or several control methods with an understanding of the dynamics of the ecosystem (Groves 1989). Mechanical or manual cultivation, or the manipulation of fire to benefit indigenous species, can sometimes be effective on lands with small infestations (Groves 1989). Short-term application of herbicides targeted at individual species also can produce beneficial results. However, mechanical and chemical control practices tend to simplify invaded systems, reducing diversification and increasing susceptibility to future invasions by other exotic plants (Groves 1989). Efforts to promote the growth of indigenous plants and suppression of invasive plants through plant competition can offset this, but the techniques are inadequately researched.

Many weed infestations occur in natural areas that are inaccessible to control equipment. In other instances conventional methods may be economically impractical and/or environmentally disruptive, or involve species that simply do not respond well to available control techniques. The limitations of conventional control practices, coupled with the rapid rates of colonization by many exotic plant species, have generated increasing interest in biological weed control (Story 1992).

Biological Weed Control

Classical biological weed control is the deliberate use of natural enemies such as insects, mites, nematodes and pathogens to reduce weed densities to tolerable levels (van den Bosch and Messenger 1973). In nature, biotic and abiotic factors determine the distribution of a plant species. In turn, plant distribution and abundance influence the population dynamics of specialized natural enemies. Ideally, these interactions provide a self-sustaining, balanced system. Insects commonly are used as control agents because of their high degree of host specialization (Story 1992). When matched successfully in a country with a troublesome invasive plant species, they can provide an effective, long-lasting, cost-effective and environmentally sound weed control program.

Incorporation of biological control into modern weed management has not been well endorsed or financially supported (Tauber and Baker 1988). Skepticism concerning the safety and effectiveness of exotic insect introductions remains prevalent among the general public, administrators and even scientists. However, the history of weed control using insects and the rigorous protocol required by the U. S. Department of Agriculture's Animal and Plant Health Inspection Service for screening foreign insects prior to their release into the U. S. makes such skepticism unwarranted (Malecki et al. 1993).

Three highly host-specific European insect species were introduced to North America in 1992 to control purple loosestrife. These were a root-mining weevil, *Hylobius transversovittatus,* which attacks the main storage tissue of the plant and two leaf-eating beetles, *Galerucella pusilla* and *G. calmariensis*. All three species overwintered and reproduced successfully at wetland sites across North America. In 1994, we plan to introduce two additional flower-feeding beetles, *Nanophyes brevis* and *N. marmoratus,* which severely reduce seed production.

Our strategy is to achieve long-term control of purple loosestrife through provision of a simple, yet diverse collection of natural enemies. Purple loosestrife now is a naturalized weed that will be a part of most North American wetlands forever. However, the introduction of this select group of insects should result in replacement of monotypic stands of loosestrife by native vegetation and an overall decrease in the occurrence of the plant. We predict a reduction of purple loosestrife abundance to approximately 10 percent of its current level over approximately 90 percent of its North American range.

A number of control programs targeted at other noxious, exotic weeds are in various stages of development and implementation. Notable are those for control of spotted and diffuse knapweed (*Centaurea maculosa* and *C. diffusa*), leafy spurge, musk thistle (*Cardus nutans*), Canada thistle (*Cirsium arvense*) and St. Johnswort (*Hypericum perforatum*) (Story 1992). Unfortunately, most of these programs are underfunded and consequently slow in being implemented. Due to the duration of a biological control program (i.e., often 10–20 years to be effective) and the involvement of international cooperators, as well as specialists from various disciplines, programs require a team effort to become economically and ecologically efficient. Efforts to control purple loosestrife were guided by an international advisory group throughout all phases of the program. This approach has proven successful (Malecki et al. 1993).

Conclusion

By itself, classic biological control may not be the sole answer for any given weed species, but when integrated with a sound control strategy, coupled with improved land management practices, it has the potential to make the difference between success and failure. If we continue to delay in recognizing the need for and support of new control initiatives, such as insect biological weed control, we could very well be jeopardizing the future integrity of natural systems as we recognize them today.

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Managing for Biodiversity in a "Special Interest" World

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Introduction

Driven by public demand and increased technical complexity of resource management, fish and wildlife agencies are working hard to provide quality hunting and fishing. Open to question is whether simply providing good hunting and fishing will be enough to satisfy the public in the future.

Among changes in public attitudes is a new and burgeoning interest in preserving "biodiversity." Among the general public, this can be equated to an appreciation for nature, and often is perceived as protecting "native" or pristine ecosystems.

In the past, public attention was focused almost solely on huntable and fishable species. Individual members of the public generally "specialized" in one species or group of closely related species. Organized groups have formed around promoting the management of certain species and this has shaped the backbone of wildlife and fisheries management for decades. Now, along comes public demand for "biodiversity." This also can be viewed as a special interest, because the proponents of the cause generally advocate it to the exclusion of other forms of management.

Concern for biodiversity in recent years has been fixated on places like tropical rain forests in Brazil or old growth forests inhabited by northern spotted owl (*Strix occidentalis caurina*) in the Pacific Northwest. In such locations, loss of species and habitat destruction provide big and visible images that television crews can document and environmental groups can champion against.

But it's not just in exotic and spectacular places that the conflicts between forms of resource management are waged.

In Texas, a long-term commitment to restore native ecosystems in a state park has taken a bizarre twist as efforts to eradicate invading cedar trees have brought park staff into direct conflict with the Endangered Species Act, which forces single-species management under penalty of law. The golden-cheeked warbler (*Dendroica chrysoparia*) may now, or in the future, utilize invading cedar trees, making the trees off limits to eradication. This has placed efforts to restore a native ecosystem on hold while the cedar "problem" grows out of control.

At another Texas state park, stocking of rainbow trout (*Oncorhynchus mykiss*), part of the state's highly popular winter "put-and-take" trout fishery, now is in jeopardy because it is possible, no matter how improbable, that a trout may eat an endangered Houston toad (*Bufo houstonensis*) adult or tadpole. Never mind that the trout are all caught or die shortly after stocking, and never mind that the small pond is artificial and not considered suitable habitat for toad reproduction, and never mind that the pond already is stocked with catfish, bass and other sunfish which all are voracious predators.

These are examples of how the special interest of managing for biodiversity or "multiple use" may come into conflict with management of a single species.

Biodiversity and Aquatic Resources

Concern over aquatic biodiversity is only now beginning to mount among anglers and the non-fishing public. Fish are a public resource (except for commercially cultured fish), which means everybody has a say in how recreational and commercial fisheries are managed and used, and in how aquatic habitats are managed. Some people are beginning to express concern over species of "nongame" fish, mussels, aquatic insects, salamanders and so on—all important to aquatic biodiversity (Williams and Neves 1992).

Emphasis is being placed squarely on managing entire "ecosystems" or "communities." There are benefits to such management, because a well-managed habitat will provide for healthy fisheries and wildlife. However, this is a major shift from enhancing sport fishing for the sake of recreation to restoring and managing for "native" communities.

Recreational fishing is better now than ever before. Fisheries are better managed and today's anglers are better equipped, better informed and better at catching fish than at any time in the past. But many of the old problems that complicate fishery management persist, such as pollution, wetlands loss, agriculture, mining, channelization and so on.

While much attention has been focused on creating better fishing, other aspects of fishery management have been underplayed. Now, with increased public and government attention on endangered species and managing for the benefit of entire ecosystems, fisheries managers are under pressure to broaden their outlook to more fully include nongame aquatic resources. This can, and probably will, have an effect on fishing.

Restoration of native fish communities could mean the elimination of some or all sport fishing in many waters. If taken to extremes, traditional fishery management agencies will have to change management philosophy to accomplish a goal of managing waters for native biodiversity.

Fishery managers realize that past practices may have contributed to loss of native fish, but managers also have been on the front line—often standing alone—in defense against the damages to aquatic habitat that have been the primary factors influencing biodiversity. Fishery managers also realize the difficulty in addressing the conflicting demands from the public and reversing the persistent legacy of environmental degradation of aquatic habitat.

Traditional Management Programs

Traditional management of lands and waters has not been effective in preventing the decline of many species of fish, especially in the face of the many threats to the health of natural aquatic habitats.

In the past, endangered species protection focused on a single species. While federal agencies and many proponents of the Endangered Species Act continue on this course, protection of individuals of a species has been in part the reason why the Act has been so unsuccessful in reversing the plight of far too many species. Strategies to preserve single species simply may not work to preserve biodiversity on larger scales.

Instead, protection of "endangered habitats" and ecological communities within which a rare species exists in what is important. Restoration and preservation of the health of biological communities and the habitat they occupy focus squarely on threats to biodiversity. Few waters in the lower 48 states remain unaffected. Threats generally are tied to economic activities:

- *physical habitat alterations,* such as filling of wetlands, construction of dams and reservoirs, channelization, siltation, and water diversion for agriculture flood control, and industrial use;
- *chemical changes,* such as pollution from agricultural runoff, municipal and industrial wastes, mine seepage, acid precipitation and global atmospheric changes;
- *introduction of nonindigenous species* of plants, animals and fish, and;
- *overharvest.*

Critics of traditional fishery management often point to hatcheries and stocking of nonindigenous fish as reasons for the decline in native fish. Other factors also play a prominent role, but there is no doubt that interactions with nonindigenous fish, whether stocked or introduced unintentionally, have taken a toll.

Western states have been especially dependent on nonindigenous species for providing fishing, because these states have few native sport fish that can stand intensive fishing (exceptions include salmonids on the west coast). Arizona is a state with successful fisheries composed almost entirely of introduced fish, such as striped bass and even Arctic grayling. However, Arizona, with 33 species of native fish, may have as many as nine candidates for listing, 18 federally endangered, 1 extinct and only 5 without threat (Cain 1993).

Some argue that the presence of spectacular sport fisheries created by stocking nonindigenous fish has overshadowed the plight of native nongame fish. Others point to the fact that state fishery management agency funding comes almost exclusively from anglers who want better fishing. They suggest that supporters of managing nongame fish should contribute dollars for management of these fish just as anglers have contributed dollars for managing game fish.

Fishing also involves people in an outdoor activity that brings them closer to the natural environment. The result should be greater support for environmental protection, because the hands-on experience of fishing is the best tool for learning about why healthy aquatic habitat is important.

Many anglers express a general appreciation for native fisheries and preservation of aquatic communities. They believe that agencies have an obligation to address native fish management because it is the "right" thing to do. In general, most fishery managers will agree that good fishing is the result of good fishery management, but good fishing is not necessarily the only reason to manage fisheries.

One of the factors too easily overlooked by those with an interest in protecting aquatic biodiversity is the fact that anglers have been the primary protectors of aquatic habitat for decades. Angler contributions in the form of license dollars and excise taxes on fishing equipment (Wallop-Breaux Program) have pumped hundreds of millions of dollars into protecting rivers and lakes from abuse, restoring fisheries and creating spectacular new fishing opportunities.

Federal Roles

Anglers make tremendous economic investments directly to state fishing and boating programs by buying fishing licenses and paying taxes on fishing equipment and motorboat fuels. This creates a direct connection between a state's anglers and state fishery management agencies. Working with anglers is simply a matter of being responsive to local citizens. Federal agencies are responsive to different forces.

Federal agencies take direction from Congress, are administered by appointees of the President and are funded by general tax revenue. Federal agencies also are subject to pressure from diverse citizens' groups at the national level, many of which are apathetic to the needs of anglers.

While federal agencies are rapidly developing a strong emphasis on managing for biodiversity, there is a growing realization that the traditional federal approaches to preserving biodiversity, such as creating refuges and parks, are not working. To address aquatic habitat protection, entire watersheds must be acquired, restored and properly managed. There is too little money available to buy enough land for preserves and parks to do the job.

In addition, removing land from private ownership and placing it into government hands provides no assurance that it will be managed to preserve biodiversity. The overall track record of the federal government at maintaining biodiversity is mixed.

For example, about a third of the U. S. is under federal management. These lands are subject to the full strength of federal laws designed to protect the environment, such as the Endangered Species Act, yet the bulk of endangered fish species are located in the Western states, the area with the greatest concentration of federal land. In total, about 70 percent of federally endangered and threatened fish are located on federal land.

Ecosystem Management is Complex

Among significant challenges in managing ecosystems for biodiversity is the enormous complexity of accomplishing it. In general, we don't know very much about managing ecosystems.

Even the definition of "biodiversity" has many shades of meaning that can cause confusion.

The term has become a buzzword, symbolizing the mantra of today's environmentalists and many politicians. The term often covers for an expression of poorly conceived concepts. The lack of clarity is not unexpected, however, because biodiversity is an exceedingly complex and far reaching topic.

Diversity of life forms and functions is fundamental to all living systems. This "diversity" exists at all levels of life from the very molecules that make up organisms, to the many classifications of organisms, such as bacteria, mold, algae, insects, trees, animals, fish and so on. Biodiversity also can be used to express the functional interactions of organisms within an ecosystem, but the definition of "ecosystem" is at least as open to interpretation as biodiversity, and this creates still more confusion. Ecosystem management seeks to make sense of the diversity of life and all the interactions between life forms, including humans.

Use of the term biodiversity now is so pervasive and has come to mean so many things to so many people, that it can mean almost whatever the user wants it to mean.

This philosophical befuddlement is not at all helpful to the fish and wildlife manager seeking to balance the needs of the public with the biological realities of aquatic and terrestrial resource management.

Many citizens and scientists who are demanding that agencies manage for biodiversity have little or no practical experience in resource management. Biologists who will be given the responsibility to do this so called "ecosystem management" are skeptical. The level of information needed and need for continued monitoring is great. Costs will be very high at a time when money is scarce.

In practice, most land and water resource managers are so swamped with day-to-day chores that they have little time to deal with complex theoretical science or vague concepts. Operational and maintenance dollars usually are tight and staffing is slim. Often the most pressing needs of resource managers are keeping the visiting public happy and seeing to general maintenance, such as ensuring vehicles will start and paperwork is kept in order. Resource managers usually are the last to get modern equipment and among the first to sustain budget cuts. When it comes to preserving biodiversity, they have what may be the most complex and important job in federal and state government. Yet, the resource manager gets all the blame when things go wrong, but has little input in the forums where decision makers conclude what is possible and what isn't.

Summary

Loss of biodiversity is a real threat to the integrity of aquatic and terrestrial systems. While attention always has been devoted to this problem, because it's professionally appropriate to do so, emphasis by constituency groups and possible litigation will force changes in what managers can and cannot do in the future.

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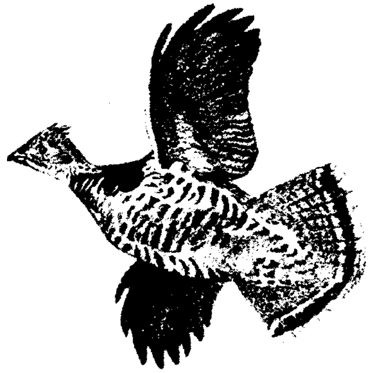

Special Session 6. *North American Gamebirds: Developing a Management and Research Agenda for the 21st Century*

Chair
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Introductory Remarks: Do We Need a National Upland Gamebird Plan?

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Wild galliforms hold a unique place in the historical development of modern wildlife science. Leopold (1933) drew heavily on galliform examples to illustrate relationships between vertebrate population abundance and variation in habitat and environmental conditions. For example, the index in *Game Management* lists more than 500 topical references to gallinaceous birds. During the past century, a rich collection of natural history literature on wild galliforms has been amassed by zoologists, wildlife managers and behavioral ecologists.

In certain cases where relatively long series of yearly population data were collected in relation to specific local conditions (e.g., Errington 1945, 1946), such information was used as a basis for expanding our theoretical understanding of vertebrate population dynamics. Elsewhere, empirical work on galliforms by early wildlife scientists such as Stoddard (1931), Leopold (1944) and Mosby (1949) contributed to the development of the habitat concept in ornithology and wildlife (Block and Brennan 1993).

Despite all the interesting things we have learned from the study and management of wild "chickens" over the years, there are some fundamental problems in the contemporary gamebird arena. I have a nagging suspicion that wildlife and natural resource professionals have been missing the mark with respect to operating as responsible stewards for these unique vertebrate resources.

Over the past half century, research and management for upland gamebirds has been characterized by a loose series of *ad hoc* studies and projects rooted largely in the classic natural history tradition started by Grinnell et al. (1918). While such studies have intrinsic value for increasing basic knowledge, they only chip away at the perimeter of how we understand mechanisms that influence population ecology and habitat relationships of these spectacular birds. Such studies also often fail to generate reliable information that can be used as a basis for informed management and stewardship.

Goal, Purpose and Scope

My goal for these introductory remarks is to point out why I think there is an array of unique challenges and opportunities in upland gamebird management and research as we move to the next millenium. The domain of modern wildlife science and conservation biology now extends far beyond traditional game species. This is good. Recent emphases on nongame resources, biodiversity, natural resource sociology, endangered species and other nontraditional wildlife issues are a positive move in the evolution of modern wildlife science.

Somehow, though, and for unknown reasons, it seems that relatively little attention is being paid to upland gamebird issues. This is especially disturbing when you consider that populations of once common species of upland galliforms have been declining in many places. For example, northern bobwhites (*Colinus virginianus*) are only about a third as abundant today as they were 30 years ago (Brennan 1991); scaled quail (*Callipepla squamata*) have undergone significant declines throughout much of their range (Brennan 1993a) and mountain quail (*Oreortyx pictus*) now are almost extinct in Idaho where they once occupied the entire southwestern quarter of the state (Brennan 1994). In most cases, however, we do not even know the current, broad-scale status of many upland gamebird species.

The purpose of this session will be to evaluate the current status of gamebird populations in North America and identify strategies for effective management under contemporary land-use regimes. The scope of the session will be to assess: (1) population trends of gallinaceous birds at regional and continental scales; (2) efficacy of current research and management efforts; and (3)to identify upland gamebird research, management and policy needs for the next decade and beyond.

I perceive a severe and critical need for a forum to address these issues. During July 1992, the Third National Quail Symposium held a National Strategic Planning Workshop for Quail Management and Research in the United States (Brennan 1993a, 1993b). This was the first time that such an activity was conducted. It received a great deal of attention from people in academia and resource management agencies. One result of these efforts was that many people requested that "the rest" of the upland gamebirds in North America receive similar attention. This session is a response to those requests.

Structure of the Session

The papers for this session were selected because they address at least two of the following criteria: (1) they have international implications with respect to gamebird issues in the context of modern wildlife science in North America; (2) they address broad-scale trends of multiple species; (3) they propose experimental designs that will serve to advance state-of-the-science management for upland gamebirds; or (4) they contain long-term (at least 10 or more years) data sets on population abundance.

The session will start with an assessment of North American gamebird research from a European perspective. Gamebird researchers in both the United Kingdom and continental Europe have embraced experimental field research and long-term monitoring to a greater degree than we have in North America. I believe North American gamebird researchers and managers, as well as the administrators and politicians that control research and management dollars, will benefit from the perspectives presented by Potts, Robertson and Lindén. Braun et al. provide an overview that contrasts how recent grouse management and research efforts in North America revolve around two extremes: (1) intensive efforts at threatened and endangered or popular species and (2) benign neglect for widely distributed species that attract only marginal attention from hunters. Droege and Sauer provide the first comprehensive overview of long-term trend data for forest and prairie grouse; Mossup applies a similar perspective to arctic grouse in northwestern Canada.

Carroll et al. outline what I think is a disgraceful lack of information on the status of Mexican quail. While disgraceful is a strong word, one must realize that Mexico contains the greatest diversity of New World quail, yet, we know less about this group than probably any other group of galliforms in North America.

The papers by Burger et al. and Leopold and Hurst provide philosophical and empirical advice about how we can design broad-scale and local area manipulative field experiments to test fundamental ideas (dogma?) about factors that regulate upland gamebird populations. Note that the themes of these papers draw heavily on and complement the themes presented earlier by Potts, Robertson and Lindén.

No contemporary session on gamebirds would be complete without an assessment of the role of non-governmental organizations (NGOs). We have seen membership in organizations like Quail Unlimited, the National Wild Turkey Federation, etc., skyrocket during the past 10–15 years. However, it remains to be seen if these organizations actually are having a positive impact on gamebird populations, research and management. Church et al. present the results of a survey that addresses how resource management agencies perceive NGOs and how NGOs perceive resource agencies. Understanding how these organizations relate to each other is an important first step in assessing the veracity of NGO programs. Finally, Gutiérrez will offer some closing remarks that will sum up and synthesize this session.

The North American Waterfowl Management Plan has been used to identify the major issues responsible for widespread waterfowl population declines. It also has served to identify strategic planning processes and joint ventures from the private and public sectors that can work toward a common goal of sustaining waterfowl populations. Can upland gamebird researchers and managers follow this lead? The success of the Accelerated Research Program for upland migratory webless gamebirds during the 1960s (Sanderson 1977) is one example that makes me think there is a positive answer to this question. The strategic planning document for quail (Brennan 1993a, 1993b) and the document developed by Sands and Smurthwaite (1992) are others. Or, are we so fragmented on the basis of state and provincial politics that a comparable effort for upland gamebirds would be a waste of time? These are key questions that we must try to answer in this session.

Acknowledgments

The proposal for this session was written while I served as a research scientist on the faculty in the Department of Wildlife and Fisheries at Mississippi State University, with support from the Mississippi Department of Wildlife, Fisheries and Parks. Many of the ideas presented here were developed while conducting research on mountain quail in California and Idaho, and bobwhite in the southeastern U.S. Teresa Pruden provided proofreading and editorial advice. Tall Timbers Research Station provided resources to further develop these ideas.

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Gamebird Research in North America and Europe: The Way Forward, a Critique and a Plea

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Introduction

Over the years, there have been many arguments over the pros and cons of pure versus applied conservation research amongst funding bodies. For far too long, we believe, theoretical or laboratory science has been valued high and hands-on conservation research valued low. It would not matter, of course, if conservation was not so urgently needed. As it is, we are faced with a steady stream of new evidence of declines in biodiversity: wildlife, including most species of gamebirds, adversely affected by developments of many kinds; developments that are driven by technological advances as well as by the cumulative aspirations of increasing human populations. When funds are diverted to conservation research it is all too often to deal with crises. Sometimes this leads to the bizarre situation where far more funds are appropriated for research on very local threatened sub-species than for equally pressed but very widespread species.

So, as gamebirds decline, is research on them adapting? Is the research on them proving useful in game conservation? This short paper addresses these questions by special reference to our experience in what many imagine must be a particularly applied field: gamebird research on farmed lands in North America and Europe. It might be expected that here we would find highly directed research producing more birds, after all, it is in the expectation of these birds that much of the research was, or could have been, funded. Instead, we found a situation in which most of the research is irrelevant to the conservation needs.

Methods

Our main sources of material for this paper are the published papers on the common (ring-necked) pheasant (*Phasianus colchicus*), the grey (gray) partridge (*Perdix perdix*) and the northern bobwhite (*Colinus virginianus*) from North America and Europe, from our own files and found in BIOSIS abstracts. In view of the vastness of the northern bobwhite literature—2,800 papers to 1984 (Scott 1985)—we included for this species only BIOSIS papers abstracted since 1984. For all three species, only papers with the species' name in the title were included.

We excluded papers in hunting magazines, those in publications with no external referees and those in our own organisation's in-house publications. We also excluded those which referred only to geographical distribution and any that were anecdotal or in the nature of short notes or reviews. The remaining 1,144 publications which we call "useful papers" were divided into four hierarchical categories as is done in the Research Planning Committee of our own organization:

1. Basic papers, descriptions, behavior, food, parasites, etc., with no attempt to explain causation.
2. As above, except that environmental factors or management techniques were related, however sketchily, to the population being studied; either through mortality or through population density changes.
3. Computer modelling from data to produce predictions, or which could have been (or in some cases were) used to design experiments, ranging from multiple regression equations through to real-time simulation models.
4. Field experiments, not necessarily replicated, but always with some kind of control. Only these publications can relate to turning research into birds, i.e., hands-on conservation.

There is, of course, a fifth category that rarely produces scientific publications; the actual use of tested research. That is why the research in our own organization is deliberately designed to proceed through categories 1–4 then finally to 5; the end use or "hands-on" conservation. We here, therefore, give considerable attention to the status of experimental research since it is crucial to the above progression; if the experiment fails, the research has to go back at least one step in the category procession.

All of our experimental research begins with a clearly defined problem to solve, the definition of it arising out of our long-term monitoring programs. We therefore examined the duration of population studies of the grey partridge, comparing the two continents over the past five decades, and relating the length of these studies to the time needed to capture the effects of important environmental changes.

Results

Table 1 sets out the origins of the "useful papers." More ring-necked pheasant research has been done in North America, and more grey partridge research in Europe.

The preponderance of recent papers was rather obvious throughout our review, but we were surprised that over half of all the "useful papers" which have accumulated so far have done so in the past 20 years (*see* Table 2).

The scarcity of papers in the "field experiments" (category 4) is perhaps the most striking feature about the "useful papers" (*see* Table 3). The even greater scarcity of category 3 papers is more easily understood since personal computers are a relatively recent research tool. It nonetheless is regrettable.

Even in the most recent decade, not more than 5 percent of papers come into category 4 and many of them originate in The Game Conservancy Trust. This organization has contributed 124 "useful papers" on gamebirds since 1987, covering several species in addition to pheasants and partridges (e.g., red grouse [*Lagopus lagopus scoticus*]). These papers were classified into the categories as follows 1—49; 40 percent, 2—36; 29 percent, 3—8; 6 percent and 4—31; 25 percent.

The duration of gamebird population studies, in general, is longer in Europe. For example, the duration of 27 population dynamics studies on the grey partridge in Europe averaged 12.15 ± 1.47 (standard error) years, whereas 11 studies in North America averaged 3.32 ± 0.62 years. We note, however, that some excellent long-term studies on North American gamebirds do occur (e.g., Roseberry and Klimstra 1984).

Table 1. Origin of "useful papers" analyzed in this paper.

Species	North America	Europe	Total
Ring-necked pheasant	351	126	477
Grey partridge	106	436	542
Northern bobwhite[a]	123	2	125
All species	580	564	1,144

[a]*See* "Methods," paragraph 1.

Discussion

Our review shows a clear and continuing preponderance of category 1 papers that, while advancing knowledge, do little for the practical management of gamebird populations. Given the rapidly increasing pressures faced by our gamebirds, we must provide a more coherent and directed program of research. Most Europeans are no less at fault in this regard than North Americans but, while the approach of The Game Conservancy Trust is open to criticism, it is worth outlining this organization's own response to these problems.

We consider research to be aimed at providing practical management solutions. It typically progresses through four stages (Potts and Aebischer 1991). First, the basic quantification of the animal's lifestyle. Second, long-term monitoring of population changes. Third, analyses and model construction to produce testable hypotheses as to the causes of population change and the factors determining equilibrium levels and variations from these levels. Fourth, experimental testing of these hypotheses in the field. This progression broadly corresponds to the classification we used to categorize the gamebird literature, it is different in one crucial respect, its employment of simulation modelling from the normal progression of research in the "declining population paradigm" (Caughley 1994). Our approach has led to significant increases in waterfowl (through fish exclusion), in red grouse (through use of medication to control parasites) and in partridges (through predator control and conservation of insect food). In each case, the management was based on experimental studies.

Table 2. Decadal production and accumulating number of "useful papers" on grey partridge and ring-necked pheasant.

Decade ending	Production per decade	Accumulating
1900	1	18
1910	5	23
1920	14	37
1930	5	42
1940	31	73
1950	64	137
1960	128	265
1970	172	437
1980	164	601
1990	331	932
2000	?	1,019 (to 1993)

Table 3. Classification of published "useful papers" according to category of research (*see* "Methods" for details of category classification).

	1	2	3	4	Total
Grey partridge	340	152	11	35	538
Ring-necked pheasant	335	117	12	17	481
Northern bobwhite (since 1984)	106	10	2	7	125
Totals	781	279	25	59	1,144
Percentage	68	24	2	5	100

How Could Research in North America Be More Usefully Directed?

Long-term monitoring. One feature of North American research, particularly on the ring-necked pheasant, has been the role of state fish and wildlife or natural resources departments in long-term monitoring. These bodies typically have a legislative requirement to monitor annual changes in game abundance, on which they then base season lengths and bag limits. They have the mechanisms for getting funds from hunting license sales and excise taxes on sporting arms and ammunition. However, to determine whether the abundance of a species has changed from the previous year requires far simpler data than those needed to determine why any such changes may have occurred. For instance, many states conduct roadside counts after crop harvest to estimate the number of pheasant hens and broods observed per mile. These data are adequate for the purpose of setting season lengths, but fall far short of those required for any serious modelling exercise or for the construction of robust hypotheses for later experimental verification. There is insufficient information to calculate more than indices of chick survival or hen success. Data such as these contain no information on density and, as a consequence, when repeated at different times of year it is impossible to obtain reliable figures on within-year changes in actual bird numbers.

Most studies containing sufficient detail for analysis of the mechanisms of population regulation have been MS or Ph D theses. By their nature they rarely continue for more than four years showing a lack of planning on the part of supervisors, institutes or funding authorities. It is worth noting that recent analyses suggest that 20 years of continuous data are required to assess accurately density dependence in insects (Woiwod and Hanski 1992), though, of course, it will be longer still for larger animals, which reproduce less frequently. In The Game Conservancy Trust's Sussex study of the grey partridge, which began in 1954, the critical change in chick survival rates that precipitated the long-term decline of the partridge through the 1970s and 1980s occurred prior to 1962. Yet, the population decline it produced only became evident in the late 1970s, suggesting again that at least 20 years were necessary to discover the cause of the trends. Most gamebird research is, from the figures on partridges given earlier, far too short even in Europe to throw light on the causes of long-term trends. In North American state wildlife agencies a better and more stable political atmosphere urgently is needed so that fund allocations are less subject to short-term expediencies.

Given the apparent "secure" long-term funding of state wildlife agencies and their staffs, they should be in an ideal position to accumulate the detailed, continuous data sets necessary for even the most basic analyses of population change. However, this role rarely is realized and the majority continue to collect unreliable and uninformative data sets based simply on an annual roadside count or indices of hunter success. This is an opportunity sadly missed.

Consequences of a failure to test hypotheses. A hypothesis that is widely accepted but based on the unsupported interpretation of data can best be described as dogma. The failure to test hypotheses is leading to the wasteful misdirection of funds for management. Europeans and North Americans are at fault equally in this regard, and many widely held views still are unsupported by experimental verification and dictated more by fashion than by any logical approach.

It is interesting to compare the ideas held in Europe with those in North America. European gamekeepers, for example, would claim that producing high-density game populations always requires predator control, whereas the North American view often is that predator control is undesirable or inappropriate given the system of land ownership and hunting rights, but there are no experiments to settle the matter. The role of nesting cover as the major limiting factor for ring-necked pheasants in North America is another example. Despite almost half the North American pheasant literature and a high proportion of the funds available for management being devoted to the subject of nesting or nesting cover management, there still is no experimental evidence that nesting cover is a key factor determining population equilibrium levels in ring-necked pheasants.

One requirement for experimentation is, of course, access to large areas of land. It seems to us that in this regard, North Americans are far better placed than Europeans because of the vast areas of land in public or federal ownership. After a feasibility study of several areas, The Game Conservancy Trust's large-scale experiment testing the effects of predator control on partridges finally was carried out on land owned by the U.K. Ministry of Defense (Tapper et al. 1991); the most suitable site available to us.

Conclusion

We highlight two shortcomings in the current approach to game management. First, a lack of the detailed long-term data sets necessary to fuel the analysis of population data and formulate testable hypotheses. Second, a lack of hypothesis testing by experiments in the field. We see these two necessarily interlinked approaches as central to a coherent research program to provide practical, tested management recommendations and to avoid a dogmatic approach to management issues. In North America, given the relatively "secure" long-term funding of many state agencies and the huge areas of land available for experimentation, the framework already is in place to move much of game research onto a higher plane. The need is urgent, as we all know.

Acknowledgment

Our grateful thanks to the American Friends of the Game Conservancy for a travel grant enabling the senior author to present this paper in Anchorage.

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North American Grouse Research and Management: A Finnish Perspective

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Introduction

Hunting usually is justified by arguing that humans are acting as natural predators. However, a natural predator lives in close interaction with its environment and is able to balance its feeding behavior with food availability. On the other hand, modern people usually lack natural and continuous contact with game animals and their living conditions. That is why we need high-quality research and management to maintain game populations at vital levels.

In both North America and Europe, particularly in Finland, declining trends in many grouse populations are prevailing (e.g., Lindén and Rajala 1981, Sauer and Droege 1994). It may be of interest to investigate and compare the general trends, differences and similarities in grouse research between these two centers of grouse interest because declining populations of game species require an extensive research strategy.

Methods

To familiarize myself with North American grouse literature, I reviewed articles in Wildlife Abstracts from the period 1971–93. In my opinion, this period represents extremely well the times of modern ecology. Thus, there was no obvious need to include older articles in my investigation. I classified all articles published in periodicals, therefore excluding abstracts, M.S. theses and Ph.D. dissertations. I divided the articles into different research areas, which required subjective reasoning at times. Usually, I placed one article with one specific theme, but there were many papers which encompassed multiple themes. The areas defined were: (1) status and distribution, (2) general biology, (3) taxonomy and genetics, (4) population monitoring, (5) population dynamics and ecology, (6) habitats and landscape ecology, (7) hunting studies, (8) energetics, (9) methodological studies, (10) veterinary medicine, (11) food and nutrition, (12) physiology, (13) behavioral ecology and ethology, and (14) predation on grouse. Altogether, 777 articles were included from the 23-year period.

From this survey, I was able to draw general conclusions about the direction and emphasis of North American grouse management. In addition, it allowed me to contrast European, particularly Finnish, grouse management with North American research and management.

Results

Volume of Grouse Research in North America

The average yearly production of grouse articles in North America was more than 30. During the study period the total production increased slightly. The annual number of articles was approximately 25 in 1970–74 and more than 45 in the late 1980s. In my opinion, these numbers showed that grouse research in North America was not a hot topic, and that wildlife research probably was more concentrated in other taxonomic groups.

My critique on the relatively small amount of research effort in North America is based on a comparison with my country, Finland, an extremely small nation with 5 million inhabitants. It is very true that Finns may exaggerate their contributions to grouse research, but using Wildlife Abstracts and the same criteria as above, Finns have produced about 20 percent of the total production of North America.

There also is one additional alarming feature I detected in my review: in the 1970s, the proportion of articles published in international peer-reviewed journals with high scientific standards was more than 50 percent, but in the 1980s, this proportion decreased to under 40 percent. Of course, these subjective indices are only indicative, and it is easy to make totally false interpretations, but nonetheless, the trend is alarming. However, during the last three years, the proportion again has been relatively high, 52 percent. It appears that a debate concerning publication forum is progressing in North America (Bart and Anderson 1981, Capen 1982, Finch et al. 1982, Scott and Ralph 1988). I add that the importance of publication in international peer-reviewed journals is a standard to which all scientists should ascribe.

A Different Approach: The Finnish Grouse Management Strategy

I will use Finland as an example of an area where very high hunting pressure of grouse potentially can cause problems. Grouse belong to the most desired game, but their numbers have declined seriously, mostly due to habitat deterioration and forest fragmentation (e.g., *see* Helle et al. in press). Hunting of decreasing populations requires careful planning of harvest, and it is an obvious disgrace that only a few preliminary studies on grouse shooting have been done in Finland (for a review, *see* Lindén 1991). Nevertheless, even if there is a gap in Finland on the amount of hunting research, the basic grouse management strategy is interesting and worthy of examination. This is particularly true because many hunted grouse populations in North America also are declining (*see* the Sauer et al. paper in this session).

Monitoring and predicting grouse abundance. Finns have monitored their grouse populations since the early 1960s. In 1988, a revised census program was started as a joint project between the Finnish Game and Fisheries Research Institute and hunters. In the new scheme, there are about 1,500 routes in the form of equilateral triangles (each triangle side 2.5 miles [4 km]). The triangles are distributed evenly across the country (Figure 1). The 7.5 mile (12 km) routes are censused twice a year: in August and in February/March. In August the census group consists of three men walking in a line 66 feet (20 m) apart, thus covering a census belt of 197 feet (60 m) (for details of method, *see* Rajala 1974). Censuses reveal both the abundance and the reproductive success of our four forest grouse species. In winter, mammalian snow tracks crossing triangle sides are counted. Lindén et al. (in press) present a detailed description of the program.

The map of black grouse (*Tetrao tetrix*) distribution in Finland gives a good example of the information derived from census triangles (Figure 1). Up-to-date distribution and densities are estimated in 964-square mile (2,500 km^2) grids. The annual fluctuations in

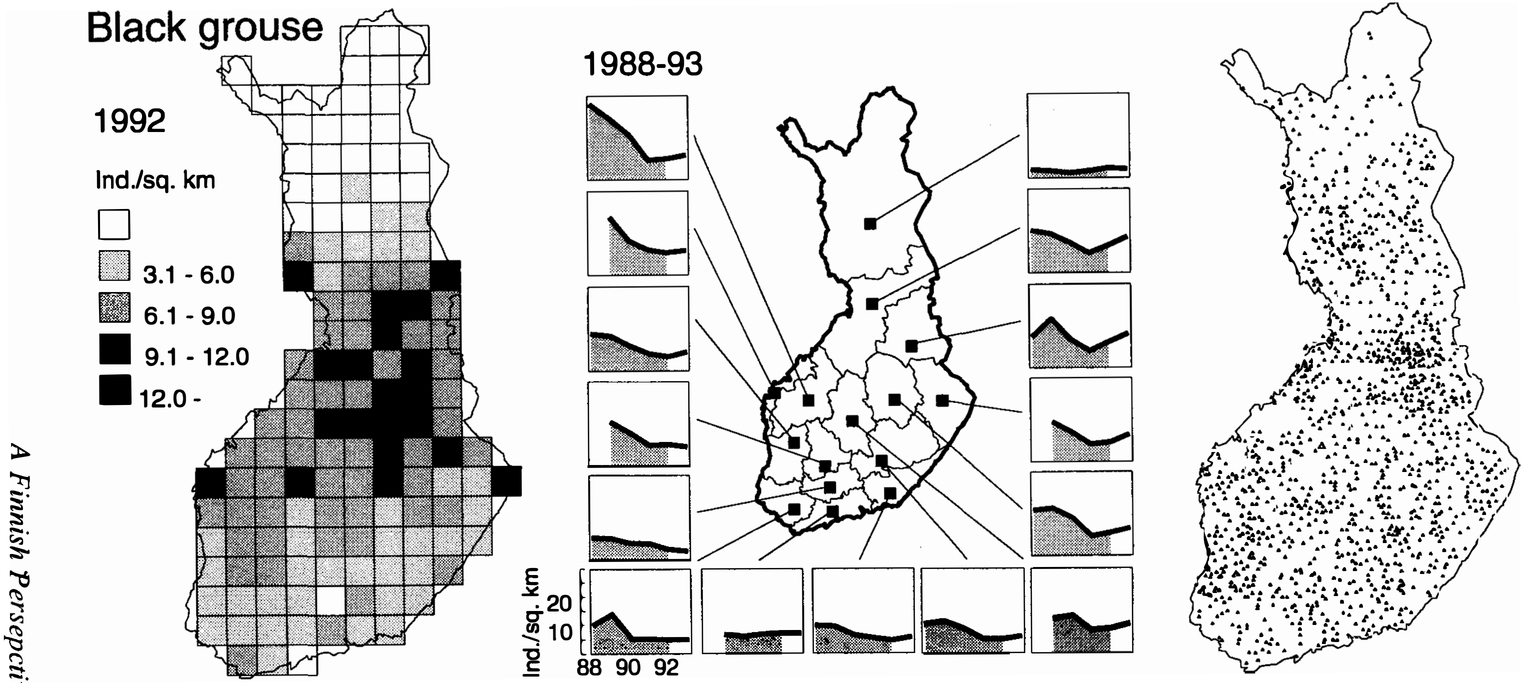


Figure 1. Density of black grouse in Finland estimated according to the triangle census within (A) 964-square mile (2,500 km²) grids in August 1992; (B) black grouse densities in 1988–92 together with the prognosis for 1993 in different provinces; and (C) distribution of wildlife triangles.

numbers are estimated by provinces (diagrams in Figure 1). In addition, it is relatively easy to predict the population density of the next year; the prediction error is, on the average, 20 percent, which is not significant for management applications (Lindén et al. 1990). The predictions are given to hunting administrators and to hunters themselves during the previous spring to ensure enough time for harvest planning.

Hunting recommendations and hunting statistics. Wildlife triangle censuses yield nearly absolute grouse densities, which can be used in planning the hunting recommendations for different parts of the country. The Finnish Game and Fisheries Research Institute informs hunters via newspapers and periodicals of the yearly status of grouse populations. For game management administrators and hunting clubs, the Research Institute sends detailed data for local populations, as well as recommendations for hunting bags in each area. Recommendations for hunting bags vary between 2–12 percent of the total August population, depending on the phase of the population cycle. Finnish grouse populations clearly are cyclic (Lindén 1989), and the reproductive parameters, as well as the tolerance for hunting, vary by phase of the cycle.

The Research Institute also keeps continuous statistics on the grouse kill in different provinces. By comparing the trends and ratios between population densities and yearly kills it is possible to establish principals of prudent hunting.

Integrated studies. In wildlife triangle censuses, each observation is located accurately on a map, which will allow spatially explicit habitat and landscape ecological studies using habitat variables adopted in Finnish forest inventories. Although our understanding of grouse habitat relationships is rudimentary, the wildlife triangle scheme, possibly linked with remote sensing of forests, will create the opportunity to develop GIS-based techniques to answer relevant ecological and management questions.

While monitoring and maintaining harvest statistics serves the utilitarian purpose of hunting administration, these activities generate an enormous amount of data. These data can be used for basic science investigations. For example, wildlife triangles give indispensable data for analysis of population dynamics.

Other studies improve our understanding of population dynamics. The long-term skull collection of capercaillie (*Tetrao urogallus*), in particular, allows estimation of adult age structure and its consequences on reproduction in different phases of the cycle (e.g., Lindén 1989). Detailed information on breeding biology is collected using nest cards. Still, I dare to mention the studies on growth and energetics of the capercaillie (Lindén 1988, Milonoff et al. 1993), which illuminated the complex problems with the evolution of sexual size dimorphism of many grouse species (*see also* Wiley 1974, Stamps 1990).

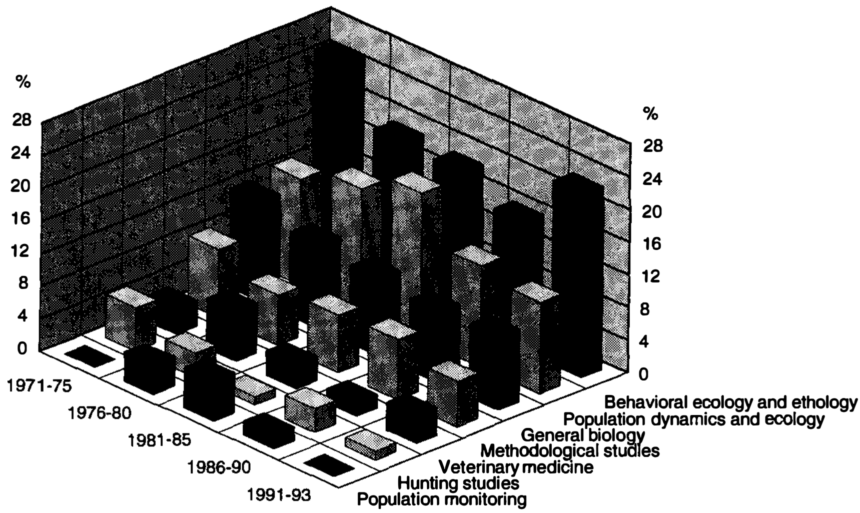
I underline the importance of integrated studies in grouse and wildlife management to achieve a complete picture of the living conditions of a species. Because we are managing and manipulating both the populations and their habitats, we also are responsible for potential threats caused to other species, or even to the species in question during a different phase of its annual cycle.

Patterns in North American Grouse Research

Grouse research in 1971–1993. In general, the variety in North American grouse research is its striking feature; the nine grouse species are treated fairly and all the fundamental aspects of grouse biology are included. When checking the chronological distribution of proportions in different research fields more thoroughly (Figure 2), it is possible to make some further observations.

Habitat studies always have been in a central position, but their number seems to be increasing during the last 10 years, perhaps the rise of landscape ecology and modern computers influenced this trend. In accordance with other fields in ecology, the importance of behavioral ecology in grouse research is high.

There are, however, some research fields with astonishingly modest research effort, at least as measured by the number of articles. Monitoring of grouse populations, studies dealing with optimal harvesting, or studies of hunting effects on populations



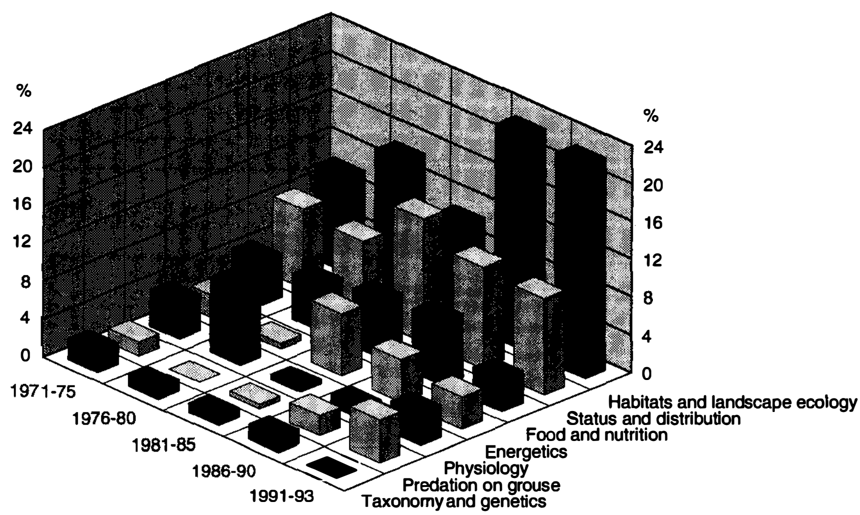


Figure 2. The proportions of different research areas in five-year periods in North American grouse literature during 1971–93, summarized from Wildlife Abstracts.

all are rare. This seems to be a worldwide phenomenon, which does not bring honor to game research in general and which does not provide any firm basis for the concept of "wise use" (*see* e.g., Dhondt 1991). Still more surprising is the nearly total absence of predation studies, which in Europe are at the core of interest among grouse researchers. Perhaps in North America predation problems are seen in wider contexts (e.g., Keith et al. 1977).

Nutritional studies such as physiology, food selection and energetics are well represented. Veterinary sciences are represented relevantly.

Declining populations require integrated studies. Hunting practices are very different between North America and Europe. For instance, in northern Finland, approximately half of the active male population hunt and most of them consider grouse as the most desired game. In addition, the vast majority of all forest areas is available for hunting. This great demand for grouse shooting causes anxiety among managers because the possibility of overharvesting exists. In addition, grouse populations generally have been decreasing (Lindén and Rajala 1981) as they have been in many North American populations (Sauer and Droege 1994).

Hunting legislation in North America also is quite different, which is reflected in the research priorities. There does not appear to be any authority responsible for the welfare of a non-migratory species over its whole range of distribution. Local management efforts concentrate too often on problems of that place, for example, improving local habitat conditions while forgetting to manage the basic reasons of the decline. It seems to be true, at least in the light of those 777 articles, that there would be a need to integrate grouse studies in order to find suitable broad-scale management techniques and strategies. This does not necessarily always mean the highest quality science, but high-quality science does foster the development of good game management.

My critique is rather straightforward, but it would be a sin not to mention some scientists who have through their personal effort benefited grouse research worldwide. For instance, Jim Bendell, Clait Braun, the late Gordon Gullion and Fred Zwickel always have had a very intimate grasp of their study objects, and Susan Hannon and her team has been an asset to behavioral ecology.

Grouse have been objects of appreciable scientific thought in North America. However, in management questions, local thinking seems to prevail. In my opinion, there is an immediate need for a large-scale monitoring program and for several experimental hunting studies, which would serve as a basis for planning a species-specific conservation scheme for the total range of the species. At the first phase, monitoring does not need to be high-quality science; the most important task is to identify the key problems. After problem identification you have to use integrated studies to reveal the different aspects for possible solution, as well as their effects on other parts of the ecosystem. With these kinds of programs, it is possible to wake up the interest of the general public. Afterwards it is relatively easy to state the reasons for habitat management practices and hunting regulations.

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North American Grouse: Issues and Strategies for the 21st Century

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Introduction

The 10 species of grouse (subfamily Tetraoninae) in North America occupy habitat types that vary from prairie, coniferous and deciduous woodlands, to alpine and arctic tundra. Prior to European settlement, grouse occurred in 47 of 49 continental states and all Canadian provinces (Aldrich 1963, Johnsgard 1973). While some species of grouse have been extirpated from many states and several provinces (especially prairie grouse species), grouse still occur in 46 of 49 continental states and all Canadian provinces.

Grouse have been hunted historically and, because of unique breeding displays, many species have been avidly pursued by photographers and bird watchers. The popularity of grouse has resulted in considerable attention by wildlife managers and researchers. Interest in hunting and conserving grouse was the impetus for the formation of one national private organization (The Ruffed Grouse Society) and several state organizations.

In this paper we give a brief review of the current status of grouse, identify issues that will confront managers in the future and identify strategies to resolve management dilemmas. We will argue for accelerating the evolution of the grouse management paradigm because we now are in serious danger of losing populations, subspecies and even species of grouse.

Current Status of Grouse

Prairie Grouse

This group includes the prairie-chickens/sharp-tailed grouse (*Tympanuchus cupido, T. pallidicinctus, T. phasianellus*) and sage grouse (*Centrocercus urophasianus*). Distribution and status of greater prairie-chickens and sharp-tailed grouse initially were enhanced (Aldrich 1963, Hamerstrom and Hamerstrom 1963) by settlement, probably because cropland created dependable sources of winter food (Schroeder and Robb 1993). Because of intensification of land use, greater prairie-chickens were extirpated from most of their acquired and presumed core ranges (Alberta, Ontario, Manitoba, Saskatchewan, Iowa, Indiana, Ohio, Kentucky, Tennessee and Arkansas). One race now is extinct (heath hen) and another is federally listed as endangered (Attwater's). Populations are precarious in Michigan, Illinois, Minnesota, Wisconsin, North Dakota, Colorado, Missouri and Texas and secure enough to permit hunting in only four states (South Dakota, Nebraska, Oklahoma and Kansas).

Lesser prairie-chickens historically occupied suitable sand sagebrush (*Artemisia filifolia*) and shinnery oak (*Quercus havardii*) rangelands of the southern Great Plains (Aldrich 1963). While they still occur in all five states of their presumed original range, their distribution has decreased by 92 percent since the 1800s because of habitat loss and deterioration (Taylor and Guthery 1980), and numbers have decreased by an estimated 97 percent (Crawford 1980).

Sharp-tailed grouse still occupy much of their historic range. However, one subspecies (*T. p. columbianus*) has been extirpated from Oregon, Nevada and California (Giesen and Connelly 1993), and exists as isolated populations inhabiting < 10–50 percent of its former range (Miller and Graul 1980). This race has been designated as potentially threatened (category 2) by the U.S. Fish and Wildlife Service (U.S. Department of Interior 1989) under the Endangered Species Act. The plains sharp-tailed grouse (*T. p. jamesi*) has been extirpated from New Mexico, Oklahoma and Kansas, and there are < 200 individuals in Colorado (Hoag and Braun 1990). Thus, sharp-tailed grouse are undergoing range restriction at the western and southern periphery of their range.

Sage grouse have been extirpated from the periphery of their range (British Columbia, New Mexico, Oklahoma, Nebraska) and greatly reduced in distribution and abundance within their former core range (Braun 1987). For example, sage grouse currently occur in only 15 of 27 formerly occupied counties in Colorado; in several counties they persist as isolated remnants (C. E. Braun unpublished data).

Reductions in distribution and apparent abundance of prairie grouse are linked to changes in land use, including increased cultivation of land for crops, grazing by domestic livestock, conversion of shrublands to grasslands, and urbanization. Remaining habitats frequently are fragmented and smaller in size because of land conversion, transportation systems, powerline corridors, and reservoir and community development. Plant succession due to fire suppression has made some habitats less useful for prairie grouse and serves as an isolating mechanism.

Forest Grouse

Three species of forest grouse occur in North America. Ruffed grouse (*Bonasa umbellus*) have the broadest distribution, spruce grouse (*Dendragapus canadensis*) the most northern distribution, while blue grouse (*D. obscurus*) are restricted to the western United States and Canada (Aldrich 1963, Johnsgard 1973). The distribution of ruffed grouse decreased with settlement and clearing of forests; today its distribution has increased because of reversion of cropland to forests, reintroductions into historic range and introductions into new habitats (Newfoundland, Nevada and North Dakota) (reviewed in Gullion 1984, Widner et al. 1988).

The distribution of spruce grouse has changed little in the latter part of the 20th century other than significant retraction and fragmentation along the southern periphery of its range in northcentral and northwestern United States (Boag and Schroeder 1992). However, forest management that emphasizes large clear cuts and single species plantations has potential to negatively impact spruce grouse populations and distribution (Boag and Schroeder 1992). The greatest threats are in the boreal forest region of Canada, Alaska and in the northeastern United States. Depending on the size of areas cut, length of cutting cycles and forest regeneration rates, spruce grouse populations may remain depressed in local areas well into the 21st century under current forest management practices.

Blue grouse are residents of montane forests in western North America and generally occupy their known historic distribution. Some local extinctions such as at Mt. Pinos in California and reductions in distribution have occurred, probably the result of human activities (Bendell and Zwickel 1984, Zwickel 1992).

Ptarmigan

Three species of ptarmigan occur in North America, although only the white-tailed ptarmigan (*Lagopus leucurus*) occurs south of the Canadian border (Aldrich 1963). This species has the most limited distribution of ptarmigan in North America, mostly the result of limited and irregular distribution of alpine habitats. White-tailed ptarmigan have been reintroduced into New Mexico, and transplanted successfully into non-native alpine ranges in California. Utah and Pikes Peak in Colorado (Braun et al. 1993). Although no significant range retraction has occurred, expansion of ski areas, roads and water developments have had local impacts on white-tailed ptarmigan, especially in the southern Rocky Mountains (Braun et al. 1976).

Both rock (*L. mutus*) and willow (*L. lagopus*) ptarmigan have circumpolar distributions in the northern hemisphere (Johnsgard 1983). In North America, both species occur across northern Canada and throughout most of Alaska (Aldrich 1963, Johnsgard 1983). While studies on changes in distribution and status of northern ptarmigan are lacking, it generally is believed their overall habitat has changed little in the last 30 years. However, mineral and oil extraction, fire, and increased human activities may have impacted rock and willow ptarmigan populations in local areas.

Current Policy

Grouse management, research and harvest regulations have been the responsibility of state and provincial wildlife agencies without significant federal guidance. Federal funding has sponsored grouse research and management through the Pittman-Robertson Act, the National Science Foundation (U.S.), or the Natural Sciences and Engineering Research Council (Canada). However, federal land-management agencies have been reluctant to alter management practices to favor grouse when alteration conflicts with commodity uses such as livestock grazing, logging and mining.

While state or provincial policies on grouse management have not been explicitly stated, past policies generally have focused on two extremes: enhancement of habitat to benefit abundant and widely distributed species to improve hunting, and preservation of rare and endangered species. Ruffed grouse exemplify the first approach. The Great Lakes Region (Minnesota, Wisconsin, Michigan, Pennsylvania) is the core of ruffed grouse range in the United States, and all states have active habitat manipulation programs to maintain a diversity of seral stages on forested land to increase ruffed grouse populations (Gullion 1984). Also, timber cutting rotations designed to benefit ruffed grouse are included in Forest Plans on numerous National Forests. The Ruffed Grouse Society has contributed nearly $2 million for research to test the efficacy of this habitat development and improvement for ruffed grouse in the last 20 years.

