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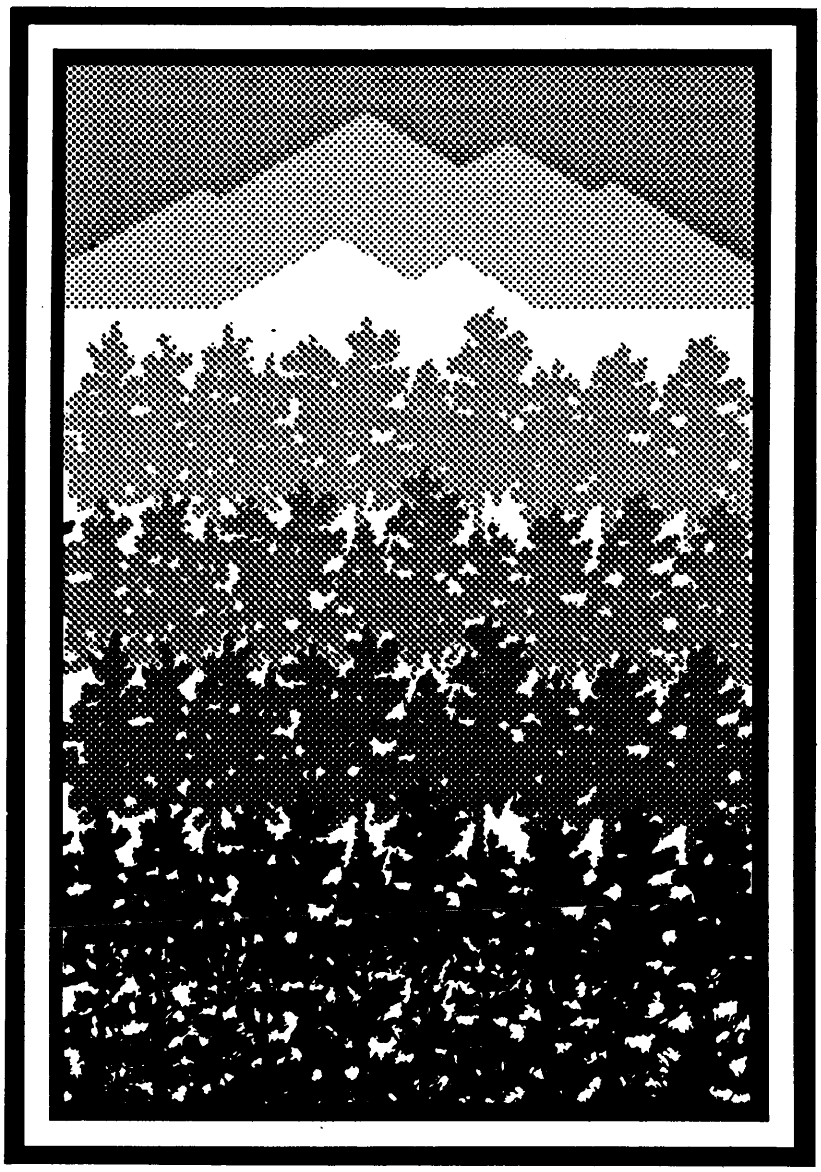
Report RM-247**

Sustainable Ecological Systems:

Implementing an Ecological Approach to Land Management

July 12-15, 1993
Flagstaff, Arizona



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Abstract

This conference brought together scientiests and managers from federal, state, and local agencies, along with private-sector interests, to examine key concepts involving sustainable ecological systems, and ways in which to apply these concepts to ecosystem management. Session topics were: ecological consequenses of land and water use changes, biology of rare and declining species and habitats, conservation biology and restoration ecology, developing and applying ecological theory to management of ecological systems, sustainable ecosystems and forest health, and sustainable ecosystems to respond to human needs. A plenary session established the philosophical and historial contexts for ecosystem management.

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Sustainable Ecological Systems:
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Technical Coordinators:

W. Wallace Covington
Northern Arizona University

Leonard F. DeBano
USDA Forest Service
Rocky Mountain Forest and Range Experiment Station

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Experiment Station
U.S. Department of Agriculture
Fort Collins, Colorado

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Preface

The purpose of this conference was to bring together scientists and managers from federal, state, and local agencies, along with private sector interests, to examine key concepts regarding sustainable ecological systems, and ways in which to apply these concepts to ecosystem management. In organizing the conference, the planning committee relied heavily on three documents. The first was the June 4, 1992, statement by Forest Service Chief, F. Dale Robertson, regarding ecosystem management. Second, was the March 8, 1993, statement of ecosystem management principles written by the Acting Director for Ecosystem Management for the Forest Service, Ann M. Bartuska. The final document was a report produced by the Ecological Society of America's Committee for a Research Agenda for the 1990's (Lubchenco et al. 1991) entitled, "The sustainable biosphere initiative: An ecological research agenda", published in the society's journal (Ecology 72:371-412).

The conference consisted of a plenary session, six concurrent sessions, three field trips, and a poster session. All papers presented in the plenary session established a philosophical and historical context for ecosystem management. This was followed by a hosted lunch and poster session featuring approximately 30 posters on a broad range of ecosystem management issues. In the afternoon two concurrent sessions where held, on "The Ecological Consequences of Land and Water Use Changes" and the other on "The Biology of Rare and Declining Species and Habitats." The second day was devoted to the three concurrent field trips: "The Biology of Rare and Declining Species," "The Ecological Consequences of Land and Water Use Change," and "Conservation Biology and Restoration Ecology." Conference evaluation forms showed that field trip participants found the trips very rewarding, not only for their content but also for the open discussion of ecosystem management concepts in the field.

The last day of the conference was dedicated to two concurrent sessions in the morning and two more in the afternoon. The morning sessions were "Conservation Biology and Restoration Ecology" and "Developing and Applying Ecological Theory to the Management of Ecological Systems." The afternoon sessions were "Sustainable Ecosystems and Forest Health" and "Sustainable Ecosystems to Respond to Human Needs." A banquet was held that evening followed by a presentation by Tom Bonnicksen on a biosocial systems perspective on ecosystem management.