Preservation of species is a policy of all management agencies, consequently, considerable attention has been given to threatened and endangered grouse. The amount of attention usually is correlated to the risk of extirpation. Typical agency responses to grouse populations at risk include: protection from or restrictions on hunting, a shift in responsibility from hunting programs within agencies to threatened and endangered programs, local research efforts, habitat acquisition and/or restoration, and reintroductions into formerly occupied range.

This policy of emphasizing both abundant hunted species and preserving endangered species ignored species and subspecies until they became endangered. This lack of proactivity greatly restricts management options, makes recovery of populations more costly and increases risk of extinction (Jennings and Scott 1993). Parochialism of agencies also has contributed to loss of populations of grouse. There is a tendency to emphasize recovery efforts on agency owned land and to minimize involvement of other land-management agencies and private interest groups. Management efforts should reflect habitat potential and not land ownership status or political boundaries. Emphasis has been on status of a species within the state, and not on the fate of local populations or global distribution. Benign neglect of local populations results in local extinctions and significant range retractions.

Perspectives for the 21st Century

Vision

Steady increases in human populations will result in even more land conversion, urbanization, mining, logging, roads and other developments which will further fragment and reduce grouse habitats. Number of grouse hunters will continue to decline. Demand for viewing grouse and ecotourism associated with grouse will increase markedly. Single-species management to enhance hunted populations will decline in favor of ecosystem management and preservation. Management decisions increasingly will be made in the glare of public scrutiny amid conflicting demands by interested parties.

Policy

Grouse management policy should evolve to include landscape-level considerations and management experiments that transcend agency and political boundaries. This will require unparalleled cooperation among management agencies and more and better research. Policy goals need to be defined for populations, subspecies and species of grouse. This will require the wisdom of Solomon, because a goal of preserving all local populations is unlikely to be feasible or even desirable in the broader context of managing for natural ecosystems and public needs and values. At the least, management should maintain current levels of grouse genetic variability and diversity at appropriate geopolitical scales, maintain biotic processes (speciation, population regulation, cycles, predator/prey relationships), retain suitable habitats to promote natural behaviors (i.e., lekking, migration patterns), and retain existing and evolving uses of grouse, such as hunting, viewing, education and ecotourism. Grouse management that achieves these goals will entail management of the places grouse live. Land management should be developed that allows grouse to be retained as components of functioning ecosystems.

Landscape and Life History Considerations

Grouse evolved in large biomes and continuous landscapes. Prairie grouse (except sharp-tailed grouse) in most areas live in climax vegetation (grass prairies, sagebrush rangelands) and do not depend on disturbance to create habitat. Several forest grouse (ruffed grouse, coastal races of blue grouse) depend on fine-grained (stand level) fragmentation, such as early stages created by wind- or fire-induced gaps in mature forests. Interior races of blue grouse breed primarily in sagebrush/aspen habitat and winter in conifer stands. Thus, they depend on patchy, coarse-grained landscapes of open shrublands, and deciduous and coniferous forest. Prairie-chickens and sharp-tailed grouse followed the plows west and north during settlement and greatly expanded their original distributions. Coarse-grained fragmentation (addition of agriculture to prairies) was beneficial (providing a dependable winter food source) until prairie remnants became too small and isolated. Crawford and Bolen (1976) estimated that areas with less than 63 percent rangeland cannot support stable populations of lesser prairie-chickens.

The process of fragmentation caused by agriculture, forestry or urbanization creates heterogeneity and discontinuity at the landscape level. This is an issue of increasing urgency. The impact of fragmentation for wildlife species varies, with some highly vulnerable to landscape change and others more resilient. The sharpness of the "edges" between habitat patches and the surrounding "non-habitat" determines the extent to which fragmentation impacts bird populations, as does patch size, distance between patches and species life history characteristics (Rolstad 1991, Swenson and Angelstam 1993). For instance, fragmentation caused by regenerating clearcuts within forests impacts forest interior birds less than fragmentation caused by similar-sized cropped fields (Rolstad 1991).

Little research on grouse has been conducted at the landscape scale in North America with the exception of Fritz (1979). Grouse may be relatively intolerant to extensive fragmentation for several reasons related to their life history, namely specialized food habits, generalized anti-predator strategies and poor dispersal abilities. Grouse are specialized herbivores that tend to subsist on large amounts of low-quality forage (Martin et al. 1993). Several species have significantly different winter and summer diets. Thus, for many grouse, removal of forest or prairie equates directly to removing their food supply.

Grouse, their nests and young provide food for a suite of avian and mammalian predators. Grouse usually comprise a small, but seasonally significant proportion of the prey biomass in many of the communities they inhabit. For example, spruce grouse comprise only about 2 percent of the prey biomass in the boreal forests of the Yukon (K. Martin unpublished data). Grouse clutches contain large eggs that develop slowly (21–27 days) in ground nests. Thus, in tundra, forest and prairie, grouse eggs may comprise a significant item in the diet of egg predators for a period of 6–8 weeks prior to the emergence of the young of other small- to medium-sized prey species. Since grouse are not capable of defending their eggs and offspring from most predators, they have developed generalized anti-predator strategies such as cryptic coloration and behavior to evade predators. Generalized anti-predator strategies probably work best when grouse comprise a small proportion of the prey biomass in the communities they inhabit. However, short-term increases in predators or reduced biomass of alternate prey can negatively impact grouse populations (Angelstam 1979, Storaas and Wegge 1987, Wegge and Storaas 1990). Course-grained fragmentation can alter the predator/prey balance permanently by exposing grouse to new predators. Fine-grained changes such as simplifying the habitat structure in forest or prairie also may increase predation risk because it is easier for predators to locate eggs and young. Increased predation likely elevates the risk of extinction in patches (Andren et al. 1985, Andren and Angelstam 1988, Gjerde and Wegge 1989, Wegge et al. 1990).

Extensive coarse-grained fragmentation can lead to metapopulations, where regional populations persist as local subpopulations. In the Adirondack Mountains of New York, Fritz (1979) found that spruce grouse occupancy of coniferous patches within a larger deciduous forest was 95 percent if patches were larger than 100 hectares, 60 percent for smaller patches and 92 percent if patches were within 10 kilometers of a colonization source. Rolstad and Wegge (1987) determined 100 hectares were necessary for persistence of capercaillie (*Tetrao urogallus*) populations. Presumably, populations in somewhat smaller patches will persist if close enough to other subpopulations or core areas for recolonization.

Juvenile dispersal ability is a key variable that interacts with distance between subpopulations as well as size and quality of patches in determining metapopulation viability (Rolstad 1991). Models incorporating dispersal abilities of vertebrates initially were developed to predict competition for space (Murray 1967, Miller and Carroll 1989). Recently, Hansen et al. (1993) extended these dispersal ability models to incorporate different patch sizes, thus providing a conservation context. Grouse are poor colonizers with relatively short dispersal distances. The natal dispersal distances recorded for grouse range from 1 to 40 kilometers, with a median of about 5 kilometers for females and 2 kilometers for males (Table 1). Furthermore, managers should focus on the most philopatric sex (usually males) when calculating probabilities of recolonizing patches based on dispersal distances. For North American grouse, it appears likely that patch recolonization and significant genetic exchange will be much reduced if fragments are more than 6 kilometers apart, especially for forest grouse. However, dispersal distances measured in contiguous habitat may not accurately reflect the dispersal potential between habitat patches in fragmented landscapes.

Issues and Strategies

Extinction of populations of grouse due to habitat loss, fragmentation, and degra-

Table 1. Mean natal dispersal for North American grouse.

Species	Location	Dispersal distance (km) Females	Males	Habitat	Reference
Spruce grouse	Ontario	<1	<1	Boreal forest	Beaudette and Keppie 1992
Ruffed grouse	Wisconsin	4.8	2.1	Deciduous forest	Small and Rusch 1989
Blue grouse	British Columbia	2.0	1.1	Coastal conifer	Jamieson and Zwickel 1983
Greater prairie-chicken	Colorado	9.2	2.7	Sandhills/grain	Schroeder and Braun 1993
Sage grouse	Colorado	8.8	7.4	Sagebrush	Dunn and Braun 1985
White-tailed ptarmigan	Colorado	4.0	1.2	Alpine	Giesen and Braun 1993

dation is, and will continue to be, the most significant issue confronting grouse managers. Species most at risk are sage grouse, several races of greater prairie-chicken and sharp-tailed grouse, lesser prairie-chicken, and spruce grouse. Policy goals must be defined for populations, subspecies and species of grouse across political and agency boundaries. Working groups should be formed with representation by state and provincial wildlife agencies, federal land-management agencies, universities, conservation groups, private landowner groups and other interested parties to identify needs, acquire information, and develop plans for habitat restoration, acquisition and reintroductions of grouse. There are several recent examples of this approach in conservation. The North American Waterfowl Management Plan strives for integrated management of wetland ecosystems on public and private lands through partnerships among federal, state, provincial, territorial and tribal governments, private conservation organizations, and individuals (Nelson et al. 1991). The Interagency Scientific Committee formed in 1989 to coordinate U.S. Forest Service, U.S. Bureau of Land Management, U.S. Fish and Wildlife Service and National Park Service plans for managing the spotted owl (*Strix occidentalis*) was a landmark in the application of population viability analysis in conservation (Harrison et al. 1993). The prompt legal challenges by conservation groups and rejection of the Committee's strategy by a federal judge suggested that management plans should be developed before species reach critical levels, and conservation groups should be included in planning and analysis from the outset. It is time to form such partnerships for sage grouse, Columbian sharp-tailed grouse, Attwater's prairie-chicken and southern populations of spruce grouse.

Restoration of grouse populations will require reintroductions in addition to habitat management. Techniques for successful reintroductions have been developed for ruffed grouse (Gullion 1984), white-tailed ptarmigan (Hoffman and Giesen 1983), greater prairie-chicken (Hoffman et al. 1992) and sage grouse (Musil et al. 1993) and are being developed for other species of prairie grouse (Toepher et al. 1990, Rodgers 1992). These techniques need to be tested broadly across habitats and species. Several points need to be considered relative to reintroductions of prairie grouse. Augmentation of existing small populations to reduce inbreeding and genetic bottlenecks (sensu Lande 1988, Lacy 1993) may not work (Caro and Laurenson 1994), and may do more harm than good with lekking species by reducing desirable or adaptive traits. Large numbers (≥100 ?) of birds from areas as close and as similar as possible to release sites should be transplanted as extreme skews in mating success among lekking grouse reduce effective population size and restrict gene flow. Releases of small numbers of individuals may succeed in the short term, but the long-term ability of populations with restricted genetic variation to adapt to changes in local conditions is reduced (Lande 1988, Lacy 1993). Lekking grouse species are most at risk and have suffered the greatest declines.

Hunting, as a possible contributing factor in extinction of local grouse populations, will become more contentious as habitats become more fragmented and, at least in the near term, hunting itself will be challenged. The extent to which hunting is additive or compensatory to natural mortality is debatable for populations in continuous landscapes (Bergerud 1988, Ellison 1991). Definitive experiments need to be conducted that evaluate the extent to which hunting is additive at different harvest rates and in different patch sizes. Pending this information, agencies should adopt conservative harvest regimes for small or fragmented populations as fragmentation likely decreases the resilience of populations to hunting pressure.

To manage grouse populations effectively in the 21st century, additional knowledge is required. We need to know how much area is needed to maintain grouse populations and population processes, and how many individuals are necessary to maintain adequate genetic diversity. We need to know how close subpopulations should be to permit recolonization. How does habitat fragmentation and degradation impact population processes? What levels of hunting are compensatory or additive, and do these levels vary with fragmentation? We suspect that heavy grazing negatively impacts grouse, but we do not really understand the process. For most species of grouse, we have the basic life history data and habitat requirements for populations living in continuous landscapes. Thus, we are in a good position to conduct sophisticated experiments to address both fine- and course-grained questions on fragmentation. We do not have all the techniques necessary to restore depleted habitats and, for some grouse, we do not know which seasonal habitats are critical. We must learn by trying. Management strategies should be implemented as experiments and evaluated (Walters 1986).

Conclusions

We should be attempting to maintain grouse populations and processes by managing habitats at both fine- and course-grained levels. To achieve this goal, policies for management of grouse and their habitats must change in the 21st century. Planning and management must encompass local populations but be implemented at landscape scales across agency and political boundaries. Increased emphasis is needed on restoration of grouse populations. This will require methodology for reintroductions and habitat restoration, and better understanding of the importance of genetics of small populations and founders of new populations. Hunting of many grouse populations no longer will occur on the scale that it has in the past, although other uses will increase. With enlightened habitat management and government policies, all North American species of grouse can be expected to persist throughout the 21st century. To expect less would indicate the public is not committed to maintaining quality environments in perpetuity.

Acknowledgments

We thank K. M. Giesen, R. J. Gutierrez and R. F. Holt for review of this manuscript. This a joint contribution from Colorado Federal Aid in Wildlife Restoration Project W-167-R, the Canadian Wildlife Service and the Welder Wildlife Foundation (contribution #432).

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Population Status and Trends of Grouse and Prairie-chickens from the North American Breeding Bird Survey and Christmas Bird Count

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Introduction

The North American Breeding Bird Survey (BBS), sponsored by the National Biological Survey and the Canadian Wildlife Service (Peterjohn and Sauer 1993), provides information on population changes for more than 250 species of North American breeding birds. The Audubon Christmas Bird Count (CBC) contains distribution data on populations of birds wintering in North America (Butcher 1990). Game managers have been examining both BBS and CBC data as possible sources of population status information for upland gamebird species (Sauer et al. in press). For example, the BBS provides useful population trend information for northern bobwhite (*Colinus virginianus*) and other quail species (Droege and Sauer 1990).

In this paper, we assess the capabilities of the BBS and CBC to estimate distributions and population trends of grouse and prairie-chicken. We document the distribution of each species from the surveys, present the trend estimates from the surveys for each species, conduct a power analysis on the existing data and review deficiencies in each survey for monitoring population status and trends of grouse and prairie-chickens. We conduct these analyses for blue grouse (*Dendragapus obscurus*), spruce grouse (*D. canadensis*), ruffed grouse (*Bonasa umbellus*), sage grouse (*Centrocercus urophasianus*), sharp-tailed grouse (*Tympanuchus phasianellus*), greater prairie-chicken (*T. cupido*) and lesser prairie-chicken (*T. pallidicinctus*).

Methods

The North American Breeding Bird Survey

The BBS (Peterjohn and Sauer 1993) is an annual, roadside survey conducted primarily in June in the United States and southern Canada. Each of the more than 3,000 survey routes consists of 50 stops spaced at 0.5-mile (0.8 km) intervals along

24.5 miles (39.4 km) of roadside. Volunteers selected for their ability to identify birds drive the route starting 30 minutes before sunrise, recording all birds heard or seen within 0.25 mile (0.4 km) of each stop during a three-minute period. The sum of the counts over the route for each species is used as an index to bird abundance. Initiated in 1966, the survey routes were allocated to ensure random coverage within the constraints imposed by the roadside locations. However, coverage has been less complete in remote areas with few roads and in sparsely populated states and provinces, which often correspond with the highest grouse densities.

The Christmas Bird Count

The CBC (Butcher 1990) is conducted each year within a few weeks of December 25. Each count is conducted within a 15-mile (24.14 km) diameter circle, and volunteer observers count birds at various locations within the circle during a 24-hour period. Because the number and skill of observers can differ greatly among circles and over time, data on effort are summarized as number of observer party-hours/circle-year. The data available for analysis covered the period 1959–1988. Unlike the BBS, CBC circle locations are not randomly located, but tend to cluster at coasts and near metropolitan areas. Because of this location bias, the statistical validity of trends estimated from the counts must be viewed with caution.

Abundance Mapping

To show survey coverage within the range of each species, we mapped the species' relative abundance using mean counts over the interval 1966–1992 (for BBS) or presence of the species over the interval 1959–1988 (for CBC) as indices to abundance. The BBS relative abundance data were smoothed using program Arc/Info (Environmental Systems Research Institute 1992) to form maps of relative abundance for each species. Kriging, a statistical procedure in which the spatial covariance between counts is used to estimate a surface from point data (Isaaks and Srivastava 1989), was used to predict regions in which relative abundances were within categories of 0.01–1, 1.01–3 and 3.01–10 birds/route. Center locations of CBC circles that contained the species then were projected onto the BBS relative abundance map to provide additional distribution data.

Population Trends

Survey coverage within the range of grouse has been very incomplete for both BBS and CBC. Survey routes and circles were not surveyed each year, observers changed, and new sites were started during the survey years. Consequently, simple averages of counts over time provide misleading views of population change (Geissler and Noon 1981). To accommodate the sampling deficiencies in the analysis, we estimated trends on each survey site using linear regression on logarithms of counts plus a constant of 0.5 versus year. For the CBC analysis, the counts were adjusted for effort before analysis by dividing them by 100 party-hours. In the BBS, observer covariables were used in the regression to allow each observer to have a separate intercept (Geissler and Sauer 1990). The trend for the route/circle was the slope associated with year, which was transformed by exponentiation to a multiplicative growth rate. Regional trends were estimated as a weighted average of these route/circle trends, with route/circle trends weighted by route/circle precision and species mean relative abundance (Geissler and Sauer 1990). We estimated variances of the regional trends using bootstrapping, and tested the significance of the trends using z tests. Trends were estimated for the portion of the species' ranges covered by the surveys.

To demonstrate regional patterns of population change, we associated the trends of individual BBS routes and CBC circles with their locations, and again used Kriging to summarize the BBS trends on survey routes into regions of increase and decline. CBC data are presented as symbols indicating increase or decline of populations at individual circles.

Finally, we evaluated the current power of the BBS and the CBC to detect changes in grouse populations within the regions covered by the surveys. Using the estimated trends and their variances, we estimated the power of the survey to detect a 2-percent yearly trend in populations with $\alpha = 0.10$. In this context, power represents the chance that the survey could have detected an actual 2-percent yearly change with the existing precision.

Results

Species Coverage

All species are encountered on BBS routes and CBC circles at low relative abundances throughout most of their ranges (Table 1). Only scattered sample sites occur in both surveys north of 51 degrees latitude, and we limit our analysis to the continental United States and southern Canada (Figure 1). Consequently, species with northern ranges tend to be poorly covered by the surveys. Spruce grouse, although widespread across boreal forests, are very locally distributed throughout the southern portion of their range and are not well sampled by either survey. Other species, such as blue grouse, ruffed grouse and sharp-tailed grouse, have large portions of their ranges in surveyed areas, whereas sage grouse and prairie-chicken ranges are contained within surveyed areas.

However, comparison of "ranges" estimated from BBS and CBC data with range maps contained in Johnsgard (1973) suggests that neither survey provides complete coverage of species even within their ranges. For ruffed grouse, the CBC provides data for Wyoming and Missouri, populations that are not sampled by the BBS. For blue grouse, there are several CBC circles with data in the central Rockies but few BBS observations. Both surveys provide poor coverage where the species occurs in central and northern British Columbia (Johnsgard 1973, Figure 1). In contrast, there are large areas of BBS coverage with no CBC data (Figure 1).

The range of greater prairie-chickens (Figure 2) and the southern portion of the range of sharp-tailed grouse (Figure 2) appear to be covered quite well by the surveys, although it is clear that neither survey is well-suited for documenting the status of small local breeding populations. For example, many local populations outside the main portion of the range are not monitored by the surveys. Sage grouse occur in portions of the west where both surveys tend to have poor coverage, leading to range maps that appear as composites of groups of individual routes (Figure 3).

Analysis of Population Change

We present trend results for the survey periods for each species (Table 1). The only statistically significant results were for blue grouse (BBS) and spruce and sharp-

Table 1. Population trends for species of grouse and prairie-chickens in surveyed portions of North America. For each species, the trend (percentage/year), power of a test of the null hypothesis that the yearly trend equals 0 for an actual trend of 2 percent per year and $\alpha = 0.10$, and mean count ($\overline{X}$) for the surveys are presented.

	BBS 1966–1992				CBS 1959–1988			
Species	Trend	N	$\overline{X}$	Power	Trend	N	$\overline{X}$	Power
Blue grouse	-4.59[a]	95	0.43	0.39	-0.27	76	0.13	0.84
Spruce grouse	–1.72	19	0.02	0.49	2.85[a]	57	0.25	0.51
Ruffed grouse	-0.55	730	0.27	0.56	0.27	919	0.80	0.95
Greater prairie-chicken	–7.91	52	0.61	0.33	-0.55	56	4.14	0.40
Lesser prairie-chicken	20.68	8	0.25	0.36	–14.21	7	2.22	0.32
Sharp-tailed grouse	1.08	184	0.49	0.46	2.22[a]	131	4.82	0.56
Sage grouse	3.01	104	0.72	0.35	–0.02	41	6.84	0.41

[a]$P < 0.05$.

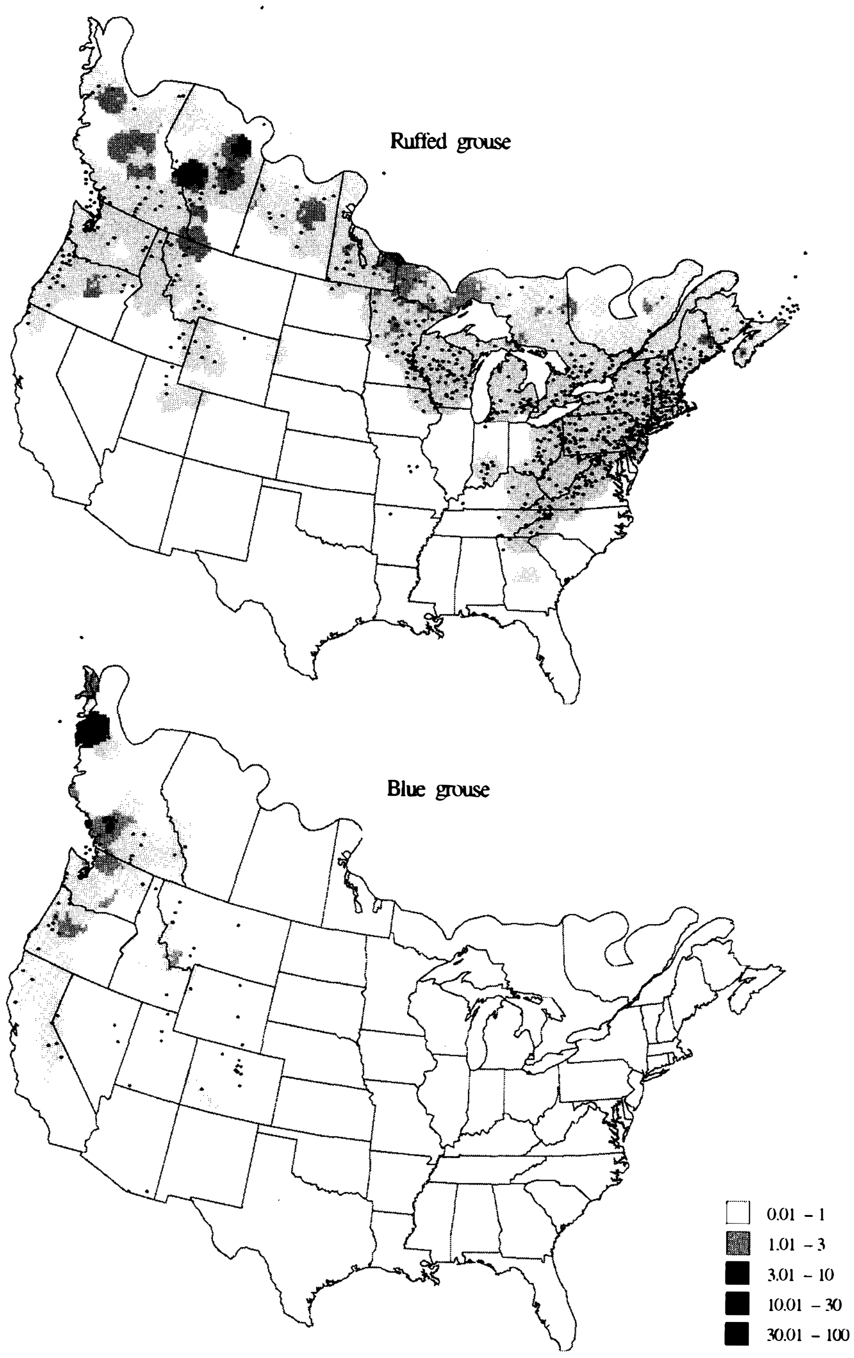

Figure 1. Distribution maps for ruffed grouse and blue grouse from the BBS and CBC. The distribution maps have relative-abundance categories for BBS data (1966–1992) and locations of CBC circles on which the species were found in the interval 1959–1988.

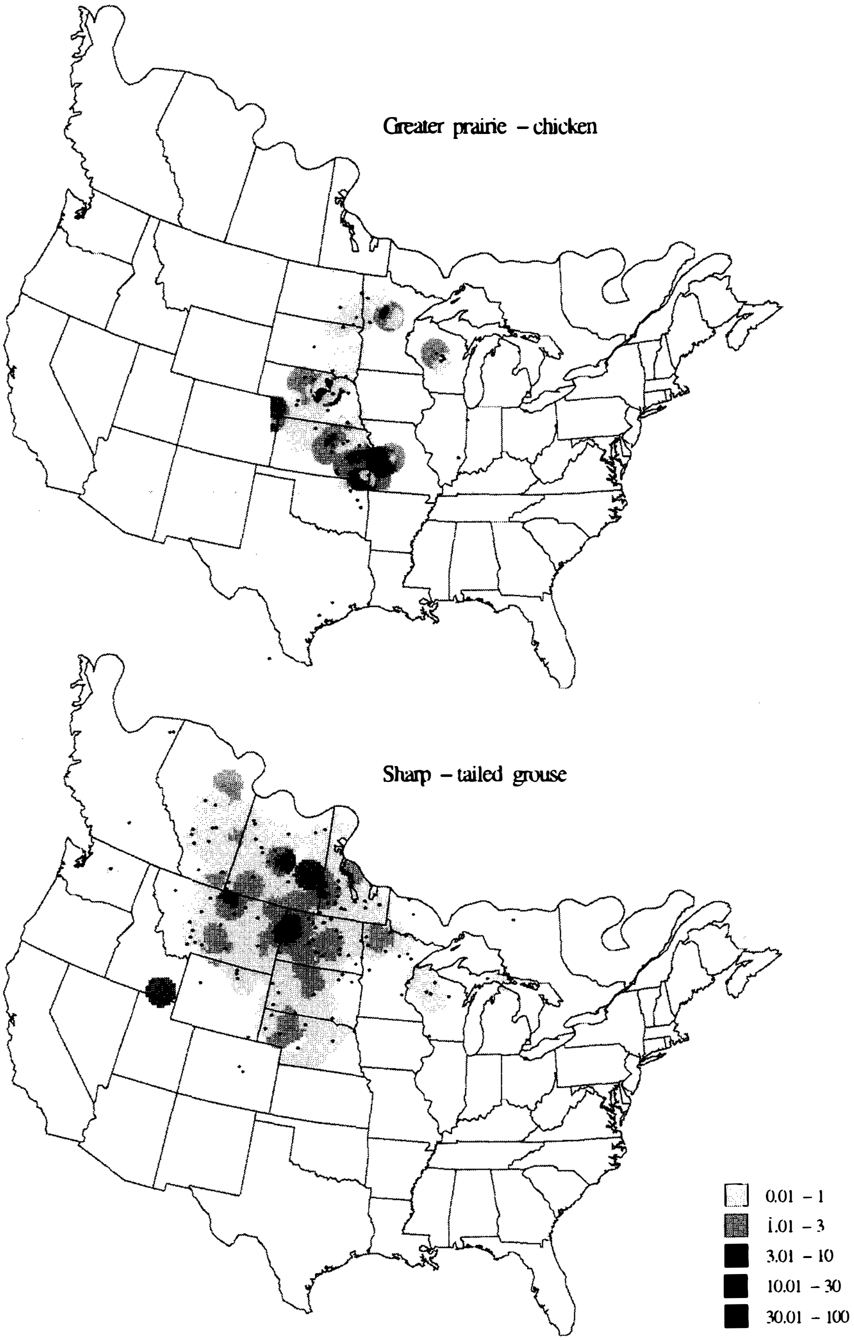

Figure 2. Distribution maps of greater prairie-chicken and sharp-tailed grouse from the BBS and CBC. The distribution maps have relative-abundance categories for BBS data (1966–1992) and locations of CBC circles on which the species were found in the interval 1959–1988.

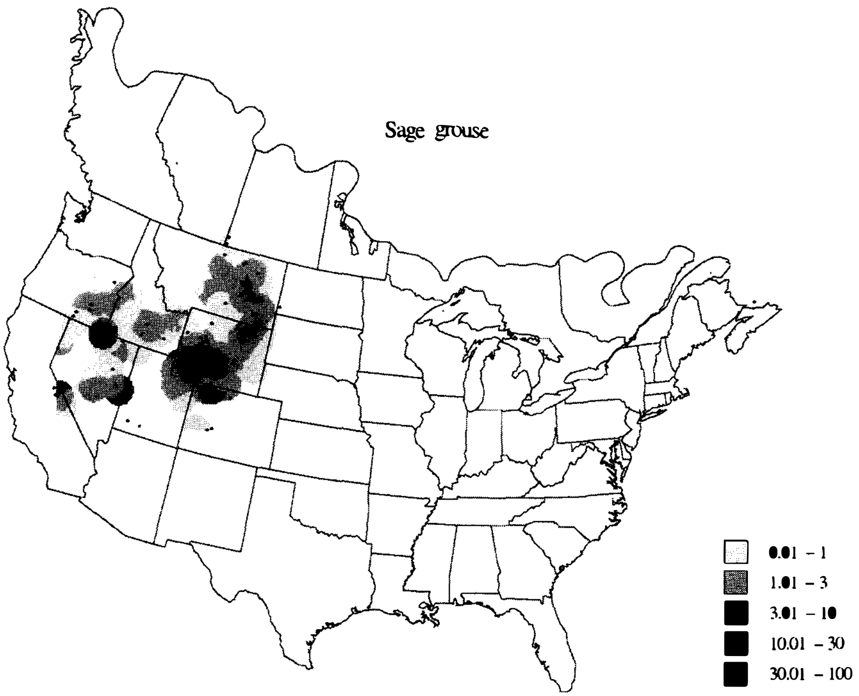

Figure 3. Distribution map of sage grouse from the BBS and CBC. The distribution map has relative-abundance categories for BBS data (1966–1992) and locations of CBC circles on which the species was found in the interval 1959–1988.

tailed grouse (CBC). However, large portions of these species' ranges are outside survey coverage. Sample sizes and relative abundances also differ greatly among species, with ruffed grouse having the largest samples in both surveys. The surveys provide little information about population change in lesser prairie-chicken results, as each survey has few samples for the species. Most results have very low statistical power to detect even large changes in populations. For example, the 0.46 power for sharp-tailed grouse indicates that there is only a 46-percent chance of detecting a 2-percent/year decline in population with $\alpha = 0.10$.

Portraying the trends geographically indicates a great deal of heterogeneity in patterns of population change. Ruffed grouse show some patterns of increase and decline based on BBS trends, but CBC results often show slightly different patterns. There are areas of consistency in trends, however, such as western Virginia and southern West Virgina (Figure 4). Sharp-tailed grouse tend to show large areas of consistency in trend from BBS data, but CBC data only provide a weak validation of the trends (Figure 4).

Discussion

We have shown that both the BBS and the CBC provide information on large-scale distribution patterns in grouse and prairie-chicken populations. Unfortunately, both

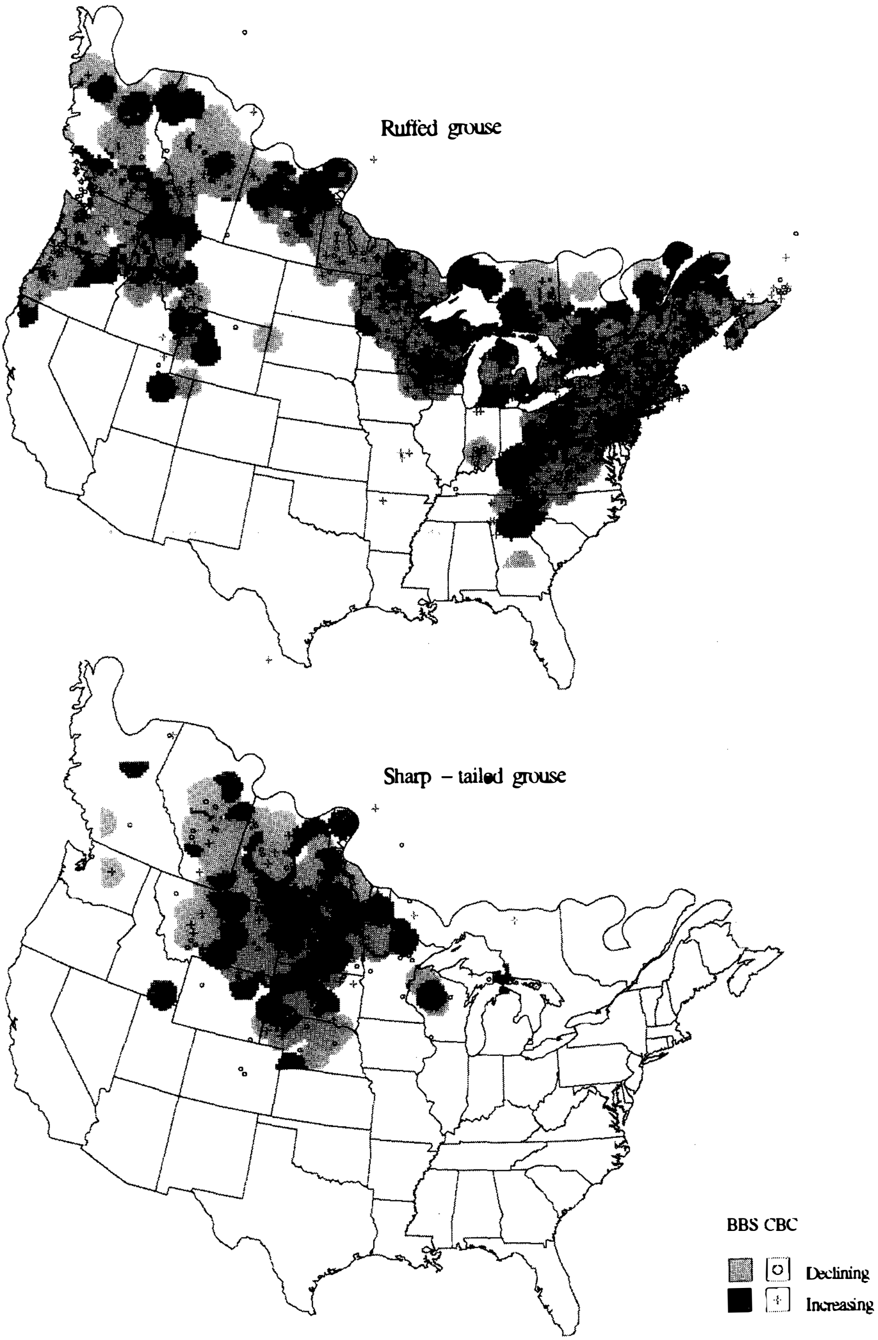

Figure 4. Maps of population trends for ruffed grouse and sharp-tailed grouse from the BBS and CBC. The trend map has contours of increase and decline from BBS data (1966–1992), and indicates increase and decline at CBC circles for the interval 1959–1988.

surveys generally are not adequate to detect population changes for the species because power of the surveys is low and the species are counted at low relative abundances. Thus, the estimated trends are imprecise and even may be positively biased (Geissler and Sauer 1990).

During winter, most grouse species would have to be detected by observers on foot, by flushing birds from their preferred habitats. The total party-hours used as an effort adjustment in the CBC could be misleading if time spent by observers on foot is not proportional to total party-hours. Unfortunately, grouse are rather secretive in winter (Johnsgard 1973), and most CBC observers probably do not accurately record number of birds present.

CBC circles are not randomly located, and observers on circles tend to "target" rare or endemic species. Consequently, if a circle occurs in an area with an isolated population of grouse, there is a high probability that the species will be observed. However, there are large portions of most grouse ranges with no CBC information. These gaps in coverage clearly suggest caution in interpreting CBC results for grouse species.

BBS data provide less regional bias than CBC results, and the greater standardization of methods in the BBS is likely to provide a more consistent index for estimation of population change. However, most grouse are sampled on relatively few routes and occur at very low relative abundances on survey routes, making them poorly suited for trend analysis (Geissler and Sauer 1990).

The main problem, in our opinion, is not the roadside survey technique, but rather that surveys are conducted after the peak display season for all species. For example, if the BBS were conducted in April or early May, many more ruffed grouse would be detected. However, the roadside bias may be a problem for blue and spruce grouse, which tend to occur in regions with few roads.

The coincidence between the estimated trends from the surveys is interesting, but because of low precision in the estimates we cannot reliably determine whether real consistency exists. Greater prairie-chickens appear to have been decreasing over most of their range (Schroeder and Robb 1993); a result consistent with both BBS and CBC trends (Table 1). Ruffed grouse appear to have trends close to 0, and the relatively high power of these tests suggest that the result may be trustworthy. Of course, estimation of a trend over a relatively short time period may not be relevant for species with population cycles, and modelling of complex population dynamics requires better data than those needed for trend estimation.

Can the Surveys be Modified to Provide Better Information?

Both the CBC and the BBS miss the peak activity times for grouse; this tends to make sample sizes and counts low, and to provide minimal statistical power of hypothesis tests about population trends. The timing of the BBS reflects its primary goal of surveying nongame birds, and the roadside nature of the survey further limits its applicability to grouse. The CBC design allows for maximal volunteer participation, but the lack of standardization introduces difficulties with both analysis and interpretation of the data (Butcher and McCulloch 1990).

The poor coverage of off-road habitats and northern areas is common to both surveys, and many nongame bird species also are poorly covered by the BBS. The Partners in Flight initiative by the National Fish and Wildlife Foundation (Finch and

Stangel 1993) has sponsored a variety of actions designed to encourage monitoring of off-road and northern populations of neotropical migrant birds, including extension of the BBS into northern regions and augmenting the survey with information from non-roadside counts. Although these actions will improve the BBS as a survey of grouse, it is clear that the survey never will provide precise information on population trends of most grouse species.

Acknowledgments

We thank the many participants in both the BBS and the CBC, who donate their expertise to assist in large-scale monitoring. Brett Hoover prepared the CBC database, which was provided by the National Audubon Society and Cornell Laboratory of Ornithology. D. K. Dawson, R. J. Gutiérrez and D. G. Krementz reviewed the manuscript.

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Trends in Yukon Upland Gamebird Populations from Long-term Harvest Analysis

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Introduction

Management policy in the Yukon Territory, as in most northern jurisdictions, has assumed grouse to be a highly resilient hunted species, superbly adapted to maintaining populations as long as their habitats remained intact. Early in the development of policy in the mid-1970s, it was decided that a tracking of harvest would suffice to judge the effectiveness of management efforts. Harvest limits would remain relatively liberal unless some obvious problem was detected. As an additional check, minimal monitoring of breeding numbers in the field was done when operational budgets allowed. Almost 15 years of harvest and field data now exist with which to examine the effectiveness of this policy.

In the Yukon, the focus for concern was on the open-habitat grouse. The forest-dwelling species were assumed to be relatively secure whereas the others, in particular because of winter flock-forming behavior, were seen as potentially vulnerable (Weeden 1963, 1972). The Yukon supports seven species of grouse. Of these, the three species of "ptarmigan" (*Lagopus* spp.) and the sharp-tailed grouse (*Tympanuchus phasianellus*) were candidates for potential mismanagement. Among the ptarmigan, the willow ptarmigan (*Lagopus lagopus*) was singled out for concern because it was thought to occupy habitats more closely associated with human development, was known to concentrate on critical late-winter habitat (Mossop 1988) and was thought to be the most heavily hunted.

A complicating factor in northern grouse is the highly cyclic nature of natural population fluctuations that are expected to occur in roughly 10-year periods. Monitoring of the critical parameters would have to continue for a very long period, and exactly how to draw conclusions from the associated data never was clear. Cycles, however, should offer a simple short-term test of the effectiveness of harvest indicators to mirror population size.

Evidence of overhunting and associated population declines in grouse now exists in the literature (Gullion 1982). Bergerud's (1988) critical assessment of how this may happen recognizes the general finding among grouse studies that production of young is the key field occurrence signalling future population levels. As well, if just to err on the side of caution, mortality from hunting must be assumed simply to add to natural mortality (Braun 1969, Zwickel 1983, Fischer et al. 1974). Harvest theoretically could produce positive or negative effects: increases should occur if spacing was increased toward more "optimum" resulting in higher production, and declines should happen if take simply was great enough to surpass that year's recruitment. So population age structure and breeding density seem to be the important parameters the manager needs to be using harvest indices to at least track in a relative way.

The Harvest Survey

The Yukon harvest questionnaire is a high-intensity annual mail-out survey of all licensed hunters. It gathers data on all hunted species, with grouse included as an identifiable subset. Information is requested by individual grouse species and although the three species of ptarmigan are pooled, the harvest can be assumed to be virtually all willow ptarmigan. The questionnaire is vigorously applied; technically, it is mandatory for all licensed hunters, in practice three mailings result in a relatively high rate of return (69–75 percent).

Initial analysis assigns harvest to small (average 1,000 km^2) geographic areas by means of a computerized gazetteer. Estimates are generated using sampling intensity and are adjusted for non-response from the ratio of compliance in follow-up mailings (Kale 1982). The key indices, determined on a per-hunter basis, are days hunting effort and harvest by area. Due to a big game bias, a flaw in the process affecting grouse data has the survey questionnaire applied before the season for three species (the ptarmigan) is closed. Bird hunters are expected to estimate their take in the rest of the season from last year's take. The lost information in the extended bird season has been assumed to be insignificant; very few hunters take advantage of the season beyond the big game season. Estimates of the key parameters are generated on an IBM (9121) main frame.

Field Census Plots

Six widely spaced permanent census areas have been established over time, and have received varying monitoring intensities. Focus has been on willow ptarmigan, with sharp-tailed grouse a second priority. Ground counts of other grouse, although attempted, have not resulted in usable data to date.

Willow ptarmigan census plots average 2 square kilometers in optimal breeding habitat. Counts have been done annually in early May during the height of territorial display and song (Mossop 1988). Total ground searches with a trained pointing dog have been assumed to produce a total count. Pairs and lone males are counted and mapped accurately on base maps of the areas. Sharp-tailed grouse field surveys are counts of males attending established leks in the spring. The number of males, along with the number of active leks in a larger study area surrounding those known persistent leks, gives the values used to monitor relative abundance.

Age Structure

In an attempt to quantify productivity, annual samples of willow ptarmigan have been taken during the late winter/early spring from one of the zones where the most reliable spring counts are made. In practice, this sample has been a combined pool of hunter-killed birds obtained at road checks, supplemented by a sample collected in late winter by management staff. Bird specimens have been aged by the pigment on the eighth primary (Bergerud et al. 1963).

Results

Cyclic Fluctuations

Field counts of ptarmigan have demonstrated striking 8–11 year fluctuations in all places where they have been monitored in the territory (Figure 1). As well, the more limited sharp-tailed grouse data suggest a similar cyclic fluctuation. A feature of these grouse cycles is the apparent synchrony in the fluctuations between species and between various parts of the territory. Densities have been consistently lower among more northerly ptarmigan populations.

Yukon grouse hunters annually have spent from 2,206 to 9,560 days afield to produce a bag varying from 2,200 to over 20,000 birds. In both cases, the variation apparently followed a 10-year cycle. On cursory examination the harvest indices of ptarmigan and sharptails apparently both tracked the population cycles (Figure 2). However, on closer observation the relationship was less than perfect ($r = 0.52$—ptarmigan; $r = 0.43$—sharptail); in fact, during the most recent decline the curves were significantly different for both species ($p = 0.05$). The harvests of ptarmigan and sharptails, although both exhibiting clear 10-year cycles, both fell off well before the populations declined. Isolating the harvest estimates by zone, the relationship was even worse.

In an attempt to clarify relationships between cyclic population changes and harvest parameters, regressions were examined for all parameters available. The ptarmigan harvest cycle was highly correlated with the overall number of hunters afield ($r = 0.95$ $p = 0.01$). Once there, hunters spend a surprising standard five to seven days afield regardless of year. The next closest relationship ($r = 0.89$) was with the harvest

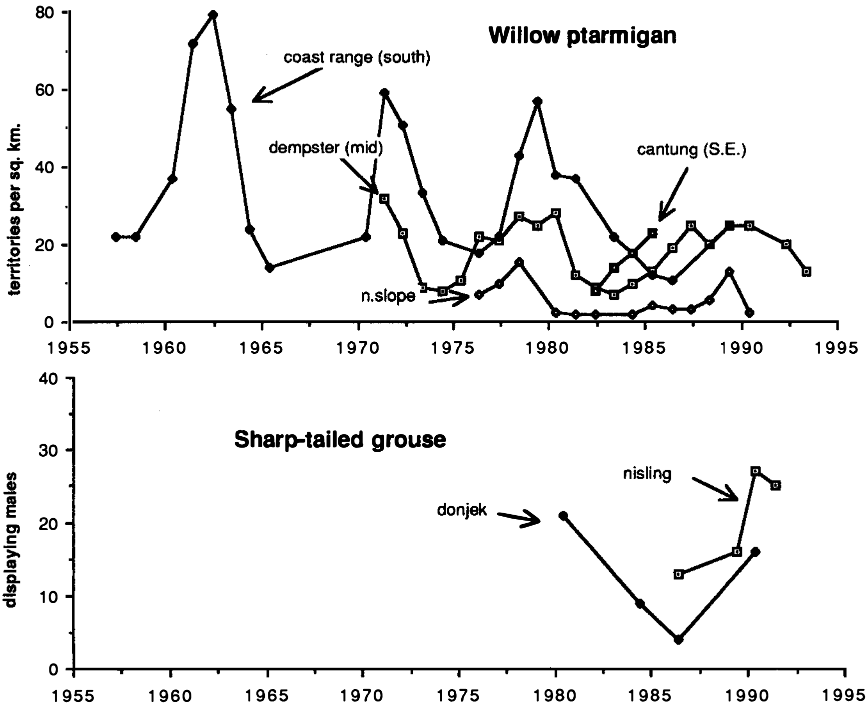

Figure 1. Population densities and trends of various willow ptarmigan and sharp-tailed grouse populations from census plots throughout the Yukon Territory, Canada.

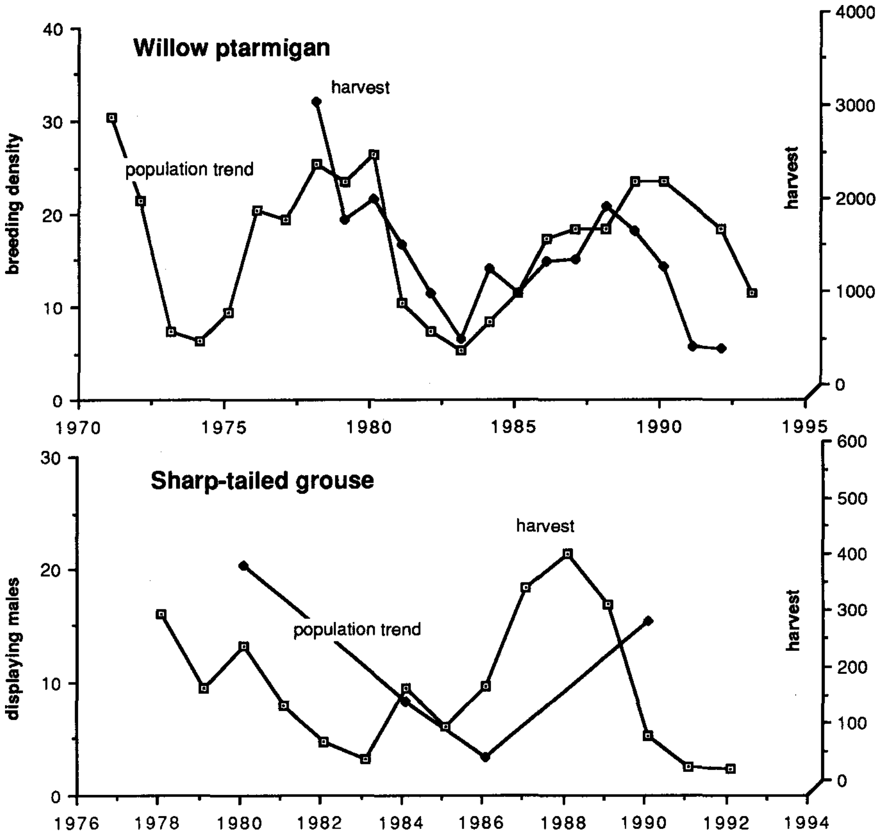

Figure 2. Relationship between population trend and relative harvest of willow ptarmigan and sharp-tailed grouse in the central Yukon Territory.

estimate for spruce grouse (*Dendragapus canadensis*)—the grouse species which is by far the most heavily harvested (Figure 3). Sharp-tailed grouse on fewer data suggest similar relationships. Clearly, the number of hunters afield must be considered the dominant harvest parameter, the number of spruce grouse taken is correlated strongly, with the harvest of the lesser taken vulnerable species most likely a second or tertiary relationship.

Long-term Trends

Only two of the willow ptarmigan study areas have long enough strings of field counts to suggest any long-term trends in addition to the regular cyclic fluctuations. (None of the sharp-tailed grouse field counts could be used in this manner.) The mid-Yukon count from the North Fork Pass has covered three cyclic peaks and shows no discernible trend. Harvest over the same period also shows no significant trend.

The southern count from the Chilkat Pass with a longer series, does seem to be suggesting a discernible decline. The overall decline has averaged approximately 1.1 percent annually—a significant decline ($p = 0.05$) even in spite of the large normal

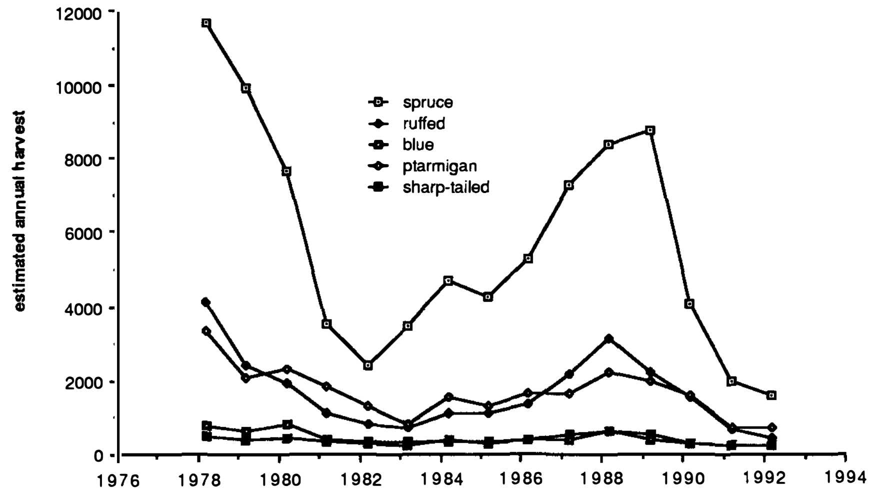

Figure 3. Relative harvest intensity of various Yukon grouse species over time.

fluctuations involved in the cycle. Interesting is the apparent way this decline is coming about: no depression in the "low" population levels of the cycle has occurred, instead the high peaks simply seem to have been steadily eliminated.

Unfortunately, there are no reliable harvest data directly relatable to this population because it occurs just across the border in British Columbia. Wildlife management staff from that province suggest a relatively heavy and steady harvest pressure for the area of at least 1,000 birds annually which apparently has been accounting for 15–20 percent of the autumn population (Gruys 1991).

Evidence of Relationship with Breeders' Age Structure

The proportion of young birds from the previous year in the breeding population did vary in a regular way over the term of the data string. In effect, the number of young was very closely correlated with the dynamic state of the population ($r = 0.92$ $p = 0.01$). Rising numbers always were associated with a high proportion of yearlings. Meanwhile, harvest showed a very poor relationship with age structure ($r = 0.43$ $p = 0.05$). If harvest somehow was affecting productivity, nothing in the data could be used to predict it.

Harvest Mortality Estimates by Region

The clearest way to view harvest level is to compare absolute numbers of grouse on the ground to the numbers taken. Grouse harvest in the Northwest generally can be assumed to come from relatively narrow corridors near access roadways (Weeden 1972, Gruys 1991). Because of relatively restricted access in the Yukon, populations being harvested (at least of the "vulnerable" species) are pretty well known. Autumn numbers in these corridors in the Yukon were estimated very roughly based on census plots and age structure. Concurrently, Yukon's management zone harvest could be assigned to these known populations (Table 1). Natural mortality of birds-of-the-year and adults between breeding seasons for sharptails and willow ptarmigan are all somewhat similar, varying between 40 and 60 percent (Bergerud 1988). Clearly, the remnant of about 50 percent of birds-of-the-year must cover the lost breeders as well as any increment to the next breeding population. From the Yukon results hunting mortality at the upper end presumably could add 20–40 percent, a value which theoretically would eliminate all but the most spectacular increases in willow ptarmigan in the years for which data exist. As far as we know, none of the Yukon populations are experiencing such high harvest mortality; the only group that may be is the Chilkat population. However, given the poor predictability of harvest relative to population trend, it must be assumed that some of these higher mortality rates eventually will ensue when population numbers are low or declining.

Among the sharp-tailed grouse groups, the situation may be more threatening. Sharptail populations are far more localized and smaller in size. Harvest already may have been running very close to productivity in at least one zone.

Discussion

This analysis suggests a basic problem with the assumption that harvest indices will track population abundance and therefore can be used to signal mismanagement. Grouse harvest in the Yukon seems to be driven by the number of hunters afield which, in turn, probably is determined not by the number of species the manager may decide are vulnerable, but by some unrelated factor—in this case, the spruce grouse population. Moreover, this conclusion may have wide application; in virtually all areas where grouse are hunted, grouse hunters pursue a suite of species (some of which may not even be grouse). If harvest data are going to be interpreted correctly in a predictive manner, the dominant species in the bag probably will be just as vital for the manager to track and predict as is the take of the species of concern.

Clearly, it becomes critical to be able to assign the actual levels of harvest at the population level; "indicator" values are probably going to remain elusive. In the Yukon Territory, harvest data were collected at a level sufficiently detailed that

Table 1. Autumn population estimates compared to harvest estimates for various grouse populations in a 4-kilometer corridor along access roadways in the Yukon Territory.

	Maximum population estimate	Minimum population estimate	Maximum harvest estimate	Approximate percentage of autumn population
Zone 2				
Willow ptarmigan	14,400	2,400	550	4-23
Zone 11				
Willow ptarmigan	8,500	1,700	350	4–21
Chilkat				
Willow ptarmigan	20,640	2,880	1,000[a]	5–35
Zone 4				
Sharp-tailed grouse	1,000	400	40	4–10
Zone 5				
Sharp-tailed grouse	900	350	135	15–39

[a]Estimated from Gruys (1991); other harvest estimates are from the Yukon harvest questionnaire.

numbers could be assigned to a relatively small region. This, in hindsight, has emerged as one of the more useful aspects of the data.

The ability to estimate local population sizes for species of concern probably is equally vital. The Yukon may be uniquely fortunate in this regard: road-corridor hunting, combined with concentration of habitat in valleys, give confidence to extrapolation estimates from census plots. Regardless of how it is conducted, field census data take on great significance. In eras of falling field budgets, managers are going to have to become innovative in defending and carrying out highly efficient population monitoring. Managers in particular need to come to agreement on the data required as a minimum to produce the critical comparisons.

Population age structure relations to harvest meanwhile remain confusing. In the Yukon, age structure of the breeding population seems to show no predictable response to harvest intensity; production tracked the cycle of abundance regardless of harvest. It is particularly interesting that the Chilkat population seems to have shown no inclination to respond to a possible over-harvest by increasing productivity. If this proves to be so, it would seem to be contrary to the spacing hypothesis for increasing production. We know harvest has sex, and possibly age, biases (Weeden 1972). However, no one has critically assessed how these biases may affect harvest-induced changes to the population. The Yukon harvest numbers are typical in that they cannot be assigned age and sex, and until researchers can clarify this relationship, it is unlikely managers will go to the expense; decisions will have to continue to rely on absolute numbers.

It will be most interesting to see if harvest manipulation generally can result in solving population problems which may become evident. An intuitive fear for the overharvest of sharp-tailed grouse emerged very early in the period of harvest data collection. (An analysis similar to the one presented here, except with far fewer data, suggested a harvest level approaching 40 percent). In 1985–86, a regulation change was implemented to severely reduce the bag limit regionally where the problem was detected. To date, that change has not significantly altered the harvest estimate, suggesting simple daily bag limit changes may not be as effective as managers have assumed. Again, harvest seems more correlated with the overall number of hunters that decide to be grouse hunters that year. Managers may have to become innovative in regulating take at the species and perhaps at the population level. An attractive test would be a change in the harvest regime in the Chilkat Pass area where heavy single-species harvest pressure is the rule.

In rethinking the value of harvest data, it is important not to ignore one of the prime purposes for harvest data which is to put tangible value to grouse populations in the overall management of wildlife species. The 2,000 to 9,500 days afield by Yukon grouse hunters is low by many jurisdictions' standards but it still ranks grouse as one of the most valued hunted resources in the territory. It will be important to exercise the care necessary to produce the proper harvest indicators without jeopardizing that use. However, managers should not assume that a simple tracking of take will suffice without direct reference to the population on the ground.

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Status of Mexican Quail

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Introduction

Mexico contains the smallest land area of the three countries in North America, yet has the largest diversity of vertebrates (Flores-Villela and Fernández 1989, Graham 1993). Recently, Mexico was found to rank among the top 10 countries in the world for the number of restricted range species of birds (103) and number of Endemic Bird Areas (14) it contains (Bibby et al. 1992). Johnsgard (1973, 1988) has suggested that the New World quail evolved in the vicinity of southern Mexico and Guatemala, and radiated to North and South America from there. Mexico has the greatest diversity of quail of any country in the New World. Of the nine described genera and approximately 32 species of quail native to the New World, Mexico has populations representing eight genera and 15 species. Despite it's importance, we know very little about populations and biology of quail in Mexico.

Mexican Quail

Species and Subspecies

Mexico contains almost half of the species of quail in the New World, of which four species are endemic and it is the central country for nine species (Table 1). Among the 11 polytypic species occurring in Mexico, 60 of 85 possible subspecies are found there (Blake 1977, Johnsgard 1988). Taxonomic status of many of these species has not been studied well and is in need of further clarification. For example, the ocellated quail (*Cyrtonyx ocellatus*) might be conspecific with the Montezuma quail (*C. montezumae*) (Sibley and Monroe 1990), and several subspecies of the singing quail (*Dactylortyx thoracicus*) described from southern Mexico are questionable (Blake 1977).

Distribution and Habitat

Quail are distributed throughout Mexico representing the range of diversity in habitat use found among the Odontophoridae. Among the genera found in Mexico, *Dendrortyx*, *Odontophorus*, *Dactylortyx* and *Cyrtonyx* are forest adapted and

Table 1. Species and subspecies of quail found in Mexico.

Species		Central country	Number of countries	Number of subspecies	Number of subspecies in Mexico
California quail	*Callipepla californica*	U.S.	2	8	4
Elegant quail	*Callipepla douglasii*	Mexico	1	5	5
Gambel's quail	*Callipepla gambelii*	U.S.	2	7	5
Scaled quail	*Callipepla squamata*	Mexico	2	4	3
Black-throated bobwhite	*Colinus nigrogularis*	Mexico	5	4	2
Northern bobwhite	*Colinus virginianus*	U.S.	3	22	16
Montezuma quail	*Cyrtonyx montezumae*	Mexico	2	5	5
Ocellated quail	*Cyrtonyx ocellatus*	Guatemala	5	Monotypic	
Singing quail	*Dactylortyx thoracicus*	Mexico	5	17	11
Bearded tree-partridge	*Dendrortyx barbatus*	Mexico	1	Monotypic	
Bruffy-crowned tree-partridge	*Dendrortyx leucophrys*	Honduras	6	2	1
Long-tailed tree-partridge	*Dendrortyx macroura*	Mexico	1	6	6
Spotted wood-quail	*Odontophorus guttatus*	Mexico	5	Monotypic	
Mountain quail	*Oreortyx pictus*	U.S.	2	5	2
Barred quail	*Philortyx fasciatus*	Mexico	1	Monotypic	

Callipepla, *Oreortyx*, *Colinus* and *Philortyx* are forest-edge or open-country adapted. Most of the forest adapted species have Central and South American affiliations and thereby are concentrated in southern Mexico. Whereas the more edge and open-country adapted species are found mainly in northern and western Mexico (Table 2).

Conservation Status

Surveys of all of the quail of Mexico have not been completed recently. Earlier, Leopold (1959) summarized much information on the biology and conservation status of quail and, to the present, offers the best assessment of quail in Mexico, although now badly out of data. Recently, McGowan et al. (1994) assessed the status of all the Odontophoridae using the best information available.

The bearded tree-partridge (*Dendrortyx barbatus*) appears to be the one species most critically endangered in Mexico. It was listed as Critical by McGowan et al. (1994) and Endangered/Situation Serious/Action Urgent by Collar et al. (1992). This species formerly occurred in cloud forests and adjacent pine-oak forests on the Sierra Madre Oriental, from southern San Luis Potosí to central Veracruz. Most of the scattered observations data before 1960. Surveys in the late 1980s and early 1990s indicated populations do occur in Hidalgo, Mexico, although with notable decreases in populations (Howell and Webb 1992). The major problem with this species is elimination of critical forest habitat.

The ocellated quail (*Cyrtonyx ocellatus*) is found in open pine-oak woodlands and brushy fields in southern Mexico and Central America. It was classified as a questionably Safe species by McGowan et al. (1994) and as a Restricted Range Species by Collar et al. (1992). It was reported as rare in southern Mexico during the 1970s, possibly due to overgrazing (Johnsgard 1988). In addition to removal of forests, grazing is an important threat to this species as its most important foods are underground tubers, such as wood sorrels (*Oxalis* spp.). Even though this species is geo-

Table 2. Habitat, distribution and status of quail found in Mexico.

Species	Regional affiliation	Habitat	Distribution[a]	Conservation status[b]
California quail	North America	Temperate forest edge	NW	Safe
Elegant quail	North America	Desert scrub	NW	Safe
Gambel's quail	North America	Desert	NW	Safe
Scaled quail	North America	Desert scrub	N	Safe
Black-throated bobwhite	North America	Forest edge/agriculture	Yucatan	Safe
Northern bobwhite	North America	Forest edge/agriculture	NW,NE, E,S	Safe
Montezuma quail	Central America	Open temperate forest	N,WC	Safe
Ocellated quail	Central America	Open temperate forest	S	Safe?
Singing quail	Central America	Cloud forest	Yucatan, E,SW	Safe
Bearded tree-partridge	Central America	Cloud/temperate forest	E	Critical
Buffy-crowned tree-partridge	Central America	Cloud/temperate forest	S	Safe
Long-tailed tree-partridge	Central America	Cloud/temperate forest	C	Safe
Spotted wood-quail	South America	Tropical/subtropical forest	Yucatan, S	Safe
Mountain quail	North America	Temperate forest/forest edge	NW	Safe
Barred quail	Central America	Tropical scrub/agriculture	W	Safe

[a]General location of distribution in Mexico.
[b]Criteria from McGowan et al. (1994), using Mace-Lande categories from Mace and Lande (1991).

graphically isolated from Montezuma quail, there is some question to species validity. The southern Mexico subspecies of the Montezuma quail (*C. m. sallei*) was recently listed as Near Threatened (Collar and Andrew 1988).

The northern bobwhite (*Colinus virginianus*) is among the most intensively and extensively studied species of bird in the world. Despite the much larger number of subspecies found in Mexico, little literature is available on the status and biology of this species in Mexico. Presently, one subspecies, the masked bobwhite (*C. v. ridgwayi*) has endangered species status in the U.S., where a small population exists on the Buenos Aires National Wildlife Refuge in extreme south central Arizona (U. S. Fish and Wildlife Service 1993). In addition to the Arizona population, another small population exists on a privately owned ranch in central Sonora, Mexico. Several recent sightings indicate that several smaller populations may occur within a 12.4-mile (20 km) radius of the known Sonoran population, although these sightings have yet to be verified.

Historic references indicate that masked bobwhites always were restricted to level plains and/or river valleys between 492 and 3,937 feet (150 and 1,200 m) elevation (Ligon 1952). These areas also are attractive for grazing livestock. Consequently, much of the suitable habitat in Sonora was effectively destroyed by the 1940s via severe overgrazing (Ligon 1952), and much of it remains in poor condition today. Overgrazing, coupled with a series of droughts, as well as the introduction of buffelgrass (*Cenchrus ciliaris*) for cattle forage, all have been implicated as the primary reasons for the dramatic decline of masked bobwhites. The one remaining population in Sonora now occupies open-subtropic summer-active grasslands within

Sinaloan and Sonoran desert scrub (U. S. Fish and Wildlife Service 1993). Preferred areas appear to be sites typified by deep soils that support high grass and forb diversity (Brown 1989).

Summer whistle counts of breeding males conducted along three survey routes on a privately owned ranch have been used to measure trends in abundance since 1968 (Figure 1). Fluctuations evident over this 25-year period likely are associated with the vagaries of precipitation as it affects habitat quality because little effort was expended to improve habitat conditions. Brush invasion was progressively enveloping the herbaceous openings necessary for quail production. Despite this limitation, whistle counts indicated that masked bobwhite abundance reached an all-time high by 1990. Unfortunately, the population declined rapidly shortly thereafter and by 1992 there was concern that masked bobwhites in Sonora were in imminent danger of extinction.

Fortunately, the landowner has implemented an aggressive, large-scale habitat improvement program since 1992. Since 1992, over 98,800 acres (40,00 ha) have be chained or range-disked, significantly improving habitat quality. Despite two dry summers masked bobwhite populations have remained stable. It is possible that the population increased slightly in response to habitat improvement. Winter flushing counts conducted with a trained bird dog in 1991 indicated a population of about 1,000 birds on the ranch. However, covey-call counts conducted during November 1993 generated an estimate of nearly 1,500 birds. More recent estimates obtained using line transect indicated a population of almost 3,000 birds on 4,000 acres (1,618 ha) of habitat. This estimate probably is inflated, as only 10 flushes were obtained resulting in a coefficient of variation of about 40 percent. The line-transect estimate was very similar to estimates obtained for bobwhites in southern Texas during a drought where that population was about one bird/9 acres (one bird/3.64 ha). There-

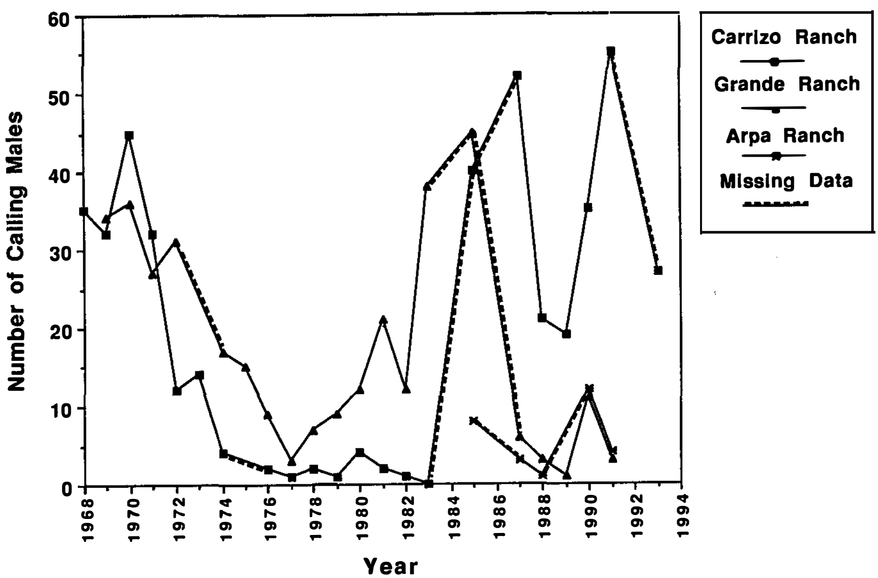

Figure 1. Masked bobwhite breeding-season whistle counts in Sonora, Mexico, 1968–1993.

fore, a more reasonable estimate for the Rancho El Carrizo population probably is 1,200 to 1,500 masked bobwhites, which is similar to the November 1993 covey-call counts. Although these are rough estimates and a number of different techniques were used to obtain them, cautious optimism regarding the eventual recovery of this masked population is warranted at this time.

From the wildlife biologist perspective, the major problem facing us for quail conservation in Mexico simply is the lack of good basic information on the species that occur there. Even major conservation assessment efforts, such as McGowan et al. (1994) are tenuous at best for most of the quail in Latin America. Without good data on populations, or even reasonable taxonomic classification, we cannot truly begin to develop conservation strategies. This is especially important for the forest species where distributions often are disjunct and isolated, and where morphological variation among populations often is quite dramatic.

Conservation and Research Strategy

The Human Aspect

The conservation status of Mexican quail cannot be addressed fully without a discussion of the socio-political situation in Mexico. Human population growth during this century has been dramatic. In 1900, the population of Mexico was 13.6 million, rising to 25.8 million by 1950 and 81.1 million by 1990 (Leopold 1959, United Nations 1992). Projections of future populations predict 154.0 million by 2025 (Vu 1985). The population also is not evenly distributed in Mexico. The densest populations are found along the Gulf Coastal Plain, the Sierra Madre Oriental and the Central Highlands.

Land-use changes have been dramatic and concurrent with changes in population, especially the conversion of natural habitats to agriculture. There is a long history of "milpas" agriculture in southern Mexico dating to the Mayans (Gradwohl and Greenburg 1988). This has resulted in a patchwork of agriculture and second-growth forest throughout southern Mexico. More recently, farming has moved well onto mountain slopes where forest destruction and erosion are serious. Even where some forests remain intact they often are grazed. Grazing also is common in grassland and desert ecosystems. In Quintana Roo there has been a loss of 40 percent of the forests since 1960 to cattle ranching. This has occurred as a result of government subsidies, even though it generally has not been profitable. Urbanization along coastal areas also has resulted in losses of habitat and further demands for agricultural products (Lynch 1992).

Among the ecosystems in Mexico, dryland and scrubland make up the largest percentage of land area (36 percent) and are the most impacted or disturbed (66 percent). This is followed by dry tropical forest (13 percent, 30 percent) oak forest (11 percent, 40 percent), grasslands (11 percent, includes natural and production grasslands), coniferous forest (8 percent, 35 percent), tropical evergreen forest (6 percent, 13 percent) and aquatic habitat (1 percent, 11 percent) (Flores-Villela and Fernández 1989). These figures already are out of date in some areas where rapid habitat conversion is taking place (Toledo and de Jesús Ordóñez 1993). For example, they suggest that <5 percent of the humid tropic zone, occurring mainly in the states of Quintana Roo, Veracruz, Campeche, Tabasco, Oaxaca and Chiapas, remains unaffected by humans (Toledo and de Jesú Ordóñez 1993).

Changes in land use in Mexico do not impact all of the quail in a similar fashion. Forest species probably have been negatively impacted to the greatest extent. All of the tree-partridges (*Dendrortyx* spp.) likely are at great risk because of removal of important forest habitats. Some other forest quail, such as the singing quail and the spotted quail (*Odontophorus guttatus*), apparently are more tolerant of habitat destruction. The black-throated bobwhite (*Colinus nigrogularis*) and northern bobwhite, which use agricultural habitat, have benefitted from land-use changes in some areas (Leopold 1959, Johnsgard 1973, Guthery 1986). A number of the grassland species can tolerate some grazing pressure, but not heavy grazing (Guthery 1986, Albers and Gehlbach 1990).

Hunting is known to be a severe problem with a number of vertebrate groups in Latin America (Ojasti 1984, Redford and Robinson 1987, Silva and Strahl 1991). The evidence of hunting impacts on quail indicates a lower use in many parts of Latin America, but is a potential problem for several species (McGowan et al. 1994).

A Conservation Perspective

The conservation of quail in Mexico requires a multi-level approach. We suggest a three-level conservation strategy.

1. Basic biology, life history and distributional research has to be undertaken. We must have a basic understanding of the distribution and life history of these species if we are to develop a conservation strategy. McGowan et al. (1994) outlined a number of research areas for Latin American quail. They suggested an extensive survey of quail distribution and populations in southern Mexico and northern Central America. Intensive surveys of the bearded wood-partridge are necessary and the Center for the Study of Tropical Birds, Inc. is initiating conservation efforts during 1994. More extensive research on the masked bobwhite also is beginning this year. Research on the biology and life history is needed on almost all of the species of quail inhabiting Mexico. We suggest priority for the three species of tree-partridges, ocellated and Montezuma quail, elegant quail, and barred quail. These species are studied little and most have rather limited distributions.
2. Human demands and perspectives need to be fully understood. We suggest research of human impacts on quail in Mexico. This could include projects dealing with direct hunting of species and impacts of changing land-use practices. We cannot overlook the importance of wildlife values, especially in rural communities.
3. Development of conservation strategies then should be accomplished using the information derived from the previous. For the Mexican quail, conservation strategies probably do not have to depend as heavily on the establishment of reserves and protection as some other groups. Although, the conservation options for reserve development based on high endemism and high diversity of avifauna, suggested by Escalante Pliego et al. (1993), would help protect a number of quail species. We suggest that a number of species are candidates for sustainable harvest and management, as with some of the quail found in the United States. The incorporation of some species into the economic structure of rural communities, as in Texas (Guthery 1986), has the potential of increasing the monetary value of conserving these species and the increase of research into their population dynamics and life history.

A View from North of the Border

Strahl (1992) outlined some strategies directed toward professional ornithological organizations to assist in research and conservation on Latin American birds. Wildlife biologists from the United States can follow the same sorts of criteria, but a number of constituency groups are available and should become involved. For example, Ducks Unlimited already has a well-developed international program involving Mexico. With respect to quail conservation, BirdLife International and the World Pheasant Association have taken a lead in the formation of the Partridge, Quail and Francolin Specialist Group. They are using the Cracid Specialist Group as a working model for quail conservation in Latin America, because of the many similarities between the two groups. Some of the very strong upland gamebird conservation groups from North America should follow this lead and increase their involvement in the region. Quail Unlimited already has included Mexico as part of its geographical area of concern. We would urge them to formalize programs in Mexico as quickly as possible.

It is also imperative that Mexican governmental agencies charged with conservation become more involved in management and research efforts. Although North American quail biologists recognize the need for strong conservation measures in Mexico, little can be accomplished without the cooperation of the Mexican government. Mexican biologists currently operate under severe financial constraints. They are aware of the lack of basic life-history information for quail; however, funding generally is unavailable for necessary research. Financial assistance from gamebird conservation groups mentioned previously would help to alleviate this problem.

In addition to monetary contributions, Mexican conservation agencies would benefit from technical assistance offered by biologists from north of the border. Establishing strong cooperative relationships among federal, state and private conservation groups in Mexico and the U.S. would help fulfill this need. For example, the U.S. Fish and Wildlife Services, Arizona Game and Fish Department, and El Centro de Ecologico de Sonora have cooperated since 1988 in efforts to restore masked bobwhite populations in Sonora. This cooperative effort successfully has maintained the viability of the one remaining population in Sonora. The cooperative approach is continuing with the initiation of a comprehensive research project through Texas A&M University—Kingsville.

Summary

Recently, Vaughan (1990) suggested that, despite the seemingly insurmountable problems found in Latin America relative to conservation issues, there is hope. Mexico's poorly performing economy is showing signs of rebounding and the expectation is for substantial economic growth in the 1990s (Shane and Stallings 1991). A strong economy in Mexico probably is requisite for long-term conservation strategies. Changes toward sustainable forestry also offer some hope for the future (Lynch 1992).

Several species of quail in Mexico probably can be utilized for sport hunting on a sustainable basis. In our search for sustainable development and wildlife values, this could become an increasingly important aspect of quail conservation in Mexico and Latin America.

The ideal of partnerships among nations in North America can provide the financial and manpower foundation for developing conservation and research programs on biota in the region. Non-governmental organizations have taken a lead in quail conservation work in the region and should continue to do so in the future. However, cooperation among governments also is necessary to ensure that the underlying socioeconomic issues which ultimately impact quail can be addressed.

Acknowledgments

We thank Keith Howman, Philip McGowan, Simon Dowell and Martin Kelsey for their contributions. The BirdLife International/WPA Partridge, Quail and Francolin Specialist Group, World Pheasant Association-International, WPA-U.K., and WPA-U.S.A. supported the Neotropical quail conservation work of John Carroll. The U. S. Fish and Wildlife Service funded masked bobwhite research by Bill Kuvlesky and Sally Gall.

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Effects of Harvest on Population Dynamics of Upland Gamebirds: Are Bobwhite the Model?

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Introduction

The effect of harvest on wildlife populations is an issue of prominent theoretical and applied interest to the natural resource profession and our society. Wildlife populations are viewed as renewable resources that provide nutritional, economic, recreational and aesthetic benefits. Compensatory mortality and density-dependent reproduction have been proposed as mechanisms that buffer harvest mortality. Traditional harvest management assumes that more animals are produced than can survive; it is believed that, up to a point, this "doomed surplus" can be harvested without affecting standing densities (Errington 1934). Relative stability of hunted populations and small differences in breeding densities between hunted and unhunted populations have been cited as evidence that hunting minimally affects abundance (Errington and Hamerstrom 1935, Marsden and Baskett 1958, Baumgartner 1944, Vance and Ellis 1972). However, despite decades of research, harvest theory of upland gamebirds remains poorly understood (Roseberry and Klimstra 1984, Robertson and Rosenberg 1988) and fundamental hypotheses regarding mechanisms of compensation remain untested (Caughley 1985).

For wildlife populations to persist under sustained harvest, corresponding reductions in natural mortality or increases in reproductive rate most occur to compensate for harvest losses (Kautz 1990). Several models have been proposed to describe the relationships between harvest, mortality, reproduction and density. At one extreme is the completely compensatory model, whereby harvest less than some threshold level does not increase seasonal or annual mortality of the harvested population (Anderson and Burnham 1976, Kautz 1990). The other extreme is the completely additive model, which suggests that any level of harvest mortality is in addition to natural mortality and reduces annual survival correspondingly (Anderson and Burnham 1976, Kautz 1990). Intermediate to these extremes is the partial compensation model, whereby harvest at any level reduces the breeding density below it's unharvested level; however, remaining individuals have enhanced survival and reproductive success and the population achieves a potential rate of increase greater than that of an unharvested population (Caughley 1985). It is this annual increase, or growth increment, that is harvested (Roseberry and Klimstra 1984, Caughley 1985, Robertson and Rosenberg 1988). The complete compensation and partial compensation models assume that reductions in natural mortality and increases in fecundity occur through density-dependent mechanisms. The completely additive model assumes that survival and reproductive success are independent of density.

Evidence suggests that the complete compensation harvest model is unrealistic and provides an inadequate basis for scientific harvest management of gamebird populations (Roseberry 1979, Roseberry and Klimstra 1984, Potts 1986, Curtis et al. 1989, Pollock et al. 1989b). However, the contribution of previous research to our understanding of the effects of harvest has been hindered by numerous factors, including insufficient sampling of marked animals, unreliable population estimates, lack of experimental control, lack of replication, insufficient understanding of the veracity of underlying assumptions, reliance on correlations and demographics, lack of independence between density and survival estimates, and improper analysis.

As our profession faces the 21st century, there is an increasing need to understand the mechanics of upland gamebird harvest to support both harvest recommendations and management practices with defensible population performance data (Murphy and Noon 1991, Nudds and Morrison 1991). Caughley (1985) suggested that we will not have sufficient data to answer questions regarding effects of harvest if we are content simply to monitor population responses to harvest intensities that we do not control. Careful manipulation of harvest regulations provides the only opportunity to rigorously test hypotheses regarding the compensatory nature of harvest mortality and reproduction. We concur with Caughley (1985) that wildlife harvest management will not become scientific until thorough experimentation has been conducted.

Our objectives are to suggest that upland gamebird management in the 21st century will require a greater understanding of the effects of harvest on game populations; describe why, among upland gamebirds, northern bobwhite (*Colinus virginianus*) may be a model species for testing compensatory harvest theory; describe potential experimental designs; and suggest mechanisms for conducting an operational multistate research effort.

Essential Population Parameters

The most commonly suggested mechanisms by which harvest mortality may be compensated for are density-dependent mortality, density-dependent reproduction and/or density-dependent emigration/immigration (Potts 1986, Robertson and Rosenberg 1988, Kautz 1990). To test hypotheses adequately concerning density-dependent mortality and reproduction, numerous population parameters, including density, survival, harvest rate and reproduction, must be estimable with little bias and known precision. Although density estimates are essential to such an endeavor, estimates of density relative to carrying capacity, although difficult to determine, would be most useful (Kautz 1990, McCullough 1990).

The extent to which hunting mortality is compensated for by a reduction in natural mortality is central to an understanding of the effects of harvest on populations (Roseberry and Klimstra 1984, Caughley 1958). Because the relationships among survival, breeding density and reproduction are complex, estimates of annual survival alone may be misleading. As noted by Roseberry and Klimstra (1984), the relationship between hunting and natural mortality prior to the breeding season is the central issue. Therefore, the seasonal timing and nature of mortality is critical to evaluating the potential additive nature of harvest mortality. Tests of compensatory reductions in natural mortality require survival estimates from early autumn to the beginning of the breeding season. However, more specific seasonal survival estimates from early autumn to the end of hunting season, from the end of hunting season to beginning of the breeding season, and from the beginning to the end of the breeding season would allow identification of mechanisms regarding the timing and nature of compensation.

Density-dependent reproduction might occur through variation in any one or a combination of the components of reproductive success. Identification of the mechanisms of compensation and testing hypotheses regarding the density dependence of reproduction require estimates of reproductive effort, clutch size, nest success and brood survival.

Why Bobwhite?

Effects of harvest are important for all hunted upland gamebrids. However, critical hypotheses of density dependence are not equally testable for all species because of differences in the magnitude of expected response, variation in expected response, environment and measurement error. McCullough (1990) identified three sources of variation, or ''noise'' that affect detection of density-dependent population responses in empirical tests. He suggested that density dependence would be detectable only if the population response is greater than the sum of variation in population response to a given level of density (alpha noise), environment (beta noise) and measurement error (gamma noise). Consequently, density dependence will be most detectable for species that exhibit a strong density-dependent response relative to the magnitude of these sources of noise. Weak population response and large variation in response, environment and measurement will make demonstration of density dependence difficult for some species (McCullough 1990).

McCullough (1990) suggested that species with high biotic potential can exhibit greater response to perturbations of density; therefore, density dependence will be more detectable for these species than for those with little physiological range in reproductive capacity, given equal noise. Few upland gamebirds possess the reproductive capacity exhibited by the northern bobwhite. Bobwhite lay large clutches (13–15) and renest multiple times (Roseberry and Klimstra 1984, Curtis et al. 1993, Suchy and Munkel 1993, Burger 1993). They also exhibit a reproductive system that is dynamic, responding to population density, weather, resource availability and physiological condition (Burger 1993). This combination of physiological capacity and dynamic mating system permits rapid and dramatic response to low densities or abundant resources unparalleled among other gamebirds.

Noise attributable to population response (alpha) and environmental variation (beta) will occur in any field test of density dependence. There is little opportunity to manage these sources of noise. However, measurement error can be minimized if precise, unbiased estimates of density, survival and reproductive success are obtainable. Line-transect (Guthery 1988), drive counts (Dimmick et al. 1982, Janvarin et al. 1991), Lincoln-Peterson (Dimmick et al. 1982) and capture/recapture (O'Brien et al. 1985) population estimators have been applied to bobwhite populations and sources of bias and variation identified. Although none of these estimators perform perfectly under all habitat conditions, bobwhite may be more easily censused throughout the year than other species of North American gamebirds.

Measurement error associated with survival and reproductive success primarily is a function of the parameter estimate and sample size (Pollock et al. 1989a). Consequently, density dependence will be most detectable for species of which very large sample sizes may be obtained. Recent studies of bobwhite have provided estimates of population parameters and associated variances and demonstrated that large sample sizes can be obtained (>100/factor level) (Curtis et al. 1989, Pollock et al. 1989b, Burger 1993). Estimates of variation reported in these studies can be used to develop harvest experiments with known power.

Bobwhite are nonmigratory, widely distributed and locally abundant. These attributes make them more suitable as a model than other upland game species (mourning doves [*Zenaida macroura*], gray partridge [*Perdix perdix*], or prairie grouse). Unlike pheasants (*Phasianus colchicus*) and wild turkeys (*Meleagris gallopavo*), both sexes are harvested. Bobwhite typically do not exhibit the cyclic population trends of ruffed grouse (*Bonasa umbellus*) (c.f. Roseberry and Klimstra 1984). The high population densities reported for bobwhite populations would reduce logistic constraints by facilitating the acquisition of large sample size in a smaller study area than would be possible for many other species. The sedentary nature of bobwhite populations would further reduce logistic constraints.

Much of the theory on which gamebird harvest has been predicated was hypothesized from bobwhite research. We suggest that among North American gamebirds, bobwhite are the most amenable to rigorous, experimental testing of compensatory harvest mortality. We are not suggesting that results from such a study could be extrapolated to other species, merely that bobwhite could be the species for which we can most realistically address this issue.

Potential Experimental Designs

Caughley (1985) stated that information from carefully designed and executed replicated experiments is necessary to substantially improve current non-scientific harvest management. He suggested that response to at least three harvest levels (including zero) should be monitored and that such experiments should run for at least two generations of the harvested species to ensure that populations had achieved equilibrium under a constant harvest rate. Caughley (1985) proposed that harvest could be plotted against density, hunting effort or resource availability to determine which harvest model is correct and the nature of the yield curve. We concur with Kautz (1990) that "the concept of compensatory mortality is useful only if it includes more explicit consideration of the mechanisms involved," and suggest that such experiments would be most useful if they not only determine which harvest model is correct, but also identify the mechanisms of compensation and quantify the relationships among not only harvest and density, but harvest, density and the individual components of population performance. Toward this end, we propose a regional study, replicated among states, that would rigorously address effects of harvest on bobwhite populations.

We suggest two potential experimental designs to accomplish this end. The common characteristics of both designs first are discussed and then the two designs are presented in increasing order of reliability of resulting information, power to detect real relationships and fiscal commitment. The proposed designs are intended to allow rigorous testing of six hypotheses regarding differences among bobwhite populations under three levels of exploitation. The hypotheses are: (1) survival from early autumn to the end of the hunting season does not differ; (2) survival from the end of hunting season to the beginning of the breeding season does not differ; (3) survival from early autumn to the beginning of the breeding season does not differ; (4) survival of adults during the breeding season does not differ; (5) reproductive effort (percentage of spring population incubating 1 or more nests, and mean number of nests incubated per adult in the spring population) of both males and females does not differ; and (6) nest success does not differ. The testing of these hypotheses will identify timing and nature of compensation and density-dependent reproduction.

Specifically, rejection of hypothesis (3) and an inverse relationship between harvest rate and survival would demonstrate the additive nature of harvest mortality and imply a depressing effect of harvest on breeding density. Rejection of hypothesis (1) with lower hunting season survival under higher harvest and hypothesis (2) with higher post-harvest survival under higher harvest would imply compensatory mortality and identify the seasonal timing of compensation. Rejection of hypothesis (3) with lower autumn to spring survival under high harvest and hypothesis (4) with higher breeding season survival under high harvest would imply that harvest mortality is not fully compensated for during the autumn-spring period, but additional compensation may occur through density-dependent reductions in breeding season mortality. Rejection of hypothesis (3) with lower autumn to spring survival under high harvest and hypothesis (5) or (6) with higher reproductive effort or success under high harvest would imply that harvest mortality is not fully compensated for during the autumn-spring period but subsequent reductions in breeding populations may be offset by density-dependent reproduction.

General Considerations

The ability to completely control timing, intensity and distribution of harvest is paramount to any design. Participating states each would conduct one replication of all treatments. Within each state, three harvest rates (0, 20 and 40 or 0, 15 and 30 percent), including unretrieved kill, would be randomly assigned to three study areas. Study areas, within states, should be approximately similar in habitat composition and carrying capacity, and located within close proximity to be exposed to similar weather conditions, yet distant enough to minimize movement of animals between areas. Within each area and state, density, harvest rate, survival, reproductive effort and reproductive success would be monitored using standardized procedures. Complete research protocol would be developed by a project steering committee composed of research representatives from participating states. However, we suggest that radiotelemetry is the only method available to adequately estimate seasonal and annual survival and reproductive success. Harvest rate should be estimated both by percentage of prehunt population removed and cause-specific mortality rates from telemetry survival data (Heisey and Fuller 1985, Burger 1993). Either line-transect (Guthery 1988, Buckland et al. 1993) or drive-count (Janvarin et al. 1991) methodology could be employed to estimate density and radio-tagged individuals could be used to evaluate validity of census technique assumptions. Bird dogs and closely spaced multiple counters could be used to increase the effective strip width of line-transect estimates (Guthery 1988, Buckland et al. 1993). Area- and census-specific detection rates of radio-tagged coveys or individuals could be employed to correct drive count estimates (Janvarin et al. 1991).

McCullough (1990) noted that density-dependent response is likely to be greatest, and therefore most easily demonstrated when populations are either near carrying capacity or very low. Monitoring of low populations presents special difficulties in obtaining adequate sample sizes to estimate population parameters with acceptable precision. Therefore, we suggest that populations on all study areas first should be monitored for three years under a zero harvest regime. This would permit the collection of baseline population performance data for each population to facilitate pre- and post-treatment comparisons. Second, it would reduce residual effects of previous harvest-management regimes and put each population in a more similar position relative to carrying capacity. Finally, this equilibration period would allow each population to approach carrying capacity where density-dependent response to population reductions may be demonstrated more easily.

Randomized Block Design

Under the randomized block design, harvest treatments would be randomly assigned among the three areas within a state and bobwhite population density and performance measures would be monitored on each area for three successive years after the initial three-year equilibration period. Three years of data under a specified harvest regime would allow populations to equilibrate and permit estimation of annual variation in population response to a consistent harvest intensity. The relationships between density and harvest rate, density and reproductive effort and success, and density and survival could be tested within each state using appropriate regression methodology. Overall, post-treatment differences in survival and reproduction could be tested among harvest levels by treating states as blocks in a randomized block ANOVA. This design would take six years to complete.

Cross-over Design

The design that will allow the most rigorous test of hypotheses concerning compensatory mortality and density-dependent reproduction involves long-term studies on multiple populations where every treatment is sequentially imposed on each population for three-year treatment periods (Kautz 1990). Such a cross-over design would remove the effect of variation among areas by allowing each treatment to be imposed on each area. We propose a latin square design that would allow tests for main effects of harvest intensity, as well as residual effects of previous harvest level (Cochran and Cox 1957: 133–139). In this design, each state would constitute a 3 by 3 latin square with columns representing study areas and rows representing the mean of three-year treatment periods. Within each state, each harvest level would be imposed on each area for a three-year period and treatments would be rotated between periods. After nine years of treatments, all areas would have received each treatment for one three-year period. A balanced design for testing of residual effects would require the number of participating states to equal a multiple of two. As in the previous design, a three-year zero-harvest equilibration period is recommended prior to implementing treatments. As noted by Kautz (1990), during the first six years, this design is equivalent to the randomized block design and would provide partial results in this time period. This design would require 12 years to complete (three-year pretreatment no harvest equilibration period plus three years of each of three harvest levels), but would control for variation among experimental units within a state and permit tests of residual effects. This design would substantially increase the power of the experiment to detect treatment effects.

Precision and Power

The precision associated with Kaplan-Meier survival estimates derived from radio telemetry will be a function of both survival rate and number of animals radio marked at the end of the interval of interest (Pollock et al. 1989a). Kaplan-Meier annual survival estimates in the range of 0.10 to 0.20 will have associated standard errors of ≤0.013 to 0.025 if at least 50 animals are at risk at the end of the interval. Seasonal survival in the range of 0.20 to 0.50 can be estimated with standard errors of ≤0.025 to 0.05 given a similar sample size. Previous radio studies of bobwhite suggest that at least 100 birds/area/year must be radio marked to maintain adequate sample sizes during the autumn-to-spring period (Burger 1993). An additional 50–75 birds must be marked prior to the nesting season to produce 25–35 nests from which reproductive success can be estimated (Burger 1993, Curtis et al. 1993, Suchy and Munkel 1993).

Determining the power associated with a potential research design requires an estimate of the magnitude of expected-treatment effect in relation to anticipated within-treatment variation. Roseberry and Klimstra (1984) reported a 0.114 difference in mean autumn-spring survival between a hunted (0.45 harvest rate) and an unhunted area in southern Illinois. Curtis et al. (1988) reported a 0.196 difference in mean annual survival between a hunted area in North Carolina (0.14 harvest rate) and unhunted area in Florida. In calculations of anticipated power, we used a conservative estimate of 0.10 as the expected treatment effect between 0.0 harvest rate and 0.40 harvest. We assumed that the 0.20 harvest treatment effect would be at the midpoint between 0.0 and 0.40 harvest. To estimate expected within-cell (block by treatment) variation under a constant harvest regime, we used the standard deviation (0.047) of 14 years of annual survival rates reported in Pollock et al. (1989b). Based on these assumptions, and at least six states participating in the study, the proposed randomized block design would have a power of ≥0.666 and the latin square ≥0.983 to detect a 0.10 treatment effect of harvest on seasonal and annual survival (Cohen 1977).

Tactical Step-down Planning

A large-scale and long-term cooperative study of this nature requires considerable planning prior to initiation. Organizations that invest in this endeavor must have some degree of assurance that their resources will be used efficiently, the inferences from the research will be reliable and the research will be applicable to management needs. Similarly, the biological and statistical reliability of this study, as is the case with any research, is crucial because inferences may have profound and far-reaching implications. Concerns about the credibility of principles and practices of wildlife management and research are increasingly prevalent in the natural resource literature (Romesburg 1981, Hurlbert 1984, Stauffer 1993). Research based on untested or incorrect assumptions could provide erroneous evidence that may unnecessarily restrict upland gamebird hunting, or conversely promote overexploitation.

To ensure that this research has a high probability of producing statistically valid and biologically meaningful information, we must examine the intellectual foundation on which it is built. Methodological assumptions inherent to radio-telemetry studies (e.g., bait-trapping as random sample, effects of radio transmitters on reproduction and survival) and density estimators (detect all birds on the transect, animals do not move away from the transect, detection probability independent of group size) must be evaluated for their potential to affect results and interpretation. In addition to methodological problems, field experiments inherently possess a high degree of variability because of lack of experimental control over factors such as weather, population structure, genetic characteristics, predator density and habitat composition. Potential sources of variability and error must be carefully identified and controlled if possible.

In order to demonstrate how the objectives of this study can be accomplished in a predictable fashion, we recommend tactical planning, as described by Phenicie and Lyons (1973), proceed study initiation. These authors present step-down research planning as a guiding philosophy of conducting research. A step-down plan would identify biological and analytical elements fundamental to the intellectual foundation of the harvest investigation. With each element the following determinations are made:

1. Is the element trivial?
2. Is current knowledge of quail or other species sufficient to answer the question at hand?
3. If not, is new research needed?
4. How much of a risk do we take if we make assumptions about the element and move to the next level?
5. Is the cost or effort to address a question too great for the knowledge gained?
6. If we cannot address this question, can we proceed with further studies?

Step-down research planning provides several benefits: (1) it will identify priority research subjects; (2) it will eliminate unnecessary duplication of efforts; (3) it will focus our efforts on mechanisms that explain compensatory responses; and (4) it will demonstrate that the ultimate objective of testing for population compensation is attainable. The plan would attempt to identify the biological and analytical elements fundamental to the foundation of the harvest experiment. Some elements identified through the planning process may be sufficiently addressed in the literature while others may need to be answered prior to the large-scale study becoming operational. Although step-down planning and associated research will require considerable time and resources, we believe it is a necessary step toward achieving meaningful results from this proposed collaborative, large-scale study.

Operating Considerations

Due to the financial resources necessary to conduct our proposed experiment, it is unlikely that any one agency will be able to provide funds to sufficiently address the objectives. Therefore, cooperative efforts among states and universities will be necessary to complete this research. Numerous examples of cooperative research can serve as models under which a workable protocol can be developed. We believe the first step has been introduced with the Bobwhite Research Initiative meeting held in conjunction with the 55th Midwest Fish and Wildlife Conference in December 1993 (T. V. Dailey personal communication). We believe that a project steering committee is necessary to guide the development of this study. The Bobwhite Research Initiative group may serve as a coordinating body. We suggest that formation of a Coordinated Research Committee within the International Association of Fish and Wildlife Agencies may provide more formal support for cooperative research activities.

We estimate funding needs for the project will approach $100,000 per year for each participating state. Obviously, this level of funding will not be feasible for all states, therefore, alternative funding sources will play a pivotal role in the development of this and similar cooperative studies. In addition, participants should coordinate efforts to develop an outside funding base. We believe alternative funding could be solicited by both state agency and university cooperators through the Federal Aid process, private foundations, non-governmental conservation organizations (e.g., Quail Unlimited) and private sector businesses, such as firearms and ammunition manufacturers.

Conclusions

Within the wildlife profession, state agencies traditionally, and rightfully, have conducted research primarily within state boundaries on questions of state-level importance. While these efforts often include cooperation among state agencies and universities, attempts to adequately test hypotheses often have been constrained by limited funds. As we move toward the 21st century it is paramount that state agency research programs recognize the need for cooperative research to address broader questions and develop the commitment to implement such programs. Within natural resource agencies, the future will require hard choices concerning the acceptability of the level of knowledge with which we now make decisions and the amount of resources that are allocated to the acquisition of knowledge. States need to grapple with the reality that the acquisition of some knowledge clearly is scale dependent.

We contend that reallocation or redistribution of resources and a willingness to cooperate in regional studies will be necessary to expand our understanding of the role of harvest in upland gamebird management. Among upland gamebirds, we believe that bobwhite offer a unique model for testing hypotheses regarding compensatory mortality and density-dependent reproduction. A recent paper by Hanley (1993) suggested that, when researchers are asked to address questions that are either too "big" or cannot be answered unambiguously, their shortcuts to practical utility will be unproductive. We contend our profession has spent decades chipping away at many questions from a standpoint of practical utility. If the science of gamebird harvest management is to progress, we will need to work cooperatively to secure sufficient funds to develop and conduct research that adequately tests hypotheses regarding relationships between harvest and population performance.

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Experimental Designs for Assessing Impacts of Predators on Gamebird Populations

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Introduction

Predators have been an important component of man's environment since prehistoric times. They once were a real danger to prehistoric man and primitive cultures. As man developed culturally, predators became less of a danger and more of a competitor. These efficient killers of valuable and, at times, scarce food supplies were generally viewed as vermin. Charles Darwin (1859) wrote "that there seems to be little doubt that the stock of partridges, grouse, and hares on any large estate depends chiefly on the destruction of vermin. . . ." In North America, as settlers cleared land and became more urbanized, the philosophy that predators were useless competitors and vermin pervaded.

During the 1600s, many American colonies established bounties for predators, following the practice of their European counterparts. By the early 1900s, large carnivores, such as mountain lion (*Felis concolor*) and timber wolf (*Canis lupus*), were extirpated from the eastern United States. Predator control was prevalent in the west, primarily as a result of ranching interests, and was viewed as a viable wildlife management technique (Leopold 1933) until the mid-1900s when biologists began to question its viability (Leopold 1964, Cain 1972).

Research on predator control indicated that it was not economically nor biologically feasible in the long run (MacDonald and Jantzen 1967, Balser et al. 1968, Knowlton 1972, Beasom 1974b, Trautman et al. 1974, Connolly and Longhurst 1975, Guthery and Beasom 1977). However, other studies reported positive responses of gamebird populations to predator control (Lignon 1946, Beasom 1974a, Potts 1986, Tapper et al. 1991). Consequently, during the past two decades there have been efforts to identify alternative methods, such as habitat management, to reduce predation on gamebirds and mammals (Duebbert and Kantrud 1974, Barrett 1981, Kenward and Marcstrom 1982, Metzler and Speake 1985).

Many states are experiencing expanding predator populations, in part a result of the success of the anti-trapping and anti-fur programs or policies. Many wildlife biologists now are examining the interaction of predators and gamebirds. Additionally, many gamebird populations, including the northern bobwhite (*Colinus virginianus*) (Brennan 1991), other quail species, the eastern wild turkey (*Meleagris gallopavo sylvestris*) (Palmer et al. 1993) and the ring-necked pheasant (*Phasianus colchicus*) (Petersen et al. 1988), have experienced population declines and/or poor reproduction. This has forced biologists to quickly, and perhaps erroneously, find "quick and dirty" solutions to sportsmen's demands for sustaining huntable populations. Consequently, the call again has risen, although subtlely, for predator control, improved habitat management or both.

Therefore, it is imperative that before any management activity is initiated to counter declining gamebird populations, specifically predator control, biologists and managers must understand better the relation of predators to their prey populations. Numerous reviews (e.g., Sih et al. 1985, Peek 1986, Reynolds et al. 1988, Newton 1993) concerning predation and gamebird population dynamics provide diverse viewpoints concerning how predators impact their prey. There is only one definitive conclusion: the process of predation and its relation to game-management objectives are complex processes and require more definitive research. It is time for comprehensive research programs to be initiated that address the question, "What do we really know about predators, predation and gamebird populations?"

In this paper, we develop a generalized experimental design regarding research on the interrelation of predation and gamebirds, although much of the material will apply to any research program regarding predation and game animals. We do not provide a specific design, but a generalized framework for biologists to follow, thereby ensuring consistent, regionally applicable results and conclusions.

Case Study

The framework of our proposed research protocol is based on two long-term studies of wild turkey population estimates and dynamics, habitat use, and other aspects of turkey population biology including harvest by man in central Mississippi. The first project began in 1983 and the second began in 1986. Both projects remain active. Based on results of the first study, we determined that predation, particularly predation on eggs, poults and reproductively active hens (i.e., incubating, brooding), was the main factor affecting turkey density (Seiss et al. 1990, Palmer et al. 1993).

Consequently, a five-year program studying the main predators (e.g., raccoon [*Procyon lotor*], opossum [*Didelphis virginiana*], gray fox [*Urocyon cinereoargenteus*], coyote [*Canis latrans*] and bobcat [Felis rufus]) was initiated in 1988 on one area. A second area has been identified and, as funding becomes available, a turkey predation study will be initiated. Telemetry studies are being used to document predator habitat use and preference, movements and activity patterns, survival rates, and population status sympatric with wild turkey hens and the subsequent reproductive efforts of those hens.

We also are conducting hunter surveys to assess their opinions of management options on the research area and statewide. We maintain weather stations throughout the study area to monitor local weather patterns. Annual vegetation surveys and small mammal monitoring are performed to assess annual changes in habitat quality. Finally, an extensive geological information system (GIS) has been developed for the 30,000-acre study area and for another 30,000 acres of adjacent land, public and privately owned.

Basic Theory of Predation

Before we discuss experimental designs required for predation research, we must review basic concepts that will have application, and therefore underlay our research design and philosophy. These concepts have been reviewed and discussed thoroughly elsewhere (Errington 1946, Sanderson 1972, Taylor 1984, Newton 1993). We will discuss them briefly to provide a framework from which to develop a research protocol.

Predators can either (1) limit or regulate prey populations (Errington 1943, Mech 1970), (2) increase vigor of a prey population by eliminating sick or unfit individuals (Markley 1967, Hudson 1986), (3) maintain prey wildness (Leopold 1944, Hornocker 1970), or (4) maintain community stability (Connell 1970, 1972, Paine 1980). The first three effects are of interest to the game manager, with the first being most important. Regarding the first effect, it is important to identify whether predators regulate or limit a prey population. Regulation implies that predators are removing the "surplus" that otherwise would die from other mortality factors. Limitation only implies that a predator (or predators) serves as an important mortality agent, but does not imply that the prey populations are kept within available resources (prey carrying capacity). Obviously, for the wildlife manager, a predator that regulates its prey population is of greater importance than a predator that simply limits the prey.

The concept of regulation versus limitation extends to another aspect of predation: whether predation is density dependent and, therefore, a stabilizing effect or density independent and, thus, destabilizing. Generally, predation is considered a density-dependent process with predation rate (functional response) and predator numbers (numerical response) increasing with increasing prey density (Solomon 1949, Holling 1959); however, there are studies indicating the opposite (Hudson 1992, Kenward 1985, Newton 1992).

One final aspect of predation is whether it is additive or compensatory to other mortality agents of prey. Most studies indicate that predation is compensatory (the "doomed surplus" concept of Errington [1946]) to a point. Predation, or any mortality agent, becomes additive above a threshold. For example, Baumgartner (1944) hypothesized that between 20 and 55 percent of northern bobwhites in Oklahoma may be removed without impacting the subsequent breeding population. Thus, predation losses exceeding 55 percent would be expected to be additive. Additionally, for ruffed grouse (*Bonasa umbellus*), Palmer and Bennett (1963) stated that 50 percent of the population could be harvested. The problem arises when the "surplus" is placed in the hunter's bag rather than the predator's stomach.

Predation is a function of many interrelated factors. In fact, many of the factors discussed cannot be easily separated and evaluated; they are interdependent. Bailey (1984) summarized the components that affected predation, including speed of numerical response of predators to changes in prey abundance, prey diversity, biotic potential and longevity of prey and predator, habitat preferences of predators and prey, intrinsic regulation of predator numbers, level of interspecific competition among predators for prey, habitat quality, prey health and vigor at high densities, dispersal rates of subordinate prey animals in poor habitats, adaptability of prey, degree of mutualism within prey species, prey and predator learning, and degree of cooperative hunting by predators. Obviously, this lengthy list reflects the complexity of the process of predation. Additionally, these factors only concern the predator and prey, and do not include effects of other environmental factors, such as weather, habitat modification and hunting. It is easy to see why much uncertainty exists regarding predator/prey relationships.

Research Priorities

Past research on predation, in the theoretical and applied contexts, provides a sound basis for management decisions. However, biologists are faced with more complex issues other than the impact of predators on gamebird populations. Broader issues, including biodiversity, community function, ecosystem management and landscape ecology, are becoming more prevalent. Additionally, a more urbanized society is sensitive to animal rights issues, and biologists and sportsmen must view predators as viable components of ecosystems, rather than as non-game animals potentially removing the "harvestable surplus." The terms "vermin" and "furbearers" are no longer acceptable.

Therefore, future research designs must incorporate a broad spectrum of environmental components. Past research generally examined simple predator/prey (game) systems (Figure 1). Perhaps the remaining uncertainty regarding effects of predators on game populations is not due to poor research designs, but from too narrow research designs. Research biologists now must incorporate additional variables that expand beyond the one or two predators of interest and the game animals, and include an array of environmental factors that have been cited by numerous authors as playing an important yet unassessed role in predation (Figure 2). It is time to examine animal/plant communities, or even ecosystems, and include into future research designs, an important yet under-studied variable: man as the main predator, and the effects of his/her hunting activities.

There are many objectives required to adequately assess predator/prey interactions. Although many past and current research projects have addressed these objectives, they often fail to include an important variable or variables. Important objectives that should be addressed include:

1. Importance of buffer species on predation rate of gamebirds.
2. Relation of land-management practices on predator and prey movements, habitat usage and reproduction.
3. Interspecific competition of predator species and its relation on predation rate of gamebirds.
4. Importance of weather on predator and prey movements, habitat usage, reproduction and its relation to predation rates of gamebirds.
5. Impact of predator control on all predator and prey species (both game and suspected buffer species), including both target and non-target predators.
6. Relation of hunting by man and by natural, free-ranging predator species on gamebird populations, in the context of compensatory versus additive mortality.

Experimental Designs

Determining if predation is an ultimate or proximate mortality factor is foremost to any research program on this topic. Future research designs will be complex. Biologists cannot afford to expend inordinate amounts of time and funds assessing predation when in fact, predation may be a symptom not a cause of low gamebird numbers. For example, high predation rates may result from poor land-management practices and minor changes in land-use patterns may be the solution. Therefore, pilot studies to determine pre-treatment conditions may be necessary to first identify ultimate mortality factors, including diseases, habitat quality, etc. If predation is identified as a key mortality factor, then testable hypotheses may be developed and research studies initiated.

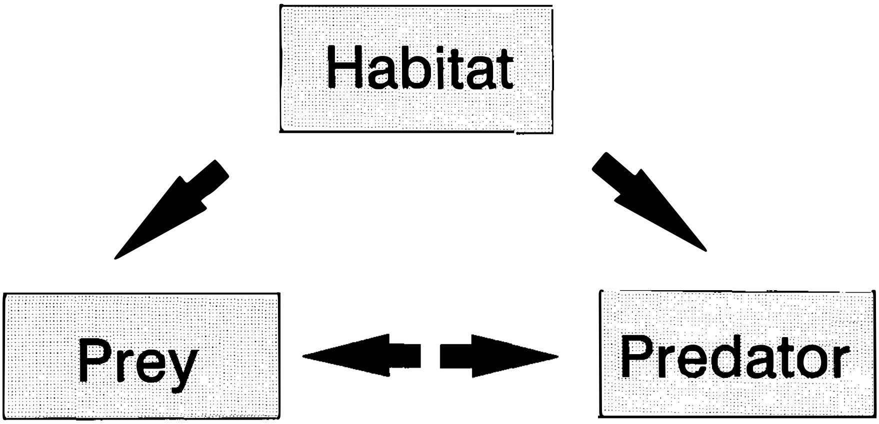

Figure 1. Format of traditional predation studies of examining simple single predator/prey systems.

Any study regarding a predator/prey interaction requires the collection of some basic data. Minimally, this data set should include for both predator and prey populations: a temporal (annual, seasonal) assessment of numbers (census), annual reproductive rates, indices of health, and age/sex specific mortality and harvest rates (including trapping or removal rates of predators). Our review of past research programs included some measurement of the above variables. Studies that failed to measure these variables usually ended up hypothesizing about them from casual observations (e.g., a drought in 1985 probably reduced seed abundance and thereby might have caused the larger movements of rodents), thereby subjecting their conclusions to criticism.

Ideally, study of a predator/prey interaction should include additional components. They are discussed below.

Time

Because of budgetary constraints, most studies extend for two to three years, but rarely beyond five years. For example, the Pittman-Robertson funding schedule of state game agencies is on a five-year cycle. It is important that all possible variables be evaluated within a maximal range of values. A two- to three-year study usually fails to assess animal responses to a diverse range of weather variables. It is imperative that studies extend beyond this three- to five-year barrier. Additionally, the need for assessing predator/prey interactions over the long-term is enhanced when we consider that numerous studies have found that predators and their prey cycle (regardless of whether it is a cause-effect process), and these cycles may be from 3–4 years to 9–10 years in length. It therefore seems logical that studies concerning a predator and its prey should extend at least 10 years.