ACKNOWLEDGMENTS

The conference coordinators thank the many individuals who contributed to the success of this conference. The organizing committee consisted of Gerald Gottfried, Bob Hamre, Charlie Hardin, Dave Patton, Mert Richards, and Wayne Shepperd. In addition to those presenting papers and posters, we wish to thank the session chairs for their diligence in selecting speakers and moderating the discussions. The field trip leaders did an excellent job in presenting each of their topics in the context of ecosystem management and sustainable ecological systems. The conference attendees represented dedicated groups of managers and scientists who participated wholeheartedly in each of the sessions. Finally, we wish to thank Pamela Barber and her staff for the excellent logistical support throughout the planning and implementation of this conference.

SUSTAINABLE ECOLOGICAL SYSTEMS—PHILOSOPHICAL, HISTORICAL, AND CULTURAL CONTEXT:

Plenary Session Summary

W. Wallace Covington, Chair

This plenary session began with a presentation by Thad Box entitled, "Sustainable Ecological Systems and Cultural Changes," in which he discussed the changing demands of different cultures for natural resources from wildlands. He cautioned today's constituents from judging too harshly the resource management and use not only of their contemporaries, but also of their predecessors. He closed with an admonition that changes in resource demands will occur at increasing rates and that continued learning is essential to successful adaptation to these new circumstances. The next plenary session speaker was Baird Callicott who presented a historical perspective on American conservation philosophy. In his presentation he traced conservation thinking from the days of Thoreau, Muir, and Pinchot through Aldo Leopold's attempts at reconciling the differences among these early conservation thinkers by advocating humans living in harmony with nature. Callicott concluded that Leopold's vision of man in symbiosis with nature can serve as a foundation for contemporary concepts of ecosystem management.

The next paper was by Susan Flader. Flader enlarged upon this theme by pointing out that Aldo Leopold is the only individual cited in Forest Service Chief Dale Robertson's directive on ecosystem management. She went on to describe Leopold's development of an ecological approach to management based on his observations while he was a forest officer with the Southwestern Region of the Forest Service. She closed by echoing Leopold's concerns regarding potential administrative stumbling blocks to implementing an ecosystem management approach on national forest lands.

Mike Soule presented the final paper in the plenary session. Soule began with an overview of key concepts of conservation biology including both functional, or mechanistic postulates, and normative, or ethical, postulates. He then moved on to making a strong case for taking an ecosystem approach to the preservation of rare and declining species. He closed his presentation with an enumeration of some central questions which must be addressed in ecosystem management.

Sustainable Ecological Systems and Cultural Change

Thadis W. Box[1]

Abstract — Sustainable ecological systems are linked to the cultural demands on the land. While the potential ecological carrying capacity of the land remains relatively stable, the cultural and social demands made on land constantly change, causing actual carrying capacity to fluctuate widely through time. Therefore, the overriding concern for those attempting to manage ecological systems for sustainability is coping with change. The key issues are 1) determining what society wants from its land with inevitable changing values, changing demographics, and changing economics, 2) achieving orderly transition and community well being in an internationalizing and amenity oriented economy, 3) developing common language, measures, and forums for identifying and evaluating trade-offs: ecological, economic, fiscal, human, and social, 4) shifting from reductionist and disciplinary work to synthesis and interdisciplinary analysis of systems, and 5) defining the issues to reflect fairly the wants and needs of society while protecting the land base. If what people want from the land is not compatible with the ecological base, cultures cannot succeed. Balancing land capability and cultural demands will be controlled by what we can imagine, creativity, and vision. And all these are enhanced by education.

INTRODUCTION

This paper today generates strange echoes in my psyche. One echo is from the very first professional paper I ever presented. It was in 1958 before the Texas Academy of Science, entitled "The Multiple Use of Rangelands: A Problem in Ecosystem Management." I have been told it was the first paper ever to link multiple use to ecosystem management, but because it was given by a brash graduate student at a state academy meeting, it attracted little attention or comment. Now, 35 years later, we meet in a major symposium to celebrate the USDA Forest Service's changing to ecosystem management as a tool to meet its multiple use mandate.

The second echo is because an earlier version of the paper I present today was given here in Flagstaff on March 30 as the Seventeenth William P. Thompson Memorial Lecture. I am indebted to the School of Forestry at Northern Arizona University for asking me to pull together my thoughts for their honor's week. Since changes from the Thompson Lecture have been minimal, many of you may think this program has gone into rerun.

I am especially pleased to have experienced foresters and resource managers in the audience because my paper, unlike many of the others to be presented here today, is not data rich or backed by sound experimental design. Instead, it is based on my experience of the effect that cultural change and public attitudes have, both on the land and on those of us who dare call ourselves natural resource managers.

Three years ago I retired as Dean of Natural Resources at Utah State University. I was teaching a graduate seminar on sustainable systems at New Mexico State University. One week end I climbed the Organ Mountains east of Las Cruces and sat in a rock shelter near the place where the oldest corn in the United States was found. As I gazed out over the lush irrigated farms, the housing developments, and the intersection of two busy interstate highways, I wondered if our civilization would also go the same way as those who made the petroglyphs in the shelter I had invaded.

[1] *Thad Box is professor emeritus, College of Natural Resources, Utah State University and Gerald Thomas Professor, College of Agriculture and Home Economics, New Mexico State University. He lives in an adobe house in Mesilla, NM.*

Coming down the mountain, I took a short cut across a dry, barren, west facing slope. There, with no trees anywhere in sight was an ancient stump with weathered axe marks still showing. As the sun went down I sat and wrote:

Stump Near Solidad Canyon

on desert ridge
bare
save yucca
   cacti
   and woody scrub
a stump clings
relict of a gentler time

viejos
cannot remember cedar
on that dry west facing slope
though centuries
   the tree grew
   it fell
in modern times

it stood proud
against drought
   and twisting wind
a rare
dark green dot
on a purple hill

a pioneer
climbed that hill
   swung his axe
removed the life
that clung to stone

did it make
   vigas for adobe hut
   spokes for wagon wheel
   fire to warm a newborn babe
   in rare Mesilla snow

the axeman
judge him not
he was a product
of a harsher time

The Native Americans who first tilled our soil, the pioneers who brought European ways, and those of us using land today are all products of our times. Land use is dictated, in part, by our cultural and social values, but our culture is changing, and our land base will be subjected to new uses.