An additional aspect of time concerns public support of long-term research programs. Often, the attention span of the public regarding environmental issues is relatively short. By the time biologists adequately respond to public concerns through management and/or research, public sentiment has changed to another issue. This incredibly short attention span by the public relates to their lack of support for

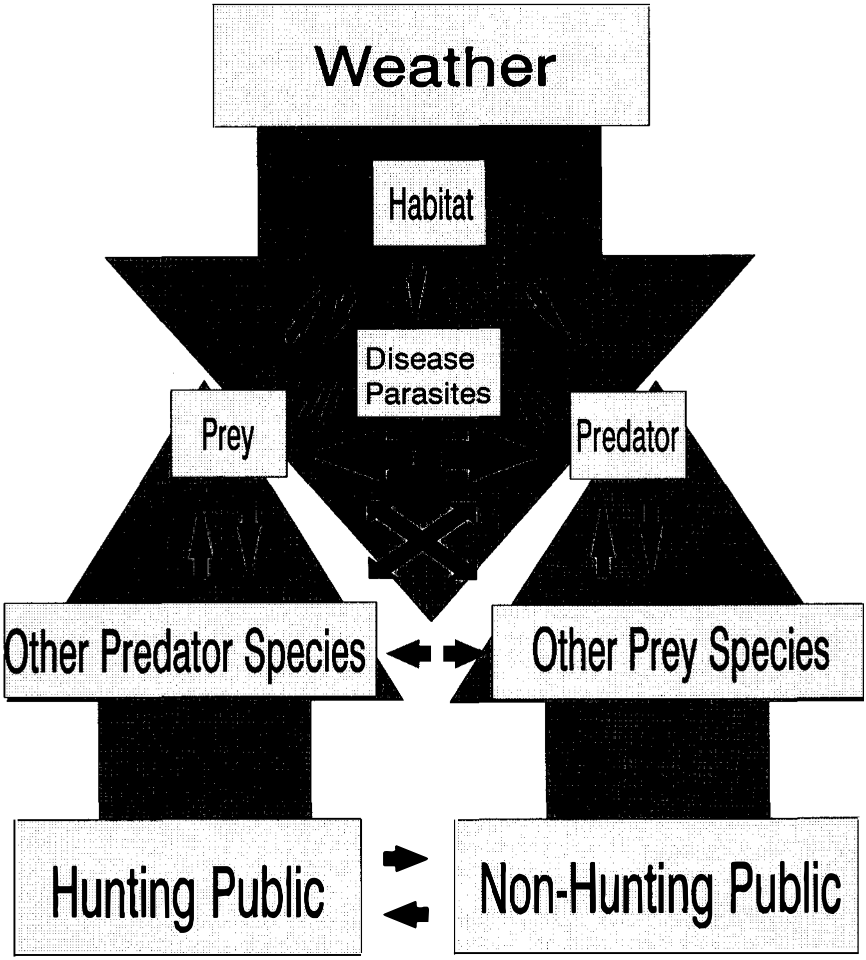

Figure 2. Proposed format of predation studies, including biotic and abiotic components of the environment.

long-term research studies and biologists must consider this problem when addressing the need for 10–15 year studies.

The Animal/Plant Community

Research scientists should include all major animal/plant species that may affect the predator/prey process when formulating a project. Numerous biologists (starting with Leopold [1933]) recognized the importance of buffer species. However, most failed to coincidentally monitor abundance, survival and reproduction of alternate prey species, and how seasonal and annual fluctuations of their numbers affect the predator/prey system. Numerous studies, many with surprising results, have demonstrated the importance of assessing species in addition to the predator(s) and prey (gamebird) of interest (e.g., Schnell 1968, Angelstam et al. 1984, Kenward 1985).

In addition to monitoring dynamics of major prey species, dynamics of predator species also should be monitored. An individual predatory animal not only interacts with its prey species, but with other predator species as well. It therefore is imperative that future research designs include monitoring systems for all major predators. Additionally, it is not sufficient to merely examine predator numbers, but also predator population health, specifically diseases. Most nest predators of gallinaceous birds (red fox [*Vulpes vulpes*], grey fox, skunk [*Mephitis* spp.] and raccoon) are very susceptible to distemper and rabies. It is important to assess the impact of these diseases on predator populations and subsequently gamebird survival and reproduction.

Finally, most prey species are granivorous (rodents) or herbivorous (lagomorphs). It therefore is important to monitor phenology of important plant species, both in the context of plants as a food supply and as structure for escape and breeding cover. Both of these components associated with the plant community are temporally dynamic, and should be examined seasonally and annually.

Abiotic Factors

Weather is an important factor that affects animal and plant distribution, abundance and productivity. Any comprehensive study of predator/prey interactions should monitor basic weather variables including precipitation, temperatures, relative humidity and wind velocity. Additionally, such variables as time of sunrise and sunset, passage of frontal systems, cloud cover and moon phase (not to assess cycles but to examine predator and prey movements) should be examined. These variables will enable biologists to explore the interaction of weather variables and plant phenology, animal movements, survival and reproduction, and even hunter success. Last, most gamebirds are ground nesters and it is important to assess microclimatic and foliar aspects of the nest site and its relation to probability of predation.

Treatments, Controls and Replication

Treatments should include areas with and without (control) predator removal. Past studies attempted to eradicate all or most of the predators from the treatment area (Beasom 1974a, Beasom 1974b, Trautman et al. 1974, Tapper et al. 1991). To better assess the functional relationship between prey (gamebird) density and predation rate, treatments should include (1) varying levels of predator removal, and (2) varying densities of the gamebird population. Biologists should recognize that complete predator control (eradication) is not feasible. Studies should determine what predator densities are tolerable and determine the appropriate level of predator control, assuming that the gamebird population is responsive to predator removal.

Regardless of whether a study concerns predators and prey, controls (areas with no treatment effects) are necessary to assure that observed changes are from the treatments applied, rather than from random processes. Replicate areas of the same and differing vegetative cover types are essential if biologists want to apply results over a broad geographic area. Therefore, study areas should be replicated locally as well as regionally to ensure a concise understanding of predator/prey interactions.

Crossover designs also are important and have been used effectively by many biologists studying effects of predator control. Crossover designs are those in which the control becomes the treatment and the treatment becomes the control after a specified time. An important consideration is when the crossover should occur, and as previously indicated, 5 years is minimal, implying that the study should last 10 or even perhaps 15 years to ensure adequate assessment of any cycles if they should arise.

Finally, distance between treatments should be long enough to ensure that predator and prey species cannot move between them, thereby confounding results. Ideally, distance between study areas should be at least as great as the maximal radial movement of all animal species under study. Some studies have addressed this by using islands; however, interpretation of densities of predators and prey should be cautious as dispersal is not possible and unnatural crowding may occur.

Habitat Quality

Habitat factors are integrated with previous components (assessing abundance of major prey species, monitoring phenology of plants, weather factors, etc.). However, research biologists need to ensure that habitat quality is assessed, both seasonally and annually. Additionally, habitat quality should be assessed on the macro scale (habitat types, edges, land management, etc.) and the micro scale (quality of the nest site, brood habitat, drumming areas, lek areas, etc.). Habitat quality also should be assessed on the landscape scale, as it is possible that factors affecting predator abundance are a function of activities or occurrences in a distant locale (avian predators moving southward to escape heavy snowfall and associated low prey availability). Finally, it is important to include an assessment of the impacts of land management practices and policies, such as the cropland and wetland reserve programs (CRP and WRP, respectively) or clearcutting versus uneven-aged management of forestlands, and how these affect habitat quality and therefore predator/prey interactions.

Hunting

We should not forget that humans also are a major predator, often far more efficient than other predators. Predation studies should include as treatment effects hunting intensity and harvest rate, and how they are affected by weather, human density, region, social attitudes, etc. Most importantly, it is critical that studies assess the compensatory nature of hunting and predation on gamebird populations. Unfortunately, this aspect of gamebird predation research will be the most challenging (which may explain why it has not been seriously addressed in past studies). There are two difficulties with evaluating the interrelation between natural and hunting mortality: (1) need of areas that are hunted and *not hunted*, and (2) assessing the interdependence of mortality from hunting, predators, disease and parasites, and accidents.

The importance of multiple study areas and replication now should become readily apparent. The predation process must be evaluated at different levels of hunting intensity in conjunction with varying levels of predation and prey density. To address effects of hunting and predation mortality on gamebird populations will require innovative methodologies constrained by suitable study areas and economic resources.

Social Issues

A final set of parameters regarding predation research concerns societal philosophies. Sportsmen represent a decreasing percentage of the population in most states. Society is becoming more urbanized and is less attuned to rural philosophies and traditions. Therefore, most citizens will not appreciate the need to increase gamebird populations through predator control (assigning aesthetic and ecological, rather than recreational and utilitarian, values to wildlife). To a growing component of society, predators are as valuable as the prey. To appreciate this concept, one simply has to examine recent issues such as hunting mountain lions in California and wolf control in Alaska. Therefore, predation research programs must include a thorough assessment of societal values regarding predator populations and examine social tolerance to hunting, predator control and/or what levels of these activities will be acceptable. This component is linked to why we suggest that rather than examine effects of complete predator removal, biologists must examine effects of varying levels of predator control. Society may be more responsive to a 30-percent reduction in predator numbers than a 80–90-percent reduction.

Summary

Predator populations are increasing and most biologists predict that they will continue to rise, while some gamebird populations are coincidentally decreasing. Biologists and laymen alike are once again inferring a cause-effect relation between predators and prey (gamebirds). Trapping for sport or meat is all but gone, and therefore management of the habitat/predator/prey/human interface must be reexamined and explored for alternatives. This mandates a new research philosophy.

Excellent research concerning predator/prey relations on gamebirds exists. However, most were short (two to three years) in duration, lacked sufficient replication, failed to incorporate a multi-species (predator and prey) approach, did not monitor micro- and macro-habitat variables, failed to examine the importance of abiotic variables, or ignored effects of hunting.

We admit that our proposed experimental design is demanding (for resources, beyond just monetary). However, we must depart from studying simple predator/prey systems and realize that these systems are a part of a complex animal/plant community. To fully assess this process will require thorough and more complex research designs.

Components of our research design should be used as a guideline to assure that minimal information is collected to allow biologists to make regional comparisons. This implies a regional approach to predator/prey (or any large scale) research problem. State and federal research agencies must cease to act autonomously, pool resources and talent and work cooperatively regarding gamebird population declines and predation.

The components we discuss must be performed simultaneously to achieve maximal benefit regarding management alternatives. It serves little purpose to first study foxes, then decide that is may be disease, then study disease, followed by studies concerning importance of buffer species. The cost is higher, but we feel that the benefits will far exceed costs in the final analysis.

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The Role of Non-governmental Organizations in Gamebird Conservation

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Introduction

The responsibility of managing upland gamebird resources in North America primarily is that of state and provincial conservation agencies (Belanger 1988). To accomplish this, agencies interact with the general public and, more recently, with numerous non-governmental organizations (NGOs). NGOs represent a diverse and increasing component of the conservation community. This movement reflects a growing interest by individuals to influence specific aspects of conservation. Wooley et al. (1988: 250) stated: "We believe that future preservation and management of upland wildlife lies in developing a cooperative relationship between organized constituency groups and the local, state, and national agencies whose collective endeavors impact habitat in agricultural regions." For example, the Bureau of Land Management (BLM) recently developed a strategic plan for gamebirds on BLM lands by encouraging input from NGOs (Sands and Smurthwaite 1992). Likewise, Pheasants Forever, Inc. responded to recent attempts to introduce the Sichuan pheasant (*Phasianus colchicus strauchi*) (Prince et al. 1988) by urging state agencies to take a cautious approach (D. R. Lockwood personal communication).

The direct involvement of individuals and citizen's groups in natural resource management is a long-standing practice (Belanger 1988), and has become more formal in recent years (Manfredo 1989). Presently, NGOs have the potential to make significant contributions toward gamebird conservation (Edwards 1988, Brennan 1993). Edwards (1988) suggested wildlife managers and administrators need to recognize the current vigor of the NGO movement and use that energy to manage gamebirds. The extent to which this occurs will depend, to a large degree, on the relationship between NGOs and government agencies.

The purpose of this paper is to describe the interaction between NGOs and state agencies involved in upland gamebird conservation in North America. Our specific objectives are to: (1) determine the frequency and "quality" of the interactions between NGOs and agencies on both management and research for upland gamebirds; (2) describe the species and associated issues that NGOs and agencies most commonly discuss; and (3) identify characteristics of NGOs that facilitate interaction with agencies. We conclude with recommendations for improving the interaction between NGOs and government agencies.

Methods

Two self-administered, mail-back questionnaires were developed to assess interactions between NGOs and state wildlife agencies involved in upland gamebird conservation in North America. The first questionnaire was mailed on October 13, 1993 to 50 state conservation agencies responsible for management and research on gamebird populations or habitats. This state questionnaire was designed to provide information about the interaction between NGOs and government agencies from the agencies' perspectives. Where appropriate, questions were asked with a Likert-type scale (McIver and Carmines 1981). All responses were confidential.

The second questionnaire was mailed October 26, 1993 to 48 NGOs thought to be directly, or indirectly involved in galliform management or research that were listed in the Conservation Directory (National Wildlife Federation 1993). This questionnaire sought the NGOs' perspectives on their interaction with state agencies. A reminder questionnaire was sent to nonrespondents on November 22, 1993.

Frequency and Quality of Interactions

Agencies were asked to list all NGOs with which they interacted on gamebird management issues during the last three years. For each NGO listed, we asked them to rate the frequency (>36 times/year, 12–36 times/year, 3–11 times/year and <3 times/year) and quality (very supportive, supportive, neutral and negative) of the interaction from the perspective of support of agency management programs. We then asked agencies to repeat the process for all NGOs with which they interacted in a research context.

Similar questions were asked with respect to management and research on the NGOs' questionnaire. NGOs were asked to list all state agencies with which they interacted during the last three years. Frequency and quality of interaction with state agencies were assessed in a manner similar to that for agencies, except NGOs were asked to rate the interaction from the perspective of support for their organization.

Species and Issues Discussed

State agencies were asked to list all upland gamebirds they discussed with NGOs during the last three years. Then, for each species listed, we asked them to rate the frequency of discussion (more than any other, more than most others, less than most and less than any other), specify up to three issues and identify the associated NGO that initiated the discussion. A corresponding set of questions about species, issues and state agencies was asked on the questionnaire sent to NGOs.

Characteristics and Actions of NGOs

To assess how the actions of NGOs meshed with the needs of state agencies, we asked agencies to list the three most useful actions a NGO could take to support the agency's mission. Agencies also were asked to list the three NGOs most supportive of the agency's management program. The same questions were asked from a research perspective. To assess the overall impact of NGOs, agencies were asked to indicate whether NGOs had a significant, slight or negligible impact on the state management and research programs.

In comparison, we asked NGOs to describe their mission, and to rate (very important, moderately important, not important) a list of 15 activities relative to how important those activities were to accomplishing their mission. We also asked them to estimate the percentage of their organization's resources spent on various activities (education, research, administration, habitat management, political action and other).

Results and Discussion

We received responses from 19 NGOs (40 percent response rate) ranging from international land-based conservation groups that operated independent of government agencies to species-specific constituency groups designed to augment agency activities (Wooley et al. 1988). We limited our analysis to NGOs with activities that directly affected gamebirds in more than one state.

We received responses from 45 state conservation agencies (90 percent response rate). They interacted with a total of 21 and 12 organizations relative to gamebird management and research, respectively. The NGOs most frequently mentioned by state agencies were The National Wild Turkey Federation (n = 34), Quail Unlimited (n = 28), The Ruffed Grouse Society (n = 23) and Pheasants Forever (n = 21). Combined, these species-specific constituency groups have greater than 200,000 members (National Wildlife Federation 1993). In addition, there were 15 NGOs that interacted with relatively few state agencies ($1 < n < 10$).

NGOs and state agencies indicated interacting with each other an average of twice a month regarding gamebird management issues, but less than monthly concerning research (Table 1). These frequencies likely reflect the predominantly management-directed missions of the four common constituency groups (cf. Wooley et al. 1988), as well as the preponderance of management activities within state agencies. Additionally, we believe relatively few NGO staffs have sufficient research experience. As a result, most NGOs lack an understanding of the benefits of research and find it difficult to communicate the long-term value of research to their constituents. When interaction occurred, the relationship between NGOs and conservation agencies was mutually supportive for both management and research (Table 2).

Most NGOs and agencies discussed an average of 3.2 species with each other during the past three years. Ring-necked pheasant (*Phasianus colchicus*), wild turkey (*Meleagris gallopavo*), northern bobwhite (*Colinus virginianus*) and ruffed grouse (*Bonasa umbellus*) were the species most often mentioned as discussed more than others. These also are the most widely distributed and popular gamebirds, and directly identified with the largest species-specific constituency groups. The issues discussed generally were related to habitat management of species abundance and distribution. Agencies also expressed a need for financial assistance from NGOs for management activities.

Agencies felt NGOs had an occasional (51 percent) to significant (31 percent) impact on their gamebird management programs, but only a negligible (55 percent) to occasional (29 percent) impact on research. The National Wild Turkey Federation (NWTF) was considered the most effective in supporting agency management and research programs. Similarly, NWTF ranked the highest when the relative frequency

Table 1. Frequency of interaction (percentage) for management and research programs between non-governmental organizations (NGO) and state conservation agencies.

Organization	Program	Very frequent	Frequent	Occasional	Rare
NGO (n = 19)	Management	32	47	5	16
	Research	11	16	21	53
Agencies (n = 45)	Management	22	44	29	4
	Research	4	22	33	40

and quality of the interaction were considered on both a per state and nationwide basis. We believe this favorable ranking is due in part to the establishment of a Technical Committee which consists of a representative appointed by each state conservation agency, and meets with scientists from various universities at the NWTF annual convention. This interaction of professional managers, researchers and NWTF staff provides a well-balanced perspective of short- and long-term issues. The Committee is responsible for directing NWTF management and research programs, and allocation of its resources.

Agencies regarded direct financial support and public advocacy as the actions by NGOs that most benefited state management and research programs. Education and information actions by NGOs also were considered important for management programs. Likewise, NGOs deemed education, habitat management and research activities as the most valuable for accomplishing their own missions (Table 3). As a result, NGOs allocated their resources to education, research and habitat management, respectively (Table 4). Our findings suggest numerous opportunities exist for mutually beneficial activities. However, we believe the key to deriving the greatest benefits are more likely when NGOs and state agencies have a clear understanding of each other's mission.

Conclusions and Recommendations

Virtually every state conservation agency interacts with at least one constituency group that identifies directly with either wild turkey, pheasant, ruffed grouse or quail. In general, these groups seem to prefer highly visible habitat management projects related to local gamebird populations. We believe many of these projects may be based on unsubstantiated wildlife principles and too limited in scope. State agencies also interact with other NGOs representing a variety of interests in gamebird conservation. Frequently, the species affected by these organizations may have relatively limited distributions, or the activity in which they are involved may address a rather

Table 2. Quality of interaction (percentage) for management and research programs between non-governmental organizations (NGO) and state conservation agencies.

Organization	Program	Very supportive	Supportive	Neutral	Negative
NGO (n = 19)	Management	47	37	11	5
NGO (n = 9)	Research	56	33	11	0
Agencies (n = 45)	Management	42	47	9	2
Agencies (n = 38)	Research	29	47	18	5

Table 3. Relative value of activities important to accomplishing a non-governmental organization's mission.

Activity	Rank
Providing landowner education	1.3
Providing youth education	1.4
Providing member education	1.5
Managing habitat on private land	1.5
Conducting public information meetings	1.6
Conducting research	1.7
Managing habitat on public land	1.8
Sponsoring technical conferences for biologists	1.8
Lobbying for political action	1.8
Lobbying for regulations and law enforcement	2.3
Acquiring land for public access	2.4
Sponsoring hunting/shooting events	2.4
Acquiring land for access by members only	2.7
Releasing pen-reared birds for hunting	2.8

specific need. These "lower-profile" organizations often are of significant local, regional or national importance.

Whereas a few NGOs provide strong support for research, there nevertheless appears to be disproportionate lack of financial support and advocacy for state research programs by NGOs. Romesburg (1981), and now Potts and Robertson (1994), have urged the wildlife profession to resist the easy, short-term perpetuation of management based on dogma. Instead, they suggest building a solid foundation based on results from field experiments designed to test hypotheses.

Our findings show direct financial support of gamebird research as one of the most needed contributions a NGO can provide for conservation agencies. Yet most NGOs find it much easier to articulate the effects of a highly visible local management action (e.g., establishing nesting cover), than to explain the less direct value of a long-term experiment (e.g., effects of harvest) (Kurzejeski et al. 1994). As a compromise between management and research, we recommend NGOs advocate and financially support Adaptive Resource Management strategies (e.g., Johnson et al. 1993). This approach will provide NGOs with highly visible projects and state conservation agencies some much-needed research activities.

NGOs have the potential to significantly influence the upland gamebird resource of North America, but in most situations that potential is not being realized. We

Table 4. Allocations (percentage) of resources by non-governmental organizations.

Activity	Percentage
Education	29
Research	23
Administration	20
Habitat management	18
Political action	6
Other	4
Total	100

believe there are many opportunities for developing important and mutually beneficial management and research programs between state agencies and NGOs. However, for this to occur, communication between these groups must improve. Therefore, we recommend the following: (1)NGOs clearly identify their interests and priorities through technical committees; (2) agencies clearly communicate their needs to NGOs through interactive short- and long-term strategic planning; (3) agencies coordinate programs among NGOs with similar goals; and (4) NGOs publicly and financially support sound management and research activities, and provide thoughtful criticism of poorly designed programs.

Acknowledgments

We thank the representatives of the nongovernmental organizations and state conservation agencies that took the time to complete the questionnaires. We also are grateful to L. Brennan and R. Gutiérrez for the invitation to explore this important topic and report our findings.

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North American Upland Gamebird Management at Crossroads: Which Road Will We Take?

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Introduction

The future of North American gamebird management is at crossroads. North American quail and pheasant are declining throughout much of their ranges (Droege and Sauer 1990, Brennan 1994). Braun et al. extend this litany of decline to many populations of grouse, although Sauer et al. report trends in most grouse are not possible to detect with current monitoring. Additionally, Carroll et al. illustrate the crisis faced by Mexican quail. How wildlife managers respond to these declining population trends will determine, in part, not only the fate of these gamebirds, but also their future as game resources.

Almost all wild species in North America are affected by human induced environmental changes and the pressures resulting from a burgeoning human population. Gamebirds also experience the ebb and flow of landscape changes. Sometimes these changes are beneficial as Braun et al. point out. However, most of these beneficial changes are short-lived as agriculture shifts from primitive to intensive or as vegetation succession advances.

The apex of gamebird research and management in the context of natural history investigations occurred when gamebird populations flourished across the continent. Thousands of upland gamebird research and management projects occurred between 1940–1965. During this time, many details of gamebird natural history and basic management (e.g., counting and catching techniques) were established. Perhaps this period of relative security and abundance of our gamebirds and our success at "managing" facilitated the following period of benign neglect. The period of benign neglect, the decades of the 1960–1980s, was marked by the population declines noted above, the appearance of endangered and relict populations, and, as noted by Potts and Robertson, the failure of researchers and managers to pursue critical research. Of course, other reasons for this period of benign neglect existed. There were funding shortages, the increasing need to expand conservation efforts to all species and changing human demographics.

Regardless of the reasons for this recent lack of attention to gamebirds, we are facing the 21st century in North America with no more of a salient gamebird strategy than we had three decades ago. It is for this reason that Dr. Brennan and I hope to raise an awareness among professionals about the problems. The time to act is now.

State of North American Gamebird Research and Management

Drs. Potts, Robertson and Lindén have conducted some of the most elegant and practical (i.e., "useful") research on gamebirds in the world. I point out, as a measure of their intellectual honesty, that their critiques encompass both North America and Europe. Potts and Robertson note that relatively little of the most useful types of research (i.e., modelling [based on empirical data] and experiments) are conducted in North America. Clearly, natural history studies are important and should continue, but the ability to understand cause and effect relationships or draw strong inference is dependent on controlled field experiments. This is standard practice in many rapidly advancing fields of science (e.g., *see* Platt 1964). Experiments are more costly than observational studies but the potential rewards, almost always, far exceed anything that can be derived from pure natural history. The bottom line as Potts and Robertson point out is that if one wants more gamebirds in an era of competing land uses then experiments are necessary to direct appropriate management activities on intensively managed land. Experimental research also will benefit declining or small populations of gamebirds.

The acquisition of "reliable knowledge" (*sensu* Romesburg 1981) should be the goal of all applied research. Thus, despite the paucity of "useful" gamebird research as defined by Potts and Robertson, it is encouraging that some North American gamebird researchers are attempting to attack management problems in more creative ways. The papers by Burger et al., and Leopold and Hurst express a vision more broadly defined than historic gamebird research. Burger et al. propose an integrated and geographically broad research program on the northern bobwhite (*Colinus virginianus*) to study a fundamental issue in gamebird management: the effect of exploitation on populations. This study would incorporate the tenets and philosophy, particularly replicated experiments, espoused by Potts and Robertson.

The effect of predation on the natural regulation of gamebird populations is an issue of fundamental concern to our European colleagues. Lindén points out this is little studied in North America. There are many reasons for this, including the complications resulting from political and social pressures. Nevertheless, predation should be studied because of its importance to the dynamics of gamebird populations. Leopold and Hurst illustrate its importance with respect to wild turkeys (*Meleagris gallopavo*). They are to be commended for raising the issue of testing the importance of predation at a time when it is seemingly "politically incorrect."

Lindén notes an increasing tendency among North American researchers to publish in non-refereed journals which indicates, perhaps, poorer quality research. A lively debate is occurring in the wildlife profession on the nature of publishing (e.g., Bart and Anderson 1981). In addition, Lindén points out that despite the widespread decline in North American gamebird populations, very little effort is devoted to monitoring gamebird population trends. From the Finnish perspective, monitoring is not only the prudent course of action but also an obligatory one for a management agency.

Lindén's observation is supported by the findings of Sauer et al. that current bird-monitoring programs at a continental scale are not effective for most grouse species. Because of grouse breeding phenologies, habitat selection patterns and distributions, neither the Breeding Bird Survey (BBS) nor the Christmas Bird Count (CBC) data are effective in discerning even small changes in population trends. Interestingly, both the Finnish and North American bird monitoring schemes are conducted by volunteers. I have had the opportunity while working in Finland to observe Finnish hunters complete transect work. In my opinion, the Finnish volunteers (who are hunters) execute their responsibilities in a professional and serious manner.

In contrast, Christmas Bird Counts are more a contest of birding skills between individuals and groups than a standardized method for gathering biological information judging from those in which I participated. The BBS appears to be more consistent and effective than the CBC. There are a great many hunting clubs and organizations in this country which, perhaps, could function in a role similar to that of their Finnish counterparts. In any event, it is obvious that a more effective monitoring scheme needs to be developed for many of these gamebirds.

A great deal of management attention has focused on monitoring and control of harvests, but few studies scientifically assess the efficacy of managing gamebirds by monitoring harvest. Mossup presents such a study for northern Canada which suggests that there are so many factors which influence harvest (including grouse population cycles, accessibility of grouse and the presence of other hunted species) that monitoring grouse harvest is not a reliable method for predicting future grouse harvests.

Both Finland and the United Kingdom (UK) are small countries relative to the U.S., Canada and Mexico. Yet, researchers there have executed some of the world's best gamebird research and management. In the former case, research is funded primarily by the government while in the latter, private funding is exceedingly important. There is even a private group (American Friends of the Game Conservancy) in the U.S. that supports research endeavors in the UK! Government funding for gamebirds is declining throughout North America. This partially explains the lack of rigorous research and effective gamebird management. The decline in license sales often is used as a reason for lower funding levels. Similarly, the changing attitudes among the "non-consumptive" wildlife enthusiasts are used to justify lower gamebird budgets. However, I will argue that gamebird research and management and, hence, hunters are getting short changed. Brennan (1993) summarized the allocation of Federal Aid In Wildlife Restoration (Pittman-Robertson) funds. Gamebird research, particularly quail, receives relatively little funding. Federal Aid funds are generated by a tax levied on sporting arms, ammunition and equipment. These funds then are reallocated to the states on the bases of hunting license sales and cost sharing. In 1990, approximately 16 percent of Federal Aid in Wildlife Restoration funds was spent on non-game/endangered species projects, while less than 1.25 percent was spent on quail. Approximately 15 percent was spent on all upland gamebirds (both migratory and non-migratory). While I believe that use of Federal Aid money generated by hunters for non-game/endangered species is appropriate, I fail to see the justification for not supporting upland game research, particularly in view of declining populations, habitat deterioration and rising anti-hunter sentiment. The dire situation faced by North American gamebirds as a result of human population pressure should justify greater attention and funding by agencies. It also argues for a reassessment of funding priorities as well as the development of additional sources of funding for non-game programs to alleviate the pressure on Federal Aid funds.

Braun et al. argue that more government/private partnerships will (or should) develop in the future to offset the consequences of reduced budgets and the predicted decline of gamebird hunting. Historically, it has been hunters who have contributed most to the conservation of all American wildlife. So it remains to be seen whether hunters or others will provide the impetus for these partnerships if hunting opportunities decline drastically, especially if those declines are exacerbated by agency neglect. Even more disturbing is the survey conducted by Church et al. concerning the role of non-governmental organizations (NGOs) in gamebird conservation. If their survey is an indication of the future of public/private partnerships (only 40 percent response rate from the NGOs) for gamebirds, the future is gloomy indeed. However, I believe that we should use Church et al.'s work to motivate wildlifers to initiate dialogue and change our relationships with NGOs. I also think it is important for NGOs to have several qualified scientists and managers on their boards as scientific advisors. As a strategy, at least one advisor should reside outside the area of interest and have no vested interest in the outcome of decisions.

In summary, it is clear that research and management have not met the challenge of managing our upland game resources. Our research has declined in quantity and perhaps quality. This assessment is shared by our respected European colleagues. Even though grouse and quail are declining, no effective continental strategy to monitor these changes has been suggested or developed. The consequence of this lack of information is an inability to explain the causative mechanism of these declines (e.g., habitat change, disease, predation, hunting) which makes gamebird hunting opportunities vulnerable to "animal-rights" activists. Funding for gamebirds is being diverted to other important needs even though gamebird hunters often are responsible for generating those funds. Yet, there are bright spots among this gloom. Some gamebird researchers have received commitment from their agencies to pursue critical questions in a rigorous scientific manner. There is a network of NGOs specifically concerned with the fate of these birds. There still remains widespread interest in these gamebirds, in spite of research and management neglect. By bringing this session to the North American, I hope we can give more momentum and encouragement to this small movement. We are at crossroads in gamebird management and research, the road we take will determine the future of gamebird hunting.

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Special Session 7. *Proactive Prescriptions for Wildlife Management in a New Era*

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Changing Agencies, Changing Expectations: How the Public Sees It

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Introduction

In years past, state fish and wildlife agencies occupied a relatively obscure aspect of state government. Most residents of a state knew little about the agencies, nor did they really care. In fact, about all the agency was required to do was keep a low profile, keep wildlife and fish in the hills and streams, and keep the cost of the license down. The most visible member of the agency was the local game warden who made the occasional "pinch" and wrote the offender up for taking a few too many trout. The public did not much care about the state fish and wildlife agency. They existed in the backwater of state government.

This idyllic version of the state fish and wildlife agency long since has disappeared. It is highly unlikely the game warden of 40 or 50 years ago would have envisioned the high-tech, big-issue, highly trained and often controversial agencies that exist now. In this paper, the authors will address how the public's expectations of state fish and wildlife agencies are changing, and suggest how increased female participation, leadership, communication and good management can improve the agencies chances of succeeding and meeting public expectations better in the future.

Who Is This "Public" We Keep Talking About?

A term that often is abused in state fish and wildlife agency day-to-day operations is the "public." The "public," it turns out, is not some homogenous group with the same thoughts, attitudes, values or beliefs. In fact, there are many "publics." All have differing attitudes, values or beliefs, and all are valid; all represent important constituencies that fish and wildlife agencies serve in some capacity. While fish and wildlife agencies perhaps are most comfortable with the traditional hunting and fishing groups, they are finding that there are many others. They range from the suburban housewife who enjoys birds in her backyard birdfeeder to the animal rights activist who trembles at the thought of any animal meeting its demise. These are real constituents who feel they have as much stake in wildlife as the licensed hunter or angler. Too often, they are dismissed as not being as important, or perhaps more dangerously, as a threat to the agency.

Agencies often have confused traditional sporting constituencies with the "public," while ignoring other public groups' valid stake in fish and wildlife management. Unfortunately, this may have led to an almost terminal case of "preaching to the choir" and only hearing what it is that the agency is accustomed to and wishes to hear. Unfortunately, agencies in the past have done too much business with relatively few, at the expense of the vast majority of state residents. Duda (1994: xi) reported that: ". . . citizens and wildlife viewers support wildlife conservation efforts more than wildlife recreation efforts by agencies. Every survey on the subject verifies that people prefer programs that benefit wildlife more than the support programs that benefit outdoor recreationists."

What Does This "Public" Think of Us Anyway?

As state fish and wildlife agencies begin to recognize that there are other constituencies they must deal with, it is important to establish what the public's expectations are.

In recent research conducted by the Responsive Management Project, the results are not very encouraging. Duda (1994: xiii) writes: "Research indicates fairly low levels of public knowledge of fish and wildlife agencies. In general, most people really don't know much about their fish and wildlife agencies. In some states a majority of the population cannot even identify their fish and wildlife agency. However, the closer individuals get to agencies, the more likely they are to have higher opinions of the agency."

In a further study done by Responsive Management on the Maryland Division of Wildlife, an overwhelming majority (98.9 percent) did not know that the Maryland Division of Wildlife was responsible for wildlife management. Even when asked whether there was a generic fish and game agency, the results still were very poor (96.2 percent did not know who was responsible). When the same participants were asked to differentiate between state or federal (U.S. Fish and Wildlife Service), those knowing the difference were almost nonexistent (less than 1 percent).

It also is interesting to note that when asked who was responsible for managing and protecting wildlife in Maryland, the response included a wide range of professions, from park rangers, game wardens, the Environmental Protection Agency, to the Humane Society. Similar results in other studies indicate that those "publics" do not really know, much less understand, what fish and wildlife management agencies do.

There are numerous historical and current reasons for this being the case. Rather than try to trace the reasons for the public's lack of recognition and expectation, the authors would rather focus on providing some suggestions on how state fish and wildlife agencies might improve on the public's recognition and expectations.

Agencies Should Reflect the Public

As government employees, state fish and wildlife agencies, for better or worse, serve the public, and really are small versions of the public. Agencies are not isolated entities with no connection to the real world ''public''; they do not exist in a vacuum. Too often in the past, agencies failed to recognize this and thus became isolated and unconnected with those served. Therefore, it is not surprising when the ''public'' knows very little about state fish and wildlife agency existence or mission beyond the relatively small traditional user groups.

An appropriate question to ask is ''Has the public changed?'' If it has, ''Have agencies changed to keep in step with these changes?'' The answer is increasingly likely to be no—state fish and wildlife agencies may not be reflecting some of the changes that are occurring in our society and in our culture.

First, it is important to look at some of the changes occurring to the many publics served by state fish and wildlife agencies. Demographic changes across North America are occurring that are greatly changing the traditional state fish and wildlife agency constituency. All will result in differing needs and different delivery systems to meet those needs. Race Thompson (1993) reported: ''The percentage of the U.S. population living in metropolitan areas has increased from 57% of the population in 1950 to 78% in 1990. Also, increasing urbanization is insulating many American's contact with wildlife as it is primarily through the media (e.g., nature shows, cartoons, etc.).

- Aging influences participation in wildlife-related recreation.
- The amount of leisure time the average American possesses has decreased 37% since 1973. With increasing competition for leisure time, wildlife recreation must compete with other activities.
- Changing family structures is having an adverse effect on hunting initiation and continuation.''

Gasson (1993) wrote about the so-called ''Nintendo factor.'' When a suburban child is asked if he or she would rather get up early and freeze waiting to see and shoot wildlife or sit in the comfort of his or her living room and play Nintendo, today, there is not much question what that child would rather do.

The important thing to note from the above-mentioned demographic changes is that, if agencies indeed are going to ''serve'' the public, they must anticipate how those changes fit with current operations. The question must be asked: ''How can you serve someone you do not know very well?'' Everyone is aware of how awkward it is when giving a dinner party and you do not know if the guest is allergic to something you have served. Agencies very well may profit from having a better understanding of who their constituents are and what their constituents want prior to serving it up. In fact, much evidence exists that many agencies have been serving meals that fewer and fewer wish to eat; and instead of changing the menu, they want to change the diner.

An important question that all agencies must be asking themselves if they want to be successful in the future is "Are they changing to reflect the same changes that are occurring to their customers?" Budgets have grown, employment has grown and interest in wildlife has increased dramatically. The agency's management has changed. In the past, it was not uncommon for agency heads to stay for years. In the last ten years, this has changed dramatically. In the last five years, there has been a large turnover in the directorship of state fish and wildlife agencies. The current average tenure of a state director is likely to be only three years.

The agency employee has changed. In the past, the individual most likely to work for a state fish and wildlife agency would be a white male, from a largely rural background, with an intense passion for hunting and fishing, who chose his career, more often than not, based on being able to hunt and fish.

Today, the typical fish and wildlife employee probably is a far cry from his predecessor. It is far more likely that the agency's new employee may even be a woman. They will likely have come from a suburban background and have a relatively limited exposure to hunting or fishing. They are almost certain to have at least a bachelor's degree, if not a graduate degree. Despite these changes, state fish and wildlife agencies still are dominated by white, middle-class and middle-aged employees.

The current workforce in state fish and wildlife agencies simply does not "reflect America," to use the current phrase of choice. The question then arises: "Is not reflecting the public necessarily bad?" It certainly can be if it results in failing to understand the public's needs and expectations at the very least, and at worst the development of a "bunker mentality" by the agency, unwilling to listen to any constituency group other than traditional hunters and fishermen. Bunker mentality has proven to fail every time; sometimes it just takes longer. In the short and long run, our agencies will not be able to support themselves through traditional means of financial and political support.

Reduced or nonexistent public support means little or no political support. This likely will lead to missing opportunities for future growth into expanding arenas, as well as expanding the agency's support base. The bottom line is, if we do not enhance ourselves, somebody will do it for us.

Organizational Response

There are a number of ways that state fish and wildlife agencies can respond to these changes. The authors would suggest five major areas in which the state fish and wildlife agency can, as an organization, respond to these changes: having a diverse workforce, focusing on leadership, improving communication, practicing good biology and embracing change.

Diverse Workforce

There is no better example of not reflecting the public than the area of female employment in state fish and wildlife agencies. Although no hard figures exist, it is estimated that less than 10 percent[1] of the top state fish and wildlife agency man-

[1]This figure was derived by reviewing the Conservation Directory and recording the number of women in management positions with titles such as Director, Deputy Director, Administrator or Chief in state fish and wildlife agencies.

agement positions are held by women today. Although there are growing numbers of women in the lower levels of agencies and in areas such as aquatic resource education or in non-game wildlife programs, the number of women in upper management in state fish and wildlife agencies is very low.

Why should agencies be concerned about poor representation of women in state fish and wildlife management? Because, by grappling with women as part of the "publics" and by utilizing women in the state fish and wildlife agency, agencies are taking the first step toward being able to serve better a wide constituency.

It was reported in the *Workforce 2000 Report '87* that numerous changes were taking place in the economy and the composition of the workforce (Hudson 1987). Since then, many studies and articles have appeared about glass ceilings, workforce diversity, etc., mainly focusing on private industry. The Report sent a message—changes are imminent. Diversity in the workforce would happen. Society would have to deal with a workforce that slowly would become older, more female, more disadvantaged and more ethnic. If it is imminent, then what are we doing to facilitate it and make it work to our advantage? It must be noted that diversification is not to be confused with Affirmative Action (Thomas 1990). Diversification is evolutionary, while Affirmative Action is reactionary. Diversification of the workforce can provide the synergy to release creative thinking for problem solving; it cannot be achieved when everyone is alike. The results are mutually beneficial and not exclusive to women (Young 1991). With increased attention given to the recruitment and retention of women in the fish and wildlife agency, there inadvertently will be greater attention cast on the delivery of the workforce already present. The consequences of such valuing of employees is another powerful key to unlocking the potential of the agency.

Agencies can begin to compose their own reasons why they are not attracting and retaining females, but it is just plain naive to assume that the reason for so little diversity, particularly in terms of female employees, is because there are not any out there or because they do not want to work in this profession. Therefore, a closer, deeper look at this issue is timely. Research specifically aimed at finding out what exactly are the problems in federal and state fish and wildlife agencies with regard to diversity now is beginning to emerge.

There are those who will question why we should even be concerned. There are several pressing reasons. First, as current senior management retire and "rightsizing" or "downsizing" is endured, familiar faces, institutional knowledge and expertise may be missed, but it presents an opportunity to take advantage of the times by bringing in new blood and tapping their creativity, brain power and perspectives (Bembry 1992).

It is just plain good business to be able to anticipate this opportunity, adjust and use it to the agency's advantage. By addressing some of the specific questions regarding bringing more women successfully into the field, other aspects of reflecting the public in our agencies can be answered.

An agency hoping to improve its recruitment, utilization and retainment of women should ask itself:

(1) Is there a gap between perceptions of what has been accomplished and what still needs to be done in the way of cultural and gender diversity?

(2) Are we attracting a strong enough applicant pool of women for positions within our agency? If not, why? Can we do anything about this?

(3) Are there things that we could be doing that would help agencies to keep good women professionals once they have them working? Is there anything to be learned from what the private sector is doing to retain women professionals?
(4) Where can a woman go professionally in an agency? Will she encounter a glass ceiling? Are women placed in line-authority positions within the agency?
(5) Does the agency really lend itself to tapping into women as part of the decision-making force?
(6) Do we provide mentors for our female employees? Have we asked ourselves "Why aren't we doing this?" What about the new, non-traditional women in the agency—the I & E types, the non-gamers, the planners—are they being given the same opportunities for career advancement as game personnel?
(7) How do folks in management in agencies view their employees' need to be managed based on today's economic, social and work-related conditions and not on the standards of the past?
(8) Is recruitment of women in our agency's workforce a part, a real part, of our strategic plan, and is there a clear statement from top management articulating the goal of creating a culture where women and minorities can work to their full potential?

An agency wishing to attract and retain professional women may want to ask these questions.

Broadening outreach to women as a constituency group will assist agencies in many ways; however, the "what's in it for me" question must be asked by the fish and wildlife agency before planning strategies. Is the agency seeking increased license sales, increased support for the agency's programs or increased credibility? Is it a proactive approach to dealing with animal rights activists?

As agencies begin to explore these opportunities, they will need to know how to determine our female constituents' needs. The authors suggest that when trying to determine what the needs of our female constituents are, that we simply ask them.

Leadership

Although perhaps almost passe, looking to leadership as a solution is a good option. A definition of leadership is not straightforward. Leaders are professional influencers, as they have the ability to exert influence over others within an organizational context. This working definition underscores the importance of the organizational context. For a leader to be successful, they must attend to the task at hand as much as they attend to their subordinates. This means the leader must be aware of the organization's goals and continually interact with the wide number of constituencies. The concept of leadership is like a jigsaw puzzle with many different pieces. And there are a lot of pieces (Yukl 1981).

It is safe to say that a leader takes people from one place to another (DePree 1987), and that certainly is the case in a state fish and wildlife agency. There are many characteristics of good leadership that enable a leader to be successful. And these characteristics can be measured and developed through training.

Demands of the '90s require the full potential of the human resources that an agency can muster. The idea of maximizing individual effectiveness meshes well with current demographic and economic changes. Translating mission statements into reality, and enabling agency personnel to change and move upward into higher levels of effectiveness is a matter which requires leadership—and the magnitude of today's challenges requires not only more leadership but newer forms of leadership (Conger 1993).

"Learning disabilities are trying in children, but are fatal in organizations" (Senge 1990).

Identifying, evaluating and developing leadership within fish and wildlife managers offer a creative human potential for the agency worthy of being tapped to the fullest. Of the 21 factors identified by McMullin (1993) as indicators of fish and wildlife agency management effectiveness, the factors of leadership and management skills of leaders were ranked by agency directors, ex-directors, commissioners and legislators across the board as being among the five most important.

A consequence of leadership development is that it puts leaders in closer touch with their subordinates. And, research reveals that even in the most effective of fish and wildlife agencies, a discrepancy exists between what the Director feels is reality within and outside his agency, and what the rest of the staff feel is reality (McMullen 1993).

Embrace Change

Agencies are much like living organisms. They need input, use energy, think, move and sometimes create waste. Oftentimes, the agency only is concerned about survival. Agencies can tend to focus on distinct aspects of agency operation without looking at how the agency's operation is changing. Change often is viewed as problematic rather than opportunistic. In fact, agencies may go through extensive review, analyses and planning, which when done, are not really taken seriously. To be able to adapt to the changing world, agencies must recognize the need to change, and recognize that change is not an abstraction, something that can be put off until some point in the future. Change is imminent; it is a fact. Too often, agencies have chosen to avoid the recognition of change occurring. Good business dictates that managers must be proactive and always anticipate the next change. Agencies must be able to accurately recognize what is changing in their sphere of influence and take an unvarnished and candid view of how to proceed. A successful agency will avoid presenting a "pastel" view of the world to its publics. Changing situations are reality checks on a constantly changing landscape. Agencies that recognize this and embrace change likely will be successful in the long run. State fish and wildlife agencies are changing even if they do not wish to. Their constituencies, missions, resource base, personnel and management all are in a constant state of change. As mentioned earlier, they no longer are in policy backwaters. Every action comes under intense scrutiny from the many publics it serves. It is important to note that embracing change is not always easy in the short term. New approaches, alternatives and strategies can be difficult and wrenching at first. However, if done with a clear understanding of the landscape and the public to be served, the agency likely will be in a better position to accommodate the demographic and attitudinal changes occurring in the public.

Increased Communication

In addition to workforce diversity, leadership improvement and embracing change, the area of communications with the public is an area in which state fish and wildlife agencies have an opportunity to meet public expectations more effectively. State fish and wildlife agencies have a long history of communication with their traditional constituencies. Means of communication have tended to be news releases. Oftentimes, however, communication has tended to be one way: the agencies telling their constituencies what was good for them. Fortunately, this seems to be changing. Many states have instituted wildlife conferences or "congresses" to find out what their constituents want from the wildlife in their state. Some questions that agencies wishing to be in better touch with their publics need to ask are:

1. Is the agency making a real commitment to their Information and Education section; are they tempted to make it the first to be cut in tough fiscal times?
2. Is agency communication with "publics" real, not just a form of placation? Does the agency make every effort to use the input gathered from the public so that the public feels it has a real say in agency operations?
3. Does the agency try to find out what the various client groups want through good stakeholder analysis and other planning techniques?
4. Does the agency recognize that communication works both internally and externally? Do agency personnel have all of the information they need to give out the "company line?"
5. Does the agency recognize that they really manage people as much as they manage wildlife? Agencies are far more comfortable with surveying wildlife populations rather than surveying human attitudes about wildlife management. Agencies can benefit from investing in modern attitude sampling techniques and methodologies.

Practice Good Biology

Although the agency's practice and support for good biologically supported management goes without saying, it must be impressed upon everyone from the Director to the field biologist that this is critical to the survival of the agency. By practicing good, credible and defensible biology, the agency meets the public's expectations of its mission; it is nothing less than the cornerstone of agency success. Agencies must never be tempted to fudge on biology. However, agencies also must recognize that biology alone does not always win the day. Management decisions based on good biology, not on mythologies or traditions (e.g., we've always done it that way) will be able to be defended when the heat is on. It also is critical that agency personnel buy into good biologically supported management. Without internal support, management decisions likely will not be implemented or carried out.

Conclusion

Agencies must come to recognize that, to a large extent, the future of state fish and wildlife agencies lies with those who currently may not even be aware that the agency exists. It is likely that agencies will not succeed only with the support of traditional constituents. Nor will they succeed in making all of the public hunters or fishermen, but we can work to gain their trust and try to reflect their interests as much as possible in programs and operations. State agencies can gain their support by presenting an agency that offers diverse programs to meet diverse needs.

What will all of this lead to? Hopefully, it will lead to agencies that recognize that changing public expectations are an opportunity to be taken advantage of, not the dismal end of state fish and wildlife management that some naysayers are predicting. Recognition that change is not always bad can have a very positive effect on an agency. That is, if the agency recognizes the value of a diverse workforce, good leadership and communication, and embraces and thrives on change.

Agencies which are in step with their "publics"—philosophically, culturally and in their personal makeup—likely will be successful and actually able to meet increasing and changing public needs and expectations. As this paper has noted, organizational responses to a more diverse workforce, better communication with publics, good leadership and good biology will lead to a successful and efficient agency with successful personnel who are able to anticipate changing public and agency needs and to proactively meet the needs of both; an agency that is not frozen in fear of the future, but anticipates anxiously the opportunities that the future will present.

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Perspectives of Traditional Constituents on Changing Resource Management Agencies

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Introduction

"Sportsmen are unquestionably the world's prize optimists. They can sit by and see fish and game conditions grow steadily worse, nationally speaking, basking in the sublime faith that some miracle is going to restore unlimited sport. They can placidly look upon the ever-tightening restrictions that are closing in from every quarter with a childish hope that some mysterious defender is going to rise up from nowhere and cut the tightening fetters. Just where these saviors of sportsmen's rights are coming from, no one seems to know.

"That being the case, it may not be amiss to state the situation plainly. There are many agencies ... that can do much in the way of directing the defensive strength of the sportsman party. But mark this: The real power to fight off the encroaching oppression now lies dormant in the sportsmen themselves. After a real catastrophe there is a fine display of indignant protest. The losers wail, but no warning of disaster seems to rouse advance action." (Foster 1935)

While those words seem pertinent to situations today, they were penned nearly 60 years ago by William Herndon Foster in an editorial in *Hunting & Fishing* magazine. To my mind, it remains an apt description of modern sportsmen. Today's hunters perceive deteriorating opportunities and threats from a number of fronts, including animal extremists. Yet, they often seem like deer in the headlights, unable or unwilling to act. Yet, they must act if they wish to salvage their preferred outdoor activity.

The Traditional Constituent

Although animal extremists would like the public to believe otherwise, hunters are just like other Americans. They come from all age groups, occupations and geographic regions. They are well educated and reasonably affluent, increasingly so on both counts. And hunters spend an increasing amount of money to engage in their activity, over $14 billion in 1992 (U. S. Department of Interior 1993). And cliche though it may be, hunters pay for conservation. Through excise taxes, license fees and stamps, hunters, (fishermen and trappers) enable the lion's share of wildlife research, management, habitat acquisition and improvement, enforcement, and hunter education conducted by state agencies. This says nothing of the hundreds of millions of dollars unselfishly contributed to habitat-oriented conservation organizations, a significant portion of which finds its way back into agency budgets as well.

Once while travelling to one meeting or another, my airplane seatmate commented that if you ever were uncertain about who you work for, simply look at the name signed in the bottom right corner of your paycheck. That was back in the days before machine-signed checks, and the point was not lost on me, and I hope not on you.

Hunters and other responsible users of wildlife resources are the ultimate constituency of state wildlife agencies because they pay for the work. They may not pay for *all* the work, and the percentage of the budget varies from state to state, but the fact is that sportsmen pay for the majority, predominance, preponderance, or whatever superlative you prefer, of the wildlife conservation work performed by states.

Sportsmen are the primary constituency of state wildlife agencies. And if you doubt it, consider what would happen if there were no sportsmen and no sportsmen-contributed dollars. There aren't many, if any, states with such budget surpluses that losses of license revenue could be made up easily. Without those dollars, would there be any research or management? Who would enforce the wildlife statutes? How large a reduction in force would be required immediately? And this says nothing of those private entrepreneurs who depend upon sportsmen for a portion or for all of their livelihood.

Non-constituents

There are a number of interests which claim to be constituents of wildlife agencies. Some claim and have a legitimate interest in nongame species, yet they contribute little if any for programs. Others claim some emotional attachment or vague interest in or relationship to wildlife. They *may* be well intentioned, but often they are motivated by a larger philosophic agenda. In either case, they feel no responsibility to share in the costs of wildlife conservation, feeling that their contributions through the income or sales taxes they pay are sufficient to garner equal consideration with sportsmen (who also pay the same sales and income taxes). Those who challenge and dispute sportsmen's use of wildlife resources I call extremists. I feel we should not cater to them because I believe they have little *real* interest in the future of wildlife. Certainly their organizations do not invest in habitat protection or enhancement, or believe that man uses animals in any responsible manner.

If you doubt that animal extremists are motivated less by a concern for wildlife than by a broader agenda, events in New Jersey this past summer should be of interest. During a three-day conference at Rutgers University Animal Rights Law Center, attorney William Kunstler and center director Gary Francione, according to Putting People First, advised participants to play down ties to the animal welfare and conservation establishment in order to co-opt feminists, minorities, gays and lesbians, etc., avowing that we "are a movement of the left." Perhaps this is why traditional sportsman constituents become concerned when agencies seek to entertain or embrace new constituencies in the name of fairness or conservation.

Perceptions

Regardless of the facts or absolute truth or evidence, juries decide court cases based upon their perceptions, and trial lawyers excel at manipulating perceptions. Therefore, in life as well as in court, perception is reality. Unfortunate and unfair as it may be, it is perception with which we must cope. Animal extremists try to create the perception in agencies and the public that agencies are responsive only to hunters and manage only for game species to the detriment of all other species. Of course, as scientists, we know that this claim is untrue, but nevertheless agencies expend resources and effort to overcome the perception.

So, just what do hunters believe is happening in wildlife agencies and to agencies' relationships with their traditional constituents? Remember that these are generalizations and should not be taken as an indictment of any particular agency. But please check for the fit of the shoe.

Hunters feel there is reduced and diminishing opportunity to hunt. They know it is harder to find a place to hunt. This may be because expanding urban areas put hunting lands farther away, because absentee landowners post their property, because landowners have chosen to charge a fee for access or opportunity or to increase such fees to levels which preclude many hunters, because agencies have failed to secure lands or access, or because agencies have closed or restricted formerly open public areas.

Of particular concern is diminished opportunity for young or beginning hunters. Compared with the multitude of recreational opportunities available to young people, initial involvement in hunting is extremely difficult. Requirements for hunter education, marksmanship and expensive equipment are more difficult to endure when it is known that hunting opportunities are extremely limited or non-existent. Very few state agencies are providing anything other than hunter education for youth, leaving opportunity and in-the-field training to local sportsmen/conservation organizations. Agencies must begin to consider that there will not be a constituency, traditional or otherwise, if today's youth are ignored.

There also is the perception that agencies are imposing more and higher fees for hunting. This cannot be denied, and in most cases was and is necessary in order to provide enhanced management for a number of important species. Sportsmen, their conservation organizations and agencies have been partners in initiating habitat- or species-specific stamps in order to improve management. Yet hardly, if ever, has the introduction of a mandatory stamp been accompanied by a reduction in the fee for general license. And when opportunities for new stamps have been exhausted, general license fees must again be raised to meet increasing responsibilities. Generally, sportsmen have been good soldiers in bearing the burden of these increased costs, but in increasing instances, they are bordering on the edge of burdensome and hunters will begin to opt out because they will perceive the financial burden has become too great. This is particularly so as regulations become increasingly complex.

The increasing complexity of regulations is troublesome. Complexity comes about, I believe, because increased knowledge about communities and populations leads us to believe we can micromanage our resources or manage them with extreme precision. However, sportsmen are not as biologically sophisticated as agency personnel and can become confused about what regulations are intended to accomplish. They also may feel they might too easily be in violation. Sometimes regulations are proposed to solve problems which may not even exist. The recent proposal to ban shotgun cartridges longer than three and one-half inches is an example. In short, make every attempt to solve management difficulties in the simplest and most easily explainable way possible.

Hunters sometimes perceive they are being either ignored or taken for granted. Their concerns often are brushed aside as ignorant or inconsequential, while concerns of other interests are listened to and acted upon based on the argument that agencies must serve *all* citizens of the state. While in a strict legal sense this may be true, the agency has a perhaps greater responsibility to conserve the wildlife of the state. These other interests often demand that greater attention be paid to nongame wildlife and endangered species. However, they seldom provide the financial means for such programs, preferring that license revenue be diverted. Agencies, I believe, deal with these interests believing that once a few concessions have been granted they will be satisfied. Nothing could be further from the truth. A prominent anti-hunter has declared that hunting "is an unnecessary form of ecological degradation contributing to the endangerment of the continuation of life on this planet" (Dommer 1989). Does that sound like the view of someone who will be contented as long as any hunting is permitted? Agencies may have an obligation to listen, but they do not have an obligation to give in to demands. Hunters sometimes perceive that agencies are restricting or eliminating opportunity without any biological justification, but simply because anti-hunting interests demanded it.

Sometime during the early 1980s, in its zeal to pass some particular legislative initiative, the Reagan administration was accused of cutting deals with its enemies to get their votes while yielding no concessions to its friends. This is how hunters sometimes perceive agency dealings with anti-use interests. Often the concessions are made in the saccharine name of biodiversity or ecosystem management. But these other groups will not support agencies in times of need; they have not yet, and will not in the future, for their interest is not the well being of wildlife but the elimination of human use of animals, and more.

Conclusion

In *The American Hunting Myth,* anti-hunter Ron Baker (1989) said that, to restrict hunting, no new or expanded seasons should be allowed, no new lands must be opened and increasing acres must be declared sanctuaries. Is any of that happening now? He further suggested that "there must be a shift away from the financing of state wildlife programs by hunters, trappers and fishermen" (Baker 1989). Among the litany of other things advocated to eliminate hunting are reducing the lands available for hunting, tightening hunting eligibility requirements and restricting wildlife management activities. Aren't these exactly the kind of things that non-hunting interests are demanding? And aren't some agencies doing these things little by little? It is time agencies actively support hunting, expand opportunity and encourage the development of young hunters.

My advice and my plea to agencies is to remember who have been your faithful partners in conservation, who have a real self interest in healthy wildlife populations and in ensuring their future. Perhaps it was better said by an unknown, ancient Texan when he advised "Dance with the one that brung you."

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Differentiating among Wildlife-related Attitudinal Groups in Alaska

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Introduction

During the last two decades, human dimensions research increasingly has attempted to identify and characterize subsets of wildlife user groups; for example, hunters (Wright et al. 1977, Kuentzel and Heberlein 1992) and birdwatchers (Applegate et al. 1982). This is due to a realization that these groups are not homogeneous; their memberships include a wide range of specialists and generalists with different environmental attitudes, perceptions and behaviors (Virden and Schreyer 1988).

Alaskans are a subset of the American public that exhibits a unique relationship with wildlife. Compared with other regions of the United States, a greater proportion of Alaskans hunt, sport fish, trap, use off-road vehicles, birdwatch, backpack, study wildlife as a hobby, keep wild pets, belong to conservation organizations (particularly for sportsmen and wildlife preservation) and read wildlife-related material (Kellert 1980a). Alaskans are less likely to visit zoos and natural history museums, hunt primarily for sport or to get close to nature, express anti-hunting sentiments, own pets or raise livestock (Kellert 1980a). Regionally, Alaskans also have the highest levels of wildlife-related knowledge; affinity for a broad spectrum of animals, including wolves and other predators; and concern for wildlife and natural habitat protection (Kellert 1980b, 1985). In fact, as a group, Alaskans generally scored higher in these categories than any other demographic or specific wildlife user groups (e.g., birdwatchers) except those with a graduate education. Finally, compared to other regions, Alaskans had the highest scores on Kellert's (1980b) naturalistic, ecologistic and dominionistic attitude scales and the lowest scores on the moralistic (animal rights), humanistic (affection for individual animals, particularly pets), utilitarian and negativistic (avoidance of animals due to indifference, dislike or fear) attitude scales.

These differences are not merely academic. Attitudes often affect the way people act. During Alaska's recent attempt to develop a comprehensive, statewide management plan for wolves (*Canis lupus*), most Alaskans were tolerant of a wide range of management alternatives. Strategies in the final plan ranged from complete protection in some areas to limited wolf control in others. The controversy and threatened tourism boycott was fueled primarily by large numbers of residents of other states who believed that shooting or trapping wolves to increase ungulate populations was an unacceptable alternative anywhere.

In the 15 years since Kellert's nationwide survey, the Alaskan population has changed. For example, from 1980–91, the average number of emigrants from other states and abroad equaled 9.9 percent of the state's population each year (ADL 1993). During the same period, the proportion of urban residents increased from 64 to 71 percent, and the proportion of females increased from 46 to 47 percent. A person's background environment, residency, gender and other demographic characteristics can significantly influence wildlife-related attitudes and behaviors (Kellert 1984, Kellert and Berry 1987).

This study is the first attempt since Kellert's benchmark research to understand better Alaskan attitudes toward wildlife. Our objectives were to identify groups of Alaskans with different wildlife-related attitudes, attempt to characterize the groups demographically and consider the management implications.

Methods

Mail Survey

The Alaska Department of Fish and Game surveyed three sample populations (Alaskan voters, resident hunters and nonresident hunters) by mail questionnaire in early 1992 (Miller and McCollum 1993a, 1993b, 1993c). Only the survey of Alaskans registered to vote in 1990 was used in this analysis (284,444 registered voters, or 75 percent of the state population ≥ age 18).

The 27-page questionnaire was pretested using nine focus groups. It included 28 Likert-scale attitudinal questions, 12 true/false questions to measure wildlife-related knowledge, 43 questions to determine wildlife viewing preferences and experiences, 65 economic and willingness-to-pay questions, and nine demographic questions.

A random sample of 4,725 names, stratified by legislative district, was drawn from the state's voter registration list. An introductory letter was sent to each, and the questionnaire was sent to every person whose introductory letter was not returned by the Post Office as undeliverable (4,141). Second and third copies were mailed to nonrespondents. In total, 2,370 votes responded, a 57-percent response rate.

Respondents' gender, age, location of residence (legislative district) and purchase of 1991 hunting license were compared with those of the original sample of Alaskan voters to identify the potential for response bias. Respondents were significantly different (Pearson chi-square, $P < 0.05$) only with regard to location of residence, and this difference was small.

Cluster Analysis

Sixteen of the 28 attitudinal questions were used to define wildlife user groups (Table 1). These questions were selected because they dealt primarily with attitudes toward hunting and wildlife viewing, and they showed sufficient diversity in responses to discriminate between potential attitudinal groups. Questionnaires that had one or more missing responses to these 16 questions were dropped from the analysis, making the sample size for analysis 2,131.

The data were analyzed with SPSS/PC + (Version 5.0) software. First, agglomerative hierarchical cluster analyses, performed 50 times on random samples of 5-percent of the completed responses, identified three or four potential clusters. *K*-means cluster analyses performed on the entire sample produced usable results for both three- and four-cluster groups; we chose the four-cluster grouping to gain a more detailed understanding.

Table 1. Attitudinal statements and questions used to define wildlife user groups and measure group attitudes toward consumptive uses of wildlife, wildlife viewing and hunters.

Variable	Attitudinal statement
	Statements used in cluster analysis to define wildlife user groups
FINDWILD	I am interested in knowing more about how to find and watch wildlife.
STOPVIEW	I would probably stop or slow down to look for wildlife if I saw a sign along the highway indicating good wildlife viewing.
HABITAT	I think more concern should be given to protecting the land and water where wildlife live.
HUNTMEAT	In general, I approve of hunting wildlife for meat.
TROPHY	In general, I approve of hunting wildlife for trophies.
TRAPPING	In general, I approve of trapping wildlife.
HUNTSAY	I think hunters have too much influence on wildlife management.
ENVIRNSAY	I think environmentalists have too much influence on wildlife management.
OUTSAY	I think people living outside Alaska have too much influence on wildlife management.
MOREVIEW	I think more areas in the state should be managed and developed for wildlife viewing.
VIEWFIRST	I think more areas in the state should be managed and developed for wildlife viewing, even if that means closing some areas to hunting.
MOREWARY	In general, I think it is more difficult to see wild animals in areas where those same animals are hunted than in areas where they are not hunted.
CLOSEHUNT	I believe more areas in the state should be closed to hunting.
EATGAME	I like to eat game meat.
SPORTFISH	I like to go sport fishing.
	Statements and questions used to measure other attitudes
KILLWOLF	I support killing wolves in some areas of Alaska to increase the numbers of moose and caribou.
HUNTSAFE	In general, I feel it is safe to be in a hunting area during the hunting season.
HUNTCONS	I think most hunters are considerate of other people they meet in the field.
ORV	I prefer to watch wildlife in areas where off-road motorized vehicles are not allowed.
PRFUNDS	The state gets about a third of its money for wildlife management from federal taxes on certain hunting equipment. How much of that money should be spent on programs for wildlife viewing or other wildlife programs which do not involve hunting? (none, a little [≤25 percent], some [26–49 percent], a lot [≥50 percent], no opinion)
BEARBAIT	Some people think baiting or attracting black bears with food allows hunters to be more selective in choosing which bear to kill. Do you support allowing hunters to use bait to hunt black bears?
HUNTERED	In many states, hunters must pass a certified hunter education course before they can hunt. Which one of the following statements best describes your opinion of requiring hunters to pass a hunter education course to hunt in Alaska? (should not be required, only for hunters hunting for the first time, all hunters should be required, no opinion)

On some questions, the answers of early respondents differed significantly from those of late respondents, which may indicate a nonresponse bias (Brown et al. 1981). We did not contact nonrespondents to ascertain whether any nonresponse bias was statistically significant. To test for potential nonresponse bias, we examined response rates of the four attitudinal groups with regard to when the responses were received (i.e., after the first, second or third mailing) and found no significant differences (Pearson chi-square, $P = 0.30$). Although this does not ensure that our sample is representative of all Alaskans, group attitudes and other attributes can be compared across groups. Voter registration and turnout is higher among older, white, higher educated, wealthier and less transient segments of the population (Jennings 1991), and response to questionnaires also is influenced by demographic characteristics. If expending effort to complete a questionnaire is indicative of those Alaskans who are likely to vote on wildlife-related issues, then our results have management implications in a democratic society.

CHAID (Chi-squared Automatic Interaction Detector) analyses were performed on the four attitudinal groups identified by the cluster analysis to ascertain any unique demographic characteristics. CHAID divides a population into mutually exclusive segments based on the most statistically significant predictor (Magidson 1992). The splitting process continues until no significant predictors remain; in this case, $P > 0.05$. Independent variables included gender, age, years of Alaska residency, income, education, race, rural/urban and regional residency, military occupation, and whether respondents had ever gone on an outing that included wildlife viewing as a planned activity, ever purchased a hunting license or purchased a hunting license in 1991. Urban areas included all communities with ≥2,500 inhabitants (ADL 1993); although some Alaskan urban areas are so remote (i.e. in rural areas far from the road network) that this national definition is misleading. Alaska had 20 communities with ≥2,500 inhabitants in 1991; only three had more than 20,000 inhabitants. The questionnaire did not ask voters if they had purchased a 1991 hunting license. That information was obtained from the 1991 list of licensees. In many instances, we were unable to positively identify whether a respondent had purchased a 1991 hunting license due to inconsistencies in the use of initials or possible data entry errors. Thus, this estimate (12 percent of respondents) is low compared to the known number of Alaskans purchasing hunting licenses in 1991 (>23 percent), but it allows us to compare the relative proportion of licensees in each group.

Results

The cluster analysis identified four distinct attitudinal groups. Group responses to the attitudinal statements sometimes are widely divergent and sometimes a continuum of overlapping attitudes (figures 1 and 2). Table 2 simplifies comparison of group attitudes toward hunting, wildlife viewing and hunters by weighting responses according to their degree of agreement/disagreement and averaging all individual scores into a single score. Each group has a different set of demographic characteristics and wildlife-related attributes (Table 3).

Group A

Attitudes. This group (21 percent of respondents) differed in almost every respect

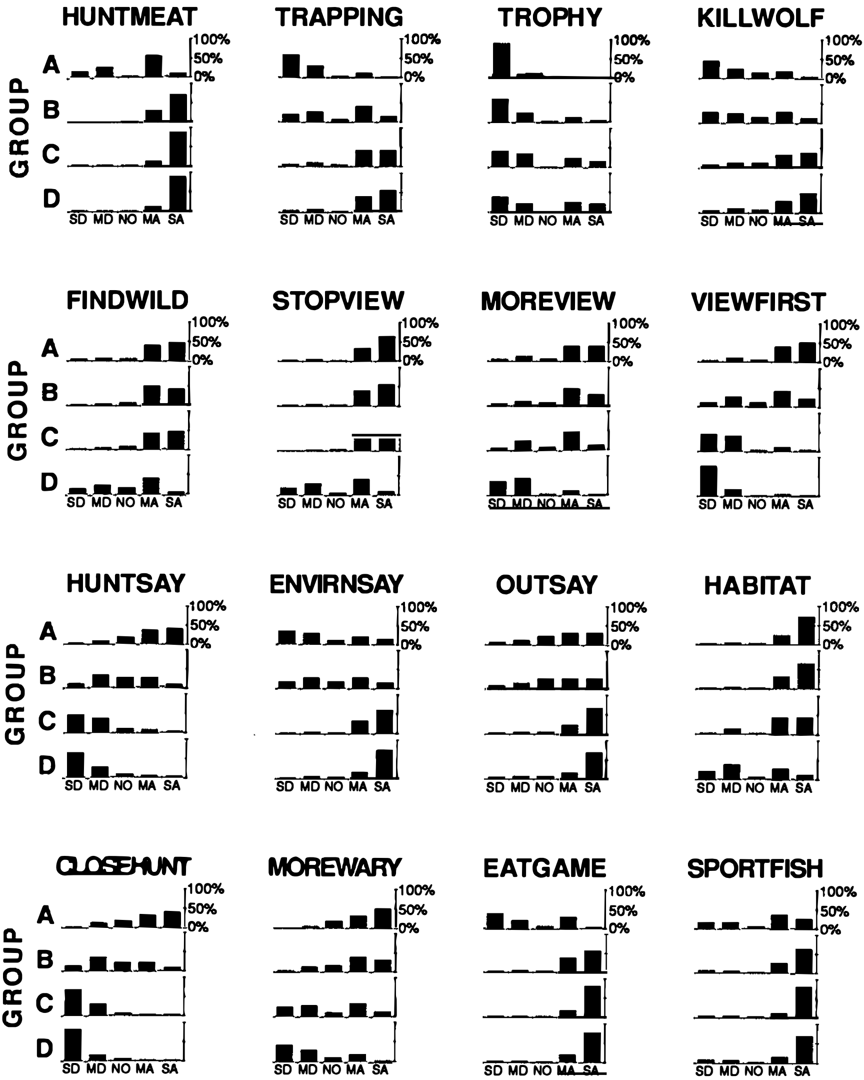

Figure 1. Responses of four Alaskan wildlife user groups to 16 attitudinal statements (Table 1) used to categorize the groups by cluster analysis. SD = strongly disagree, MD = moderately disagree, NO = no opinion/don't know, MA = moderately agree, SA = strongly agree.

from the other three (Figure 1, Table 2). Attitudes toward consumptive uses and hunting generally were negative; however, attitudes toward meat hunting and sport fishing were slightly positive. On several selected management issues (Figure 2), this group was the strongest proponent of mandatory hunter education for experienced, as well as novice hunters, and of using Pittman-Robertson funds for wildlife viewing, but opposed the practice of bear baiting. Groups A and B expressed nearly identical attitudes about use of off-road vehicles.

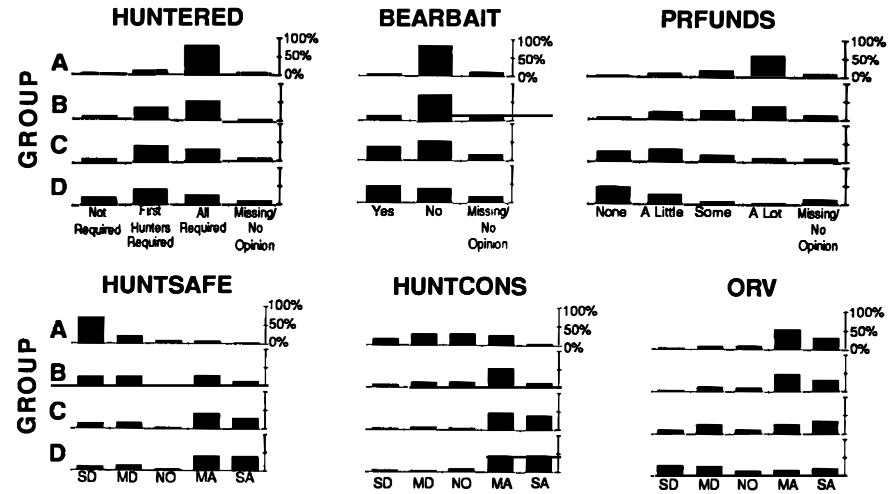

Figure 2. Responses of four Alaskan wildlife user groups to other attitudinal statements and questions (Table 1) with wildlife management implications. SD = strongly disagree, MD = moderately disagree, NO = no opinion/don't know, MA = moderately agree, SA = strongly agree.

Demographics and activities. Group A had the highest proportion of females, whites and urban residents, shortest Alaskan residency, and highest median education and incomes. This group had the smallest proportion who had hunted and the highest proportion of wildlife viewers. Group A had the highest proportion that had never purchased a hunting license or taken an outing where wildlife viewing was a planned activity (Table 3); this was the only category in which Group A did not differ significantly from Group D. The most predictive characteristic identified in the CHAID analysis was the high proportion of this group who had never purchased a hunting license. Of these, 97 percent were non-Native. Of the 27 percent in Group A who had purchased a hunting license in their lifetime, 91 percent had a median education or better. A very low proportion of Group A were current hunters (i.e., purchased a 1991 hunting license). Group A probably includes most of Alaska's anti-hunters and many nonhunters.

Group B

Attitudes. The attitudes of Group B (32 percent of respondents) clearly were consumptive, but less so than Groups C and D. Group B differed from Groups C and D most in attitudes toward trophy hunting, trapping, closing more areas to hunting and the amount of influence hunters have on wildlife management (Table 2). This group's attitudes toward hunters were the closest to neutral of any group. The attitudes of Group B toward wildlife viewing were almost as positive as those of Group A. Their interest in wildlife viewing was slightly higher than Group A's, but they were less likely to want to exchange hunting areas for wildlife viewing areas.

Demographics and activities. The demographic characteristics of Group B are closer to those of the other consumptive groups than to Group A. Among the three consumptive groups, Groups B and D are least alike. Demographically, Group B is

Table 2. Scores of four Alaskan wildlife user groups based on attitudinal statements (Table 1) regarding consumptive uses of wildlife, wildlife viewing and hunters.

Variable	Attitudinal scores[a] Group A	Group B	Group C	Group D
	Attitudes toward consumptive uses			
HUNTMEAT	0.243	1.711	1.865	1.862
TROPHY	–1.818	–1.230	–0.597	–0.288
TRAPPING	–1.360	0.030	1.017	1.259
HUNTSAY	–1.001	–0.009	1.163	1.456
CLOSEHUNT	–1.018	0.235	1.554	1.667
EATGAME	–0.657	1.446	1.709	1.604
SPORTFISH	0.330	1.341	1.565	1.273
KILLWOLF	–0.846	–0.237	0.768	1.044
Total	–6.127	3.287	9.044	9.877
	Attitudes toward wildlife viewing			
FINDWILD	1.115	1.289	1.271	0.394
STOPVIEW	1.170	1.224	1.348	0.270
ENVIRNSAY	0.370	–0.079	–1.161	–0.952
MOREVIEW	0.964	0.980	0.662	–0.983
VIEWFIRST	1.083	0.586	–1.051	–1.643
MOREWARY	1.061	0.906	0.045	–0.723
Total	5.763	4.906	1.114	–3.637
	Attitudes toward hunters			
HUNTSAFE	–1.468	–0.275	0.617	0.839
HUNTCONS	–0.264	0.490	1.126	1.199
HUNTSAY	–1.001	–0.009	1.163	1.456
Total	–2.733	0.206	2.906	3.494

[a]Scores based on average group response to five-point rating scale, where –2 = strongly disagree, –1 = moderately disagree, 0 = no opinion/don't know, +1 = moderately agree, and +2 = strongly agree, except for HUNTSAY, ENVIRNSAY and CLOSEHUNT, where values are reversed to portray positive and negative attitudes. Maximum possible totals are ±16 for consumptive uses, ±12 for wildlife viewing and ±6 for attitudes toward hunters.

the most similar to the overall sample in most of the categories in Table 3. Group B includes a higher proportion of wildlife viewers and a lower proportion of hunters than Groups C and D and the statewide average. The most predictive characteristic identified in the CHAID analysis was having gone on an outing that included wildlife viewing as a planned activity, followed by purchase of a 1991 hunting license.

Group C

Attitudes. The attitudes of Group C (28 percent of respondents) are closer to Group D than to Group B in most respects; however, this group differs from Group D in that it is more positive toward eating game meat and sport fishing, less positive toward trapping and killing wolves, and more negative toward trophy hunting (Table 2). Group C shows a high level of interest in wildlife viewing, except when viewing is perceived as adversely affecting hunting opportunities. Group C has the most negative attitude concerning the influence of environmentalists in wildlife management (Table 2).

Table 3. Demographic characteristics of four Alaskan wildlife user groups (registered voters ≥18 years old) compared with the total sample population and Alaskan public.[a]

Demographic characteristic	Group A (n = 442)	Group B (n = 677)	Group C (n = 598)	Group D (n = 414)	Alaskan sample	Alaskan public[b]
Age (mean years)	*42*	*41*	*42*	44	42	39
Gender (percentage female)	66	48	41	34	47	47
Education (median)	college grad	*some college*	*some college*	*some college*	some college	some college
Family income (median)	$52,500	*$47,500*	*$47,500*	*$47,500*	$47,500	$41,408
Race (percentage)						
White	89	*78*	*75*	*75*	79	79
Native American	3	*15*	*18*	*18*	14	14
Residency[c] (percentage)						
Urban	82	*71*	*64*	*68*	69	71
Rural	14	*24*	*32*	*28*	25	17
Years in Alaska (mean)	16	19	21	25	20	—
Hunting license (percentage)						
Ever	27	62	*81*	*82*	64	—
1991[d]	2	9	18	24	12	>23[e]
Viewed wildlife[f] (percentage)	83	77	68	52	71	—
No wildlife viewing or hunting experience[g] (percentage)	*14*	*9*	6	*11*	10	—

[a]All row values significantly different ($P < 0.05$; Pearson chi-square, except t-test for means) between groups, except italic values, which are not significantly different from one another.
[b]≥18 years old (ADL 1993), unless noted otherwise; "—" = not available.
[c]Urban includes communities with ≥2,500 inhabitants (ADL 1993); respondents residing outside state not included; groups B and C are significantly different from one another, but not from Group D.
[d]Respondents whose names were positively identified from list of 1991 hunting license purchasers; very conservative estimate due to ambiguity introduced through variable use of initials or possible data entry errors.
[e]Alaskans aged 16–59 who purchased a resident hunting license in 1991; this is a low estimate of total resident hunters because those aged ≤15 and ≥60 are not required to purchase a hunting license.
[f]Percentage answering yes to the question "Have you ever gone on an outing which included wildlife viewing as one of the things you planned to do?"
[g]Never purchased a hunting license or went on an outing that included wildlife viewing as one of the things they planned to do; groups A and B are significantly different from one another, but not from Group D.

Demographics and activities. The demographic characteristics of Group C tend to be between those of Groups B and D, except that the proportion of white and Native Americans is similar to that of Group D, and Group C has the smallest proportion of urban residents. Group C members are just as likely to have hunted in the past as Group D members; however, they are less likely to have purchased a hunting license in 1991 and are much more likely to have viewed wildlife. This group has the lowest proportion that had never purchased a hunting license or taken an outing where wildlife viewing was a planned activity. The most predictive characteristic identified in the CHAID analysis was past purchase of a hunting license (81 percent). Of these, 71 percent had a median education or lower. Of the 19 percent of Group C who had never purchased a hunting license, 35 percent were Native Americans.

Group D

Attitudes. The attitudes of Group D (19 percent of respondents) often were at the opposite extreme compared with those of Group A. Group D had the highest affinity for consumptive uses and hunters. Interestingly, their overall attitude toward trophy hunting was slightly negative (Table 2). Group D had the most negative attitudes toward wildlife viewing, particularly where it might affect hunting opportunities. On several selected management issues (Figure 2), this group was the strongest proponent of bear baiting, but opposed mandatory hunter education for experienced hunters and using Pittman-Robertson funds for wildlife viewing.

Demographics and activities. Group D tended to have the lowest proportion of females and longest Alaskan residency. The median education and income levels for Groups B, C and D were similar. Groups C and D had the highest proportions of rural residents and Native Americans. Group D had the highest proportion of hunters and lowest of wildlife viewers. The most predictive characteristic identified in the CHAID analysis was having gone on an outing that included wildlife viewing as a planned activity. Of the 52 percent that had done so, 88 percent also had purchased a hunting license in their lifetime. Of the 48 percent who were not wildlife viewers, 75 percent had purchased a hunting license in their lifetime. Group D included the second highest proportion of people with no hunting or viewing experience, though it was not significantly different from Groups A and B.

Discussion and Management Implications

Differentiating only four broad wildlife attitudinal groups is not a very sophisticated breakdown. The economic component of this survey had a higher priority, limiting the number of behavioral questions that might have helped identify more specific user groups. However, this study has given us valuable insights, and future research in Alaska can compare attitudes and demographic characteristics of other populations (*e.g.*, birdwatchers, wildlife activists, wildlife professionals) with Alaskan voters by measuring responses to the same 16 statements. Even at this rudimentary level, some interesting comparisons can be made.

The CHAID analyses indicated attitudes were best predicted by what a person did, rather than who they were. Specifically, a history of wildlife viewing or hunting was the best predictor in this study, followed by race and education. Gender and years of Alaska residency were significantly different between groups, but these and the other measured demographic characteristics were unable to predict attitudinal groups.

Most Alaskans engage in both consumptive and nonconsumptive uses of wildlife and fish. Most other states have high proportions of nonconsumptive users (USDI/USDC 1993). Alaska is unique in having the highest proportion of hunters, and a high affinity for consumptive uses, especially meat hunting and sport fishing, is expressed by three of the attitudinal groups (79 percent of all respondents).

The proportion of Alaskans (aged 16–59 years) purchasing hunting licenses declined 3 percent from 1978–1991, although it still is considerably higher than the national figure. Kellert (1980a) found 25 percent of Americans had hunted at some point in their life. About 64 percent of our respondents had purchased a hunting license in their life.

Some of this decline in the relative proportion of hunters is attributable to commitment of individual hunters over time. Slightly more than half of the American public who had hunted no longer did so (Kellert 1980a). Comparing our conservative estimate of voters who purchased a hunting license in 1991 (>23 percent) with the proportion of voters who had ever purchased a hunting license (64 percent) suggests a similar ratio.

Relatively few Alaskans express anti-hunting sentiments. Only 3 percent of Alaskans ≥ age 18 were opposed to hunting for meat in 1978 (Kellert and Berry 1980), compared to 8 percent of our respondents. Our questions were comparable, suggesting that the proportion of Alaskans who do not approve of hunting is increasing, although still lower than the 1978 national figure (14 percent). Group A is significantly less consumptive than the other three groups. However, this group appears to be characterized more by nonhunters than anti-hunters. A large proportion (27 percent) had purchased a hunting license at least once. This may explain Group A's relatively high approval of meat hunting, eating game meat and sport fishing (Figure 1). If their wildlife viewing needs are met, most members of this group are likely to continue to consent to meat hunting by other Alaskans.

Alaskans have maintained a high rate of participation in nonconsumptive activities associated with wildlife. In 1991, Alaskans had the highest participation rate for primary nonresidential nonconsumptive activities (39 percent) compared with other states, and tied with Vermont for the highest participation rate in all primary nonconsumptive activities (62 percent) (USDI/USDC 1993).

Alaskans have a lower frequency of humanistic, moralistic and negativistic attitudes than the American public (Kellert and Berry 1980). Respondents to this survey who had no history of hunting or wildlife viewing (10 percent) may have represented Alaskan voters with strong humanistic, moralistic or negativistic attitudes. The proportion of Alaskans with these attitudes is likely to increase as the proportion of the population living in urban areas increases (Kellert 1984).

A relatively high proportion of Groups A and B, which together comprise 53 percent of respondents, expressed no opinion when asked if (1) hunters, environmentalists or nonresidents have too much influence on wildlife management in Alaska; (2) it is more difficult to see wild animals that also are hunted; (3) more areas should be closed to hunting; and (4) they supported limited wolf control (Figure 1). Their level of uncertainty suggests that many members of these groups had not made up their minds on these major wildlife-management issues. When they did express an opinion, members of these groups tended to agree with Groups C and D that nonresidents have too much influence on wildlife management in Alaska, a common complaint of Alaskans. The attitudes of Groups C and D toward hunting and wildlife viewing tend to be more extreme and less uncertain than the attitudes of Groups A and B (Figure 1). Presumably, Groups C and D are less likely to be influenced by information and education programs or media coverage that conflict with their established attitudes.

This survey was conducted before Alaska's wolf-control controversy of 1992–1993. Attitudes can change when people receive additional information or experience. Undoubtedly, many Alaskans who were unsure about wolf control during this survey would express strong opinions now. It is possible that the Alaska Department of Fish and Game, through its inability to anticipate the level of unfavorable public reaction and media coverage (particularly from other states), lost an opportunity to communicate with many members of Groups A and B before the controversy reached crisis proportions. On the other hand, the much larger pool of anti-hunters and nonhunters with strong humanistic and moralistic attitudes outside of Alaska may overwhelm any statewide effort to obtain informed consent on controversial wildlife management issues.

The largest attitudinal difference between Groups C and D involves habitat protection (Figure 1, Table 2). About 58 percent of Group D did not think more concern should be given to protecting the land and water where wildlife live. This is a remarkable lack of concern for the group with the highest proportion of hunters. Only a small portion of Alaska has been developed, and this attracts those with a desire to make their fortune on the "last frontier." Group D may include a large share of Alaskans who promote and participate in oil and gas, timber, mining and construction industries. This attitude may persist, in part, because wildlife professionals have failed to convince hunters (and others) that habitat is one of the key factors in fish and wildlife conservation, and habitat protection is not an impediment to a healthy economy.

Often, our understanding of wildlife-related attitudes and behaviors and our ability to communicate with the public lags considerably behind our understanding of wildlife ecology. One of the true/false questions in the questionnaire was, "In Alaska, deer find more food during winter in forests that have never been logged than in those that have been logged." Beginning about 25 years ago, the Department of Fish and Game repeatedly has explained the adverse effect of heavy snow cover on availability of Sitka black-tail deer (*Odocoileus hemionus sitkensis*) foods in clearcuts to hunters and other residents of southeastern Alaska (which includes most of the state's deer, deer hunters and logging) in reports, meetings, journal and magazine articles, and news releases (McKnight 1979). Despite the poor showing of hunters overall in answering wildlife-related questions in our questionnaire, 40 percent of southeastern Alaska respondents who had purchased a hunting license in their life answered this question correctly (those with 1991 hunting licenses did even better—55 percent correct), compared with 21 percent of southeastern Alaska respondents who had not. We are making some progress with the group we have the most experience communicating with, but we still are not communicating well with the nonhunting public who comprise the majority of registered voters.

Acknowledgments

We thank T. A. Heberlein, R. C. Bishop, W. Martin and the entire staff of the Information Management Program, Division of Wildlife Conservation, Alaska Department of Fish and Game. Contributing agencies included the U.S. Fish and Wildlife Service Federal Aid Program, USDA Forest Service (Region 10), National Park Service and Bureau of Land Management.

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Leading from Behind to Solve Natural Resource Controversies

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Introduction

Many resource efforts stall or fail because of improperly handled controversy. Controversy stems from differing perspectives, information gaps or misunderstandings, but regardless, it fuels solutions to resource problems. While some view it as the enemy, effective leaders recognize and harness its power for resource efforts and understand the value of extremists. Such leaders "neither seek nor shun controversy" (Bleiker 1990), but know that non-controversial issues may die for lack of interest. Effective agency and non-agency leaders are essential in partnerships if people with strong and diverse opinions are to be guided toward consensus.

This paper describes "lead-from-behind" leadership (Nelson et al. 1993), a "non-controlling" agency tool for building partnerships. Its results can be seen in the Heron Lake Area Restoration Project (Nelson et al. 1993) and Carlsbad cases (Rideout 1993). Leadership roles at different levels, agency climate, roles of non-agency leaders, traditional role reversals, and attitudes toward controversies and extremists will be discussed. It describes a current understanding of the leadership web necessary to maximize public involvement and solve controversies.

Case Histories

Water Management and the Watershed Board (Board)

Heron Lake once was more than 8,250 acres (3,339 ha) of pristine Type IV prairie wetland, an autumn migrational stop for up to 700,000 canvasbacks (*Aythya valisineria*) and nesting habitat for 50,000 Franklin's gulls (*Larus pipixcum*), but, drainage activities in its 472-square mile (1,222 km^2) watershed have caused extensive flooding and damage in its four sub-basins.

The 1969 dynamiting of the state-owned Heron Lake dam after spring floods was an extremist attempting to control water levels. By 1989, the controversy turned into a public/private partnership for an integrated resource management effort which focused on the watershed's water quality, soils, fish, wildlife, education, local economy and recreation (including public access). Formation of the Heron Lake Area Restoration Association (Association) and its consensus plan are parts of a process for others to follow. The Board is one of 18 voting members of the Association.

Empowering local people was a fist step. It began with regular attendance at various local meetings. This led to the 1986 cooperative $380,000 repair of the dynamited dam by the Minnesota Department of Natural Resources (DNR) and the Board. Next, operation of the state dam was delegated to the Board by an agreement developed with local public participation. Operation of the dam remains controversial because its role in controlling water levels is not fully understood.

DNR cooperated with the Board when irate landowners demanded that private dikes along a channelized stretch of Jack Creek (tributary to Heron Lake) be removed. Despite their flooding concerns, the dike's owner refused to lower or remove them. The Board asked for help and DNR facilitated many on- and off-site meetings with individuals, the Board and the Jackson County Board. Upon consensus, DNR bought the diked land as a wildlife management area (WMA) and asked landowners for specifics on dike removal and water management. Landowners negotiated with each other and presented DNR with signed recommendations (to be review in five years).

Facing many issues, the Board wanted ecological expertise but lacked funding. The Nature Conservancy (TNC), North Heron Lake Game Producers Association (Game Producers) and DNR (through North American Wetland Conservation Act match money) provided the Board with two years of salary for a watershed ecologist (hired in 1993).

DNR is complying with the Board's request to partner on redirecting their flood control efforts from "steep and deep" impoundments to wetland restoration and flood plain buffer strips. An undeveloped impoundment site acquired by the Board many years ago will be acquired by DNR for use as a WMA. DNR will support the Board's request to redirect its $200,000 in legislative funding. The Board also is partnering on perpetual buffer strips on watercourses with the Board of Water an Soil Resources.

Since the dam was destroyed by persons unknown, a lead-from-behind approach has helped to defuse the control issue. In its place are productive partnerships and resource enhancement.

North Heron Lake Surface Use Plan

Though prized nationally for waterfowl hunting and abundant wetland wildlife, a 2,000-acre (810 ha) sub-basin (North Heron Lake) was difficult for the public to reach. Legal access was limited to riparian landowners and guests or those boating 2 miles down Division Creek form South Heron Lake. Landowners also signed a 1906 agreement restricting their own hunting use. Though a biologically sound waterfowl refuge, legislators and the public opposed "spending public money on a private hunting area."

The issue came to a boil when an access was acquired 1 mile upstream along Jack Creek. The Game Producers invited DNR to three meetings where open-access alternatives were discussed. All wanted the "refuge," but the Game Producers opposed public use opportunities.

The Game Producers talked to the DNR Commissioner. The most extreme member said "even one outside person on the lake is too many" and threatened legal action. The Commissioner refused to close off access and suggested more discussion of alternatives.

Early in 1993, the Game Producers hosted three public meetings on North Heron Lake's value. Then, they suggested dividing Heron Lake into six parts, each represented by a subcommittee of 7 to 11 members. It involved 55 local people varying widely in perspective and location, and 6 nonvoting agency personnel. A 13-person review committee, including 8 local and 5 nonvoting agency members, were to coalesce the six recommendations into one.

Each subcommittee met one to three times (they decided). The review committee reached consensus at its third meeting and presented findings at a public Association meeting. Management needs for guiding future committees emerged as a by-product.

The Game Producers published recommendations and sponsored a second public input meeting. Most of the 75 participants reacted favorably to the proposal. It was endorsed by the Association and sent to the Jackson County Board and DNR for action in early 1994.

This controversy turned into grassroots solutions for the access issue and also a list of future management issues. The Game Producers agreed to limited public uses and the public agreed that wide-open use of the lake would be detrimental to its special resources. Both cases illustrate solutions developed by the public with a role reversal—the public leads and the agency becomes advisory.

Leading from Behind

Leading from behind is a positive non-controlling and cooperative approach. Resource leaders using this approach believe that an informed public is the real power for solving resource problems and are the antithesis of the "I'm from the government and I'm here to help you" approach. Non-arrogant and leaving egos at the door, they careful listeners and advisors, and encourage non-agency partners to lead. They are open, flexible, creative, innovative, have a sense of timing and make and keep public promises (DePree 1992). Their words, actions and body language are low-profile, but they are essential parts of successful efforts. They value controversy and an informed public as a long-term source of consensus strategies. They know "publicly owned" plans succeed and plans "sold" by agencies usually don't. If asked to compromise their mission or handle a crisis they must switch to leading from in front.

Who Can Lead?

Unique qualities are required to lead from behind. The right attitude must be combined with the proper attributes, skills, experience and sense of timing. Agency managers often lack background for recognizing and rewarding lead-from-behind leaders.

Leaders learn their art by doing (DePree 1992), and lead-from-behind leaders are no exception. They usually are not agency- or self-appointed, but emerge *along* with trust levels in early project stages. They live near the developing effort and *earn* the necessary trust and support inside and outside the agency. Private local leaders are selected by peers and trusted by the resource leader.

Leading from Behind Takes Time

Lead-from-behind leaders must trust and be trusted to be effective, and realize honesty and integrity are the foundation for trust and the heart of principle centered effectiveness (Covey 1989). They are patient and persistent in the years needed to build two-way trust and provide information for sound consensus decisions and strategies. Later, time is required to maintain two-way trust.

Agencies may waste time appointing a leader, devising a plan without public involvement or using a SWAT team approach to sell agency plans. At the least, action waits while the resource leader spends years earning trust. At the worst, the "outsiders" will be resented enough to kill the effort before it gets started.

Planning in Action or Planning Inaction?

"Planning in Action" happens when the public trusts the agency, has ample information, initiates its own planning and chooses strategies. Plans developed by the public are more likely to be realistic, have easily understood goals and objectives (clean air, clear water, soil conservation) and be attainable. Agency planners (often the resource leader) must be close to the project and be a part of helping the public reach consensus.

Plans developed without public input run the risk of being presumptuous, insulting, and leading to "planning inaction." Action, not plans, is the product. Lech Welesa said, "There is a declining world market for words." Pure agency plans are just words.

The Control Issue

Whether launching an initiative or facing a suspicious or disgruntled public, authoritarian agency resource managers ar likely to be ineffective because of their tendency toward control. The struggle for control is an unproductive battle, an agency's "stone wall—its own Pickett's charge" (Duda 1992). The public, sensitive to agency control, prefers instead to be given facts and allowed to plan its own strategies. Resource management doesn't happen until it happens on the ground, and it won't happen until agencies defuse the control issue and work in public partnerships.

Getting through the control issue depends on the resource leader to: (1) listen, learn and empathize; (2) provide information as often as necessary; (3) participate in meetings regularly, but sit as equals, don't chair them and don't vote; (4) when discussing project progress, give credit to partners and use "we" instead of "I"; (5) avoid jargon whether spoken or written; (6) invite and welcome "radicals" to participate; (7) draft plans only if asked; (8) always be honest; (9) negotiate everything but your mission; (10) make and keep public promises; and (11) be patient and persistent.

Upper and Middle Management Climate for Lead from Behind Effectiveness

Proactive. Upper management shares the lead-from-behind attitude and offers proactive support and credit to creative on-the-ground resource leaders. Agency leaders empower, encourage risk taking, reward and fill in behind resource leaders so they don't have to do two full-time jobs. They inspire and train others with leaders from the "trenches." Creative people feel welcome.

Neutral. Upper and middle management are benign and tend not to celebrate good projects originating in the field. Creative resource leaders are allowed to take risks and accomplish projects but get little support unless it is requested. There is more agency planning than action and internal discussions are dominated by those who haven't demonstrated skills in on-the-ground efforts. Creative leaders feel less than welcome and accomplish on-the-ground resource enhancement despite agency "complicators."

Hindering. Management defends traditional approaches even if they aren't working (Duda 1992), and often resort to directing rather than coaching. Middle managers are threatened by successful projects not conceived by them or central office staff. Status quo managers are irritated by creative resource leaders with "new" problems because they require new solutions. Creative leaders leave the agency due to frustration.

Conclusions

Leading from behind was valuable in dispelling the control issue in the described controversies and many other unmentioned portions of the Heron Lake Area Restoration Project. Trust levels remain high and the project continues to move forward having focused over $6 million on watershed management since 1986. The power of controversy and the value of extremists was recognized as important to shaping resource management strategies. They are highly valuable in shaping long-term efforts.

Acknowledgments

I wish to thank Thomas Isley, Dr. Alfred Berner, Thomas Conroy, Spencer Amend, Verlyn Ebert, Lee Carlson, Randy Markl, Jay McAninch, Charles Mitchell, Art Olsen and Ken Varland for their help in preparing this paper, and the dozens of partners who have made the case histories successful, including the late James Thompson, member of the North Heron Lake Game Producers Association, who died of a heart attack before the public process he suggested for resolving the North Heron Lake access issue could be completed.

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Special Session 8. *Panel: Strategies for Conservation of Biodiversity on the National Wildlife Refuge System*

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Cochair
GLENN A. CAROWAN, JR.
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Opening Remarks—Conservation of Biological Diversity for the National Wildlife Refuge System: Challenges and Issues

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As the National Wildlife Refuge System approaches its 100th birthday in the year 2003, there is much discussion about the mission of the refuge system and how the U.S. Fish and Wildlife Service should be managing its magnificent 92 million-acre system. Although the system faces many challenges—including severe underfunding and continuing criticisms of incompatible uses on refuges—this special session focuses on one question: "What should the role of national wildlife refuges be in managing for biological diversity?"

There are a number of recent initiatives both within and outside of the Fish and Wildlife Service that are focused on this question. Foremost, Secretary of the Interior Bruce Babbitt and the Service's new Director, Mollie Beattie, have both identified conservation of biological diversity and ecosystem management as primary goals for the Service. Second, Director Beattie has stated that she intends "to make the [national wildlife] refuges the anchor points for biodiversity and demonstrations of ecosystem management, and to provide a reserve system of the nation's typical ecosystems." Third, the Fish and Wildlife Service has issued the draft "Refuges 2003," the environmental impact statement and management plan that aims to guide the refuge system into the next century. The conservation of biological diversity and ecosystem management play pivotal roles in several of the proposed alternatives. Last, bills have been introduced to the Senate and House proposing organic legislation that would provide the general framework and philosophy for managing the system. The emphasis of this legislation is on ecosystem management. Given these numerous and diverse initiatives, there are some major policy changes on the horizon for the National Wildlife Refuge System. It also is clear that the conservation of biological diversity and ecosystem management will play prominent if not the preeminent roles for defining the future mission and goals of the system.

Recognizing these new challenges, the Wildlife Management Institute, in cooperation with the Division of Refuges, U.S. Fish and Wildlife Service, initiated a project in Spring 1993 to gather information relating to the appropriate role of national wildlife refuges in the conservation of biological diversity. To accomplish this task, we developed a series of questions for obtaining information on the relationship between management programs and biological diversity on refuges. We conducted site visits at six refuges across the country, representing varying refuge management programs. There were two major objectives for the project, including: (1) determine if our questions could provide refuge managers a framework for evaluating the potential role of their refuges for contributing to biological diversity, and (2) identify the current contributions these refuge management programs were making to the conservation of biological diversity and identify potential constraints.

Focusing on this second objective, we found that all the refuge units we visited currently were making very significant contributions to the conservation of biological diversity. Even at units such as Sacramento National Wildlife Refuge, which many regard as a refuge intensively managed for only waterfowl, we found that there were substantial management efforts being directed at a wide variety of sensitive natural communities and taxa, including rare vernal pools and riparian habitats, neotropical migrant birds, and rare invertebrates and fishes. Furthermore, refuge staff eagerly supported the goal of conserving biological diversity. However, a number of common issues emerged from our refuge visits, including:

1. A new funding initiative is critical if the conservation of biological diversity is to be a primary goal for the NWR System. Existing resources are simply too marginal to shift existing resources to new program initiatives.
2. There also is a need to prioritize biodiversity goals at the national and regional levels and to provide consistency to these goals over more than one fiscal year. Washington and regional offices of the Service need to provide leadership for incorporating biodiversity objectives into refuge management programs.
3. Managed areas play a key role in managing for biological diversity as do natural areas. Many units simply are too small for a "hands-off" management policy, and refuge units are affected by land-use activities on adjacent lands.
4. The long-term maintenance of refuge biota depends on partnerships with private and public landowners in the region. Again, refuges are too small and budgets too limited for the Service to succeed without partners. Additionally, management programs must take into consideration existing conditions and opportunities considering the regional landscape matrix and socio-economic conditions.
5. Regional or ecoregion efforts need to be made to provide managers with access to information and modern technologies (i.e., GIS and satellite image processing) to facilitate local and regional planning.

Although refuge staff identified a variety of other issues, these five were consistently identified as being major constraints to implementing an effective national program for the conservation of biological diversity. To that end, the participants in this session will share with you their perspectives on many of these policy and management issues relating to implementing biological diversity management programs in the National Wildlife Refuge System.

Changing the Way We Look at the Land

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Introduction

The National Wildlife Refuge System (System) is a unique assemblage of lands and waters which, collectively, provide vital habitat for fish and wildlife throughout the United States. These units, ranging from less than 1 acre to more than 20 million acres, were brought into the System through a variety of legislative and administrative processes. The purposes for which individual refuges were established range widely; there is, in fact, no overriding, legislated purpose for the System.

The Refuge System faces enormous challenges as it approaches its 100th Anniversary. The diverse and outspoken "customers" that share intense interest in the future of this System differ widely in their views on how the System should be managed. Refuges are challenged by the accelerating abuse or loss of surrounding habitat and by competition for scarce resources, such as water, on which they depend. The list of mandated responsibilities of facing today's refuge manager continues to grow with each new Congressional session. But these challenges pale in significance when compared to the chronic shortfall in operational and maintenance funding that plagues this System, making it impossible for individual refuges to fully achieve the purposes for which they were established or to embrace an expanding mission.

Role of the Refuge System in the Conservation of Biodiversity

The Refuge System already plays a significant role in the conservation of this nation's biological diversity, at the species, community, ecosystem and landscape levels. Nowhere in the System is this more evident than in Alaska. But it is equally true on small refuges providing essential habitat for critically endangered species or on refuges that protect remnants of imperiled ecosystems in other states and territories. Even the more "traditional" refuges that protect habitat for migratory waterfowl contribute substantively to national species conservation objectives.

Notwithstanding its current value, there is enormous potential for the Refuge System and other protected lands to play a more significant role in the conservation of biological diversity. Indeed, the Refuge System can be a fundamental cornerstone for the effective management of ecosystems. There are numerous examples, particularly in the last decade, where individual refuges have risen to this occasion. Yet, to institutionalize the Refuge System role will require a profound change in the way we look at the land. It will require all refuge managers to view their role in an ecological context, to expand their thinking to include not only the species for which they manage, but also the ecological processes that sustain them. The significance of administrative boundaries will diminish as refuge managers implement innovative ecosystem conservation strategies in concert with other U.S. Fish and Wildlife Service (Service) professionals and a wide variety of public and private partners.

Signs of Change

There are encouraging signs of change that reflect a growing commitment by the Service to more fully embrace the principles of ecosystem management and to expand the role of the Refuge System in the conservation of biological diversity. It is useful to illustrate this point with a brief review.

Service Policy

The Service has been progressing on several fronts that reflect an evolution in policy. President Clinton's signature on the Rio Biological Diversity Convention in June 1993 marked a significant change in U.S. policy. The Service continues to play a lead role in defining the Department of the Interior's strategies to implement provisions of the Convention. The Service also is a founding member of the Interagency Ecosystem Management Coordinating Group, involving over 20 federal agencies working together to define and pursue common goals.

After several months of internal effort and work in partnership with other agencies and organizations, the Service also developed a "Strategic Plan for the Conservation of Biological Diversity." While it remains in draft, this Plan has helped to shape ongoing discussion by the Service Directorate relating to ecosystem management. The most recent product of this discussion is a March 1994 concept paper entitled "An Ecosystem Approach to Fish and Wildlife Conservation." This paper describes a partnership strategy to conserve natural animal and plant diversity through the perpetuation of dynamic, healthy ecosystems.

Evolving Service policy with direct bearing on the Refuge System is best reflected in the ongoing development of "Refuges 2003—A Plan for the Future." This combined System Plan and Environmental Impact Statement was released in draft to the public in March, 1993. The role of the Refuge System in the conservation of biological diversity was among the issues of most concern to the public. More than 22,000 comments were received on the Draft Plan/EIS. A Final Plan/EIS, scheduled for release in fall, 1994, will provide clear direction for the Refuge System into the next century. The Service also has been working closely with its diverse customers and Congress in the development of "organic" legislation for the Refuge System. The Service strongly supports passage of constructive legislation, first and foremost because it will establish clear, legislated purposes for the System as a whole. Conservation of biological diversity will be one of those purposes.

Service Procedures

Clear policy, in and of itself, will not ensure a fundamental change in the way lands in the Refuge System are managed. Several initiatives are underway to change the procedural guidance relating to both acquisition and management of refuge lands. As an example, the Service's Land Acquisition Priority System recently was modified to incorporate more effective consideration of the biodiversity values of lands under review for acquisition. The new "biodiversity target" enables Service regions to submit acquisition proposals that can compete successfully with other land values (e.g., endangered species, waterfowl habitat, etc.). As we improve the data on which this process depends, the role of the Refuge System in protection of imperiled biological communities and underrepresented ecosystems will be enhanced.

Procedural guidance also is in preparation relating to the inventory of species and communities on refuges. We are working closely with Gap Analysis researchers to assess the application of their approach to terrestrial vegetation classification for refuges. We also are working with The Nature Conservancy in developing guidance to support community inventories on refuges. Finally, we are drafting procedural guidance and criteria for prioritizing wildlife inventory activities on refuges. Collectively, these initiatives will enable refuge managers to plan and assess the effectiveness of their management strategies in an ecosystem context and focus their time-consuming biological programs where they will do the most good.

Most refuges have not completed station management plans that establish clear objectives and strategies to achieve them. Many of the plans that have been completed lack the benefit of substantive public involvement when they were developed. Several plans do not reflect serious consideration of the ecological context in which the refuge is found. To correct these deficiencies, procedural guidance is being developed for inclusion into the Service Manual. If enacted, pending organic legislation also would strengthen the policies and procedures for refuge planning.

Service Programs

The Service recently has embarked on numerous nationwide "programs" designed to conserve biological diversity at a landscape scale. The Refuge System plays an integral part in each of these programs. The various "Joint Ventures" of the North American Waterfowl Management Plan each include numerous refuge wetland enhancement projects, in conjunction with significant contributions by states and private partners. More than 15 Habitat Conservation Plan permits have been issued by the Service since 1983, all pursuant to the Endangered Species Act. Refuges typically play a very important role in these partnership initiatives.

The Service's "Partners for Wildlife" program is focused on the restoration of wetlands and associated habitats on private lands, in cooperation with landowners. Refuge staff participate actively in the planning and implementation of these projects. Developed habitats on private lands function in concert with refuges and other habitats under management by states and other organizations. Another partnership program with diverse wildlife benefits is the "Partners in Flight" initiative, designed to promote conservation of neotropical migratory birds. Again, refuges figure prominently in the habitat conservation strategies of this program.

The protection of coastal wildlife habitats is critically important to the conservation of this nation's biological diversity. The Refuge System is an important component of two Service programs underway in coastal areas. "Coastal America" was begun in 1991 as a multi-agency initiative to preserve coastal environments. To date, more than 20 federal agencies have cooperated with more than 100 nonfederal partners to fund projects in 15 states. The "Bay-Estuary" program is a classic example of habitat conservation at the watershed level that has proven enormously successful in mobilizing the diverse capabilities of many partners. Beginning with the Chesapeake Bay initiative, this program has grown in the last decade to include nine major bays and estuaries. Again, refuges contribute substantively to habitat management, landowner extension and education strategies within each bay-estuary project.

One new initiative with major implications to the ecosystem role of the Refuge System is the North American Free Trade Agreement (NAFTA). As a result of the signing of NAFTA, planning is actively underway to promote the conservation of ecosystems and species that are shared with Mexico. Wildlife inventories and habitat restoration activities will accelerate on several border refuges in support of NAFTA.

Service Projects

There are numerous individual projects in progress that illustrate the evolving role of the Refuge System in the conservation of biological diversity and, in particular, the ecosystem approach to resource management. Eight individual projects are described briefly that will demonstrate the wide variety of approaches in planning.

Connecticut River Planning Project. The Silvio Conte National Fish and Wildlife Refuge Act authorized the Service to study the entire Connecticut River basin and evaluate the potential for establishment of a new wildlife refuge. The planning area transects four states. Anticipated conservation strategies will involve urban, agricultural and natural areas. To gather input from the diverse agencies, organizations and private citizens in the area, the Service has made more than 100 presentations and held 19 public meetings. ''Issue Workbooks'' have been distributed to identify concerns, values and vision for the future of this river basin. Several workshops will be held this year to identify alternatives to conserve and enhance the natural resource values of the basin. At this point, it is not a foregone conclusion that there will even be a national wildlife refuge in the traditional sense, but the land protection concept has provided an important catalyst for basin-wide conservation planning.

Blackfoot River Watershed Project. This project integrates a variety of land protection strategies to conserve the natural diversity of species and habitats within a 125-mile-long watershed in northwestern Montana. This 250,000-acre basin includes glaciated wetlands, riparian corridors, native prairie and forested habitats. Threats to the area include overgrazing, logging, mining and, most recently, residential development. It was the last of these threats that prompted local landowners to seek the assistance of the Service. This coalition was built from the ground up.

The underlying strategy in the valley has been to utilize diverse tools to protect habitat without fee acquisition of additional lands. When complete, the project will link Refuge System lands (waterfowl production areas and easements) with state Wildlife Management Areas, Nature Conservancy easements and over 70 projects on private lands, implemented under the Service's Partners for Wildlife program. In FY1993, $245,000 in Service funding were used to leverage $770,000 for habitat restoration and protection activities in the watershed.