An old Chinese curse says, "May you live in interesting times." This is both an interesting and frustrating time to be alive. Only two years ago I was filled with optimism. The Berlin wall had tumbled down. The Soviet Bear had been declawed. Peace was upon us. Opportunities were greater than in my lifetime.

Now, just two years later, the world economy is in recession, tribalism pits neighbor against neighbor, and we strangle on our own waste. We kill for oil. Bosnians kill one another, and Somalis starve while we fiddle around at the margins.

I could be discouraged, but I have heard two comments recently that give me hope. First, from home-grown cowboy humorist, Baxter Black, who tells his audiences that America can undertake such missions as feeding starving children in Somalia only because agriculture has been so successful.

And second, Charles Kuralt recently told us that Domingo Arroyo, our first military casualty in Somalia, was the first soldier in history to die just for the purpose of feeding starving children.

Together these comments tell us that we have the tools and the desire to make the world a better place. If we can replace Audie Murphy with Domingo Arroyo, we can combine technology with social justice.

Since my retirement I have thought a lot about my own career, about changing cultural values, about how land will be used, and about the new worldwide concern for sustainability.

I welcome this opportunity to have my fellow resource students and professionals hear my ideas. I have tried my ideas in classes and seminars where I had a captive audience. I have shared my understanding of changing views of society during the past three decades with service clubs and my own professional society, but I welcome this opportunity to be with foresters.

I have thought much about my 30 years of frustration as a teacher, department head, and dean. During this time students changed, research changed, and funding for education changed. Society itself changed; new public attitudes about conservation and land use came and went, each having an effect on the land.

Cultural changes didn't just affect land use, they altered people's lives. For instance my last PhD student, head of the Botany Department at the National University of Somalia, recently wrote me, "Thanks to God, only one of my children has starved."

For decades the white man's legal system has tried to resolve the disputes between Hopi and Navajo over land use. The suggested settlement caused screams from the white recreational culture at the very idea that land might be returned to Native Americans for religious purposes

All these examples fit into my topic for today, land use and cultural change. My thesis is that we in natural resource management walk a narrow line between the demands of a given culture with its social values and what it wants from the land, and the need to leave options open for evolving cultures. Our goal is sustainable land use that will support present and future cultures.

Today I will offer you my view of what I think the real sustainability issues are and how they relate to cultural change. I will not present my college prof lecture on natural processes and ecosystems. I will not argue the cultural case for Hopi ownership of San Francisco Peak or the white man's case for keeping the land under Forest Service management.

Instead, I will paint with the broad brush of societal needs and sustainable land use.

I will review the history of conservation and how we came to a global concern about sustainability. I will discuss the human motivation of the past two decades and how it has led societal concerns and cultural change. I will examine the concept of sustainable development as it relates to cultural change. I will look at some factors affecting future land use, and close with some ideas about sustainability, sustainable development, and cultural change.

HUMAN EMOTIONS AND LAND USE

Many times emotional, but marginal issues, sidetrack natural resource managers from sustainable land use and social justice. We divide society over environmental issues rather than seek a sustainable future. For instance, sophisticated groups are trying to remove commodity use from the public lands.

Slogans such as "Cattle free by 93" or "no more moo by 92" attract headlines. The spotted owl becomes a surrogate for old growth timber. Animal welfare groups and wild horse enthusiasts expend time and resources to protect feral horses that in turn destroy public rangeland. In the end we spend more on feral horses than on battered women. What sets our priorities?

At a recent conference on endangered species in Phoenix, I wrote the following. I call it

Endangered Arizona

Mexican spotted owl
Gila trout
Northern goshawk

creatures
cowboys
timber beasts
threatened

Arizona snowbirds
arrive on dead dinosaurs

Braceros
die in boxcars

A rabbitbush makes
the endangered list

Chrysothamnus molestus

Chrysothamnus molestus, there should be a way to balance conservation with social justice. And there are some encouraging signs.

From amongst all the emotion, a "new land management" is evolving...using land for societal values. There is a call to shape the future conditions of landscapes for a full diversity of life, ecological processes, human values, and resource uses.

This will mean balancing science with social values, economic feasibility, institutional traditions and political muscle.

The "new land management" is a recipe for sustainable land use, but in this country it has largely been associated with protectionist causes. It has most often been directed to concerns dealing with wetlands protection, endangered species, or biological diversity.

It has not become the watchword, as many of us had hoped, for agriculture, natural resource management or world aid organizations. We have yet to relate new land management and sustainability concerns to cultural values, equity issues, and social justice. To have sustainable development we must take the next step.

SUSTAINABLE LAND USE

Where does "new land management" or sustainable development fit into the lives of Forest Service Officials and colleagues at Northern Arizona University? Land sustains our bodies, our children, our life style—we belong to the land.

The land use objectives for all of us are the same...wise resource use...but we may differ on what wisdom to use. To a Hopi, San Francisco peak is a holy place..a place for spiritual renewal to sustain his culture. To a white recreationist it is a ski slope, a summer cabin, or wilderness..a retreat to sustain another very different set of values. To a forester, it is a place to grow trees; to a shepherd, it is a place to grow sheep. It is a place to support commodity production to satisfy yet another culture of consumers.

Wise use for each group is to sustain the use that perpetuates its cultural values.

Only when we are forced to think globally and beyond our own culture does wise use include managing for options to be kept open for new or future uses...in other words, to think about sustainablility.

The quest for sustainability is a grass roots movement. It has many definitions, but all definitions have four central concepts:

1). There must be equity for today's land stewards. Farmers and foresters must be able to make a good living. If they do not have a high standard of living, there will be no tomorrow.

2). There should be equity for future generations. We must leave options open for our grandkids. We must not close out future uses. Sustainability means that people today embrace a dream for tomorrow.

3). Long term sustainability must take precedence over short term profit. We must keep the land productive. We must learn to live on the interest without depleting the principle. If possible, we should increase the bank account.
4). The fourth central concept is environmental enhancement. We must improve what has been given us. We need to leave the world better than we found it. This dictates that we become active in land improvement.

These are not new concepts. They are the same as those that Aldo Leopold and Hugh Hammond Bennet wrote about sixty years ago. They are the same as those I used thirty-five years ago when I talked to the Rising Star Methodist Church on Stewardship Sunday.

The concepts are the same. But the world has changed. And I have changed. Then I had yet to earn my PhD. I did not even know where Somalia was. I had never looked into the eyes of a starving child. And I had never had a friend write "Thanks to God, only one of my children has starved."

A BRIEF HISTORY OF AMERICAN CONSERVATION

Our culture, like me, has changed because of what has happened to it. I believe that the history of conservation in the United States gives us an insight into the current quest for sustainability.

When the first European settlers arrived in North America, we entered an "Era of Exploitation." To conquer the wilderness was right and honorable. Development of the new land was the public policy. Forests were cut. Prairies were plowed. Buffalo were replaced with homesteaders. The railroad connected the Atlantic to the Pacific. A new nation raced to become an industrial giant and a world power.

About the middle of the 19th Century, a few people began to call for saving plants, animals, or land. We entered an "Era of Preservation." Yellowstone became our first National Park. Forest Reserves were established. Public policy still embraced growth and development, but room was made to save a little of our resources.

Between the first and second world wars, we entered an "Era of Reclamation." Because of the dust bowl days of the thirties, the soil erosion service was formed and the Taylor Grazing Act passed. Make-work projects of the Great Depression constructed terraces, planted forests, built campgrounds, established windbreaks, and tried to rebuild that which had been depleted.

After WWII, as the Cold War made us aware of the possibility of nuclear destruction, we entered the "Era of Environmental Concern."

Rachel Carson's book, Silent Spring, focused our fears and our attention on ourselves and our families. This book changed the conservation movement forever. What had been a land based movement became a concern for personal health and safety. This new environmental awareness became the moving force for clean air, clean water, organically grown food, and other demands of urban based citizens. It changed the demands made on the land.

I think we may have now entered a yet another era, the "Era of Sustainablility." It grew out of the Earth Day movement of the 1970s. Originally, the movement was led by powerless kids who distrusted society and its leaders. It was reactive in nature and spawned laws such as NEPA, NFMA, FLMPA, etc. Solutions were based on rules rather than reason. Litigation and lawyers dominated the conservation scene.

The Earth Day Movement became muddled with an unpopular war, new sex mores, free speech movement, changing gender roles, communal living and other evidences of cultural evolution. It lost its power in many diverse, but related movements for which conservation leaders were not ready. Cultural values were being challenged in our society, but most did not view it as a fundamental new direction for conservation. At best, they saw it as simply an extension of the environmental movement.

RECENT CULTURAL CHANGE IN RESOURCE STUDENTS

The examples of cultural change I have seen in natural resource students over the past three decades cause me to believe that we are indeed moving into the beginning of the era of sustainability.

During the 1970's I taught an intimate little class of over 340 students. The title was "Natural Resources and Man's Future." It was so popular that I, like the airlines, overbooked it by 15%, thinking that I would have enough absences for any given lecture to allow all to be seated. Instead, I would often have students sitting in the aisles and standing at the back of the auditorium.

The 1970's kids were filled with idealism. They cared. They wanted to save the world. They did not want a job...at least the kind we had to offer in forestry. Foresters and wildlife managers were bad guys who cut trees and wallowed in blood and guts. The students went into the Peace Corps, they demonstrated and marched against injustice, and the resisted those in power.

Today, these very people are in power. They are now in leadership positions from the district ranger's office to the White House.

In 1970 students were driven by a fear of nuclear holocaust and found release in such events as Earth Day. Their distrust for authority and their relative powerlessness filled them with frustration. There is evidence that these concerns have carried over today when they are in authority. We can only hope that maturity has leavened their cynicism and distrust.

The students of the 1980's also saw in "the bomb" a real likelihood that their dreams could be cut short. Class size in the beginning natural resource course dropped to less than 100. Students sought material wealth and wanted to make lots of money. They were willing to take any job if it paid well enough.

They were not concerned with social issues or the land. In the 1980's fear of extinction led to a "let's get it now" attitude. Concern for personal wealth replaced concern for society. BMW's and MBA's were dominant.

Military Science as a major subject was more popular than forestry and all other fields of conservation combined. Rambo ruled. God lived on Wall Street and drank Perrier water.

The ethic for the 80's was, "greed is good, rules are for fools, and he who has the most toys in the end wins."

The bubble burst in October 1987 when the stock market experienced its greatest one day loss since the crash of 1928.

What cultural values will the next generation bring now that the threat of nuclear destruction has diminished? I don't know. But whatever they are, the new cultural values will determine how we use our land base to meet societal needs. Sustainable development depends on what we demand from the land.

SUSTAINABLE DEVELOPMENT AS A GLOBAL ISSUE

Sustainable development became a world issue with the awakening of global economic and environmental interdependence. As groups of nations moved toward "economic communities" they found that trade and national economies can be regulated only with great difficulty. Environmental regulation was even more difficult. Environmental disasters knew no national borders. They cut across political subdivisions.

We see the problem in microcosm in the south Mesilla Valley of New Mexico where I live—Juarez, Chihuahua and El Paso, Texas overshadow what is done in Las Cruces, New Mexico. What Las Cruces does dictates policy for the historic village of Old Mesilla. While the town council of Mesilla strives to preserve the historic flavor of their town, actions in Washington and Mexico will ultimately determine how the resources the village uses are allocated.

A number of international organizations are addressing the problem of sustainable development. The World Commission on Environment and Development issued a report as early as 1983 on global development problems. The Commission was an independent entity of United Nations, chaired by Norwegian Prime Minister Gro Harlem Bruntland. Some of their findings were optimistic: human life expectancy was increasing, infant mortality was decreasing, adult literacy was climbing, scientific and technical innovations were promising, and global food output was increasing faster than population growth. However, they also reported that topsoil was eroding faster than it formed, forests were declining, air pollution was smothering our cities, ozone protection was diminishing, and toxic substances were more abundant in water supplies.