South Carolina Coastal Ecosystems Project. This project encompasses a coastal area in excess of 4,500 square miles, including more than 50 distinct natural communities and 18 listed species. Efforts are concentrated within five ''focus areas'' ranging from 350,000 to 725,000 acres. Each focus area was established on the basis of watersheds, their individual assemblage of plants and animals, and their different sources of threat. Coordination of actions within each focus area is the responsibility of a task force representing agencies, organizations and private landowners. The goal of the focus areas is to achieve long-term habitat protection through perpetual voluntary conservation easements, habitat management on selected lands and through regulatory processes. There presently are five national wildlife refuges (and one in the planning stage) that participate in the project.

One of the focus areas is the ACE Basin, a flagship project of the North American Waterfowl Management Plan. This project has extensive involvement by private landowners and nongovernmental organizations. Currently, 100,000 acres are protected in perpetuity with another 100,000 acres projected. The ACE Basin National Wildlife Refuge is an important cornerstone of this land protection strategy.

Cameron County Agriculture Wildlife Co-existence Committee. When refuge staff look beyond their boundaries to address problems at the ecosystem level they often may find themselves embroiled in conflicts between wildlife conservation and agriculture. One such conflict erupted when the Environmental Protection Agency and the Service began consultation on the impact of pesticide use in Cameron County, Texas on the endangered Aplomado falcon. At one point, the use of 40 pesticides was scheduled for elimination in the county, an action that would have destroyed a $100 million/year agricultural industry.

The Agriculture Wildlife Co-existence Committee was formed to find solutions to this conflict. Steve Thompson, manager at Laguna Atascosa Refuge, was the Service's principal representative on the Committee. Thompson worked with the agricultural community to explore alternative pesticide application methods and rates as well as the use of alternative chemicals. Gradually, the conflicts dissolved and options were developed that allowed farming to continue without sacrificing protection of endangered wildlife. The work of the Committee has become a model for cooperation between wildlife managers and farmers nationwide.

Klamath Basin Ecosystem Restoration Project. This interagency project began in response to the conflicts over water and, in particular, the strategies implemented to protect the habitat of two species of endangered suckers. The long-term objective is to develop partnerships between government and private entities to facilitate habitat restoration on private and public lands. Riparian losses, forestry and agricultural practices, and rangeland abuse all have impacted the quality and quantity of water in the basin. Five national wildlife refuges are within the planning area and will participate in the project. The interdisciplinary approach to this project is reflected in the appointment of staff to the Ecosystem Restoration Office from the U.S. Fish and Wildlife Service, Bureau of Reclamation, Bureau of Land Management, USDA Forest Service and from Klamath Tribes. This office will coordinate the federal agency role in the development of habitat conservation and planning strategies, research investigations and management of Klamath Basin digital data sets.

Upper Mississippi and Missouri River Restoration Project. Sometimes the catalyst (and opportunity) for change in the way we look at the land results from natural disasters. The Great Flood of 1993 was one such event. In response to the flood, several public agencies and private organizations are assessing the alternative strategies to restore habitat impacted by this flood. The Service is an active player in this interdisciplinary effort, in part because eight national wildlife refuges were directly affected by the flood.

A FY1993 supplemental appropriation of nearly $25 million will support the refuge repair work. Prior to the expenditure of funds on Service land for levee repair or reconstruction, managers are evaluating options that will (a) relocate or redesign flood prone facilities, (b) ensure that facilities do not contribute to local flooding, (c) reduce the reliance on levees that may prevent desired restoration of natural ecosystems, and (d) utilize strategies that favor self-sustaining native plant and animal communities. The Service also is mobilizing its other technical assistance, planning and project review functions to keep water and soil in place in the watershed and enhance the natural functions of floodplains.

Sandhills Management Program. The Sandhills is a contiguous 19,600 square mile sand dune formation covered by grasses, located in northcentral Nebraska. Approximately 1.3 million acres of wetlands, formed by groundwater discharge, are scattered throughout. This ecosystem supports a wide diversity of wildlife and plant communities, as well as a strong cattle ranching economy. Yet, both are threatened by lowering of groundwater, exotic plants, grassland conversion and other factors. The goal of this program is "to build a partnership with landowners and other agencies to enhance the Sandhill wetland-grassland ecosystem in a way that sustains profitable private ranching, wildlife and vegetative diversity, and associated water supplies."

To initiate this Program, the Service facilitated appointment of a 14-member Sandhills Task Force, representing various agencies and the ranching community, to develop a Sandhills Management Plan. This Plan envisions a combination of land protection strategies, education and technical assistance, legislative initiatives and financial support. Even though there are three national wildlife refuges within the Sandhills, significant expansion of fee title refuge lands in the area was not a viable option, because of enormous cost and landowner opposition. Instead, the program will involve extensive use of voluntary conservation easements and lease agreements. The easements will compensate landowners for specific rights purchased, such as the right to drain or fill wetlands and convert grassland to cropland. This approach is dependent upon the building of partnerships to achieve common goals and will be a model that can be emulated in many areas of the country.

Alaska Maritime. Preserving the function and integrity of natural ecosystems often may involve aggressive approaches to prevent the invasion of non-native species. One such example is the Alaska Maritime National Wildlife Refuge, where the accidental establishment of rat populations on remote islands would permanently disrupt the ecology of critically important seabird nesting colonies. Refuge staff are working closely with the Coast Guard, National Marine Fisheries Service, Department of Agriculture, state agencies, fishing and shipping industries, and local communities to develop a comprehensive prevention program. In the Pribilof Islands, training programs are underway, poison bait stations have been established and local ordinances have been passed to prohibit ships with rats from entering some harbors. On Shemya Island, where rats were introduced during World War II, Air Force and refuge staff have embarked on a three-year eradication program. If successful, Shemya will be the largest island in the world from which rats will have been eliminated.

Looking to the Future

While the many "signs of change" are encouraging, the Refuge System is some distance away from the institutional course correction necessary to fully embrace the conservation of biological diversity as a primary purpose. The opportunity lies in our ability to build on our successes, learn from our failures and explore new and innovative strategies in the future.

Adopt an Ecosystem View

The Refuge System does not exist in a vacuum. Most refuges outside of Alaska are small islands that function within a highly altered landscape. Our vision of conserving the nation's biological diversity through perpetuation of healthy, dynamic ecosystems will release us from a view constrained by white boundary signs. But it also will require increased sensitivity to the diverse interests and responsibilities of the many other players who own and manage far more real estate than does the Service.

Refuges need to be part of coordinated efforts to protect and/or restore the function, structure and species composition of the ecosystems within which they are found. Indeed, in many ecosystems, refuges and other protected areas can be the "cornerstone" of the ecosystem approach to conservation. This concept should be reflected in the earliest planning efforts relating to new land acquisition projects, as well as in the management programs of long-established refuges.

Many refuges of tomorrow may look a bit strange by today's standards. The central "core" of selected refuges may be rigidly protected and free from manipulation. Surrounding lands would be managed more intensively to achieve specific habitat objectives. On some refuges, degraded lands will be the focal point of experimental restoration efforts, including the replanting of native plant species, reintroduction of native vertebrates, prescribed burning to mimic natural processes and control activities to eliminate non-native species. Where better than a national wildlife refuge to experiment with evolving techniques to restore degraded ecosystems?

The Refuge System of tomorrow also will make greater use of land protection alternatives to fee acquisition. Examples include cooperative agreements, non-development easements, economic incentives to landowners and technical assistance. While these strategies are in use today, what is typically lacking is a well-coordinated, landscape-level approach where the combination of techniques is choreographed to best protect and restore the structure and function of the ecosystem.

Mobilize the Service Toolbox

The Service brings a unique mix of land protection tools to the table but, historically, has not effectively mobilized these diverse capabilities in a coordinated fashion. Service programs often have worked in parallel or, worse yet, conflicting directions as they exercise their mandated authorities. The result has been confusing to the public and far less effective than is possible.

The Service now is wrestling with the difficult transition from a program-focused organization to one which will promote effective team approaches to ecosystem management. The Refuge System of tomorrow will reflect this transition. Lands for inclusion in the Refuge System will be selected to function in concert with one another, with other state-managed conservation areas and with habitat on private lands. Refuge management activities will be undertaken that complement the regulatory, technical assistance, habitat restoration and information transfer roles of the

Service, all within an ecosystem context. The cross-fertilization of Service expertise will benefit the Refuge System directly. This talent pool is a wellspring for creative change, if tapped effectively.

Clarify our Vision

Completion of "Refuges 2003" and the enactment of pending Refuge System organic legislation will serve to clarify the Refuge System role in the conservation of biological diversity. One part of the vision should be to represent that diversity within the boundaries of the System. At the landscape level, the Refuge System should represent each major ecosystem type sufficient in size and condition to allow ecological processes to continue. At the community level, the Refuge System should strive to protect the full range of variation within ecosystems. At the species level, the System objective should be to maintain viable populations of native species in their natural patterns of abundance and distribution. Of course, to achieve these objectives, refuges must be established and managed so as to function more effectively in concert with other lands in public ownership and with habitat on private lands.

Equally important to defining a vision is the task of planning strategies to achieve that vision and measures to document progress. Again, "Refuges 2003" will provide the Systemwide planning framework. Ecosystem-level planning initiatives, such as the Connecticut River Valley Project, will be needed throughout the country to set broad goals for large, ecologically related blocks of land. Within this context, these goals must be translated to the ground through accelerated management planning on individual refuges.

Commit to Science

Elevating the role of the Refuge System in the conservation of biological diversity will require a commitment to enhance the variety, quality and utility of data upon which strategic land protection and management decisions are made. Among the additional tools needed will be a standardized vegetation classification system, refined community inventory and classification techniques, technical guidance for monitoring a wider array of species and hardware and software to expand use of spatial data. National standards to guide the periodic review of refuge biological programs also will be needed to ensure we are collecting information that is most relevant to our expanded mission.

The transition in biological programs will require a commensurate commitment to enhanced training of refuge staff. It also will require broadening of our rather narrow existing perspective on the types of skills, education and experience necessary to manage a refuge. Understanding and managing resources in an ecosystem context will necessitate that our talent pool expand to include more conservation biologists, restoration ecologists, invertebrate zoologists, plant taxonomists, hydrologists and soil scientists, to name just a few.

Promote our Partners

Now more than ever, refuge managers must look beyond their boundaries to protect the integrity of the lands they manage and to work as partners to preserve and restore the structure and function of whole ecosystems. The Service does not have legal authority, technical ability, public support or funding to do it alone. The refuge manager of the future may play a variety of roles, from facilitator to cooperator to leader, depending on the circumstances. He/she must display flexibility, demonstrate awareness of the priorities and needs of partners, and communicate more frequently and openly.

Spread The Word

The Refuge System is uniquely suited to educate the American public about the critical link between the sustainability of ecosystem function and the exploitation of natural resources. Interpretive programs and facilities on refuges provide a relatively low-cost vehicle to enlighten the visiting public about the natural world. Refuges in expanding urban areas will play an increasingly more critical educational role for city dwellers who otherwise would lack exposure to wildlife. Yet, refuges also must serve to demonstrate the evolving techniques to conserve the natural diversity of species and to restore the function of degraded habitats.

Summary

The prospect of institutional change always is unsettling and particularly so when the results are uncertain. The Refuge System role in the conservation of biological diversity has been a lightning rod in the continuing debate over "Refuges 2003" and pending organic legislation. Yet the growing number of successful ecosystem-based projects involving refuges illustrates that this is a natural and desirable evolution in Refuge System policy that is well underway.

Progress to date also illustrates that this transition need not pose a threat to those who have helped build the Refuge System and are concerned about the effect on historic priorities and traditional, wildlife-oriented uses. Neither should this transition be of concern to those agencies with principal responsibility to manage resident species nor private landowners whose first priority is to reap their living from the land. This evolution in Service policy represents a recognition that protected lands must be part of, not separate from, ecosystem-based conservation strategies. Ultimately, the success of this initiative is dependent upon the effective coordination and cooperation of diverse partners who are equally dependent upon the perpetuation of healthy, dynamic ecosystems.

A Regional Perspective for Conserving Biological Diversity in the National Wildlife Refuge System

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As the only system of federal public lands established specifically to conserve wildlife, the National Wildlife Refuge System should be at the forefront of the nation's efforts to conserve biological diversity—a term defined recently by the U.S. Fish and Wildlife Service (Service) as "the variety of life and its processes, including the variety of living organisms, the genetic differences among them, and the communities and ecosystems in which they occur" (U.S. Fish and Wildlife Service 1994). To meet this challenge, the Refuge System must broaden its mission from the more traditional and narrow focus of individual refuges and pay closer attention to the regional context in which management decisions are made, land acquisition priorities are established and cooperative relationships are fostered.

This paper describes in general terms a regional perspective for conserving biological diversity in the Refuge System and presents two examples where the Service has enthusiastically implemented this approach and two examples where such an approach is lacking.

Myths About Conserving Biodiversity

Before we discuss what the Refuge System already is doing to conserve biological diversity and what additional actions it can take in the future, it is important to explain first what biodiversity conservation should *not* entail. First and foremost, conservation of biological diversity should not be interpreted to mean hands-off management. Prescribed burning, manipulation of water and other forms of active management—including farming, grazing and forestry in some situations—will continue to be important management tools. Aggressive management to eradicate or at least contain invasive exotic species also is an essential component of biodiversity conservation. For example, in Alaska, eradication of foxes (*Alopex lagopus* and *Vulpes fulva*) and other alien predators from islands within the Alaska Maritime National Wildlife Refuge is a major priority that involves active management.

Equally important, biodiversity conservation should *not* entail management to maximize the total number of species or communities on any one plot of land. In particular, management to enhance species that are common elsewhere to the detriment of rare species for the sake of total species diversity is ill founded.

Finally, the Refuge System should *not* abandon management to achieve the specific purposes for which refuges have been established.

Direction to Conserve Diversity

Currently, the Refuge System is managed less as a system than as a collection of individual units. Most of the nation's nearly 500 refuges were established with a narrow set of purposes, such as protection of a particular endangered species or conservation of certain migratory birds. With the passage of the Alaska National Interest Lands Conservation Act in 1980, Congress gave each of the Alaska refuges a purpose to "conserve fish and wildlife populations and habitats in their natural diversity." Few refuges outside of Alaska have such a specific concern for diversity as part of their charter.

As "organic" legislation is debated for the National Wildlife Refuge System, it appears likely that Congress will supplement the traditional refuge purposes with a broad directive to conserve the full diversity of fish, wildlife and plants and the ecological processes that sustain them. If Congress takes this step, the Refuge System then would join the National Forest System as the only federal public land networks with explicit legislative direction to conserve diversity.

With the passage of the National Forest Management Act in 1976, Congress required that the conservation of biological diversity be a guiding management objective of the National Forest System. This statute requires that plans be developed for individual national forests that "provide for diversity of plant and animal communities. . . ." Implementing regulations require the USDA Forest Service (Forest Service) to ensure that "fish and wildlife habitat shall be managed to maintain viable populations" of vertebrate species on National Forests.

The Service already has indicated an interest in making conservation of diversity a major objective of the National Wildlife Refuge System. The Service's Director, Mollie Beattie, has articulated a vision in which refuges are "anchor points for biological diversity" and are "demonstration areas for management of biological diversity" (Beattie remarks to Biodiversity and the Law: Challenges and Opportunities, conference of Defenders of Wildlife, March 11, 1994). Realizing this vision will require some fundamental changes of perspective.

A Regional Perspective for Refuge Acquisitions

To a certain extent, the Refuge System is captive to its history. As a result of its traditional focus on migratory birds, for example, the System is relatively well endowed with certain habitats, such as emergent wetlands, but lacking in others, such as southern old-growth pine forests. The System currently contains 31 million acres of wetlands of which 1.6 million acres (1.7 percent of the System) are actively manipulated (U.S. Fish and Wildlife Service 1993a).

Broadening the focus of the System to address conservation of biological diversity will require an aggressive land acquisition program which should emphasize areas with outstanding assemblages of rare, declining and poorly protected species and natural communities. This is already occurring. For example, 58 refuges have been acquired specifically to conserve a particular threatened or endangered species or group of listed species. Refuges have been established for scores of listed species—from golden-cheeked warblers (*Dendroica chrysoporia*) to loggerhead sea turtles (*Caretta caretta*) Columbia white-tailed deer (*Odocoileus virginianus leucurus*) to West Indian manatees (*Trichechus manatus*), Iowa Pleistocene snails (*Discus macelintocki*) to Ash Meadows blazing stars (*Mentzelia leucophylla*) and dozens of other imperiled plants and animals.

Beyond protecting critical areas for the nation's most imperiled species, the Service should adopt a regional perspective in establishing priorities for land acquisition. Several land classification schemes have been developed to describe the North American continent. For example, Bailey has mapped the continent's ecosystem types (Bailey 1983). The Service recently has unveiled another such classification scheme (U.S. Fish and Wildlife Service 1994). Whichever scheme is chosen, protection of those areas that are critical to the integrity of the nation's major ecological units should receive strong emphasis in the Service's land acquisition program. For example, acquisition of the substantial inholdings in many of the large Alaska refuges is a major priority to enable true ecosystem management of these refuges.

Michael Scott and others have initiated an ambitious effort to identify areas of high species richness which currently are not protected (Scott et al. 1987). The Service used this methodology to determine priorities for land acquisition at the Hakalau Forest National Wildlife Refuge in Hawaii—a refuge established to conserve endangered forest birds. A logical future use of the information generated from Gap analysis is to acquire the "gaps" for inclusion in the Refuge System. This presupposes that these lands will be managed from a regional perspective.

Refuge Inventories to Determine Management for Regional Biodiversity

An effective program to conserve biological diversity must begin with a thorough inventory of the species and natural communities on refuges. The most recent information available on refuge inventories was collected in 1991 by the Division of Refuges as part of Refuges 2003—the Service's *Plan for the Future of the National Wildlife Refuge System*. Unfortunately, of 478 refuges reporting, relatively few indicated that they had completed wildlife inventories (Table 1).

To successfully implement a regional approach to conserve biological diversity, the Service must not only know what resources are found on refuges, it must also have a thorough understanding of the natural systems around refuges. To determine which species, natural communities and ecological processes are regionally rare or declining, and therefore in need of special attention, will require close coordination with state natural heritage programs and organizations like The Nature Conservancy. Unfortunately, in the Service's 1991 review of the Refuge System, only 85 (18 percent) refuges indicated that they were incorporated into a state natural heritage program.

Regional Perspective on Refuge Management Decisions

What should the Service do with lands within the Refuge System? Simply stated, management for biological diversity should take special care of those native species, natural communities and ecological processes that are regionally or nationally rare, declining or poorly protected.

At one end of the diversity spectrum are those species that have been listed as threatened or endangered or are candidates for such listing by the federal or state government. At least 180 federally listed species and 350 candidates for federal listing occur on refuges; listed species have been introduced on 17 refuges and 51 refuges

Table 1. Refuges conducting wildlife inventories.[a]

Taxa	Refuges with inventories (percentage reporting)
Birds	286 (59)
Mammals	152 (32
Plants	145 (30)
Fish	97 (20)
Reptiles	95 (20)
Amphibians	88 (18)
Invertebrates	24 (5)
Natural communities	138 (29)

[a]Data from U.S. Fish and Wildlife Service *Refuges 2003* database.

contain designated critical habitat for listed species (Young 1993). Refuge management plays an important role in the recovery plans of a number of species.

But too often even listed and candidate species have been harmed by secondary uses on national wildlife refuges (Eaton and Waltman 1992). For example, the Service acknowledged that public recreation at Crystal River National Wildlife Refuge in Florida was causing "a majority" of West Indian manatees to leave the refuge (58 *Federal Register* 28382, May 13, 1993). The Service also acknowledged that grazing at Turnbull refuge in Washington was not only degrading nesting habitat for waterfowl and other migratory birds but also was damaging several rare plants including water howelia (*Howelia aquatilis*), a candidate category 1 plant (U.S. Fish and Wildlife Service 1990).

Audubon and several other conservation organizations filed a lawsuit to force the Department of the Interior to suspend harmful secondary uses that were allowed on refuges (*National Audubon Society v. Lujan*, Civ. No. 92-1641). Fortunately, the Department of the Interior and the Service have made a commitment to prevent incompatible uses in the future and we were able to reach a settlement of the litigation. But the Service should be more ambitious than merely achieving the vague legal standard of compatibility. Any "take" of listed species on a national wildlife refuge is unacceptable, unless done for scientific purposes or to enhance the survival of the affected species, and the Service should adopt a policy that says as much.

Beyond threatened, endangered and candidate species, management priorities become less clear. The Forest Service's controversial implementation of its diversity mandate has demonstrated how this seemingly simple concept can be interpreted in a number of different ways. For example, at the Ottawa National Forest in Michigan, the Forest Service defended itself from critics concerned by the impacts of forest fragmentation on area-sensitive songbirds by arguing that "[t]he 'negative' edge effects claimed by [these critics] are indeed positive effects with respect to the plant and animal diversity of the Ottawa. Our intent is to provide habitat for all native vertebrate species, including nest predators such as the blue jay and nest parasites such as the catbird (sic). . ." (Wilcove 1988).

The Forest Service has been roundly criticized for justifying clearcuts and other logging prescriptions within National Forests as beneficial to diversity (Noss 1983, Wilcove 1988). But traditional wildlife management practices often have emphasized techniques to maximize the numbers of species on a given preserve by maximizing the diversity of successional stages and habitat types—including edge habitat (Giles 1971). Such practices, whether implemented by the Forest Service or managers of wildlife reserves, fail to adopt a regional perspective.

Rather than maximizing species richness—the number of species on any one refuge—the Service should focus refuge management to conserve and restore those native species, natural communities and ecological processes that are regionally or nationally rare, declining or poorly protected.

Habitat Networks to Conserve Biological Diversity

While many of the large refuges in Alaska encompass complete ecosystems or substantial parts thereof, most of the refuges in the remainder of the Refuge System are too small to represent self-sustaining systems and are strongly influenced by land uses around them. This should not, however, make the conservation of biological diversity any less of a priority. In the language of landscape ecology, refuges can be viewed as protected "nodes" of diversity. An effective strategy to conserve biological diversity will require that these nodes be buffered, interconnected and permitted to interact with surrounding natural habitats (Noss and Harris 1986), although there is a justifiable concern that poorly planned corridors may have unintended negative effects by allowing passage of exotic species and disease (Soule and Simberloff 1986).

Wherever possible, the Service should attempt to develop integrated conservation strategies with other federal and state land agencies, conservation organizations, and private landowners. The various federal programs designed to restore habitat and foster sound management on private lands, such as the Service's private lands habitat restoration program, the wetlands reserve program and conservation reserve program can serve as important buffers for refuges and other reserves, and provide networks of protected habitat.

The Service recently adopted a controversial but commendable policy that requires that restoration projects under its "Partners for Wildlife" program "re-establish the original natural community or a successional sequence of natural communities that will lead to the re-establishment of the original community on at least 70 percent of the project site" (U.S. Fish and Wildlife Service 1993b). The Service should be careful, however, to ensure that actions on private lands don't become an end unto themselves rather than a means to achieving predetermined regional goals. In addition, resources should not be diverted from important refuge management programs to implement actions on private lands.

Case Studies

Below we discuss two examples where we believe that the Service is using the various conservation tools at its disposal in an effective manner to conserve biological diversity and two examples in which the Service apparently has failed to adopt a regional perspective.

Grasslands Ecological Area

The Bureau of Reclamation's massive California Central Valley Project and other agricultural developments have destroyed much of the native habitat in the Central

Valley. The Valley has lost 96 percent of its historic wetlands—including 66 percent of the vernal pools, more than 99 percent of its perennial grasslands and nearly 90 percent of its riparian woodlands; of the riparian vegetation that remains, less than 1 percent is in natural high-quality condition (California Nature Conservancy 1987, Reiner and Griggs 1989). Of California's large list of threatened and endangered species, 55 percent live in the Central Valley. The San Joaquin Valley—one of several major watersheds in the Central Valley—supports the largest number of federally endangered vertebrate species in the country (California Nature Conservancy 1987).

But what is left of the former Central Valley still is extremely important for fish and wildlife. The Valley long has been recognized as a critical wintering area for waterfowl and other migratory birds, and approximately 60 percent of the waterfowl in the Pacific Flyway winter there (Bellrose 1976). The area has been recognized as one of 15 critical areas in the Western Hemisphere Shorebird Reserve Network.

The Grasslands Ecological Area (GEA), a 30-mile long, 25-mile wide area in the heart of the San Joaquin Valley, contains four of the San Luis National Wildlife Refuge Complex's five refuges. The GEA includes approximately 60,000 acres in public ownership, including approximately 33,000 acres of refuge lands, and 100,000 acres in private ownership. The GEA contains over one third of the Central Valley's remaining wetlands, 20 natural communities, 235 bird, 56 mammal, 7 amphibian, 20 reptile and 270 plant species. More than 60 federal and/or state listed and candidate species are found in the GEA, including the California tiger salamander (*Ambystoma californiense*), vernal pool fairy shrimp (*Branchinecta lynchi*) and Aleutian Canada goose (*Branta Canadesis leucopeseia*).

The Service has pursued a very ambitious land acquisition program to protect additional habitat in the GEA. Since 1990, 17,000 acres have been added to the refuges of the Grasslands Ecological Area. Restoration of wetlands, riparian vegetation and native grasslands on these areas is a national priority. A long-term goal is to restore natural floodflow regime on federal lands along San Joaquin River.

In addition to fee title acquisition, the Service has an aggressive program to purchase perpetual easements on private lands in the Grasslands Ecological Area. To date, the Service has acquired easements on 44,000 acres out of a target of 75,000 acres. Wetlands, native grasslands and riparian communities are being restored on some of these private lands through the Service's "Partners for Wildlife" program and by private conservation organizations.

The Service has entered into strong partnerships with the state of California, National Audubon Society, The Nature Conservancy, Ducks Unlimited, California Waterfowl Association and private landowners under the framework of the North American Waterfowl Management Plan's Central Valley Habitat Joint Venture. But far more than just benefitting waterfowl, the objective of these groups is to conserve the full diversity of fish, wildlife and plants in the San Joaquin Valley.

Lower Rio Grande Valley

Irrigation and flood control projects in the middle decades of this century brought major agricultural and urban development to the Lower Rio Grande Valley (Valley) in southern Texas. Today, 95 percent of the Valley's original native brushland and 90 percent of its riparian habitat has been cleared (Jahrsdoerfer 1988). The area's dramatically expanding human population further threatens its natural diversity.

Despite major habitat conversion, the Valley still contains an extraordinary diversity of flora and fauna distributed among 11 biotic communities that include sabal palm forest, desert shrub, mesquite thorn forest and bottomland hardwoods. These 11 communities harbor a total of 1,200 species of plants, 700 vertebrates and 330 types of butterflies, and provide stop-over habitat for migrating birds of the Central and Mississippi flyways (Wildlife Corridor Task Force 1994). The Valley is home to 48 state and/or federally listed threatened and endangered species, including the entire U.S. population—estimated at 80–120 individuals—of the endangered ocelot (*Felis pardalis*) (M.E. Tewes personal communication: 1994). The area also is an important wintering habitat for redheads (*Aythya americana*) (Belrose 1976).

The Lower Rio Grande Valley National Wildlife Refuge is a new kind of refuge. The goal of the refuge is to link together remnants of once vast brushland, wetlands, coastal prairie and palm forest along a 200-mile long "wildlife corridor" (U.S. Fish and Wildlife Service 1993c). The refuge will link together two existing refuges—Laguna Atascosa and Santa Ana, state parks and wildlife management areas, and private sanctuaries, including National Audubon Society's Sabal Palm Grove Sanctuary. At the end of the 1994 Fiscal Year, nearly 70,000 acres of the 132,000-acre proposed refuge had been acquired.

The Service and its partners in the Wildlife Corridor are restoring native vegetation on cleared areas with a goal of restoring a functioning remnant of the historic regional diversity.

In addition to land acquisition and restoration of refuge lands, the Service has acquired conservation easements and leases to obtain management rights throughout the region. The Service and its partners also have initiated an ambitious program to plant native species on private lands that have been cleared for agriculture.

Recognizing that efforts to preserve, restore and connect remaining native habitats of the Rio Grande Valley will ultimately be unsuccessful if they are not further coordinated with efforts in Mexico, Service staff have forged agreements with Mexican environmentalists, biologists and concerned citizens to restore and protect habitat in Mexico (Best 1994).

Monte Vista/Alamosa National Wildlife Refuge Complex

The Monte Vista and Alamosa National Wildlife Refuges include over 25,000 acres (2 percent) of southern Colorado's San Luis Valley—a broad intermountain valley which stretches 80 miles long from north to south and 50 miles across at its greatest width (Bureau of Land Management 1991). Natural communities in the Valley include desert scrub, consisting of saltbrush (*Atriplex* spp.) and greasewood (*Sacrobatus vermiculata*), Great Basin grasslands which include blue grama (*Bouteloua gracilis*), western wheatgrass (*Agropyron smithii*) and other species, and wetlands consisting of wet meadow, marsh and riparian areas (Povilitis 1993).

The San Luis Valley is considered the most productive waterfowl area in the state (U.S. Fish and Wildlife Service 1993d) and the Monte Vista and Alamosa refuges were established under the authority of the Migratory Bird Conservation Act to perpetuate these values. The refuges also provide important oases of protected habitat for a variety of other species within a region generally dominated by grazed grasslands and wetlands and irrigated pasture. For example, 474,000 acres (91 percent) of the

Bureau of Land Management's 520,000-acre San Luis Resource Area currently is grazed (Bureau of Land Management 1991).

In 1990, to the consternation of many conservationists, the Monte Vista and Alamosa refuges instituted a high-intensity, short-duration grazing program of refuge grasslands, wetlands and riparian areas. The program was invented by Alan Savory and has been called "Holistic Resources Management" (Savory 1993). The grazing program, which includes cows, sheep and goats, has been strongly criticized by gamebird biologists from the State of Colorado's Division of Wildlife and the Colorado Cooperative Fish and Wildlife Research Unit—a unit of the new National Biological Survey. Long-term research has demonstrated a detrimental effect of even "light" grazing on waterfowl nesting density and success at the refuge. Non-gamebird biologists, grassland ecologists, soil scientists and plant biologists with Colorado University and other institutions also have strongly criticized the grazing program.

Ironically, the refuge has argued that the grazing program is intended to benefit biological diversity. Specifically, the refuge has argued that by creating "heavily grazed areas with little residual cover" they will promote diversity of species (U.S. Fish and Wildlife Service 1993d).

Like the Forest Service in our earlier example, the Service seems to have equated maximizing the *number* of species at the refuge with conserving biological diversity. The Service has failed to recognize that species that might benefit from grazing are most likely thriving on abundant grazed areas available in the immediate vicinity of the refuge, while those that require ungrazed conditions, including many species of neotropical migratory birds (Bock et al. 1993), may be rare and declining in the region and thus may be dependent on ungrazed refuge habitats. In short, the refuge has not adopted a regional perspective.

Bald Knob National Wildlife Refuge

In 1992, the Service began acquiring land for the Bald Knob National Wildlife Refuge, a proposed 14,000-acre unit that borders the Little Red River in eastern Arkansas. The refuge is being established for migratory birds, and is a priority of the Lower Mississippi Valley Joint Venture of the North American Waterfowl Management Plan (U.S. Fish and Wildlife Service 1991).

The 14,000-acre acquisition area was most likely once completely forested with water oak (*Quercus nigra*), Nuttal oak (Q. *nuttallii*), sweetgum (*Liquidambar styraciflua*), hackberry (*Celtis laevigata*), bald cypress (*Taxodium distichum*) and other species. Today, the area contains approximately 11,600 acres of agricultural lands, 1,100 acres of fallow fields and only 1,300 acres of woodlands (U.S. Fish and Wildlife Service 1991).

The Bald Knob refuge is located in the Mississippi River Alluvial Plain, an expansive area that extends over 700 miles from Illinois to Louisiana, passing through a total of seven states in all. This system once contained 21 million acres of forested wetlands, the largest such tract in the United States (Creasman et al. 1992). But the Corps of Engineers' flood control and drainage projects have dramatically altered the system. More than three fourths of the historic forested wetlands have been converted to agriculture or lost to development and in Arkansas, 85 percent of the historic forested wetlands are gone (The Nature Conservancy 1991).

With this regional context in mind, it is not surprising that an important objective of the Bald Knob refuge is to restore bottomland hardwood forest. What is surprising is that the Service has proposed to restore the native forested habitat to obtain only 5,000 total acres of this community type (36 percent), while keeping 5,000 acres in agriculture (36 percent) and managing 4,000 acres (28 percent) in moist soil units. This is in sharp contrast to the Service's policy regarding its activities on private lands under the Partners for Wildlife program. There is no doubt that the latter, manipulated habitats can provide significant benefits to wildlife, particularly waterfowl. But such restoration plans are inconsistent with management recommendations for other wildlife, such as some species of neotropical migratory birds that require large contiguous tracts of bottomland forest (Pashley and Barrow 1993). It is at least questionable whether the Service's plan for such high proportions of the refuge to be maintained in agriculture, while only a third of the refuge is to be restored to native habitat, is consistent with a strategy to conserve regional biological diversity.

Conclusion

The National Wildlife Refuge System can, should and has played an integral role in the nation's efforts to conserve biological diversity. To increase its contribution, the Refuge System should adopt a regional perspective when making management decisions, establishing land acquisition priorities and fostering partnerships with other agencies, organizations and private landowners. The goal is not much more complicated than Aldo Leopold's simple premise that "to save every cog and wheel is the first tenant of intelligent tinkering."

Although the Service has shown that in areas where it focuses its various conservation tools toward that end it can accomplish significant gains for biological diversity, it is apparent that such a perspective has not been universally adopted by the Service.

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National Wildlife Refuges: Contributing to the Conservation of America's Biodiversity

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Introduction

Biodiversity: buzzword or opportunity? If we have given any attention to what has been happening in our profession in recent years and what has been said at this Conference, we know that the term is both; it is a buzzword currently in vogue but also a huge opportunity to start moving away from narrow, single-species management to a much more inclusive approach based on ecosystem- and landscape-level thinking. This paper will explore what biodiversity could mean as it relates to the management of the National Wildlife Refuge System (System) and consider some of the realities of implementing such a philosophy.

To understand better how biodiversity or any other significant operational focus can become a part of refuge management, we must consider what the system is and how it operates.

The National Wildlife Refuge System

By executive order in 1903, President Theodore Roosevelt established the first refuge, what is now the Pelican Island National Wildlife Refuge (NWR) in Florida. At present, there are over 490 designated refuge units, plus thousands of other land parcels with System interests applied to them. While segments of the Refuge System are numerous, outside of Alaska they generally represent only bits and pieces of habitats, not ecosystems. Individual refuge units range in size from the 19+ million-acre Arctic NWR and Yukon Delta NWR here in Alaska to the tiny 0.6-acre Mille Lacs NWR in Minnesota. The System total is more than 91.5 million acres, with the non-Alaskan refuges averaging approximately 22,000 acres. Habitats represented in the System range from those found in Puerto Rico and the Virgin Islands to those of the arctic, the desert southwest and the atolls and islands of the south pacific. The Refuge System is administered by the U.S. Fish and Wildlife Service (Service), a bureau of the U.S. Department of the Interior.

Establishment Authorities

Although it is referred to as the National Wildlife Refuge System, it actually exists in name only. There is no single legislative mandate for the acquisition of land or the management of the System. Refuges are established not under a single rule but under a myriad of authorities and mandates, some of the most important of which are: Endangered Species Act (16 U.S.C. 1531–1543), Fish and Wildlife Act (16 U.S.C. 742a–742j), Land and Water Conservation Fund Act (16 U.S.C. 4601–4–460–11) and the Migratory Bird Conservation Act (16 U.S.C. 715–715d, 715e, 715f–715r). It also is common to have purposes legislatively directed at the time a refuge is authorized. Thus, the primary goals and the resultant allotment of operating resources can vary greatly from area to area depending on the administrative emphasis of the moment. Some refuges have been acquired for specific reasons when endangered species or species of special significance, such as bison or brown bear, are involved. But over the years, the acquisitions have been largely opportunistic, not the result of a specific national or regional plan.

There is at present in Congress proposed legislation "to amend the National Wildlife Refuge System Administration Act of 1966 to improve the management of the National Wildlife Refuge System and for other purposes" (S.823). This legislation, introduced by Senator Bob Graham, would define System purposes, provide overall policy and management guidance, specify planning schedules, and accomplish other actions that would remedy many of the fractions problems that now exist.

Past Refuge Field Management

Many refuges and some forms of refuge management predate much modern wildlife/wildlands knowledge. It is important to acknowledge that many past actions on refuges that seem strange today were based on agendas that of necessity were short on science but long on what was available—political influence and administrative direction. These management approaches were much the same as those being employed concurrently on most other public lands. There was strong emphasis on a few "important" wildlife species with a bias toward those that were huntable. Little consideration was given to the "lesser" species. Extractive activities such as grazing, farming and lumbering were encouraged or permitted often in the name of better community relations or as a means of accomplishing some needed management action in the face of inadequate staff and funding. Many of these activities became local traditions that, because of their political support, could be changed only with great difficulty. Some persist to this day.

It was not until the 1930s that technically trained people were widely available for refuge manager positions. The intuitive accomplishments of the earlier managers were remarkable when the limitations under which they worked are considered.

Present Refuge Field Management

Present refuge professionals at all levels are highly qualified and remarkably dedicated, but most of their formal training has been concentrated on management techniques for single species or groups of similar species. Specific training in biodiversity or landscape planning at the senior level still is unusual but most are at least aware of current trends and some have installed truly innovative management schemes designed to broaden past approaches.

Change is underway. In recent years, refuge staffs have increasingly benefitted from younger and ever better trained people. For these folks, the concept of biodiversity and landscape-level planning is not foreign but fundamental. A fertile seedbed is present.

Possibilities and Problems

In anticipation of embracing the concept of biodiversity and ecosystem management, the Service has been developing administrative guidelines, but as this is being written they are not available for public review.

As a working thesis for this paper, biodiversity is defined as a concept that considers, for management purposes, all of the life forms that naturally should exist in a particular ecotype or ecosystem and the essential ecological functions required to sustain those populations and systems indefinitely. The assumption is that biodiversity will become a basic management tenet for the entire System in addition to the approximate 5 percent of refuges where it already is specified in the purposes.

Because it already exists, the System, with one or more designated units in each of the 50 States, has the potential to be a major contributor to the development of applied biodiversity technology and knowledge. A logical approach would be for the Service to select a series of refuges that contain diverse habitats and apply the necessary resources to develop the full range of information needed for biodiversity management.

By administrative direction, only a fraction of the plant and animal species that are or could be present on individual refuges have received significant management attention. A biodiversity approach may uncover many potentially fruitful opportunities to enhance the maintenance of diversity in refuge system habitats. However, it will become evident immediately that much necessary information on which to base such broad management is not available and will require new funding and additional manpower resources to remedy the situation.

Under the present conditions of significant reductions of funding and staff it will be extremely difficult even to start obtaining the data needed to make informed decisions. The emerging National Biological Survey (NBS) may be of some assistance in obtaining needed information and such action is strongly encouraged, but it will still require additional resources and time. At present, there are no indications as to where biodiversity inventories, status surveys and monitoring will fall in the priority list of Service and NBS cooperative activities.

In any organization, major changes in management direction, no matter how worthy, are viewed first with suspicion and then alarm. To be effectively implemented, biodiversity-oriented management and landscape-level planning must be given high priority at the top of the administrative pyramid and then vigorously and continuously supported through policy directives and appropriate resource allocations to the implementing field. Any other approach will ensure that change will occur with glacial swiftness.

It is inevitable that holistic management on some refuges will come at the expense of presently favored uses. The Service and refuge managers can expect intense political pressure any time the status quo is threatened. The quality and persistence of the Service response will be a gauge of true commitment.

The present administrative structure of the Service places the System at a great disadvantage when it comes to obtaining sustaining levels of operating resources, resisting outside pressures, and making and maintaining significant, long-term management changes. As mentioned previously, there is no single legislative mandate for what is referred to as the National Wildlife Refuge System. In actuality, there is an amalgam of land parcels that have been acquired and are being managed under different authorities and mandates. To complicate matters additionally, refuges are administered nationally as merely one of four functions under one of five assistant directors. At the regional level there is similar subordination. There is a constant and often futile struggle within the Service to obtain minimum base level refuge staff and funding. These circumstances make the implementation of new directions difficult to coordinate. This is in sharp contrast to other land-management agencies, such as the National Park Service and the USDA Forest Service, where management of the land base is the primary focus and the agencies have greater control of their budget, policy and compliance functions.

The lack of a single legislative mandate for the Refuge System, coupled with the obvious need for changes, has encouraged some dramatic administrative tinkering in the past. This most often has been correlated with the four- or eight-year national political cycles. The movement toward biodiversity and landscape thinking, while an absolutely supportable approach, represents another dramatic change from the recent past. Service personnel can be forgiven for seeking new and additional commitments of resources before jeopardizing past management programs that are known to benefit many wildlife species and publics.

Recommendations

The National Wildlife Refuge Association, a private organization founded in 1975 and dedicated to the protection and perpetuation of the National Wildlife Refuge System, recommends that:

1. The Service move assertively to implement a policy of biodiversity-oriented management on national wildlife refuges;
2. The Service move toward landscape-level planning, especially where refuge habitats are integrated with other public and private lands to meet ecosystem and ecoregion needs;
3. Selected units of the System be utilized to develop ecosystem data collection, monitoring and management techniques;
4. The Service utilize the inventory data of the National Biological Survey to identify wildlife communities and habitat types not included in the System and include qualifying representative examples in its acquisition priorities;
5. All who are vitally interested in a more efficient and effective National Wildlife Refuge System should actively support the passage of the Graham Bill, S.823. This organic legislation for the System would correct many of the existing problems; and
6. The Service establish a Deputy Director for Refuges. This would make it possible for the System to develop and retain the policy, direction and resources necessary to implement the best management approaches to ecosystem management.

Intensive Wetland Management: A Key to Biodiversity

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Wetlands are among the most productive ecosystems in the world. Primary productivity of coastal marshes and estuaries normally are greater than terrestrial or deepwater systems. Wetlands also support a rich fauna at higher trophic levels. For example, more than 50 percent of the 800 protected migratory birds and 80 percent of the breeding bird population in the United States rely on wetlands (Wharton et al. 1982). Other taxonomic groups, including plants, fishes, amphibians, reptiles and some mammals, also are dependent on wetland habitats at least some time during the annual cycle. Thus, it is apparent that wetlands are a critical component contributing to biodiversity at many different scales, including local, regional and continental.

Although wetlands support myriad wildlife species, many vertebrate populations dependent on wetlands have exhibited declines over the past three decades. For example, of the 261 vertebrates listed as federally threatened or endangered in 1990, more than 45 percent are dependent on wetlands to provide resources necessary to complete at least one life-cycle event (e.g., breeding, hibernation). Declining waterfowl populations probably are the best documented and publicized example. Estimates of breeding ducks during the past 30 years clearly indicate the downward trend of many populations (U.S. Fish and Wildlife Service 1993). However, the distribution and population sizes of other wetland-adapted species also have been severely reduced, although the decline is less well documented.

Wetland Status

Undoubtedly, declines in species populations partially are attributable to the general loss and degradation of wetlands (Table 1). Although the extent of original wetland area is not known, estimates of losses in the conterminous states since European settlement exceed 50 percent (Dahl 1990). However, such estimates only reflect vague changes in wetland area because they are summarized by region and broad categories (e.g., palustrine emergent, forested), whereas the impact of losses on the disruption of riverine corridors and wetland complexes, fragmentation of vast pristine wetland systems, and disproportionate losses of certain wetland types have received only minimal consideration. Remnant wetlands frequently are isolated from other wetlands and often exhibit modified hydrology, high rates of sedimentation and increased levels of contaminants, including organics, heavy metals and trace elements. Faunal species vary in mobility and require different types of wetlands to complete annual cycle events (Laubhan and Fredrickson 1993). Consequently, the interspersion, juxtaposition and quality of habitats, in addition to overall wetland loss, are extremely important in determining the consequences resulting from the loss of a specific wetland basin. Shallow flooded wetlands with the shortest duration of flooding are most prone to destruction, whereas wetlands that are deeply flooded for long periods are difficult or costly to drain. For example, forested wetlands with different flooding regimes have been destroyed at different rates. Forests with the shallowest flooding for the shortest duration have disappeared more extensively than deeply flooded sites with an extended duration of flooding. Thus, the absolute impact of wetland loss on wildlife populations tends to be obscured because some key habitats may no longer have a desirable distribution or the total remnant area is insufficient to provide the required resources for some species. These more subtle perturbations often are as important as total wetland loss in causing disruption of wetland functions and decreasing biodiversity and/or population size (Table 1).

Declining Habitat Conditions

The quality, abundance and availability of resources, as well as the spatial arrangement of different wetland types that provide such components, are critical factors that determine abundance and biodiversity of wetland wildlife. Changes in wetland wildlife communities from pristine conditions often are related to changes in functional habitat size. Typically, remnant wetlands now have altered timing, depth and duration of flooding. These hydrologic changes influence plant community composition and structure, thereby affecting the abundance and availability of foods and cover (Figure 1). As such changes occur, habitat diversity tends to decrease and essential life requisites often become more widely dispersed. If habitat fragmentation increases to the extent that resources do not occur within the geographic area that can be exploited by a species, the survival and/or reproductive potential of that species may be compromised. Because tolerance to habitat disruption varies among species,

Table 1. Suggested effects of wetland loss, modification and perturbation on biodiversity.[a]

Effect	Change in habitat	Reduced population size	Reduced species richness	Mortality/ extirpation/ extinction
Wetland loss				
	Reduced size			
	Fragmentation	I,F,H,B,M	I,F,H,B,M	B,M
	Disruption of wetland complex	F,H,B,M	I,F,H,B,M	I,F,H,B,M
	Less diverse wetland types	I,F,H,B,M	P,I,F,H,B,M	P,I,F,H,B,M
	Habitat no longer present in historic proportions	I,F,H,B,M	P,I,F,H,B,M	M
Modified hydrology	Structure and distribution	I,H,B,M	I,F,H,B,M	
Perturbations				
Herbicides	More monotypic	I,F,H,B,M	P,I,F,H,B,M	P
Pesticides	—		I,F	I,F,H,B,M
Fertilizers	More monotypic, change in structure		I,F,H,B	
Sedimentation	More monotypic, change in structure	I,F,H,B,M	I,F,H,B,M	

[a]P = Plants, I = Invertebrates, F = Fish, H = Herps, B = Birds, M = Mammals.

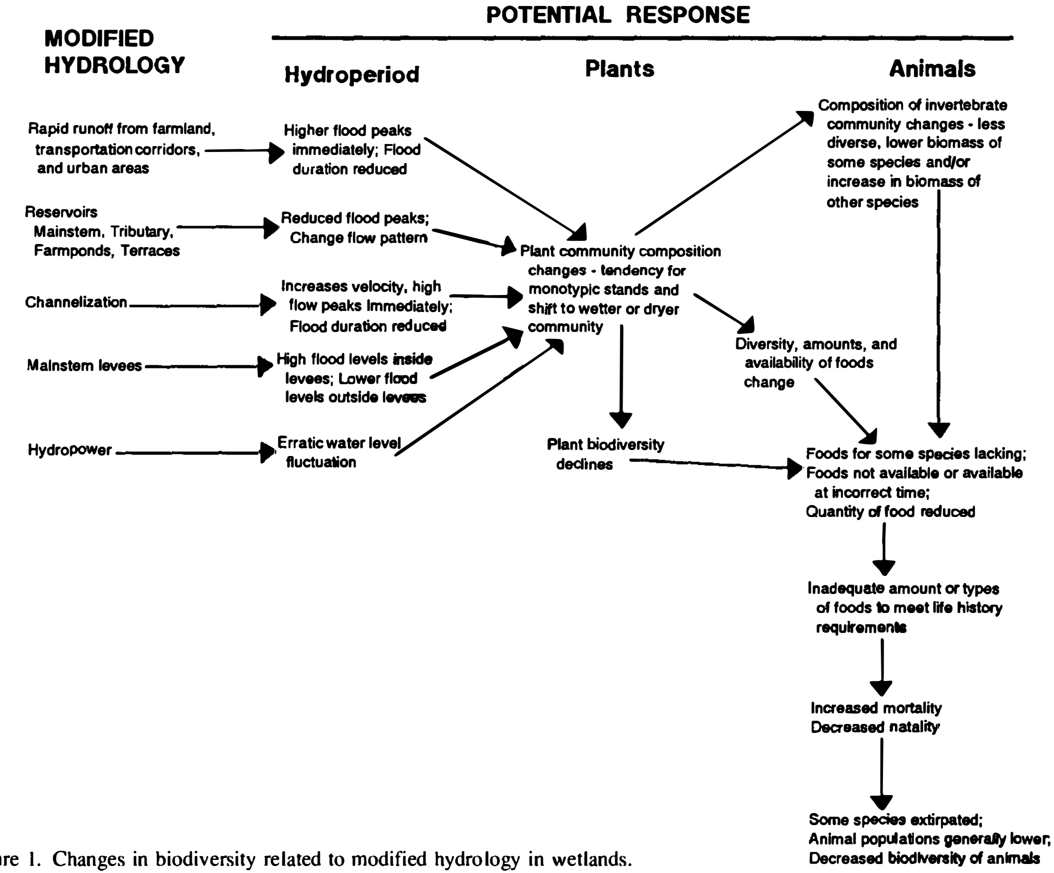

Figure 1. Changes in biodiversity related to modified hydrology in wetlands.

the loss and/or degradation of wetlands differentially influence the survival and reproductive potential of individual organisms.

Although home range size is not known for a majority of species, evidence suggests that the home range of organisms belonging to the same taxonomic group (e.g., birds, mammals, reptiles) is positively related to body size (Peters 1983). For example, mallards (*Anas platyrhynchos*) can exploit a larger geographic area than clapper rails (*Rallus longirostris*). Thus, biodiversity and abundance of wetland wildlife likely is negatively related to the extent of wetland loss and severity of hydrologic modification relative to the species composing the community. Further, the modification of a single wetland basin likely will influence the reproductive and survival potentials of species with small home ranges or limited mobility to a greater extent than species capable of exploiting larger geographic areas. In fact, many wetland species currently listed as threatened or endangered have limited mobility and small home ranges, indicating that local or small-scale perturbations are as important in reducing the diversity of wetland fauna as are large-scale changes.

The Historical Focus on Wetland Management

In the past, wetland management has focused on waterfowl, including the provision of hunting opportunity (Heitmeyer et al. 1989). Thus, management under certain circumstances has been directed toward providing habitats at a geographic scale that accommodates primarily large-bodied waterbirds that exploit large areas. In addition, because hunters exhibit preferences in harvesting waterfowl, management often has been further refined to provide resources that attract only select species. Such management is not necessarily bad, but the consistent application of the same techniques at similar times for many years usually results in less dynamic conditions that tend to reduce wetland productivity. Further, the spatial arrangement and quality of habitats, although capable of providing resources to large birds, may be inadequate to support populations of smaller or less mobile species with different food or cover requirements (Soule 1991). There is much confusion over this historic approach, and wetland management primarily for waterfowl now is considered inappropriate by many individuals and agencies. However, the goals of management, rather than the applicability or value of management, should be scrutinized.

Wetlands can be intensively managed to improve conditions for many species. For example, properly timed drawdowns can increase the availability of invertebrates for shorebirds and, at the same time, stimulate germination and growth of food-producing plants consumed by waterfowl. However, the ability to manage for multiple species varies depending on site conditions, including configuration of wetland basins and degree of hydrologic control. A majority of national wildlife refuges and state wildlife areas were established when little was known about factors that control wetland functions and values (e.g., hydrology). Although research and experimentation have greatly improved our understanding of wetland and waterbird ecology, the application of this knowledge is hindered because physical structures (e.g., levees, water-control structures) were located, designed or installed incorrectly when many wetlands originally were developed or restored (Fredrickson and Reid 1990). This problem is exacerbated because funds often are not available for renovation of existing wetland areas or proper development of newly acquired land; in addition, some managers may be poorly informed on wetland dynamics and wetland wildlife requirements. Thus, wetland managers often are constrained in their ability to manage areas for multiple species.

Developing Strategies for Biodiversity

Reversing the decline of wetland wildlife populations and improving conditions for a greater diversity of wildlife will require implementing programs based on ecological principles. Foremost is the recognition that the resources necessary to successfully complete annual cycle events, as well as the strategies and geographic area used to acquire such resources, vary by species (Fredrickson and Reid 1986, 1988, Fredrickson and Laubhan 1994). Individual wetland basins and wetland complexes must be evaluated based on the type, amount and quality of surrounding habitats to determine the potential type and density of species that can be supported. In areas that have suffered extensive loss or degradation of wetlands, such potential will be lower compared to relatively pristine areas. Often, however, disrupted systems can be improved through management. The rationale behind intensive management is that more resources, and thus more species (or a greater abundance of target species) can be accommodated more consistently on a smaller area. Although management successfully can improve both wetland productivity and biodiversity, the correct combination of techniques must be applied at the appropriate time. Otherwise, human intervention designed to improve conditions for wetland wildlife actually may accelerate degradation.

Active management obviously will favor some species or groups of species. However, the decision to manage is not necessarily wrong. A single wetland basin cannot provide all resources to all species, regardless of whether the basin is protected or managed (Laubhan and Fredrickson 1993). Consequently, an insufficient number of wetlands or inadequate composition of wetland types in close juxtaposition, not intensive management, often is the factor that precludes improving species richness.

Achieving greater species richness on such areas likely will require increasing the number or type of wetland basins. Several federal (e.g., Wetland Reserve Program) and state programs have funds dedicated to increasing the amount of wetland area. However, the types and spatial arrangement of wetlands restored or created must be considered carefully (Figure 2). Existing wetland areas should be evaluated to identify the resources compromising biodiversity. Priority should be given to wetland types that provide these limiting resources in close juxaposition to existing wetlands. Such an approach would promote the formation of wetland complexes that have greater benefit for multiple species and also increase management flexibility. Accomplishing this task will require unprecedented cooperation among agencies, as well as individuals at different administrative and managerial levels within the same agency.