They concluded that the gap between the rich and the poor was increasing and that the land use in 1983 was not sustainable.

Now, a decade later, the Bruntland Commission findings are still valid. Several major conferences, the latest last year in Brazil, have tried to get global consensus on a plan of action that will allow the world to develop in a sustainable fashion. Unfortunately, the United States, the world's largest per capita consumer of resources, has yet to lead to developing global sustainable development requirements.

Sustainable land use means implementing a policy that meets the needs of people today without destroying the resources that will be needed in the future. Development cannot be sustained on a deteriorating environmental base. For national economies to grow and be profitable, the natural resource base must be maintained.

SUSTAINABLE LAND USE AND CARRYING CAPACITY

Sustainable land use depends, in part, upon determining the ecological carrying capacity of the land, determining what people want and need from the land, and a political and economic system that matches what people want and need with the land's ability to produce the desired goods and services.

While the potential ecological carrying capacity of the land remains relatively stable, the cultural and social demands on the land are constantly changing, causing the actual carrying capacity to fluctuate widely through time. Therefore, the overriding element for those attempting to manage development is coping with change. The key issues are:

1) determining what society wants from its lands with inevitable changing values, changing demographics, and changing economics.
2) Achieving orderly transition and community well being in an internationalizing and amenity oriented economy.
3) Developing common language, measures, and forums for identifying and evaluating trade-offs: ecological, economic, fiscal, human, and social.
4) Shifting from reductionist and disciplinary work to synthesis and interdisciplinary analysis of systems.
5) Defining the issues to reflect fairly the wants and needs of society while protecting the sustainable land base.

The concept of sustainable development in rich countries is most often embraced by conservation groups and organizations. At the same time, these rich countries are using a disproportionate amount of the world's resources. Unless sustainable development is linked to basic issues of equity, social justice, and community stability for poor people, sustainable development will fail.

But can we in America relate to sustainable human lifestyles when we are hooked on a consumption society? Our foreign policy suggests we will fight for cheap oil with enthusiasm, we will feed hungry children when we see them on the TV screen, and we will give US aid to political friends.

Our development is not sustainable because our domestic policies are too closely linked to low food prices, artificially cheap oil, and consumption encouraged to promote growth. With the collapse of the Soviet Union and recessions in Japan and western Europe, we are once again cast as the world's leader. Opportunities have never been better for us to lead the world onto new plateaus — if we relate all new issues to sustainability. However, we can only lead if we make our own lifestyles sustainable.

FUTURE ISSUES AND SUSTAINABLE LAND USE

No one knows exactly what the future holds. However, I am certain that there will be several issues that will dominate much of our attention in the future. I will discuss four of these.

Demographics

First, the human population, especially its demographics, will affect sustainable land use. The primary question for my generation was, "Who will feed the hungry world?" We have made major accomplishment in this area, to the point where we have an embarrassment of surpluses in some countries. But the problem of feeding the human population remains, only the time frame has changed. Even with our abundance of food, some 40,000 people die each day of starvation and disease. In a month more people than live in New Mexico will die from nutrition related ailments.

Sustainable land use, equity in this generation, means feeding those less fortunate than we whether they are in Somalia, or Bosnia, or Northern Arizona.

There are other pressing demographic issues besides balancing human numbers with food supply. The gap between the haves and the have nots is increasing. Poor countries are growing 4 times as fast as rich countries; 4 out of 5 babies are born into poverty. Sustainable land use must relate to this poverty gap.

Economic Trends

Rich countries are not growing...they are growing old. Poor countries have young populations of mostly non-white people. In the United States, wealth is concentrated in a few, usually older people.

The United States has moved from the world's largest creditor nation to its largest debtor nation. The global financial center has moved from Wall Street to somewhere on Pacific Rim. Our markets and our labor are in poor countries, but they are unable to buy. Their standard of living must be raised if they are to participate in sustainable development.

Sustainable land use, equity for future generations, depends on world peace and world trade.

Material Science and Technology

We live in a world where designers imagine a product, engineers specify the characteristics of the components, and chemists create the building materials from polymers, graphite, ceramics, or whatever combination of elements can produce the required strength and aesthetic qualities. No longer does the designer buy two by fours and then let them determine the final product; the final product is based on the creators imagination and skills. The demand for producing natural building products such as wood, wool, and cotton will not necessarily determine land use. But the new synthetic materials must be constructed from existing elements, and all will require increased levels of energy input.

Alternate energy sources must be a high priority if we are to have sustainability. We cannot have world peace if we continue to bet our future on cheap oil. The continued burning of hydrocarbons will contribute to global warming.

To achieve sustainable land use, the development of alternative energy sources needs to be global. If in this decade all rich countries stopped burning hydrocarbons, still the increase in coal burning as China and Eastern Europe expand their industrial output would likely keep global warming on its present upward trend.

Sustainable land use, long term stability, means adapting new materials and adjusting land use through a combination of ecology, economics, and technology.

Philosophical Trends

In the past, world development, sustainable or otherwise, has largely been the product of western thought patterns of growth and development. Of the world's 10 largest countries only 3 are "Christian" in philosophical thought. The philosophical implications of a global change away from Judeo-Christian attitudes about development will have profound effects on sustainable land use.

The most obvious trends are an increase in animal rights activities and a wider acceptance of vegetarianism. However, much more important changes will occur with different concepts of equity, beauty, property ownership, productivity, and work.

Even now, work is not what we do, but is what we can imagine.

Vladimir Horowitz, one of the greatest pianist of all times, died a couple of years ago. A clever computer programmer can make a synthesizer play Horowitz, Chet Atkins, Alabama, the Grateful Dead, or even Bob Wills. But she cannot make the computer imagine the music.

Science fiction writers tell us of transferring material directly from one brain to another. In their world you can transfer Russian from a disk to your brain. If you are going on a pack trip and do not know how to ride a horse, you can transfer cowboying from your uncle's brain. The android, Mr. Data, on Star Trek has all the past knowledge and human experience stored on his computer chips, but he lacks human emotion and a philosophical base. A Mr. Data could provide all the information needed to make the world better, but he could not define better for us.

Sustainable land use, environmental enhancement, will depend on what our concept of "better" is. Our philosophical base will be the key element, not our technology.

Notably absent from my list of important future conditions is climate change. I did not leave it out because I do not think it important, or that I do not believe it will happen, but because it is a symptom...not a cause. To have sustainable land use, we should concentrate on causes, not effects.

SUSTAINABLE SYSTEMS AND CULTURAL CHANGE

Now for some final thoughts on sustainable land use and cultural change. Sustainable land use ANYWHERE is linked to cultural demands. Cultures change. Land use changes. If what cultures want from the land is not compatible with the ecological base, cultures cannot succeed. Balancing land capability and cultural demands will be controlled by what we can imagine, creativity, and vision. And all these are enhanced by education.

Education and Sustainable Land Use

There are some simple steps in education...the creation of a vision:

1) Make people aware of the situation and give them the facts.
2) Give them problem solving skills.
3) Give them a bag of tools.
4) Inspire them to do something about the situation.

There have been two great, successful experiments in American education that are worth noting. The first is the establishment of the Land Grant Universities through the Morrill Act. These new colleges took the rural poor and the mechanic class, taught them liberal arts, taught them discovery techniques with research, and developed continued learning through extension. This new type university changed our country and made it the envy of the world.

The other great experiment was the GI Bill. Although military service was not designed as an education program, it dragged kids from the farm and the ghetto, taught them simple sanitation and discipline, made them work together, expanded their vision, and after the war, our country paid them to go to school.

America has received immeasurable dividends from these two experiments in vision creation and education...and the dividends continue. These and other successful models of education can be tied to developing sustainable land use.

What we need to do is to create new visions of what the world can be. We need to tie science and application together in the simple steps of education:

1) Identify the problem. What is causing non-sustainability on our planet or in our back yard garden?
2) Set priorities. What problems should we tackle first that will really make a difference? For example global warming and the greenhouse effect may be a policy problem rather than one for individual action. If everyone planted a tree, if all the rain forests are saved, and we do not change fossil fuel use, all will be for naught.
3) Improve our bag of tools. Do good science. Synthesize and integrate. Tie ecology, economic development, and social justice together. Accept social sciences as land management tools. Improve our application of science.
4) Inspire. Inspire to make something happen. Inspire to create new visions of what may be.

We are having trouble creating new visions because we are unable to relate to new cultures and social values. Our traditional approaches have been based on the cold war fears, protect ourselves from world communism and promote consumption for continued growth and economic gain.

With the diminished threat of nuclear destruction, our new social values turn toward sustainability....of our income, our land, our lifestyles. New advocates for sustainability may come from diverse groups with varying immediate goals. Environmental groups demand natural resource protection. Commodity groups want sustained production. The underemployed, the hungry, the have-nots wish for social justice and a sustained fair wage.

Overseas, our new allies are attempting to apply market economies where there are no institutional support or past experience. Our new friends have inherited a landscape spoiled by past misuse.

The new support is not always scientifically credible. We often get bogged down defending practices or positions that are equally incredible. We mix scientific credibility with social acceptance or political correctness. We try to apply past solutions to current problems.

We forget that we, like the pioneer who chopped down the cedar tree, are products of a different time. Our success will be determined by our ability to adjust, change, lead.

Some say that in our quest for sustainable land use we are going round in circles. A friend reminded me we are not going in circles. We are in a spiral. Concern for conservation is coming round again, but we are on a higher level, like the next step on a spiral staircase. We are on a new plateau. We are no longer living in fear of communism or the bomb. Sustainability is a grass roots movement. Our science is better. Limited peace is upon us. We realize that people are an important part of this new sustainable land use.

If we concentrate on education, creativity, application, we can move to a yet higher plateau.... a higher plateau where social justice is balanced with resource use, where development is truly sustainable.

New cultures that develop in the future will be able to reach their potential if we in this generation remember:

- equity for today's generation
- a better life for our grandkids
- leave options open for those who follow us
- leave the world better than we found it.

But we will not have, indeed we do not deserve, public support if we continue business as usual, continue to organize our programs of development around narrowly drawn issues such as cheap oil or saving an endangered species. We will fail if we underestimate the worldwide support for sustainability.

The quest for sustainable land use is doomed if we ignore equity and social justice in our sustainability equation whether it is keeping Ahmed Elmi's children alive in Somalia, getting single moms off welfare in Flagstaff, or keeping gang members from killing one another in Tucson.

As educators we have a special responsibility to make sure our culture is sustainable. As natural resource professionals we have stewardship of the ecological base for sustaining our culture. I pray that we in forestry and natural resources in this generation set the stage for a better world.

I feel incredibly fortunate to have been involved in development. I am proud to have been a university teacher and a conservationist. I am struggling to be a different kind of teacher today. I appreciate your kindness in letting me spout my biases. Thank you for asking me to participate in this symposium today.

A Brief History of American Conservation Philosophy

J. Baird Callicott[1]

Abstract — Conservation as wilderness preservation originated with Ralph Waldo Emerson and Henry David Thoreau and was popularized by John Muir. Conservation as resource management was articulated by Gifford Pinchot. Subsequently, North American conservationists were split into two factions—nature preservationists versus resourcists. The Pinchovian philosophy dominated state and federal agencies, such as the Forest Service. The Muirian philosophy dominated private conservation organizations, such as the Sierra Club. Aldo Leopold is usually represented as having begun his distinguished career as a member of the Pinchot camp and, influenced by the new science of ecology, gradually going over to the Muir camp. But Leopold actually articulated a third philosophy of conservation. He advocated a human harmony with nature. Leopold envisioned ecosystem—as opposed to resource—management and sustainable development, if by sustainable development is meant development limited by ecological as well as by economic exigencies. Resource conservation is untenable because it is founded on an obsolete, pre-ecological reductive scientific paradigm. Conservation via wilderness preservation is equally flawed. Conservation conceived as a mutually beneficial symbiosis between the human economy and the economy of nature is destined to be the philosophy of conservation for the twenty-first century.