Interagency Cooperation

Because of continued wetland loss and degradation, concern for wetland protection has increased and resulted in the passage of legislation that governs wetland delineation, acquisition, protection and management (Table 2). A complicating factor, however, is that different organizations are responsible for the interpretation and

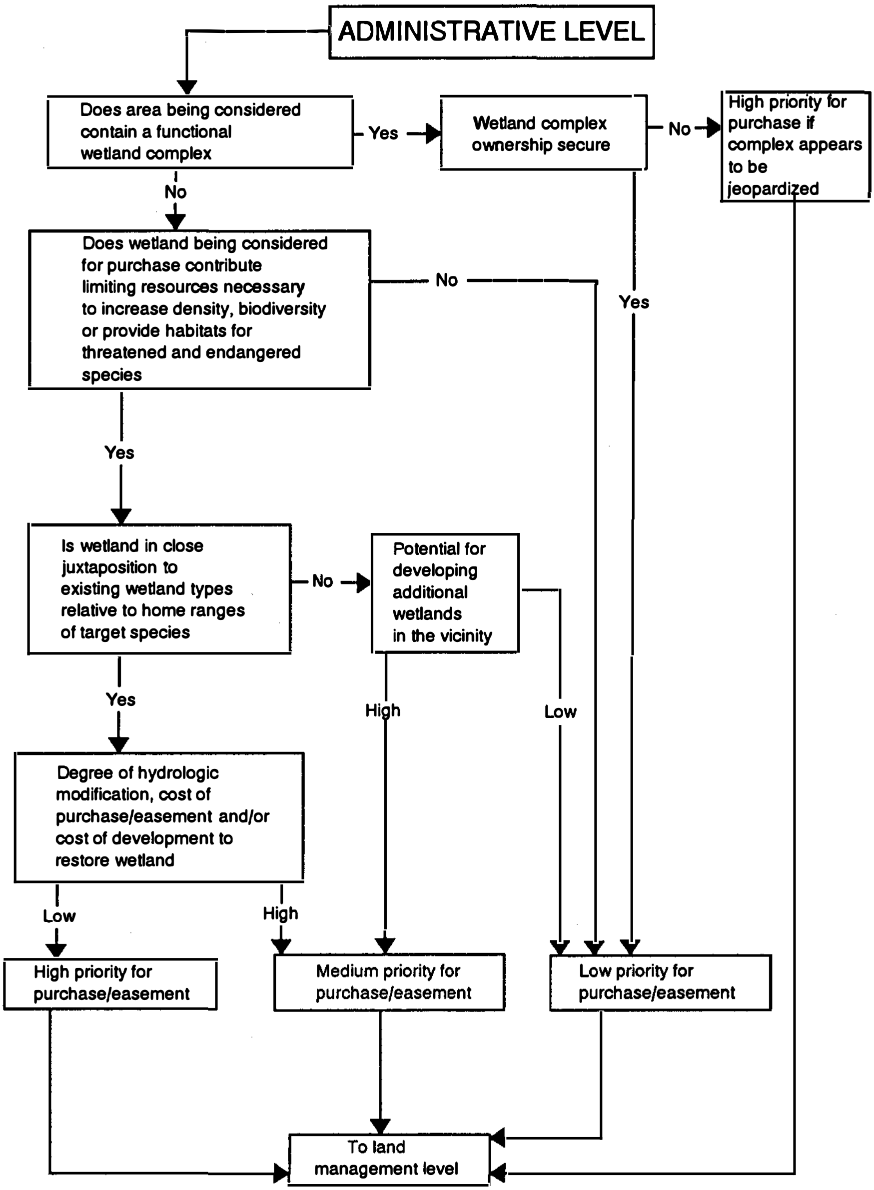

Figure 2. Selected factors for designating wetlands for purchase or easement—Administrative level.

enforcement of such regulations. As a result, the diverse goals of private enterprise and various state and federal agencies often are not differentiated. Although the intent of Section 404 relates to wetland protection, interpretation of the Act does not assure such protection (Bean 1988). In contrast, Section 404 of the Clean Water Act originally was intended to curtail wetland disruptions caused by agricultural or development interests. For example, Section 404 recently has been used to prevent intensive soil and water manipulations on man-made wetlands administered by agencies mandated to provide resources for wetland wildlife. Such wetlands were designed and are dependent on intensive management to maintain productivity. Further, most remaining wetlands have been severely modified by past perturbations and the only opportunity to provide resources to a diversity of wildlife is dependent on intensive management. If legislation, such as Section 404, is strictly applied without consideration for the types of manipulations being conducted and their intended purpose, regulations designed to protect or increase the amount of wetland area ultimately may result in the reduction of habitats that benefit wildlife. To avoid further confusion, personnel from regulatory agencies must cooperate with state and federal natural resource management organizations to develop an enforcement strategy that accomplishes the intent and purposes of legal mandates.

Improving biodiversity on public lands also will require cooperation among various state and federal agencies charged with managing natural resources. Although the rate of wetland loss has been reduced considerably during the past 10 years, almost all wetland systems in the conterminous states have been adversely affected (Dahl and Johnson 1991). The National Wildlife Refuge System (administered by the U.S. Fish and Wildlife Service) is responsible for only 0.5 percent of the landbase of the continental United States (Salwasser 1991). Because wetlands in private ownership often are managed for specific purposes (e.g., duck hunting), the limited wetland area in government ownership often must be managed to assure that resources not available on privately owned wetlands are provided to a variety of wildlife species. As a result, government lands often are expected to provide a range of resources on small parcels. This problem can be diminished greatly if management personnel on federal and state areas located in close juxtaposition cooperate to develop coordinated plans for the

Table 2. Selected events in wetland and land-use legislation that impact the protection of habitats or have the potential to affect biodiversity.

1972	Clean Water Act authorizes the Environmental Protection Agency to create and enforce water-quality standards and guidelines for permitting draining and filling of wetlands (administered by the Army Corps).
1973	Endangered Species Act authorizes the U.S. Fish and Wildlife Service to list as threatened or endangered species, to designate critical habitat areas, and to develop recovery plans.
1977	Executive Order 11990 mandating that all federal agencies work to minimize impacts on wetlands.
1985	Food Security Act establishes the Wetlands Reserve Program administered by the U.S. Department of Agriculture's Soil Conservation Service to provide funds to farmers who keep wetlands out of production.
1986	Emergency Wetlands Resources Act enacted.
1988	The National Wetlands Policy Forum sets a goal of "no net loss" of wetlands and Presidential candidate George Bush endorses the goal.
1990	Water Resources Development Act passed.
1992	Central Valley Project Act sets aside 800,000 acre-feet of water for fish and wildlife protection and establishes a Restoration Fund with an initial $35 million.

acquisition, lease and management of wetlands that take into account the number, type and condition of wetland habitats at different geographic scales relative to the home range and mobility of species, rather than managing habitats based on political boundaries. Cooperation among agencies and the private sector will not only maximize the value of individual basins and enable more consistent provision of resources to a greater number of species, but also will allow for greater management flexibility.

Intraagency Cooperation

Many state and federal agencies originally acquired wetlands for the purpose of protecting and/or enhancing habitats used by migratory birds. However, such areas now are recognized as providing critical habitats for other species and the goals and objectives of wetland management have undergone dramatic changes to reflect this increased awareness. Oftentimes, the political entity (administrative and operational framework) relative to the status of wetland conditions precludes meeting such objectives. Resolution of this problem requires cooperation among personnel at both administrative and managerial levels within an agency.

Initially, existing wetlands should be evaluated to determine realistic goals regarding biodiversity relative to current conditions with a region (Figure 3). Information concerning the type, condition and juxtaposition of wetlands, as well as historical data and results from annual monitoring, must be assessed to determine the types and densities of wetland wildlife that might use the area. Following the establishment of goals and objectives, the area should be monitored to determine if goals are achieved. If goals cannot be met, the type and timing of management actions should be evaluated (figures 3 and 4). For example, basins originally thought to be unmodified may have been disrupted and will require management. On areas that already are managed, the correct type of physical structures or equipment may be lacking to manage the area correctly. In either case, the solution is to renovate the area and provide the capability to initiate appropriate management strategies. The primary concern should be developments that allow hydroperiods to be emulated correctly on various wetland types (Figure 4). In addition, mangers should received periodic training to improve their understanding of wetland and wildlife ecology. Otherwise, the expenditure of funds to improve management capability will result only in marginal improvements.

If management strategies are deemed appropriate, the area should be evaluated to determine the juxtaposition and interspersion of existing wetland types (Figure 3). If all necessary resources are provided within the home range of target species, the ability to increase species richness or density may be limited to other factors, such as human disturbance (e.g., roads, recreational uses) or contaminants. Site-specific research may be necessary to identify and resolve such problems in an effective and cost-efficient manner. In contrast, if all resources are not available, the inability to realize goals may be related to the lack of a specific wetland type or incorrect spacing of existing wetland types rather than inappropriate management. Additional wetlands should be restored or created to form a more comprehensive wetland complex that includes the appropriate type, number and configuration of wetlands necessary to accomplish stated goals. State and federal programs designed to increase wetland area currently exist, but the emphasis of such programs should be altered to recognize the importance of wetland complexes, rather than simply maximizing flooded area.

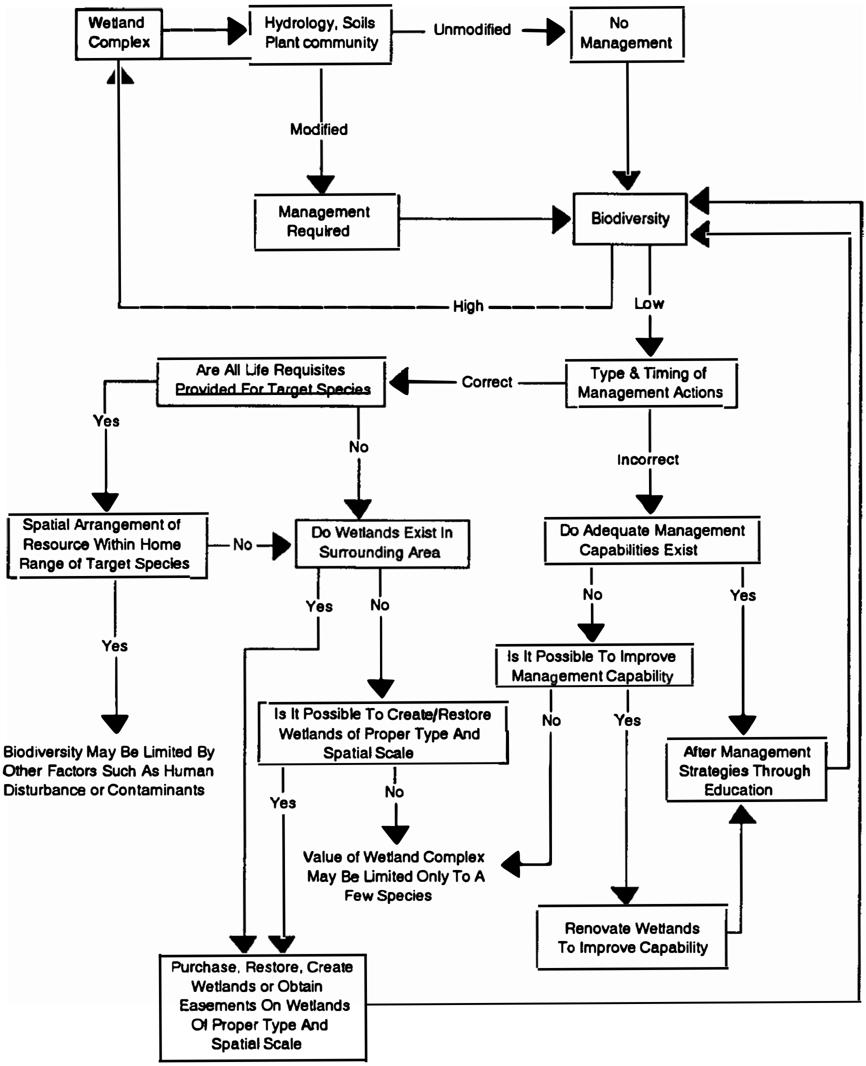

Figure 3. Potential strategy to identify possible factors limiting wetland biodiversity.

Summary

Many agencies and private environmental organizations currently consider intensive wetland management as favoring individual species or contributing to the degradation of remaining wetland habitats. Such concerns are valid and must be addressed. Wetlands with unmodified hydrology should be protected. Unfortunately, the hydrology of many wetland basins in the conterminous states has been disrupted and intensive management is the only viable option available to provide benefits to

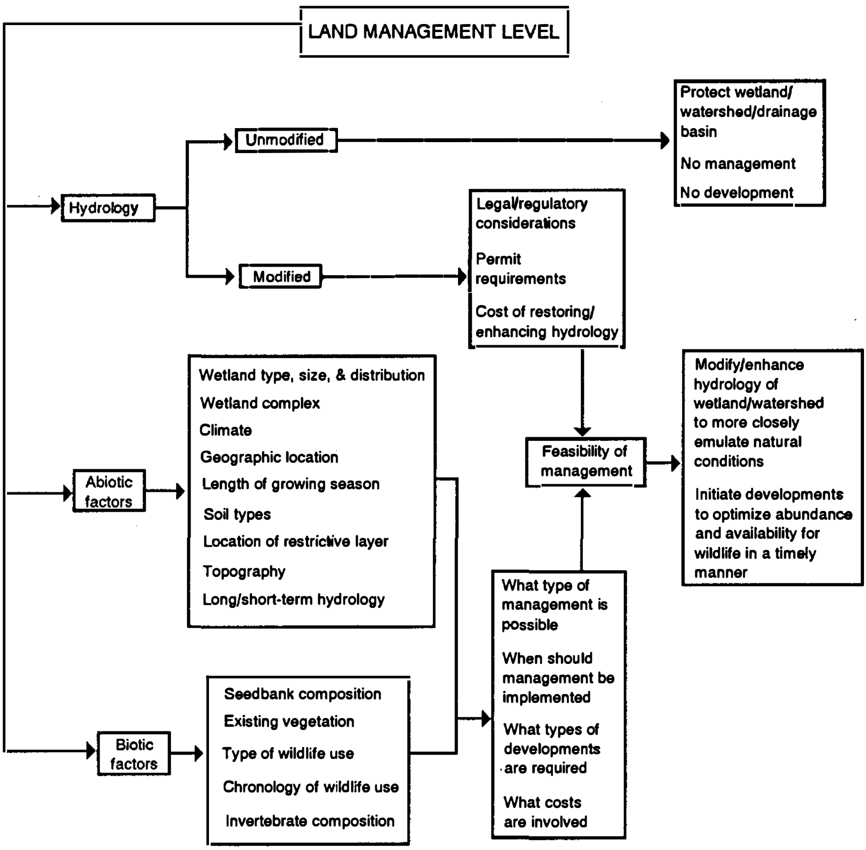

Figure 4. Selected factors in determining appropriate management actions—Management level.

wetland wildlife. Consequently, the challenge often is not to determine the need for management, but the most appropriate method of management. The historical approach of managing wetlands primarily for waterfowl obviously will not maximize the value of wetlands for all species. Rather, a more integrated approach that considers individual basins as forming wetland complexes that provide habitat for myriad species must be embraced. Successful application of this approach will require cooperation among and within agencies. Managers must be informed of emerging techniques that benefit multiple species. Administrators must acknowledge that additional funds may be required to purchase/restore additional wetlands, rehabilitate existing areas or acquire equipment necessary to implement new techniques.. Decisions regarding the expenditure of available funds for acquisition or easement must be based on increasing the value of existing wetland areas rather than maximizing the amount of wetland area per unit cost. Finally, cooperation among agencies and private individuals will be essential. Foremost is the ability to communicate effectively the underlying strategies and concepts of wetland ecology. If management is appropriate, agencies charged with management responsibilities must inform regulatory personnel of the need and rationale for using specific techniques. Similarly, regulatory programs must recognize that managers use techniques commonly associated with agriculture (e.g., disking, water manipulation) to emulate, not destroy, wetland functions and maintain productivity. Further, realistic goals should be based on ecological criteria rather than political boundaries. Wetlands continue to be a declining resource and the difficulty of providing habitat for all wetland- dependent species is becoming increasingly arduous. Management of adjacent wetlands or wetland areas as discrete entities no longer is possible and often reduces productivity of the complex as a whole.

Acknowledgments

We kindly acknowledge critiques by Diana Cross, David B. Hamilton and Dale Humburg. This is a contribution from Gaylord Memorial Laboratory (The School of Natural Resources, University of Missouri-Columbia and Missouri Department of Conservation, cooperating), Missouri Agricultural Experiment Station Journal, Project 183, Series No. 12, 088.

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The Application of Gap Analysis to Decision Making in the U.S. National Wildlife Refuge System

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Introduction

Conservation of biodiversity is an issue of growing concern to scientists, land managers and citizens. Globally, the world system of protected areas in an important means of conserving biodiversity. Within the United States, the National Wildlife Refuge System is an important element in the existing inventory of protected areas. Administered by the U.S. Fish and Wildlife Service (USFWS), the objective of wildlife refuges is primarily to conserve fish and wildlife (Curtin 1993, Sierra Club 1987). In this respect, they are unique among lands managed by federal agencies. In 1993, there were 494 wildlife refuges covering over 40 million hectares (USFWS 1994). Refuges range in size from one-half hectare to more than 7.5 million hectares, though more than half are less than 4,000 hectares in size. Approximately 85 percent of the land area of the refuge system is contained within 16 refuges in Alaska.

Hence, the U.S. National Wildlife Refuge System can make a significant contribution to the conservation of biological diversity within the U.S. However, wildlife refuges and the biodiversity they contain face threats from a variety of human actions (Amato 1991, Curtin 1993, Sierra Club 1987). A 1989 federal report estimated that activities harmful to wildlife are occurring on nearly 60 percent of National Wildlife Refuges (GAO 1989).

Increased attention to the biodiversity values of the National Wildlife Refuge System is necessary in the face of human activity (such as economic development) and ecological challenges (such as small refuge size and boundary influences). Successful strategies will require: (1) ecosystem-scale management that extends beyond refuge boundaries and involves other agencies and landowners; (2) better understanding of human actions that impact biodiversity; and (3) analytical techniques and practical tools for making land-management decisions that impact biodiversity. One such technique is *gap analysis.*

Gap analysis is an interdisciplinary technique for examining the protection status and vulnerability of biodiversity. A geographic information system (GIS) is used to map elements of biodiversity, such as vegetation types and species distributions. An overlay of protected areas is used to identify unprotected or under-protected elements of biodiversity. These represent "gaps" in a coherent conservation strategy (hence the term "gap analysis"). Next, a variety of socioeconomic indicators (including measures of population growth and economic development) are entered into the GIS. Based on a model of human impacts on biodiversity, these indicators are combined to determine the relative vulnerability of the identified gap locations to future biodiversity loss.

Gap analysis is intended as a coarse filter approach to biodiversity protection, complementing the traditional techniques of protection for individual rare species (Jenkins 1976). Gap analysis currently is underway in 32 states, and a national gap analysis program (GAP), administered by the National Biological Survey, has been initiated. Coordination with other programs such as those of the Environmental Protection Agency, state conservation data centers, the National Heritage Program, Multi-State Inventory and the National Wetlands Inventory is critical to the GAP program. The purpose of this paper is to provide a brief introduction to the technique of gap analysis, and suggest its potential application to the U.S. National Wildlife Refuge System.[1]

Biological Aspects of Gap Analysis

Gap analysis most often is initiated at the state level.[2] Three principal biological components are mapped: existing vegetation, vertebrate distributions and areas managed for biodiversity. These components provide the basic information necessary for determining gaps in biodiversity protection.

Vegetation can be considered a useful integrator of many factors in the physical environment. The vegetation map provides information on the distribution, extent and management status of existing vegetation.[3] Using visual interpretation and image processing techniques, land areas with similar spectral reflectance are delineated. The resulting polygons are identified and labeled using aerial photography, airborne video (Graham 1993), existing vegetation maps, digital elevation models, expert opinion

[1]For more detail on the gap analysis technique, see Davis et al. (1990), Edwards et al. (1993), McKendry and Machlis (1993), Machlis et al. (1994), Scott et al. (1993) and Butterfield et al. (in press).

[2]The national GAP program ultimately intends to make evaluations with biologically defined areas, such as ecoregions, to assist decision makers in assessing resource management needs.

[3]The map is based on Landsat Thematic Mapper (TM) satellite imagery with a pixel size of 30 X 30 meters on the ground. In many areas of the country, the smallest mapped vegetation polygon is 100 hectares, although some individual states are mapping to 40 hectares or smaller when appropriate to meet the needs of local cooperators in the national GAP program.

and field visits. The classification scheme is at the series level of detail (UNESCO level 5) and is consistent with work in progress by The Nature Conservancy, which generally is based on the work of UNESCO (1973) (Jennings 1993).

Vertebrate distribution maps incorporate species habitat preferences from a variety of sources, including regional field guides and published literature. The maps are produced by combining this information with available data on species sighting records, museum specimens, habitat associations and vegetation maps.[4] In some areas, wildlife habitat relationships already have been modeled (e.g., Thomas 1979). Information on elevation, temperature and soil type is used to refine species occurrence models when these data are appropriate. The availability of such data varies dramatically from state to state. When available, data on invertebrates also are being incorporated into the databases (Scott et al. 1993).

An inventory of areas managed primarily for biodiversity protection is developed from maps of land ownership and management status. Ownership categories include both private and public lands. Public lands are described by management agency and designation, including U.S. Forest Service wilderness areas, U.S. Fish and Wildlife Service refuges, Bureau of Land Management Areas of Critical Environmental Concern and U.S. National Park Service administered lands. Land is classified into broad management classes such as private land, protected public land and unprotected public land (see Scott et al. 1993, Wright et al. 1994).

Gaps in land managed for biodiversity are identified where under-protected species or vegetation communities occur. Operational criteria are established. For example, a particular vertebrate species might be considered under-protected if it does not have three areas of at least 10,000 hectares each of protected habitat (a gap). By combining information on gaps for many species, it is possible to identify those geographic areas that are critical to biodiversity conservation (gap locations). These can be spatially depicted on a map, showing their location in context with other ecological and socioeconomic criteria. Several other methods for determining gaps and gap locations in a biodiversity conservation strategy are possible, depending on the set of species or vegetation types of interest, the scale and the objectives of the analysis.

Socioeconomic Aspects of Gap Analysis

Because human actions may increase the vulnerability of gap locations to biodiversity loss, socioeconomic factors are critical to gap analysis. Hence, the gap analysis technique is being extended to include indicators of human activity relevant to biodiversity conservation. A conceptual model identifies the major paths by which human actions affect biodiversity (Figure 1; for a detailed description *see* Machlis and Forester 1994). Social, economic and political factors are considered the driving force behind changes in how people use resources. Changing resource use leads to impacts on ecosystems, some of which may result in biodiversity loss.

To examine human impacts on gap locations, socioeconomic zones of influence are delineated around each identified gap location. These "critical zones" are geo-

[4]Of particular concern to the developers of the vertebrate models are issues of map scale, resolution of sighting data and degree of knowledge of particular species. In some instances, museum records and species sighting data are only spatially documented at the county level (Scott et al. 1993, Csuti and Scott 1991).

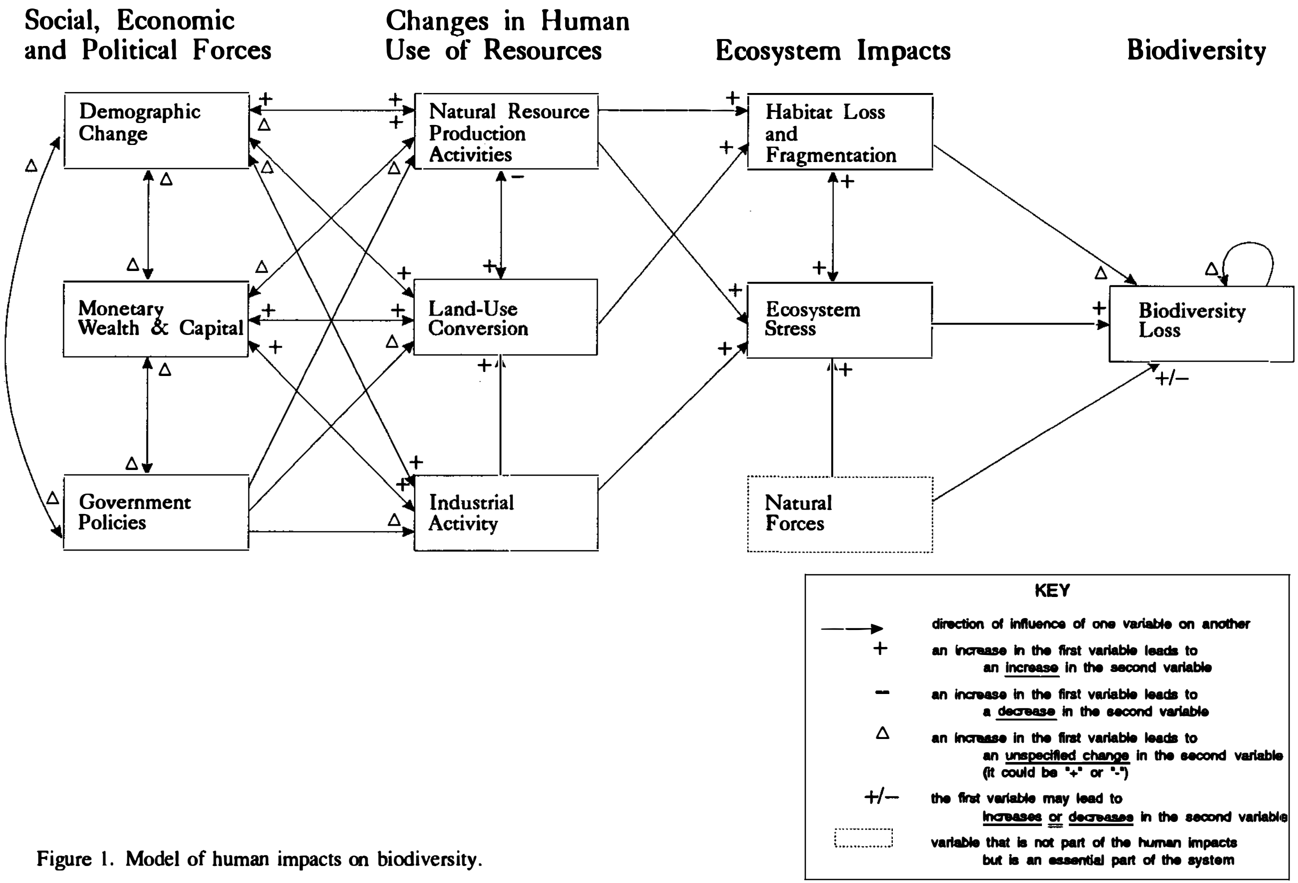

Figure 1. Model of human impacts on biodiversity.

graphic areas (an example is a collection of counties) in which socioeconomic forces are likely to affect the biodiversity in specific gap locations.

Based on the model, indicators of human action are collected for the socioeconomic zones of influence from a variety of sources, such as the U.S. Census Bureau and other federal and state agencies (see Table 1 for examples). Data are entered into the GIS database. Related indicators are combined into indices, again based on the model. An overall index of vulnerability is created and each gap location is given a relative index score. The results are displayed in map form; the maps may be useful to managers, landowners, resource agencies, advocacy groups and interested citizens (McKendry and Machlis 1993).

A Gap Analysis of Idaho

The potential for using this interdisciplinary technique was tested through a pilot project in Idaho (Machlis et al. 1993). Native vertebrate species richness (excluding fish) was used as the basis for the analysis. Data for the state were aggregated by 635 square kilometer hexagons developed for the Environmental Protection Agency's Environmental Monitoring and Assessment Program, as a convenient way of making the analysis.

Gap locations were selected using a specific algorithm. The hexagon with the highest number of species was identified, followed by the hexagon that added the highest number of species not already in the first hexagon. This procedure continued until all native vertebrate species were included in the set of hexagons. The result was the minimum number of hexagons containing all native vertebrate species in the state (Kiester et al. unpublished manuscript, Machlis et al. 1993). Five hexagons were selected for analysis. Together, the selected hexagons contained approximately 95 percent of all native vertebrates in Idaho.

Each of the five hexagons was identified as a gap location potentially important to biodiversity management in Idaho. A map of these gap locations was overlaid with a map of special management area status in Idaho; areas were defined as having ''complete'' or ''partial'' protection based on The Nature Conservancy classification system. None of the hexagons was totally protected, though small portions of protected areas, including National Wildlife Refuges, were present in several of the gap locations.

Socioeconomic indicators similar to those listed in Table 1 were collected for the counties surrounding each gap location (the socioeconomic zones of influence). Data

Table 1. Example indicators of human actions

Air quality	Municipal solid waste
Defense lands and installations	Number of vehicles
Demographic forecasts	Occupation
Economic forecasts	Political units
Hazardous waste exposure	Population
Housing characteristics	Population density
Labor force projections	Real estate transactions
Land-use regulations	Residential construction
Location of manufacturing	Road construction

were entered into the GIS database. Four indices were constructed and mapped: socioeconomic change, government policies, land development, and ownership complexity. Figure 2 shows the results for socioeconomic change; the lower the index score, the lower the predicted level of population growth and economic development. The four indices then were combined into an overall index of vulnerability to

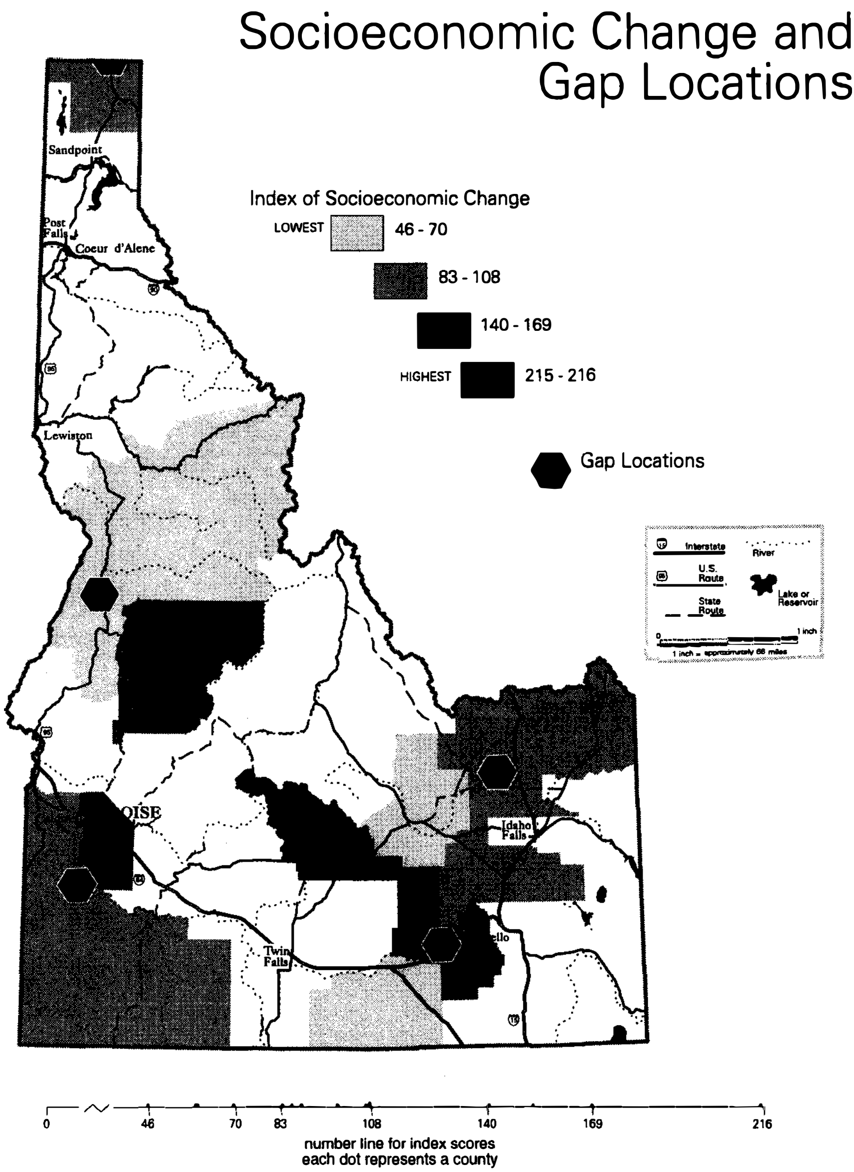

Figure 2. Map of index of socioeconomic change and gap locations.

biodiversity loss. Finally, the gap locations were ranked as to their relative vulnerability and the results were displayed in a map series (Machlis et al. 1993).

Applying Gap Analysis to the National Wildlife Refuge System

Gap analysis is both an evolving technique and an emerging multi-state program of data collection and research. As the technique and program are refined and expanded, the potential application of gap analysis to the National Wildlife Refuge System is likely to be realized. The general technique can directly aid decision makers and refuge managers by providing a framework for specific biodiversity studies and planning efforts. Also, the national GAP program eventually will produce spatial data sets for many areas of the country. Three broad categories of application, all interrelated, can be outlined: description, analysis and prediction. Several examples of each follow.

Description

The gap analysis technique can be used to describe the ecological and socioeconomic context of individual refuges. Ecosystem management is best accomplished within an ecological context, such as an ecoregion, drainage basin or watershed. Few refuges are complete in this regard; many are a small fraction of such areas. Thus, if ecosystem management is to be practiced by refuge managers, they will require sophisticated knowledge of the ecosystem(s) in which a refuge occurs and the degree to which the ecosystem's representation is complete within refuge boundaries. Such information should include the ecosystem's defining features (hydrologic units, elevation, vegetation types) and biodiversity values (such as species richness).

Wildlife refuges also function within a ***human ecosystem***, i.e., a landscape profoundly influenced by human actions (Machlis 1989). Socioeconomic factors such as land development, population and government policies are crucial to understanding this human ecosystem. Maps of these factors can help managers understand better the location, nature and extent of human activities that may have an impact on, and be impacted by, refuges. This may be especially important for small refuges heavily affected by surrounding human activities.

The gap analysis technique and data from the national GAP program can be used to describe the biodiversity value of individual refuges in terms of specific vegetation types, species richness or a selected subset of species. The distribution of biodiversity across the landscape also can be depicted. This description can be useful in determining the roles a particular refuge, as well as the refuge system, play in biodiversity conservation within a defined ecological or political unit.

Analysis

Identification of gaps in the biodiversity protection currently provided by the refuge system should be done on an ecoregion basis whenever possible. The gap analysis data set can be used to ask what elements of biodiversity currently mapped (such as actual vegetation and vertebrate distributions) are found within national wildlife refuges for a given ecoregion. The results then can be compared qualitatively and quantitatively to the known occurrence of these elements of diversity within the ecoregion. Absence or under-representation of a species or vegetation type from the refuges could represent a gap in the biodiversity coverage of the refuge system.

Gap analysis can be used to analyze threats to biodiversity in identified gap locations or in individual refuges. Generalized threats (such as population growth in a county) and point source threats (such as hazardous waste disposal sites) can be examined and evaluated both individually and in concert. Analysis of several threats may show a concentration of factors in one area that presents special challenges to refuge management.

The gap analysis technique can be used to analyze acquisition or restoration needs for individual refuges and for the refuge system. For example, after examining the ecological context of individual refuges, one may consider changing the boundaries of a refuge to coincide more closely with an ecologically defensible unit. If that is impractical or undesirable, it may be necessary to cooperate with adjacent land managers to maintain ecosystem integrity.

Maps of factors such as land ownership, land use or distance to areas managed for biodiversity can help managers to determine the feasibility and desirability of alternative acquisition or management plans. For example, the number of owners in an area may indicate the potential complexity of acquisition and the need for cooperative management arrangements. The distance to other areas managed for biodiversity may be important in considering issues of connectivity, dispersal distances of particular species, the possibility of genetic exchange between populations and corridor design.

In addition, the results of gap analyses can contribute to ecological research. For example, refuges frequently exist in a matrix of land uses that may not be favorable to dispersal or immigration of species. Combining gap analysis data sets with information such as mortality and dispersal rates, it may be possible to conduct metapopulation analysis, as well as to determine isolation factors and potential extinction rates in a specific landscape.

Prediction

The gap analysis technique can be used to predict the relative vulnerability of gap locations to biodiversity loss. An examination of the socioeconomic factors in areas surrounding gap locations, using the model described earlier, can help determine which areas are most vulnerable to biodiversity loss and which human factors contribute the most to that risk (e.g., land development or industrial pollution). The same technique can be used to examine the relative vulnerability of several refuges (such as all of those within an ecoregion).

Land-use changes in and around gap locations or refuges can be predicted using the gap analysis technique and national GAP data sets. Demographic, economic and land-use projections can provide important information for planning long-term management strategies. Different scenarios (such as low, moderate or rapid population growth) can be examined, and their potential impact on refuges evaluated.

Another example of its predictive use is how gap analysis data sets may prove an important component of global climate change research. Simulation studies of the effects of different temperature scenarios on changes in habitat and the dispersal of species can be conducted using data derived for gap analysis efforts. In the context of refuges, these studies could be especially useful in pointing out the potential value of particular refuges for conserving biodiversity under varying climate change scenarios.

Conclusion

The U.S. National Wildlife Refuge System plays an important role in conserving biodiversity in the United States. Intelligent and proactive management requires that refuges be managed in an ecosystem context, and that managers have access to information on ecological features and human activity that may impact biodiversity values. Gap analysis is an evolving technique with potential applications to decision making throughout the refuge system, and could serve to help integrate the National Wildlife Refuge System into broader efforts to conserve our nation's biodiversity.

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Partners for Natural Resource Conservation: A Strategy Addressing Natural Resource Issues through Partnerships

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Introduction

In January 1993, U. S. Fish and Wildlife Service (Service) Regional Director John G. Rogers convened an interdisciplinary task force to consider the effectiveness of the Service's Region 2 (Region) in addressing natural resource issues. Director Rogers encouraged the team to look into new ways of doing business and, specifically, to: (1) develop an integrated, cross-program approach to address issues; (2) reduce duplication and controversy among divisions and programs; (3) encourage partnerships within and outside the Service; and (4) maximize resources/benefits to wildlife within the Region.

Partners for Natural Resource Conservation

The task force proposed Partners for Natural Resource Conservation (PNRC) to emphasize the need for Service employees at all levels to develop partnerships—cooperative efforts with other federal and state agencies, private individuals and organizations, and corporations—to achieve the highest possible level of conservation. Note that this approach also will require stronger partnerships and coordinated work efforts among and within Service programs.

With limited funding and personnel a given in our agency—combined with the multitude of threats facing our nation's fish and wildlife—the task force concluded that the most effective approach to protecting and managing natural resources is through partnerships with others who share our vision. Another intent of this strategy is to increase participation of all Service employees in decisions affecting natural resources in the Region. The approach will require the Region to: (1) permit maximum delegation of authority to project leaders and their staff; (2) encourage Region employees to accept delegated responsibilities; (3) facilitate cross-program coordination and cooperation; and (4) create a functional program in response to the strategy.

The primary role of the Regional Directorate (Assistant Regional Director level) will be as ''agents'' for the Region, that is, they will take an aggressive approach to acquire funding for the Region, garner support from the Washington office, and encourage/help the field implement the strategy to achieve its goals and objectives. Implementation of PNRC will generate new ideas from the field level to improve management of Service lands, federal trust species (e.g., migratory birds, anadromous fish, threatened, endangered and candidate species, and habitats that support these species) and ecological resources of national/regional importance. It also will improve our relationship with entities outside the Service. To be fully successful, this strategy will require the support and participation of Service personnel and the community they live in.

Goal

The goal is to establish a broad-based landscape management strategy for natural resource conservation, based on ecological areas and priorities as defined by interdisciplinary resource specialists, rather than individual programs. The strategy recognizes the importance of considering and protecting culture/community values occurring throughout the Southwest Region.

Facts and Assumptions

Region 2 PNRC is a multi-divisional and multi-disciplinary landscape-based strategy for resource management. It directs cooperative development of projects among Region programs and partners. The importance of maintaining and preserving existing cultural/community values throughout the Region is recognized. Funding for programs will not change, but may be redirected within individual programs as the strategy is implemented. Funding for projects will come from various existing Service program funds and from non-Service partners. To ensure success, alliances will be forged among all divisions, applicable programs and initiatives within the Region. Cooperative and coordinated efforts of all interest and potential partners (e.g., federal, state, local and international governments, private landowners, Native Americans) are required. Attempts are made to identify (cooperatively with partners) resource issues, needs and projects from the field level and across divisional and program lines. Proper application will reduce the number of external mandates and increase the ability of Region personnel to identify and influence future mandates.

Implementation

Successful implementation of this strategy depends on accomplishing the following steps. First, establishing a temporary team to identify ecoregions and partnership opportunities within each ecoregion (task completed by December 1993). Second, an Ecoregion Development Team (EDT) within each ecoregion would be comprised of interdisciplinary specialists from the Service, universities, states, other federal agencies, Native Americans, private landowners, nongovernmental organizations, etc.

Service field personnel from all appropriate divisions will serve as initial leaders in establishing the teams. The emphasis of these partnerships is to benefit the natural resources within each ecoregion, not just Service trust resources. The EDT will develop projects at the ecoregion level that need to be completed to improve the condition of our natural resources.

The EDT would report to the Regional Implementation Team (RIT), which includes one representative from each EDT and one representative from each Service program. The RIT would be responsible for dividing the Service budget among the appropriate projects. If a project does not receive funding from the Service, this does not prevent the project coordinator from implementing the project through some other funding source.

Obstacles

The task force of field representatives identified many obstacles to completing the PNRC strategy. First, restrictions apply to various funding sources, programs and initiatives that may impede implementing the strategy. Second, Service employees may be reluctant to participate if this is seen as just another reorganization program. Many employees carry a great deal of pride and ownership with their programs and may be reluctant to work with other programs or a new program. The field team felt that both the Washington and Regional offices would be reluctant to relinquish traditional control points normally associated with their offices, down to the field level. It was felt that the current interpretation of legislative mandates could impede implementation of this strategy. In general, we felt the Service lacked sensitivity to partners, other programs and the expertise of others. We need a great deal of training in how to deal more effectively with our partners, local attitudes and "how things are done around here." In other words, what works in Arizona may not work in Texas.

Recommendations

We must (1) remove restrictions that apply to various funding sources, programs and initiatives that may impede implementing this strategy; (2) identify the real and perceived restrictions that keep us from serving the resource; (3) where feasible, recognize and advocate flexibility for those funding sources that may be too restrictive; and (4) identify programs that encourage and use partnerships.

Some Service employees will be reluctant to participate because of program ownership or refusal to share because of traditional "turf battles." Perhaps the best way to address this issue is with pilot projects to illustrate the value of such a strategy and when they prove successful, expand the PNRC strategy to other areas as appropriate. Both the Washington and Regional offices probably will be reluctant to delegate more authority to the field. Again, the best recommendations were to show by example that this type of strategy could be successful and productive.

The Service will need to reexamine legislative mandates that may impede implementation of this strategy and, where feasible, amend restrictive situations to ensure flexibility. For those situations where inflexibility is beyond repair, remedial action needs to be initiated. A need exists to remove the perception and, in some cases, the reality that we are "over-regulated." The Regional Director should maximize discretion and authority to make decisions regarding flexibility.

In many cases, the team felt the Service lacked sensitivity to our partners, other programs, the expertise of others and the local attitudes (''how things are done around here''). We recommend increased sensitivity through training, apprenticeships and details, and increased cooperation through increased awareness and coordination.

Conclusion

In coming together as teams, we rediscovered an environment of shared responsibility. We in the Service must work together as a team to discover ''what is right,'' rather than try to solve the age-old question, ''who is right?'' Certainly, we can do a better job of working together for the resources. Perhaps this strategy is one way we might envision what can be rather than what is.

Acknowledgments

We thank all the employees of Region 2 of the U. S. Fish and Wildlife Service for their comments and constructive suggestions as we developed this strategy. We thank John Rogers, Joe Mazzoni and Jim Young for having the original vision and pulling this cross-program team together. A special thanks goes out to the following team members: Charlie Ault, Debbie Davis, Jim Hutchinson, Rod Krey, Rob Lee, Ric Riester, Marie Sullivan, Thea Ulen, Laurel Kagan Wiley and the staff at Laguna Atascosa National Wildlife Refuge.

Special Session 9. *Outdoor Ethics and Recruitment*

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Into the Quagmire of Outdoor Ethics

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Welcome to the Quagmire

If you are here (those of you physically here on the last day in the last session, or those of you reading the Transactions), you probably have been in the quagmire of outdoor ethics. Anyone who has ever been involved in outdoor ethics issues knows the feeling: the helpless flailing in the philosophical debate, the dangerous undertow of panacea programs that might suck valuable resources away from your over-scheduled agenda, the force of gravity that could sink you if the ethics of practices of influential groups come into question and the nagging certainty that ignoring the issues will not make them go away.

All of this is exacerbated by the fact that many of us have the roots of our training in the natural sciences and are far more comfortable with both questions and answers that can be generated in that realm. Be that as it may, the ethical issues continue to surface periodically, even in bastions of biological thought such as the *Journal of Wildlife Management* and the *Transactions of the North American Wildlife and Natural Resources Conference* (Leopold 1943, Wagar 1945, Klein 1973, Jackson et al. 1979). Ethics continue to surface in other realms, as well. In the past 15 years, there have been two national conferences on outdoor ethics (Izaak Walton League 1980, 1987). In addition, several of the more well-known outdoor writers have kept the ethics question on the forefront of the popular literature for the past 10 years (Reiger 1985, 1986, 1988, Williams 1992, Williamson 1989). A wide variety of conservation groups, agencies and businesses also have promoted outdoor ethics through creeds and brochures, and some states have had committees whose task has been to study and report on ethics of practices and equipment in outdoor recreational activities (Ethics and Fair Chase Study Committee 1991).

The ethics questions are important to the entire resource management community because they affect the resources and the constituent groups that we manage. The face of North America is changing as our population becomes more urbanized and less likely to have ties to the land. We are in an era when regional differences in culture and acceptable behavior are being reduced by the preponderance of mass media in our lives. As a result, the image of the activities that we think of as traditional wildlife-based recreations is suffering. It is even possible that this image may be inhibiting individuals from participating in those recreations (Thomas and Peterson 1991). This session was put together to raise outdoor ethics issues to the national consciousness again. Session presenters have been to the quagmire and they hope their experiences can help move the discussion along in fruitful ways.

The initial intent of the session was to focus on the activities of traditional outdoor recreationists. Not long after the "call for papers" was published, the session chairs realized that the "ethics" issue means many different things to many different resource management professionals. A few proposed ideas for papers and were encouraged to submit. Although they did not, it is worth sharing some of their ideas so that we begin to understand that the ethics question is much broader than just the size and method of bagging.

Paper inquiries included one that focused on the importance of the ethics of the individual resource manager in her or his dealings with the public. Confidence in our whole management system hinges on that issue. A second inquirer would have presented a paper on the methods for helping landowners develop a land ethic that is conducive to and compatible with wildlife management. Surely that slant on the ethics question is critical to our resource management system as well. A third paper would have looked at the criminal justice system as a vehicle for using violators to teach others about the consequences of unethical acts that also happen to be illegal. A fourth paper would have looked at factors, including ethical treatment of women and minorities in resource management agencies, that may affect recruitment of underrepresented groups into resource management careers. As we discovered, the range of "ethics" questions is far broader than that initially envisioned.

Those papers that were submitted and selected show a range of topics as well. While they may be diverse in content, there are themes that emerge from these papers. Session presenters and chairs hope these themes will help resource managers in their deliberations about ethics.

The first theme, as already mentioned, is that the "ethics" question should not be narrowly confined to the behavior of the traditional wildlife user. It should permeate every action and interaction of everyone in the resource management stream. As resource managers, for example, our relationships with the resource, user, landowner and each other must be based on respect.

When it comes to encouraging ethical behavior in the sporting community, the second theme that emerges from this session is the need for research to determine what types of programs work. A perusal of the literature and the proceedings of various conferences, workshops and committee deliberations shows that most groups who have looked at outdoor ethics questions have advocated similar cures for our ethics ills. Almost every look at the problem has advocated a "creed." These creeds have probably taken up space on millions of reams of paper. Has anyone read them? Have they affected anyone's behavior? Nearly all recommendations advocate launching off on a new (usually national level), massive program to incorporate an education program into an existing curriculum or start a new program. But do we know whether any of these will work, have worked or would ever work? Isn't it time we asked these questions? Perhaps we do not really want to know.

Another theme that will emerge from the session is the idea that we probably will not solve all our ethics problems with national-level projects of any kind, nor even with state-level projects. Some of what needs to happen may be influenced or modeled by national- and state-level projects, but some of what needs to happen is that individuals need to aspire to being ethical. Perhaps as managers and educators we can help the sporting community flesh out what that aspiration should be, but the individual must decide whether to strive to attain those aspirations.

When it comes to the "fleshing out of what is and is not the ethical sportsperson," those of you who have been to the quagmire know that is a natty problem indeed. I spent 1990 as secretary of the Wisconsin Ethics and Fair Chase Study Committee, which was appointed by Tom Lawin, then chair of the Wisconsin Natural Resources Board (Board). That group did an extensive study of practices and equipment used in outdoor activities in Wisconsin and wrote an extensive report which it submitted to the Board. The report includes position statements on a variety of outdoor practices (Ethics and Fair Chase Study Committee 1991). The citizen committee made an attempt to "flesh out" the ethical sportsperson. Not everyone liked what they saw. There are several lessons in their efforts that may be of help to resource managers.

First and foremost, the Committee was very controversial from the outset. The press had a field day trying to dredge up controversy before the Committee even met the first time. Headlines had the Committee banning everything from fish locators to rifle scopes. While it is true that controversy sells papers, it is also true that, by and large, the outdoor media is different from other journalism fields in that most outdoor writers also are part of the sporting community and not simply passive reporters. With that in mind, the outdoor media could help to facilitate some agreement on the "fleshing out" process. Resource managers should work to bring the media into this process.

A second lesson from the Wisconsin experience is that a vehicle for citizen participation already existed in the Wisconsin Conservation Congress. Some members of the Congress saw the Fair Chase Committee as somehow usurping its role instead of helping to further an important goal of all sportspersons. If the definition of the ethical sportsperson is to gain wide acceptance, the leadership of the sporting community needs to be involved in the process. A top down approach may provide some direction, but will be slow in gaining acceptance. Not one sports group came forward to support the recommendations of the Ethics Committee when they were submitted. Part of this is the fault of the Committee, for not realizing that this was needed. Still, the ideas in the report are beginning to pick up momentum, now three years later. Sports groups are beginning to request copies of the report and at least two statewide organizations are grappling with ethics issues in a public way.

That brings us to the final lesson of the Wisconsin experience, which is also a theme of the papers in this session. There is no one instant answer to the ethics questions. No helicopter rescue will extract us from the quagmire. It will take hard work, careful consideration of the direction that needs to be taken and pulling together to make steady progress toward a goal that will continue to evolve even as we make progress toward it. The striving is crucial, for the fate of wildlife-based recreation and even traditional wildlife management may hang in the balance.

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Ethics and Subsistence

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Introduction

Wildlife management has evolved from a long history of sport and commercial hunting, and the concept that recreation and economic gain are the best consumptive uses of animals. However, this model has been largely unsuccessful in managing subsistence harvests in Alaska, partly because standards of behavior, or ethics, differ greatly between subsistence hunters and other consumptive users of wildlife.

Although considerable attention is focused on the importance of ethical sport hunting, the ethics of subsistence hunters are seldom discussed. In Alaska, subsistence is a way of life in which personal well being and cultural identity are closely associated with harvesting and sharing wild resources.

Subsistence can be a way of life for both Alaska Natives and non-Natives, although the primary focus here will be on indigenous residents of the state. This paper will consider some of the ethical differences between sport and subsistence hunters, discuss the effectiveness of existing wildlife management programs in rural areas, and suggest a strategy to protect more fully both subsistence lifestyles and wildlife in Alaska.

Sport and Subsistence Hunting

Sportsmen in North America have placed a strong emphasis on ethical hunting, and standards of fair chase were quickly established by early sportsmen's organizations. The use of motorized vehicles for spotting, herding or pursuing animals was denounced. Ethical hunting has become even more important today, as the potential for abuse is increased by technological advances (Jahn 1992) and changing social values become less supportive of harvesting animals for recreation (Heberlein 1991).

Wildlife management in Alaska is based on western conservation ethics utilizing seasons, bag limits and fair chase principles. Management programs typically have provided uniform opportunities, regulations and enforcement for all users. Although sport hunters have been well served, the state has been less conscious of subsistence interests. It is not surprising that Native residents in particular are critical of existing wildlife management systems in Alaska.

Subsistence is identified by both state law (Alaska Statutes, Title 16, Section 16.05.258) and federal law (Public Law 96–487 Section 802 (2)) as the priority use of wildlife and fish. Subsistence, recreational and commercial harvests may be allowed when resources are abundant. However, during times of scarcity, non-subsistence harvests will be restricted before subsistence uses are significantly reduced.

Subsistence is defined by the state as "customary and traditional uses by Alaska residents for food, shelter, fuel, clothing, tools, transportation, making of handicrafts, customary trade, barter and sharing" (Alaska Administrative Code, Title 5, 99.010 (2)). Both Alaska Natives and non-Natives may harvest wild resources for subsistence under state and federal laws. Alaska is unique among the states in granting subsistence a priority among all consumptive uses of wildlife.

Subsistence centers around using both efficient and effective hunting practices, sharing harvests and respecting wildlife. Poor weather and travel conditions, and brief opportunities to harvest migratory wildlife often limit or even prevent annual harvests of some species. Fall (1990) summarized several studies in Alaska which show that hunting, fishing and gathering in rural areas are part of a mixed subsistence and market economy. It is typically undertaken by family groups using small-scale technologies, such as rifles, traps and snares, small motor boats, and snow machines. Subsistence harvests are supplemented by periodic wage employment, with cash being reinvested in equipment. Fall (1990) also reviewed data for 122 Alaskan communities showing that total annual harvests averaged 300–400 pounds (136–182 kg) per capita. For these residents, subsistence is an integral part of the economy, social life and culture.

Hunting Regulations Which Accommodate Subsistence

The challenge facing Alaskan wildlife managers is to balance statewide sport hunting interests and local subsistence needs within the framework of sound conservation. Sport hunting regulations in Alaska often are inconsistent with established subsistence practices which are deeply rooted in cultural traditions. Even before the first subsistence law was enacted, the Alaska Department of Fish and Game (ADFG) took steps to protect subsistence uses by promulgating regulations favoring local users. When biologically permissible, seasons and bag limits were very liberal, including no closed season and no bag limit for caribou (*Rangifer tarandus*) in northwestern Alaska and a two-moose (*Alces alces*) bag limit in several areas of interior Alaska. User conflicts between urban sport hunters using aircraft and rural subsistence hunters using boats prompted the state to establish several "controlled use areas." It was illegal to use aircraft for moose hunting in these areas, which significantly reduced competition from non-local hunters by eliminating their principal means of transportation.

More recently, additional regulatory changes have tried to accommodate subsistence hunters. An eight-month autumn and winter moose season and another moose season in mid-summer expands hunting opportunities for local residents in two areas which have adequate moose populations. Other regulations have eliminated some bag limit restrictions on antler or horn size. For example, a bag limit of one bull moose with 50-inch (127 cm) or greater antler spread applies in several popular sport hunting areas, but not for resident hunters in most rural areas of the state. Additionally, qualified subsistence hunters may take up to three Dall's sheep (*Ovis dalli*) of any age or sex during a seven-month season in some areas of northern Alaska. Sheep bag limits typically are one full curl ram during a short autumn season for recreational hunters in most areas of the state.

Other regulatory changes have legalized some hunting methods and means that previously were disallowed. Subsistence hunters typically have killed caribou at water crossings in some areas of Alaska, historically from kayaks with spears and presently from boats with firearms. It previously was illegal to shoot any big game which was swimming, to shoot from a moving power boat or to use .22 rimfire cartridges. However, these constraints were abolished in one area of northwestern Alaska, thereby replacing western standards of fair chase with more appropriate subsistence hunting regulations. Additionally, trapping regulations were amended in several remote areas of the state to allow the shooting of beaver (*Castor canadensus*). This legalized a common practice of taking beaver during spring when meat and pelts are used locally for food and garments.

Until recently, grizzly bear (*Ursus arctos*) hunting was considered to be of exclusive interest to sportsmen. Alaska hunting regulations epitomized the sport hunting ethic by maintaining relatively short seasons, limiting bag limits to one bear per four years, requiring that a $25 tag or "trophy" fee be paid before hunting and requiring that the skull and hide be salvaged and a locking tag attached to both by the ADFG. However, some subsistence hunters do not consider bears as trophies, but as animals with unique spiritual qualities and as valuable sources of fat and meat (Loon et al. 1989, Coffing et al. 1992). The state responded to requests from subsistence bear hunters by temporarily eliminating the "trophy" fee in northwestern Alaska and by allowing a one bear per year bag limit in western Alaska. These changes were made with the intention that subsistence bear hunting would become less encumbered by sport hunting values, and with the expectation that harvest reporting therefore would improve. However, harvest reporting did not improve.