Conservation in the Old World, especially forest and game conservation, seems to have evolved gradually (Peterken 1981). No doubt, a parallel, but very different practice and conception of conservation also independently evolved in the New World as well. With the wholesale devaluation and destruction of American Indian cultures that occurred during four of the five hundred years of European discovery, conquest, colonization, and finally complete domination of the Western Hemisphere, however, indigenous New World conservation thought and practice was all but lost (Viola and Margolis 1991).

The depopulation of North America was so thoroughgoing, owing more to what might be called inadvertent biological warfare than to conventional warfare (Deneven 1992a), that the English colonists could imagine that they had settled in a wilderness (Nash 1967), not in a country once fully inhabited and significantly transformed by its indigenous peoples (Deneven 1992b). Thus, two allied myths established themselves in the Euro-American consciousness: one, that the whole of North America was a "virgin" wilderness of continental proportions; the other, that North America's natural resources, and especially its forests were inexhaustible. The second of these is conventionally called "the myth of superabundance."

While the wilderness myth has only been recently debunked (Callicott 1991; Gomez-Pompa and Kaus 1991), the myth of superabundance was abandoned around the turn of the century. With the completion of the transcontinental railroad, the slaughter of the bison herds, and the subjugation of the Plains Indians, the North American frontier palpably closed and the limits of North America's natural resources dawned on thoughtful Euro-Americans (Hays 1959). Against the background of laissez faire exploitation—unregulated hunting and fishing, logging, mining, plowing, and so on—the necessity of conservation received a good deal of conscious reflection.

George Perkins Marsh (1864, 1874) is generally credited with first articulating an American conservation philosophy in his prophetic book, Man and Nature or The Earth as Modified by Human Action. Marsh was mainly concerned about the adverse effects of deforestation on stream flow, soil stability and fertility,

[1] *University of Wisconsin-Stevens Point, Stevens Point, WI.*

and climate. His conservation ethic was an early American version of contemporary Judeo-Christian stewardship. "Man," he wrote, "has too long forgotten that the earth was given to him for usufruct alone, not for consumption, still less for profligate waste" (Marsh 1874, p. 33).

Ralph Waldo Emerson and Henry David Thoreau had not attained the essentially ecological understanding of the relationship between vegetation, soil, water, and climate that Marsh had. They were principally concerned rather with the aesthetic, psychological, and spiritual paucity of the prevailing American materialism and vulgar utilitarianism. As an antidote, they turned to wild nature—contact with which, they argued, invigorates and strengthens the body, inspires the imagination, energizes the mind, elevates the soul, and provides an occasion for transcending finite human consciousness. Because wild nature is a psycho-spiritual—as well as a material—resource, Emerson (1836) and Thoreau (1863) argued that Americans should preserve a significant portion of it undefiled.

Emerson and Thoreau thus stand at the fountainhead of the wilderness preservation philosophy of conservation. Thoreau was probably the first American to advocate what eventually became a national wilderness preservation policy: "I think that each town," he wrote, "should have a park, or rather a primitive forest, of five hundred or a thousand acres . . . where a stick should never be cut—nor for the navy, nor to make wagons, but to stand and decay for higher uses—a common possession forever, for instruction and recreation."

This philosophy of conservation was energetically promoted by John Muir (1901). Through his lively writing, thousands of American readers experienced vicariously the beauty, the physical and mental salubriousness, and the spiritual redemption that he experienced directly and personally during his many and lengthy wilderness sojourns.

Gifford Pinchot, a younger contemporary of John Muir, articulated a very different philosophy of conservation firmly grounded in utilitarian values and closely associated with the world view of modern classical science. Pinchot (1947, pp. 235-236) crystallized the resource conservation philosophy in a motto—"the greatest good of the greatest number for the longest time"—that echoed John Stuart Mill's (1863) utilitarian creed, "the greatest happiness of the greatest number."

Pinchot bluntly reduced the "Nature"—with which Marsh, Emerson, Thoreau, and Muir were variously concerned—to "natural resources." "There are two things on this material earth," he averred, "people and natural resources" (Pinchot 1947, p. 325). And he even equated conservation with the systematic exploitation of natural resources. "The first great fact about conservation," Pinchot (1947) noted, "is that it stands for development." For those who might take the term "conservation" at face value and suppose that it meant, if not nature preservation, then at least saving some natural resources for future use, Pinchot was quick to point out their error: "There has been a fundamental misconception," he wrote, "that conservation means nothing but the husbanding of resources for future generations. There could be no more serious mistake" (Pinchot 1947). And it was none other than Pinchot (1947, p. 263) who characterized the Muirian contingent of preservationists as aiming to "lock up" resources in national parks and other wilderness reserves.

In short, for Pinchot conservation meant the efficient exploitation of "natural resources" and the fair distribution of the benefits of doing so. Science was the handmaid of efficiency and macro-economics of fairness. Thus Pinchot's philosophy of conservation was wedded to the eighteenth- and nineteenth-century scientific world view, according to which nature is a collection of bits of matter, assembled into a hierarchy of independently existing chemical and organismic aggregates, that can be understood and manipulated by reductive methods. It was also wedded to the correlative social science of economics—the science of self-interested rational individuals pursuing preference-satisfaction in a regulated market.

John Muir and Gifford Pinchot were, for a time, friends and allies. Their very different philosophies of conservation, however, led to a falling out (Nash 1967). The personal rift between Muir and Pinchot symbolizes the schism that split the North American conservation movement into two mutually hostile camps at the beginning of the twentieth century (Fox 1981). Pinchot commandeered the term "conservation" for his philosophy, while Muir and his followers came to be known as "preservationists."

Pinchot's philosophy dominated conservation in the public sector of the United States—the Forest Service (of which Pinchot himself was the first Chief), the Fish and Wildlife Service, the Bureau of Land Management, and state departments of natural resources (Fox 1981). Muir's philosophy prevailed in non-governmental conservation organizations—such as the Sierra Club (which Muir founded), the Wilderness Society, and the Nature Conservancy (Fox 1981).