Grizzly bear hunting, as with other species, is governed by multiple regulations affecting different facets of the hunt. It is not surprising that attempts to accommodate subsistence practices by changing only one of many interrelated regulations were largely ineffective. Therefore, comprehensive regulatory changes recently were made in two large bear management areas. Regulations specific to these areas now allow residents to hunt either under sport regulations or new subsistence regulations. The subsistence regulations include a nine-month season, a bag limit of one bear per year, and elimination of both the "trophy" fee and the need to salvage the hide and skull. However, all edible meat must be salvaged. If the hide or skull is removed from the management area, locking tags must be attached and the trophy value of the hide destroyed.

Almost 100 permits were issued last year in each management area to hunters preferring to hunt under subsistence regulations. A subsistence harvest of 8 and 12 grizzly bears was reported for the two areas. The number of hunters obtaining subsistence permits and the reported harvests suggests a favorable response by some hunters to the comprehensive regulatory changes. However, the actual harvest undoubtedly was higher than reported (see Loon et al. 1989).

Hunters must comply with several administrative requirements by obtaining licenses and harvest tickets or permits to take big game species, validating harvest tickets after taking animals and returning harvest reports to the ADFG. Harvest tickets usually are required to hunt most big game species, such as caribou, moose and sheep. Harvest reports ask several questions pertaining to hunter residency, sex, date and location of kill, horn or antler measurements, length of hunt, weapon used, type of commercial services used, and method of transportation used. Similar questions are asked when a grizzly bear hide and skull are sealed. Because these administrative requirements were viewed as onerous by some local residents, they were simplified for hunters taking caribou in northwestern Alaska and grizzly bears in the two bear management areas.

A hunting license still is required and individuals must register to hunt both caribou and bears. However, hunters no longer were required to answer superfluous questions on big game harvest reports or grizzly bear sealing documents. Rather, the ADFG annually sends a letter to everyone registered to hunt caribou asking only how many caribou were killed in autumn and how many were killed in spring. A similar letter sent to subsistence grizzly bear hunters asks if a bear was taken, and the date, sex and location of the kill.

Other attempts to allow subsistence practices have involved limiting enforcement efforts. Waterfowl regulations, which were promulgated primarily for sport hunters, often are ignored and not enforced in remote areas of Alaska. Many Native residents historically have taken migratory birds and their eggs for food during spring and early summer in direct conflict with the 1916 Migratory Bird Treaty between the United States and Canada (Wolfe et al. 1990). Discussions to amend the treaty to legalize and regulate spring hunting in both Alaska and Canada currently are underway.

Subsistence Management and Continuing Regulatory Problems

Previous regulatory and administrative changes demonstrate that managers recognize that subsistence harvesting practices often are inconsistent with sport hunting ethics. These steps, albeit small, were taken to develop more culturally acceptable management systems. However, conflicts still exist.

The number and complexity of regulations continue to be problems. ADFG hunting regulations pertaining to terrestrial species have increased in volume by a factor of 10 since 1960, and by a factor of two and one-half in just the last decade. Area-specific management needs, objectives and problems have resulted in more and increasingly complex regulations. Although sport hunters often complain about such trends, most grudgingly accept added restrictions as necessary to maximize recreational opportunities and to protect wildlife. The trend in Alaska has been for hunting seasons and bag limits to apply to progressively smaller areas, shorter times and more specific user groups. To improve standards of sport hunting conduct, reduce opportunity for abuse and increase accountability of hunters, a myriad of additional regulations and administrative procedures have been promulgated.

The Alaska hunting regulation booklet applies statewide. This can be useful to sport hunters who may hunt in several areas, but is of little value to subsistence hunters who tend to hunt near their community (Fall 1990). Native residents are frustrated by the large volume of and difficulty in locating applicable regulations, and are confused by legal phraseology. These problems are especially acute for residents who speak English as a second language.

Subsistence regulations frequently are inconsistent with the traditional values and contemporary culture of Native people. A wildlife advisory committee in northwestern Alaska consisting primarily of Native subsistence hunters reviewed Alaska hunting regulations (Schaeffer et al. 1986). The committee found the regulations replete with stipulations detailing legal and illegal sport hunting behavior which had little relevance to Native hunters.

Regulations prohibiting taking game from mechanical vehicles frequently are perplexing to subsistence hunters who typically have hunted waterfowl from moving boats and some furbearers from moving snow machines (Schaeffer et al. 1986). Until recently, even shooting from a stationary snow machine or using the seat to steady a firearm was illegal in most areas of the state. These prohibitions become more contentious when subsistence users see extensive legal use of aircraft by sport hunters for locating wildlife, and apparent illegal use for herding animals. Low-level flights also occasionally divert animals away from subsistence hunters on the ground. Alaska regulations allow use of aircraft for transporting hunters to and from the field or for spotting wildlife, although no hunting may occur until the day after being airborne. The ban against use of radio communication for hunting likewise is an irritation. Citizen band radios are widely used in rural Alaska, and they are a convenience which increases safety and efficiency of hunting.

One of the greatest difficulties in applying Alaska wildlife regulations to subsistence hunters pertains to bag limits (Schaeffer et al. 1986). Sport hunting tends to be an individual recreational experience which can provide a trophy and meat for use by a hunter or his immediate family. Regulations limit individual harvests to avoid depleting populations and maximize recreational opportunity. However, subsistence often is a group undertaking in which harvests are conducted by experienced hunters in the most efficient manner possible, and are widely shared by an extended family of several households in the community and adjacent villages (Loon 1989).

It is not unusual for subsistence hunters to exceed individual bag limits and not report harvests (Andersen et al. 1992). Alternatively, subsistence hunters may be accompanied in the field by others who possess a hunting license but do not hunt. Excess kill then may be reported by other license holders. Large legal harvests of up to 66 caribou annually also have been recorded from northwestern Alaska (ADFG files, Nome: 1990), where the bag limit is five per day.

Closely related to problems with bag limits are regulations pertaining to possession of wildlife (Schaeffer et al. 1986). Effective enforcement of wildlife regulations required that the harvester can be identified. Therefore, when a hunter transfers ownership of unprocessed meat he also must provide a signed statement describing the names and addresses of each person who gave or received wildlife, when and where it was taken, and what parts were transferred. Although this requirement is easy to comply with when sport hunting partners share a kill, it becomes impossible within the extensive sharing and bartering network of the subsistence economy.

Harvest information often is important for wildlife management, and the ADFG administers several systems in which hunters are responsible for reporting their harvests. Although some violations occur, sportsmen generally understand reasons for these requirements. Harvest report return rates range from 100 percent for many permit hunts to 70–80 percent for wildlife species requiring harvest tickets (Alaska Wildlife Harvest Summary 1993).

However, subsistence hunters frequently do not participate in administrative harvest reporting systems (Usher et al. 1987, Andersen et al. 1992). In some remote areas, the process for issuing licenses and harvest permits is poorly developed, and even well-meaning hunters may have difficulty obtaining required documents. Rural hunters often have difficulty understanding complicated regulations. Consequently, harvests are not reported because hunters fear self-incrimination by having unknowingly committed a violation. Others basically distrust management agencies and refuse to provide harvest information, believing that it will be used to restrict them in the future.

Directions for the Future

More liberal regulations and easier harvest reporting have been established for some wildlife populations in rural Alaska. Nevertheless, many subsistence hunters remain indifferent and resentful toward western wildlife management, believing that it is unnecessary and insensitive to their needs. Effective subsistence management will continue to be an elusive goal unless local hunters can participate in what they believe is a useful and legitimate regime. The best alternative for understanding and incorporating subsistence interests into wildlife management in Alaska is through meaningful cooperation between users and managers.

Cooperative resource management has become a popular concept (Osherenko 1988, Berkes et al. 1991), although it is subject to broad interpretation (Schwarber 1992). While the degree of cooperation or sharing of authority will be determined by legal and political factors, significant participation by local residents in wildlife management offers hope for protecting wildlife populations and subsistence lifestyles in the future. Several resources, including geese in western Alaska, bowhead (*Balaena mysticetus*) and beluga (*Delphinapterus leucas*) whales, walrus (*Odobenus rosmarus*), and polar bears (*Ursus maritimus*) have been successfully managed through cooperative programs (Pamplin 1986, Carpenter et al. 1991, Huntington 1992). A cooperative management plan for a depressed caribou herd in western Alaska is nearing completion. Users and managers are taking steps to replace confrontation with cooperation by reaching consensus on goals for herd growth, harvest levels and information exchange. However, state and federal governments have retained authority to regulate.

Cooperative management requires genuine communications between local users and biologists to develop management plans, monitor harvests, improve hunter efficiency and harvest utilization, and/or promote research and education programs. Management goals must ensure conservation of the species and allow traditional subsistence harvests above minimum population levels. Subsistence users have an opportunity for their social, cultural and economic needs to be considered, and to help ensure that regulations and enforcement are appropriate. Biologists benefit from the traditional knowledge of hunters and from improved public willingness to provide harvest data and comply with regulations.

Cooperative management is consistent with principles of the World Conservation Strategy (IUCN et al. 1991). This Strategy describes a comprehensive program for sustainable use of the world's resources. It also urges that local use take priority over non-local use of resources. A critical component of the document stresses the need to protect the rights of indigenous people to harvest resources upon which their culture depends, and to ensure that they fully participate in decisions affecting them.

The greatest challenge in Alaska is to manage wildlife for both subsistence and recreational interests. The rights of each group are being increasingly contested. Natives are trying to broaden and strengthen their authority, and sportsmen are attempting to reduce or eliminate what they believe are unfair advantages enjoyed by subsistence hunters. While the courts and politicians work to resolve differences and establish criteria for allocating scarce resources, management agencies must continue to expand opportunities for local participation in managing wildlife. A strong desire to protect and strengthen their cultural heritage, and their increasing legal, political and economic power ensure that Natives will increasingly participate in managing the resources upon which they depend.

Still, subsistence hunting advocates need to be aware of the power of public opinion. Although society is becoming less tolerant of recreational hunting (Heberlein 1991), residents of Alaska strongly approve of harvesting wildlife for food (Miller et al. 1993). Several traditional pursuits, such as whaling, sealing and trapping have been severely condemned (Gentile 1987, Lynge 1992, Kalland 1993), and environmentalists have criticized special hunting privileges of American Indians (Schwarz 1987). Indigenous people nevertheless enjoy considerable support for integrating their subsistence traditions into modern society. However, occasional examples of harvests beyond need and failure to utilize carcasses further opposition to subsistence activities and criticism of those who support traditional but responsible hunting practices.

While developing more effective subsistence management programs, sport hunters and wildlife viewers must not be ignored. Most wildlife populations in Alaska are relatively abundant, and with few exceptions, habitats remain intact. Traditional practices important to the cultural preservation of Native and non-Native residents can be accommodated by many wildlife populations without reducing hunting and viewing opportunities of other users. With innovation and a commitment to work together, wildlife can be protected for sustained use and enjoyment of all people, and wildlife management in Alaska can be responsive to both subsistence and recreational interests.

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Setting an Agenda for Outdoor Ethics Education

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Introduction

Within natural resources agencies, there often is a debate over "resources versus recreation." Agencies that manage wildlife resources frequently depend on recreational license fees and must build a constituency of outdoor users. Yet sometimes the question is asked, "Do we really want *more* outdoor users?" Clearly, with the increase in population this is a moot question. There will be more and more outdoor users in the future. Rather than addressing whether or not we should be encouraging more people to hunt and fish, shouldn't we be concentrating on getting more people to be responsible users, to show more courtesy to each other and respect for the sport or activity, to realize how their lifestyles impact the natural world, and to better appreciate the needs of wildlife? In that way, we all work together *for* the resource. Kellert (1987: 18) pointed out that "a personally meaningful environmental ethic requires . . . affection for and identification with nature," and an ability to see "oneself as an integral . . . member of the ecological community." Fish and wildlife agencies have a unique opportunity to help their constituents develop that connection to nature and reinforce ethical behavior, countering the alienation and separateness that can lead to unethical behavior.

Outdoor ethics education should be playing a major role in this process. Many state and federal agencies, as well as wildlife and recreational advocacy groups, have recognized this and are attempting to address the issue. Are these attempts successful? Have efforts been made to evaluate their effectiveness?

To determine what state natural resource agencies were doing in one area, the National Wildlife Federation surveyed aquatic education coordinators to inquire as to how ethics were being addressed in angler education programs. Most respondents thought ethics were important and many were addressing them in a multitude of ways. But virtually no one knew if those methods were effective. Few had evaluated their programs. In many cases, the programs relied heavily on materials being distributed—workbooks, videos, stickers, rulers—many of which contained some code of behavior.

The National Wildlife Federation, with the help of a U. S. Fish and Wildlife Service Federal Aid grant, set out to determine effective ways of teaching outdoor ethics and how best to evaluate them.

What We've Found

From hunter educators to wilderness preservationists, the off-road industry to fisheries managers, the message from all sides is clear: we need to develop better outdoor ethics (Elliot 1992, Jackson and Norton 1979, Marshall 1993, Schmied 1993, Enck and Stedman 1992, Waterman and Waterman 1993). Most propose that education is the answer (Barker and Van Der Bie 1987, Eyman and Teeters 1987, Johnson 1987, Prosser 1987). However, the methods or strategies developed and used for outdoor ethics education all too frequently are based not on research evidence supporting their effectiveness, or even on critically accepted educational theory, but on what another program or agency is doing. Only in very few instances have attempts been made to evaluate outdoor ethics education efforts, and these have not supported the effectiveness of the ethics education approach used (Bromley et al. 1989, Jackson and Norton 1979). In this day of accountability, we must be able to show evidence that our outdoor ethics education efforts are effective and based on sound educational theory. When we do less, we do a disservice not only to those we are attempting to educate but to the resource, the outdoor activity and ourselves.

What Is It We're Really Trying to Do?

What are we really trying to do with outdoor ethics education? The desired outcome of most outdoor ethics education efforts is to influence behaviors. Whether concerned with reducing litter, eliminating user conflicts, cutting down on poaching violations, promoting catch-and-release or developing stronger environmental stewardship, the only worthwhile *evidence* of effective ethics education is through observable behavior. Influencing behavior, especially the long-term changes most outdoor ethics educators seek, is a tall order. The evidence from environmental education research generally does not support the intuitive links we often make between awareness/knowledge and doing/acting (Borden and Schettino 1979, Dwyer et al. 1993, Gigliotti 1992, Hungerford and Volk 1990, Marcinkowski 1989, McRea and Weaver 1984, Sia et al. 1985, 1986, Sivek 1989). Clearly, outdoor ethics educators need to base their methods on the best available models if they expect to be able to demonstrate results. The agenda for outdoor ethics education must include significant support for evaluating results.

Methods

Let's look at some of the methods being used, and then consider what the best evidence has to say about their potential for effectively changing behavior.

Public awareness campaigns and codes of ethics. Public awareness or promotional campaigns use catchy slogans, codes of behavior and techniques borrowed from Madison Avenue to promote better outdoor ethics. Two recent examples are the Hunter's Pledge, developed by the North American Hunting Club and the Izaak Walton League of America (Marshall 1993) and the ethical angler code developed by the Sport Fishing Institute's Future 21 program. The centerpiece for most of these campaigns, as well as for much outdoor ethics education, is a formal code of ethics. These codes often are developed by groups of concerned individuals, and have "catchy phrases . . . to capture the essence of each behavior promoted" (Schmied 1993). According to George LaPointe of the International Association of Fish and Wildlife Agencies, "the key to the code being successful is using and marketing it" (Marshall 1993). Some of the marketing techniques include bumper stickers, tackle box stickers, media public service announcements, comic books, codes on cards, videos, brochures and posters.

User education courses. Another technique used to promote outdoor ethics education is a voluntary or mandatory training course, such as hunter or sportsman education. Some of these courses rely on codes of ethics as well, typically introduced to the class by an instructor who is expected to be a role model and urge students toward the highest ethical standards.

Interactive techniques. There are a number of approaches that involve a more interactive methodology. These usually center around introducing a situation involving a situation involving an ethical dilemma. The students are asked to think critically, reason morally and choose the most ethical course of action. These interactive techniques include:

- *Dilemma discussions*, which present a situation that poses an ethical dilemma. The best of these involves more than one "right" choice, and walks the students through a judgment/decision-making model, helping the students identify choices, outline consequences and discuss the result (Martin 1985, Jackson no date b, no date c, Jackson et al. 1987).
- *Trigger films or slide shows* use audio-visual materials to set the stage for ethical decision making. As the moment of truth arrives, the projector is stopped and the discussion begins (Jackson no date a).
- *Interactive videos* use cutting-edge technology not only to set the stage for ethical decision making, but to follow through on the consequences. The video simulates the situation, the "player" chooses and the computer-operated control matches the decision with the appropriate consequence, with the player experiencing the result (National Wild Turkey Federation 1993, M. Stone personal communication: 1993).
- *Role playing/simulations* require group members to play different roles in ethical decision-making scenarios.

Outdoor heroes. One approach suggests using stories about actual people and the ethical choices they made to stimulate discussion (Knapp 1993, Kilpatrick 1993b).

Mentors. Some programs use a mentoring approach to guide youngsters and families as they encounter outdoor situations calling for ethical responses. Mentors recognize and reward ethical behavior. They guide their apprentices by observing both good and poor behaviors, and discussing the appropriate responses.

Community clubs. Another approach uses community clubs and organizations to assist youth with developing appropriate outdoor behavior. Peer counseling and teaching in these clubs is important to help participants develop ethical behavior.

Developing group and personal codes. Encouraging the development of a personal or community code of outdoor ethics differs significantly from imposing an externally derived code. This approach guides students, usually in groups and over an extended period of time, as they wrestle with ethical situations. This type of critical thinking, evaluating and reevaluating is done best in a community context.

What Does the Research Say About What Works?

Research into the effectiveness of what variously is called character education, moral education, values education or clarification, and even environmental values education offers some clear indication about what has not been demonstrated to be effective. The research is less revealing about what *is* effective; however, there are areas that show promise and are worth pursuing by outdoor ethics educators.

To summarize, we have known since the 1920s and '30s that traditional, authoritarian character education models—using codes of conduct and a heavily didactic approach involving a good deal of moralizing—can offer no evidence to support their effectiveness in changing behavior (Hartshorne and May 1928–1930, Leming 1993). In the 60-plus years since Hartshorne and May (1928–1930) conducted their pioneering studies into the effectiveness of character education, no one has been able to challenge their findings (Leming 1993).

In the 1970s, two schools of thought emerged from a resurgence of interest in moral or values education. Lawrence Kohlberg proposed a moral dilemma discussion approach in which students engage in a process of discussing dilemmas as a means of encouraging development toward higher stages of moral reasoning (Leming 1993). When Kohlberg's moral dilemma discussion approach was used over an extended period of time (at least a semester), researchers were able to demonstrate significant positive shifts in moral reasoning (Enright et al. 1983, Lawrence 1980, Leming 1981, Lockwood 1978, Schlaefli et al. 1985, Rest 1979).

The second school of thought was based on Sidney Simon's values clarification approach, in which students were supported by values-neutral instructors as they explored and affirmed their own values. Research done on values clarification has been consistently unable to find significant changes resulting from this approach (Leming 1981, 1993, Lockwood 1978).

It would appear that the moral dilemma discussion approach holds the most promise. Yet, we must note that while this approach showed positive results, the gains in moral reasoning were relatively small and took a significant period of time to achieve. Further, there is a difference between moral reasoning and behavior; and strong evidence supporting a link between the moral dilemma discussion approach and actual behavior has yet to be shown (Blasi 1980, Leming 1993).

What does this mean for outdoor ethics educators interested in changing behaviors? We know that research does not support the effectiveness of much of what goes on under the guise of outdoor ethics education. Public awareness campaigns may in fact raise awareness; but awareness is not a predictor of behavior (Borden and Schettino 1979, Dwyer et al. 1993, Gigliotti 1992, Hungerford and Volk 1990, Marcinkowski 1989, McRea and Weaver 1984, Sia et al. 1985–1986, Sivek 1989). Externally derived codes of ethics, the standard lecture method used in many hunter education courses, morality videos and teacher-directed approaches have not been shown to be effective in changing behavior (Hartshorne and May 1928–1930, Bromley et al. 1989, Jackson and Norton 1979), nor has the hero and morality story approach advocated by Kilpatrick (1993a, 1993b) (Leming 1993). The more interactive approaches involving dilemma discussions may hold promise, but have not yet been demonstrated to be effective in changing behavior, and these may take more time than is practical for most outdoor ethics education situations. Is there any hope at all?

Several years ago, when Jackson et al. (1987) searched the literature for guidance from moral educators as he and his colleagues worked to develop more effective hunter ethics education in Wisconsin, he found no consensus among moral educators. Today, this picture appears to be changing. In the emerging body of research on the new character education, there appears to be support for a number of strategies that are effectively demonstrating behavioral changes. These include cooperative learning strategies, community service and social action programs, just-communities, sex and drug education, and school climate research (Leming 1993).

There are some common threads weaving through most of these that have relevance to outdoor ethics educators. They are:

- the importance of community as the context for developing and nurturing pro-social behavior;
- teachers as guides, not authoritarian figures;
- a climate of mutual respect;
- group consensus building and ownership in group norms, including codes of moral behavior; and
- the importance of peer teaching, counseling and support.

To summarize, it appears from the research that the most prevalent outdoor ethics education efforts, the traditional hunter education programs and the public awareness campaigns, have the least support in the literature. This does not mean these approaches have no value in other areas, or even that they are ineffective in changing behavior. But the research that has been done, and some of it is quite extensive in contexts other than outdoor ethics, has failed to establish any causal relationships between these approaches and changes in ethical behavior.

There is some research-based support for the more interactive approaches to outdoor ethics education, especially those that are done over an extended period and involve building group consensus with relevant issues in a community setting. These approaches appear to hold the most promise for supporting ethical outdoor behavior.

One limiting factor is the inability of untrained volunteer instructors to use many of these techniques. Instructors need training and skill in leading these discussions and setting the stage for ethical reasoning (Beyer 1976, Jackson no date a, no date b, no date c, Jackson et al. 1987, Lickona 1983). Another obstacle involves the time needed to affect ethical behavioral changes. Another factor may be a lack of consensus about what in fact constitutes ethical outdoor behavior. These may be significant barriers, yet they must be addressed.

Setting the Agenda: Outdoor Ethics Education

As with the beginning of many social change efforts, outdoor ethics education has been based primarily on intuition and untested suppositions. It is time to move beyond this stage. We know what has not worked, and we know more about what does work every day. We cannot afford to have any sacred cows; the resource demands this much of us. In a society dominated by instant gratification and Madison Avenue-defined visions of the good life (subject to change, of course, at a moment's notice), we must resist the inclination to find a quick fix to a problem that requires far more time and effort to solve. There are no ethics pills.

Outdoor ethics education efforts must be based on sound research, or, in the absence of research, on the best evidence available. It is apparent that we will need to redefine how we approach the ethics education process. We will need to base these efforts in community contexts, involve far more social support, do them over an extended period of time and use well-trained instructors from within the communities we are targeting. And we must *evaluate*.

Clearly, we will need to be extremely creative in how we use our limited resources to achieve the needed changes. The conservation community has responded to these challenges in the past, and it must do so again.

If outdoor ethics education is important, even critical to the accomplishment of the mission of wildlife agencies and natural resource organizations, a major challenge confronts us. Are we willing to bring together the best minds, both within and outside of our organizations, and apply what we've learned to our individual outdoor ethics education efforts? Are we willing to reevaluate our outdoor ethics education programs and get rid of what does not work? Are we willing to: (1) explore the importance and meaning of basing our outdoor ethics efforts in a relevant, community context? (2) use community, family, group and peer influences to provide the social support needed to accomplish outdoor ethics education? (3) really train our instructors to accomplish real ethics education? (4) invest the time needed, through long-term, sustained effort, to make a difference in outdoor behavior? and (5) invest the time, effort and financial resources needed to evaluate our efforts, and make the needed changes?

The agenda is before us. There are far more questions than answers; yet we know where we should look for the answers. We also know what isn't working. The authors plan to continue their research, and are actively seeking programs, particularly in angler education, that would like to serve as models for evaluation purposes. Together we can hope to make a difference. We must get on with it.

Acknowledgments

The authors gratefully acknowledge the assistance Jody Enck, of the Cornell University Department of Natural Resources Human Dimensions Unit, for his comments on earlier drafts of this paper.

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Turn on the television, pick up the outdoor section of a newspaper, or talk to a stranger about hunting fishing and other outdoor pursuits, and you'll gain insight about the many images that combine to form the public's perception of outdoor recreation and outdoor recreationists. You likely will learn that even if they have never shouldered a shotgun, a fishing pole or a backpack, many individuals have a strong interest in—and opinions about—outdoor recreation and its impacts on wildlife, natural resources and society.

From newspaper reports about hunting, to television commercials advertising off-highway vehicles, to magazine features focusing on the risks and rewards of rock climbing, outdoor activities now are in the public eye. Although this increased exposure creates new problems, it also provides new opportunities to educate people about responsible outdoor recreation.

At a time when the public derives much of its information from sound bites and new briefs, first impressions count when conveying messages about outdoor recreation. The outdoor community needs to be aware that as little as 30 seconds of television time can have a dramatic impact, not only on recreationists' image but also on their sports and the wildlife and natural resources on which they depend. It also is important for the outdoor community to recognize that these media sources sometimes are the sole providers of outdoor recreation information for non-outdoor recreationists.

Members of the outdoor community already use the media and advertisers to help promote responsible outdoor recreation. For example, under the Tread Lightly! program—a nonprofit effort that seeks to reduce the environmental impacts of off-highway vehicles (OHVs)—conservationists have prompted several OHV manufacturers to show responsible use of these vehicles in their advertisements. Similarly, campaigns to promote improved hunter behavior have been broadened to focus not only on hunters, but also to inform the public that hunters, like the public, will not tolerate illegal and unethical behavior within their own ranks.

Still more can be done. While working with media representatives and advertisers, the outdoor community may need to reconsider traditional approaches to promoting safe, responsible recreation. Largely urban and more culturally diverse than ever, today's public has social expectations about outdoor use that may challenge traditional outdoor recreationists' perceptions of themselves. For example, hunters' and anglers' reliance on the heritage and tradition of outdoor ethics in their activities may be ineffective, since these themes may not resonate with people who have no current, direct connection with wildlife and natural resources. Such individuals may not inherit traditions, passed down from generation to generation, that emphasize outdoor ethics. Creating the perception and the reality of outdoor recreationists as responsible stewards of wildlife and natural resources calls for a close examination of the future—not just the past—for these activities.

Setting the Stage

The public's perception of outdoor recreation is shaped by many different influences. Two of the most powerful forces are news media coverage and advertising. The following describes how both can have positive and negative effects in enhancing public awareness about responsible outdoor recreation.

News Media

As America becomes more interested in outdoor recreation, news stories, features and photographs about recreation topics are appearing more regularly in the main news sections of magazines, newspapers, and television and radio programs. This progression provides stories about outdoor recreation with a wider, more general audience. In some ways, this move may be beneficial, because it fosters greater public awareness of outdoor activities. Increased general coverage also provides opportunities for health public debate about controversial outdoor recreation issues.

However, general assignment reporters and editors who do not cover outdoor issues regularly may not fully understand outdoor recreation topics, and inaccurate reporting and/or editing may result in misinformation. For example, reporters often do not use the term "poacher" or "wildlife law violator" to describe someone who has committed a wildlife-related crime. Instead, the term "hunter" is used, which may leave the public with the impression that all hunters regularly commit these crimes or that these criminals are regarded as "hunters"—not lawbreakers—by the rest of the hunting population.

In addition, the media often are not specific when referring to hunters, shooters and other firearms users. For example, although many sportsmen and sportswomen do not consider events such as pigeon and prairie dog shoots "hunting," the media usually does little to explain the distinction. Animal rights activists capitalize on this issue. According to Wayne Pacelle (personal communication: 1991), The Fund for Animals' executive director, "The public doesn't make such fine distinctions between hunting and shooting events as do hunters and animal rights activists. This has been an effective tactic for us."

Story and/or photograph placement and emphasis also are factors that influence public perception. Coverage of hunting in particular is scrutinized closely and raises important questions about the ethics of the media, as well as the outdoor ethics of sportsmen and sportswomen. In addition, analysis of news coverage of hunting issues may address concerns that are common to other types of outdoor recreation.

A recent example from Vermont helps illustrate these issues. On October 19, 1993, the state held its first moose hunt in 100 years, and the event sparked controversy about not only the hunt, but about media emphasis on the hunt. The *Burlington* (Vermont) *Free Press* covered the hunt extensively, publishing front-page and other articles before and after the event. One of the first articles contained both pro and con arguments about the hunt. Supporters stated the hunt was necessary to control the moose population, which has caused traffic accidents and property damage. Opponents said shooting moose was unnecessary and unsporting, claiming it is "no more difficult than shooting a parked car" (Pyne 1993a). The paper also ran its own editorial opposing the hunt (*Burlington Free Press* 1993) and a story about media coverage of the hunt (Liley 1993).

In addition, an Associated Press photo of a dead bull moose appeared in the October 20, 1993, national section of the *New York Times* and in other papers in the state. The ***Burlington Free Press*** received 14 complaints about the photo, which was run on the front page of the paper. More arresting images appeared on television when Vermont television station WCAX aired video footage of a hunter harvesting a moose during the hunt. The video was shown twice on WCAX's October 19, 1993, news program, and the station received 50 telephone calls, mostly critical, about the footage (Liley 1993).

The follow-up *Free Press* story gave an account from the hunter featured in the WCAX videotape. The hunter said he killed the moose at point-blank range, and as it continued to kick after its death, the WCAX news van appeared and started filming the incident. "They didn't say anything. They just scrambled to put the camera together," the hunter recalled. "I got so mad, I didn't know what to do. They totally invaded my privacy. Even if I'd wanted to shoot (the moose) again, I'd look like a blood-thirsty maniac" (Pyne 1993b).

Incidents such as the one described above—showing hunters behaving in a seemingly unethical manner—have made members of the hunting community understandably wary of general media coverage. In addition, they raise serious questions about the media's role in covering such events, questions that need to be addressed not only by the hunting community but also by the media. This example illustrates a clear need for immediate dialogue and understanding between the two communities.

There are two sides to the news coverage coin, however, and it is important to note that the press has not ignored good news about hunting. For example, in June 1993, the Izaak Walton League of America, with assistance from eight other organizations, released a nine-point Hunter's Pledge. Hunters who take the pledge promise to respect the environment and wildlife, property and landowners; show consideration for nonhunters; hunt safely; know and obey the law; support wildlife and habitat conservation; pass on an ethical hunting tradition; strive to improve their outdoor skills and understanding of wildlife; and hunt only with ethical hunters.

The pledge, created as a noncopyrighted, nonproprietary work, is designed to be adopted or adapted by hunting and conservation organizations, hunt clubs, federal and state agencies, and manufacturers of hunting-related products. In addition to teaching and reinforcing legal, ethical behavior by hunters, the pledge also serves to inform the public that hunters strive to behave legally and ethically and are mindful of their public image.

Media response to the pledge has been an overwhelming success. Major newspapers, news wire services, magazines and syndicated radio programs all have run stories about the pledge. As of February 1994, the League had received more than 200 news clips of newspaper and magazine stories that mention or feature the pledge.

Other positive images of hunting come to light when reporters focus on "hunt for the hungry" programs, which provide game meat for people in need, and other socially beneficial activities, such as sight-in days, hunter education programs for youth, and wildlife and natural resource education efforts.

Interestingly, media coverage of wildlife and natural resource law enforcement activities sparks dispute in the hunting community about whether such public attention is beneficial or detrimental to the sport. Some members of the hunting community believe media coverage of activities such as decoy deer operations, undercover ''stings'' and the apprehension of wildlife law violators gives additional weight to the anti-hunting community's arguments that hunters are unethical, careless, law-breaking individuals.

However, the Izaak Walton League and other members of the hunting community believe that when appropriate terms—''poacher'' or ''wildlife law violator,'' instead of ''hunter''—are used in describing individuals involved in these incidents, such news stories may have a positive impact on both hunter behavior and the public image of hunters. These stories can help make the public aware that hunters will not accept criminal or unethical activity any more than society at large accepts such behavior.

In addition to peer pressure and civic awareness programs, the possible deterrent effect of publicity about wildlife law violators ranked high in a recent survey of state fish and wildlife agency and hunting and conservation organization representatives. A respondent from a largely rural southern state noted that media coverage about violators is an effective deterrent because ''they know they are not in good standing in the community'' (Ruh 1994: 19).

Advertising

Many advertisements for products associated with outdoor recreation are designed to make viewers, readers and listeners want to jump into an OHV or onto a Jet Ski, or run to the store and buy the latest in fishing poles and backpacks. Although such advertisements can help generate greater public enthusiasm about, and possibly participation in, outdoor activities, they also can send negative messages about outdoor behavior. For example, advertisements that show unsafe shooting practices, children boating without life jackets or OHVs driving through streams promote an image that such activities are acceptable.

Publications' editors often serve as gatekeepers who determine which advertisements are acceptable, but in some cases, ads showing outdoor activities are not always examined to determine if they promote responsible behavior.

''We really just look to see if things are realistic, if they are tasteful and technically correct,'' said Peter Barrett, associate publisher of *The Fisherman* magazine. *The Fisherman* ''has never encouraged or accepted ads that encourage the killing of fish,'' he said, and the publication works closely with advertisers to create ads that promote responsible use (personal communication: 1993).

According to Jay Perkins (personal communication: 1993), *Boating World* magazine's advertising sales manager, the final decision about an ad rests with the publisher. Although the magazine is ''very adamant'' about safe boating, ''we probably would accept an advertisement with a kid without a PFD [personal flotation device]. We can't tell the manufacturer how to promote the product.'' he said.

With creative work often done by outside advertising agencies and with magazines hungry for advertising dollars, promoting responsible outdoor behavior in ads is not always a priority. Sometimes, subtle but important problems get overlooked. For example, a full-page, full-color ad for DuPont's line of outdoor clothing fabrics shows a campsite established next to a waterbody and an individual preparing to cook over an open fire (*Outside Magazine* September 1992: 17). At first glance, it's a typically pastoral scene, until a closer look reveals that the tent does not appear to fall within the 200-foot perimeter recommended to reduce camping impacts on surface waters.

In addition, a camp stove could have made an easy, low-impact alternative to the open fire, which could unnecessarily scar the land and start forest fires if not properly extinguished.

Although the company is "extremely conscientious as a corporation about the need to protect the environment," Du Pont marketing and communications specialist Liz Angstadt noted that the ad reflects "honest oversights due to a lack of knowledge" (personal communication: 1993).

Another more problematic type of advertising has appeared as the public's increased interest in the environment prompts advertisers to capitalize on the promotion of wildlife and natural resource protection. For example, a two-page, full-color ad for the Chevy S–10 truck promotes wilderness and urges recreationists to "leave it as you found it," while showing a close-up of a truck that appears to have been driven off-road. (*Field and Stream*, March 1993: 28–29) By law, motorized vehicle use is specifically prohibited in areas designated as wilderness.

Jerry Van Noord, who handles truck advertising for Chevrolet, said he received no negative input about the ad. Photographs for such ads always are shot in areas designated for OHV use or on private land that is owned or leased by the company, he said. In addition, he noted that ads are reviewed by the company's legal staff. "The trucks are used off-road; we show them off-road," he said. "I guess we can't keep everybody happy. We walk a very fine line sometimes with what some people think is right or wrong" (personal communication: 1993).

For one OHV manufacturer, Land Rover North America, Inc., walking that fine line meant making major changes in its ads. In 1989, the Izaak Walton League's *Outdoor Ethics* newsletter used the company's Range Rover "We brake for fish" ad to illustrate a front-page story about unethical ads (Merriman 1993). The *Wall Street Journal* picked up on the issue, mentioning the *Outdoor Ethics* article, Land Rover and other OHV manufacturers in its "Marketing and Media" column in March 1989 (Lipman 1989).

The attention spurred Land Rover to revamp its ads, and company guidelines now dictate that only limited retouching can be done to ads; ads must avoid showing vehicles in meadows, new forests, streambanks and lakeshores; vehicles no longer will be shown in any way that gives the impression of potential environmental damage; and inappropriate language (such as "wilderness") cannot be used.

In addition, all ads undergo a review process that includes the company's marketing and legal departments, and even the company president (B. Baker personal communication: 1993).

Greater emphasis on recreationists' responsibility and environmental awareness also is evident in advertisements for other products. Responsibility for promoting safety has struck a chord with Remington, which placed three paid ads promoting gun safety in 12 publications in 1992 (*North American Hunter*, August 1992: 70, B. Wohl personal communication: 1993). Berkley, the fishing equipment manufacturer, urges anglers to recycle their fishing line and has used paid ad space to promote catch-and-release fishing (L. Rubis personal communication: 1993).

There are other encouraging signs, such as the Times Mirror Magazines Partnership for Environmental Education, established "to do something to encourage responsible behavior and to help develop environmental leaders," according to Times Mirror Public Relations Director Linda Boff. Under the program, Times Mirror contributes to environmental programs 2.5 percent of net revenue from ads in its magazines when those ads include environmental messages. To qualify, each ad must include a public service environmental message of at least 15 words in a type size equal to or larger than the type size used for product description (personal communication: 1993).

Other help for advertisers comes in the form of a guide published by Tread Lightly! Inc. The guide urges OHV advertisers to avoid showing the following: excess wheel spin, vehicles being driven on wet roads or in streams, and vehicles being driven off designated roads or trails. The guide also warns advertisers against the use of the words "off road" and "wilderness" in ads (Tread Lightly 1992).

With a green image seen as an increasingly powerful marketing tool, outdoor product manufacturers may do well to listen to conservationists and consumers. "Overall, the closer management is to the end user, the more ethical their advertising," according to David Secunda, executive director of the Outdoor Recreation Coalition of America, which represents 350 outdoor product companies (personal communication: 1993).

Linda Rubis, advertising services administrator for Berkley, agrees, "Our business is the outdoors. If we don't protect it, we won't be here" (personal communication: 1993).

Finding Solutions

Whether they realize it or not, members of the outdoor community have many tools at their fingertips to promote responsible outdoor recreation to the public. News media outlets, heightened public interest in wildlife and natural resource protection, and greater industry-wide consciousness of outdoor issues provide an expansive array of opportunities.

These new opportunities also provide incentive for members of the outdoor community to reach out beyond more traditional approaches to public education about outdoor recreation. For example, programs that seek to instill an outdoor ethic in individuals who have little or no contact with the natural world may need to use images, spokespeople and messages that are different from those that are used to promote responsible behavior to people who are familiar with the outdoors. Promoting angling ethics in an inner-city environment may require role models and educational methods that are different from those that would be used to promote angling ethics in a small rural town.

In addition, continued dialogue with media, advertisers and industry representatives will help ensure that accurate, appropriate messages about responsible outdoor use reach recreationists and the public. For several years, the Izaak Walton League of America has focused on these needs by promoting an outdoor ethics writing contest among members of the Outdoor Writers Association of America. In addition, the League this year is proposing creation of a task force that would meet with advertising agencies who handle outdoor product accounts. The task force would provide general information and possibly consulting and review services for outdoor product ads.

The League also recognizes that any efforts to improve outdoor recreationists' image and behavior must include *all* members of the outdoor recreation community: academics, outdoor product manufacturers, conservation and recreation organizations, federal, state and local wildlife and natural resource agency representatives, and all others who have a stake in promoting safe, responsible outdoor recreation.

Federal and state agencies in particular are playing an active role in promoting responsible outdoor recreation outside the realms of news coverage and advertising. For example, each year the Izaak Walton League of America sponsors an Association for Conservation Information contest that awards prizes to state agencies that produce public information materials promoting outdoor ethics.

In addition, the U. S. Fish and Wildlife Service (USFWS) recently released a television public service announcement (PSA) that promotes hunting safety. The PSA has been issued in 12 states—Colorado, Kansas, Idado, Maine, Minnesota, New Hampshire, Nevada, Oklahoma, South Carolina, Texas, Virginia and Wyoming—and 193 of the 252 television stations (80 percent) that received the PSA have aired it. The PSA has reached approximately 20 million households and garnered more than $700,000 in free air time (USFWS PSA Fact Sheet 1993).

The project marked the culmination of extensive work to determine the public's reaction to a PSA about hunting. The research found that after being shown five possible screen treatments, 212 public service directors from television stations in the 12 pilot states stressed family and multi-cultural involvement, sportsmanship and knowing the law as being of primary importance as themes in the PSA (Professional Media Services 1993).

But with all the public relations tools available—PSAs, favorable news coverage and advertisements depicting responsible use—it may be easy to overlook the most basic and powerful instrument that can instantly improve the image of outdoor recreationists. Hunters, anglers, backpackers, hikers, kayakers and other outdoor users who set an example of ethical outdoor behavior benefit their sport's image more than anything else. In this respect, responsible behavior is its own reward.

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Who Is the Ethical Sportsperson of Today and Tomorrow?

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Introduction

The science of resource management has progressed by leaps and bounds over the past few decades. No longer are living resources viewed strictly as isolated commodities to be managed solely to maximize economic return. The direction of change in modern management is toward holism; this new direction is reflected in the shifting perspective of resource managers from commodity to community. Modern fish and wildlife managers dealing with natural systems no longer focus exclusively on maximizing populations of one or a few harvestable game species. Rather, protecting species viability and promoting long-term ecosystem integrity have emerged as legitimate and important objectives of management programs.

Most of us consider the emergence of enlightened scientific management a healthy and welcome advance in this field. At the same time, we acknowledge that science alone, no matter how enlightened and holistic, cannot solve all the problems with which managers are confronted. Even if scientific studies ultimately disclose the management options available to achieve holistic goals, they cannot tell us whether these goals alone are sufficient, for our choice of goals always reflects our values, and values are not derived solely from data. Wildlife and fisheries managers influence and are influenced by the values of hunters and anglers, and those values increasingly are being called into question by some within, as well as many outside, the hunting and angling communities.

It is incumbent today upon hunters, anglers and resource managers not only to utilize modern resource management science but also to carefully consider the ethical implications of their activities. Though slow to respond at first, wildlife professionals and sportspersons are to be commended for their recent efforts to effectively and proactively address these issues. As a result of their efforts, we finally are shaking the legacy of Descartes, the 17th Century thinker known as the father of philosophy and the preeminent representative of his era's new scientific spirit. That legacy underlies the characterization of managers and sportspersons as cold, detached rationalists, immune and unresponsive to the suffering of the animals they kill or wound. Descartes' mechanistic attitude led him to assert that to nail a dog to a table and dissect it alive in order to study the circulation of the blood is no different than dismantling a clock, and he insisted that any commiseration with the "victim" is misguided sentiment. Since, in Descartes' view, animals do not belong to our moral community, he concluded that they cannot be worthy of our moral consideration (Marshall 1992).

The Cartesian view of animals as sophisticated machines has been replaced by one that recognizes the evolutionary link between humans and non-human animals and that acknowledges, with regard to the higher animals, the important relevant similarities between us and them. In light of these similarities, we cannot categorically deny that animals, at least the sentient ones, are worthy of our moral consideration, and we are beginning to take seriously the legitimate concerns of animal activists, anti-hunters and the non-hunting public. Substantial progress has been made recently in identifying and clarifying the moral dilemmas intrinsic to hunting, fishing and related management. In short, we are well on the way to achieving the first step, acknowledging awareness and showing concern, concern over the sustainability of the resources and, yes, concern that some hunting, fishing and management practices may be unethical.

The Psychological Roadblock

Despite this growing openness and concern, we seem to have come to a psychological roadblock on the path from awareness to action. Only reactionary extremists deny that ethical hunting and angling are desirable. However, even while conceding the importance of addressing ethics in their sport, many sportspersons get plainly uncomfortable and some even hostile when the discussion turns to how we should decide what does or does not constitute ethically responsible behavior. This question of who decides and how the decisions are made has emerged as a sticking point; few among us presume to be in a position to pass absolute judgment on others, yet it seems we each inwardly fear failing the test someone else may propose. This is not surprising. Our images of ourselves as hunters or anglers may literally be integral to our sense of personal identity, as evidenced by the slogan I recently saw on a T-shirt: "I fish, therefore I am." If we are what we do and if what we do reflects our values, then our reticence to accept some prescribed program of ethical hunting or fishing is understandable. None of us wishes to be condemned as unethical because some activity we enjoy and consider to be wholesome and ethical suddenly is proscribed by the "moral authorities."

Must we each then refuse to make moral judgments on particular acts of hunting and fishing in order to avoid such scrutiny of our own behavior? Not at all. I have attended numerous conferences and meetings where I met many individuals for whom I have utmost respect and confidence as ethical hunters or anglers, yet who would each sanction at least one form of hunting or fishing that another could not personally sanction, and vice versa. The logical conclusion I draw is not that one or more of them must be mistaken or unethical, but that ethical hunting and fishing have a lot more to do with attitude, intention and deliberation than with adherence to specific codes of conduct or do/don't lists.

Autonomy is necessary for moral maturity. Autonomous thinkers listen carefully and consider the opinions of others, yet ultimately make and accept responsibility for their own moral decisions. Thus, it is possible for two anglers to consider a question such as "Is catch-and-release fishing ethical?" and come to opposite conclusions without it being the case that one of them is necessarily wrong or morally immature. To become an ethical "___-er" is to take seriously the moral implications of "___-ing," not necessarily to agree with someone else's moral perspective or position.

In order to get past this psychological roadblock, the fear of moral judgment and its potential personal implications, and to advance in our sincere efforts to promote ethical hunting and fishing, we must address and attempt to answer the question that has recently come to haunt us: Who is the ethical sportsperson?

Put simply, I believe that the ethical sportsperson is the virtuous sportsperson. By virtuous I do not mean merely assenting to principles of compassion, justice and fairplay. Even if we could agree on a set of rules and regulations prescribing every aspect of ethically responsible hunting, angling and management, and even if we could somehow guarantee full compliance, we would not succeed in our mission. There are two reasons why this is so.

First, many of the rules and regulations we agree on, such as those intended to prevent poaching, are at least in part motivated by self-interest. A hunter who turns in a poacher is not necessarily demonstrating moral courage, but may in fact be reacting to a sense that the poacher violated his "property rights" (the property being the poached animal). Following rules and regulations and assisting in their enforcement require little moral courage; far more courageous is the hunter willing to condemn the routine moral offenses, not covered by law, to which we frequently turn a blind (though perhaps discomfited) eye. Many times it is the attitude and intent of the sportsperson, not his or her particular act, that are morally questionable, and attitudes and intentions are not subject to regulation through laws and conduct codes.

Second, efforts to prescribe and enforce ethical behavior miss the point. What we should seek to foster is not just assent, but internalization of these principles by individuals who thus come to exemplify the corresponding virtues. Laws that prohibit racial or sexual discrimination may deter individuals who fear punishment from overtly discriminating, but we will not abolish racial and sexual prejudice unless individuals internalize the values embodied in the legislation. Similarly, rules and codes may dampen the allure and discourage the display of unethical hunting and fishing, but they alone will do little to encourage the development of those values that guide the ethical sportsperson.

What I advocate is a shift of focus. Traditional ethical deliberation focuses on the acts of the agent, but as I have attempted to demonstrate, this approach leaves us short of our goal. By shifting the focus from the particular acts of the sportsperson to his or her nature or character, we are presented with a new set of questions relevant to the moral dilemmas in hunting and fishing. Whereas such old questions related to acts such as "Is it proper to bait bears or hunt deer with dogs?" and "Are trophy hunting and angling ethically defensible?" have come to seem intractable, such new questions as "What personal qualities would the ethical sportsperson possess?" and "What sort of person would kill wantonly, cruelly or for trivial purposes?" invite answers. These are the sorts of questions asked at the start when ordinary people think about the morality of hunting and fishing. While this may be a new approach for sportspersons and resource managers, it has potential to open up new areas of debate and discussion and lead us around the psychological barriers currently blocking the path to effective action.

The Role of Tradition

In order to identify the ethical or virtuous sportsperson, we must not appeal only to standards of ethical behavior but also to a generally recognized standard of moral character. Virtues are understood best as part of a tradition in which we inherit them and through which we understand and evaluate them (MacIntryre 1984). Though moral standards within our society change over time, such standards nevertheless often are long-lasting and become traditions, a term that carriers honorific connotations. However, not all traditions begin honorably or remain so, as the tradition of slavery clearly demonstrates; moral traditions are subject to decay and disintegration. Traditions change in the context of a changing society, and outdated ones are replaced with those more in keeping with current values and priorities. Thus, the argument that certain forms of hunting and fishing constitute cultural traditions worth preserving is sometimes shaky, as few today would argue that cockfighting, though once established as a cultural tradition, should be revived.

Even the current language of wildlife and fisheries managers sometimes reflects a tradition of trivialization. We refer to hunters and anglers as sportspersons and to their activities as sport, yet ''sport'' implies a contest between two or more willing and more or less evenly matched contestants. Sportspersons pursue game, yet ''game'' implies a trivial pursuit intended to amuse its participants and not to be taken seriously. How often have we reminded the over-enthusiastic card player that it's just a game? I am not suggesting that we should necessarily avoid or replace these terms, but rather that we examine and replace the outdated traditions and attitudes they imply. What we are trying to do in promoting ethical hunting and angling is no less than to create a new tradition in which not only the intellectual and physical virtues but also the moral virtues will be admired and nurtured.

Who Is the Ethical Sportsperson?

By shifting the focus from particular acts of the hunter or angler and focusing instead on the character traits of the virtuous sportsperson, we protect and promote moral autonomy and individual responsibility. We also foster new habits of thought and action in the individual—not just to get the individual to make the right decision about particular acts, but to reorient the whole way of thinking (Frasz 1993). We are seeking to nurture the development of a new world view as well as to ensure that the sportsperson acts ethically. In order to foster this development, we must take the lead and set the example for attitudinal changes. These come from within; they are not externally imposed.

No doubt, many will be uncomfortable at first with the idea of focusing moral scrutiny on the sportsperson's character, for some assume that making moral judgments is tantamount to labeling others good or bad. The point, however, is not to skirt the issue of ethical hunting and fishing with personal attacks or *ad hominems*, but simply to raise a different moral question. In many cases, we find no solid ground for determining that particular acts of hunting and fishing are wrong, yet we may still find that a willingness to engage in them reflects the absence of certain character traits that we regard as morally important and wish to promote (Hill 1983).

Aristotle defines the virtuous person as one who strives for excellence, and he defines excellence as a state of character concerned with our choices (McKeon 1941). That is, excellence is a habit, and habits issue from character. Of course it is easy to see how a basketball player or pianist reaches excellence in his or her craft through practice and the cultivation of good habits. What is less clear is how we might apply this to identifying the excellent sportsperson. Yet every human activity, from the practical to the moral, admits of excellences and deficiencies. Each has its own purpose and there are corresponding excellences, developed through training and practice, appropriate to each of them.

What are the excellences or virtues of the ethical sportsperson? The characterization of the virtuous sportsperson must be developed in the context of our modern and diverse society, yet this is not as subjective and difficult a task as it may seem. Just as we acknowledge that the responsible sportsperson or resource manager has an obligation to act to protect species, habitats and ecosystem integrity for the enjoyment and well-being of other humans, so we can identify and gradually expand a set of commonly agreed-upon minimum moral obligations that extend beyond the human community and that the ethically responsible sportsperson or manager will necessarily embrace.

At a minimum, the ethical manager or sportsperson acknowledges responsibilities not only to the hunt, the hunter and the non-hunter, but to the hunted as well, and has the strength of character to act accordingly. These responsibilities impose obligations to be mindful, compassionate and humble. Mindfulness requires sportspersons to take into consideration the interests and values of those who do not share their love of hunting and fishing, though not necessarily to adopt their values. It requires them to refuse to be a slave to passions or urges of the moment and to think carefully about what they do and potential effects. Mindful sportspersons will not hunt or fish unless they are confident that they understand the game they seek and its habitat, their marksmanship skills and equipment are excellent and up-to-date, and their knowledge of all applicable regulations is thorough; that is, until they are demonstrably competent. Mindful resource managers and agency professionals might urge that a test of such competence be required before licenses are issued, since in many states today, the hunting and fishing license is little more than a tax stamp.

Compassion compels the ethical sportsperson to avoid inflicting any and all unnecessary pain and suffering on the animals they kill or catch. When in doubt as to the level of sentience of one's quarry (e.g., fish), the compassionate sportsperson would give the benefit of the doubt to the animal until research or compelling anecdotal evidence settles the issue. The compassionate resource manager takes into consideration, to the extent practicable, the interests of the individuals affected by management aimed at protecting or enhancing populations, species or ecosystems. He or she makes management decisions and chooses implements and techniques with a concern for minimizing pain and suffering. Indifference to suffering is antiethical to compassion and reverence for life.

The humble sportsperson has a solid understanding of nature and an appreciation and awareness of one's place in it. Such a person could not help being indignant about the wanton destruction of nature or about wanton, frivolous killing. To feel otherwise would be to take the arrogant attitude that only the human is important. Appropriate humility requires resource managers to view natural systems and their animal components as valuable in themselves, not just as means to satisfy short-term human interests. Such an attitude might force us to revise drastically our concept of "quality management," too often a euphemism for designing trophy factories. It might also compel us to end contest killing and trophy competitions, as we would seek to promote hunting and angling which feed the mind, spirit and body, not the ego. Humility teaches us that we cannot have an ethical relationship with anything—

fellow humans, wildlife or the land—without feeling a deep sense of respect for the subject of the relationship.

Finally, ethical sportspersons or managers actively cultivate the intellectual virtues of open-mindedness and critical thinking. They recognize that ethical dilemmas are presented by their sport or profession, they agree to discuss openly and deliberate these dilemmas, and they enter such discussions and deliberations with a clear, open mind subject to change.

Must the ethical sportsperson, in pursuit of virtue and excellence, then abandon the pursuit of pleasure in his or her sport? Absolutely not. The ancient Greeks, originators of virtue ethics, believed that sublime pleasure not only is a good in itself and the goal of all appropriate action, but is the natural accompaniment to the practice of virtue. I do not know anyone who has taken the moral dimensions of hunting and fishing seriously and, as a result, lost interest or pleasure in their sport; usually, pleasure and satisfaction are increased as a result of their personal growth.

Conclusion

The kind of guidance to ethical sportspersons provided by the character virtues of mindfulness, compassion and humility is admittedly vague; we are not told precisely how to act, but rather advised on how to think. Because this ethic must be applied in a pluralistic society in which there is no broad consensus on how we should treat nature and non-human animals, we will find disagreement among demonstrably ethical sportspersons as to the moral acceptability of particular forms of hunting, angling and resource management. But this is to be expected where moral autonomy and individual responsibility are emphasized. We do not seek uniformity of behavior or thought, but rather convergence of values and goals. Ideally, each hunter or angler will define and develop his or her own ethics with regard to these activities; each will cultivate those character traits and habits he or she believes best exemplify the virtuous sportsperson. Moreover, by abandoning the laissez faire ethics that assume prevailing moral standards are okay and by creating new moral traditions sustained by the sportspersons who practice them, we will move away from our present deadlock of impotent concern and personal confusion and skepticism and begin to act in accordance with our individual and collective sense of responsibility.

One need not fear the moral judgment of others if one has taken seriously and carefully considered the ethical implications of one's activities. Of course, we must be wary of applying ethical analysis too liberally, that is, of focusing the moral light on every little aspect of hunting or fishing instead of on only the crucial issues. Otherwise, we run the risk of being seen as clerical scolds or, worse, irrelevant (B. Gibbons personal communication: 1993). It remains for the hunter, angler, resource manager and agency professional to fully explicate the nature of the ethical sportsperson in the context of our modern and diverse society, and thus to show what its full realization in the virtues would look like. The participation of agency directors and professionals is badly needed, since the new traditions of ethical hunting and angling will grow only with the leadership and support of local, state and federal resource agencies. It is time for all of us to acknowledge our responsibilities and obligations both to the hunt and to the hunted. Embracing them will demonstrate not just what we do, but who we are.

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