Aldo Leopold was employed by the United States Forest Service for fifteen years (Meine 1988). Thus he began his career as a conservationist solidly in the Pinchot camp. Nevertheless, he gradually came to the conclusion that Pinchot's conservation philosophy was inadequate because it was based upon an obsolete pre-ecological scientific paradigm (Flader 1974). As Leopold (1939a, p. 727) put it,

> *Ecology is a new fusion point for all the sciences . . . . The emergence of ecology has put the economic biologist in a peculiar dilemma: with one hand he points out the accumulated findings of his search for utility in this or that species; with the other he lifts the veil from a biota so complex, so conditioned by interwoven cooperation and competitions that no man can say where utility begins or ends.*

From an ecological point of view, nature is more than a collection of discontinuous useful, useless, or noxious species furnishing an elemental landscape of soils and waters. It is, rather, a vast, intricately organized and tightly integrated system of complex processes. And human beings are not specially created and uniquely valuable demigods, any more than nature is a vast emporium of goods, services, and amenities. We are, rather, very much a part of nature. Further, the portrait of human beings in economic theory as single-minded consumers is a gross caricature. Individual welfare, from an ecological point of view, is inextricable from the health and integrity of both the social and natural communities to which we belong.

We tend to think of Leopold as having begun his distinguished career in the Pinchot school of conservation thought and gradually to have come over, armed with new ecological arguments, to the wilderness preservation school of thought. And indeed Leopold was committed to wilderness preservation throughout his life, though his reasons evolved from an emphasis on recreation (Leopold 1921) to an emphasis on the role of wilderness in scientific research and wildlife conservation (Leopold 1936, 1941).

But Leopold realized that the Muir-Pinchot schism had left North American conservation in an unfortunate "zero-sum" dilemma: either lock up and preserve pristine nature or efficiently and fairly develop it . . . and, in doing so, necessarily degrade or destroy it. Half a century after institutionalizing Pinchot's conservation philosophy through the establishment of the Forest Service and similar natural resource management bureaucracies, the United States Congress institutionalized Muir's conservation philosophy in the Wilderness Act of 1964. It reads in part, "a wilderness, in contrast with those areas where man and his own works dominate the landscape, is hereby recognized as an area where the earth and its community of life are untrammeled by man, where man is a visitor who does not remain" (Nash 1967, p. 5). Reflecting the unequal political strength of the conservationists and the preservationists, the contiguous forty-eight United States eventually became segregated into large development zones dotted here and there (mostly west of the Mississippi) with wilderness preserves adding up to only two or three percent of the total. Hoping to break out of this dilemma, Leopold advocated a "win-win" philosophy of conservation, stressing ways of inhabiting and using nature that are at the same time ecologically benign. As he put it, the "impulse to save wild remnants is always, I think, the forerunner of the more important and complex task of mixing a degree of wildness with utility" (Leopold 1991a, p. 227).

Accordingly, Leopold set out to define conservation in the following terms: as "a universal symbiosis with land, economic and aesthetic, public and private" (Leopold 1933, p. 639); as "a protest against destructive land use" (Leopold 1991b, p. 212); as an effort "to preserve both utility and beauty" (Leopold 1991b, p. 212); as "a positive exercise of skill and insight, not merely a negative exercise of abstinence and caution" (Leopold 1939b, p. 296); and, finally, as "a state of harmony between men and land" (Leopold 1949, p. 207).

Currently, Leopold's harmony-with-nature philosophy of conservation is called "sustainable development"—if by "sustainable development" is meant the initiation of human economic activity that does not significantly compromise ecological health and integrity; and ideally economic activity that might positively enhance it. "Sustainable development" is, however, an unfortunate phrase. "Ecological livelihood" would be less liable to misinterpretation and misappropriation. "Sustainable" is vague and often used by economists to mean passing on enough capital and technological know-how to replace exhausted natural resources and compromised biological systems with artificial alternatives. And "development" is often a euphemism for the building of high-rise condominiums, shopping malls, parking lots, and subdivisions. In calling for a "universal symbiosis with land," Leopold had in mind changes far more radical than, say, building more energy-efficient tract houses and automobiles. He was proposing, rather, a veritable revolution in the way we human beings inhabit and use the natural environment.

How should we assess twentieth-century North American conservation philosophy as we approach the twenty-first century?

Pinchot's philosophy of conservation is no longer viable, since it is founded on a reductive, pre-ecological scientific paradigm. Even the United States Forest Service is admitting that old growth forests are not just senescent stands of timber, overdue for clear-cutting and replanting to even-aged monotypical blocks of fast-growing trees. The Forest Service is finally coming around to the idea of ecological forest management.

Muir's philosophy of wilderness preservation is equally obsolete. First, no less than Pinchot's, it perpetuates the pre-evolutionary strict separation of "man" from "nature." It simply puts an opposite spin on the value question, defending bits of innocent, pristine, virgin "nature" against the depredations of greedy and destructive "man." Second, it ignores the presence and the considerable impact of indigenous peoples in their native ecosystems. North and South America, for example, had been fully inhabited and radically affected by Homo sapiens for 10,000 or more years before European discovery (Deneven 1992b). And third, it assumes that if preserved an ecosystem will remain in a stable steady-state, while current thinking in ecology stresses the importance of constant, but patchy, perturbation and the inevitability of change (Botkin 1990).

Leopold's harmony-with-nature philosophy of conservation is the only twentieth-century North American philosophy of conservation that seems likely to be viable in the twenty-first century. It recognizes that human beings are as much a part of nature as any other species. But it would urge that, like most other species, we human beings learn to live symbiotically with our fellow-denizens in the various ecosystems that we inhabit. And it absorbs the enduring conservation value and the core of truth in the obsolete wilderness idea. Wilderness areas, originally set aside for outdoor recreation, scenic beauty, and solitude can best serve contemporary conservation as habitat for populations of species that, to remain viable, require deep undisturbed forest, extensive unplowed savannah and heath, uncompromised wetlands, and so on. But such areas may require invasive management—not "resource" management but ecosystem management. Prescribed burns, for example, may be necessary to manage savannahs and certain forests so as to maintain the mix of species that compose them.

From the perspective of Leopold's harmony-with-nature philosophy of conservation, what is ecosystem management? And how does it differ from resource management? First and foremost, resource management is commodity oriented. Forests are managed for maximum sustainable yield, ideally, of commercial timber and pulp to supply the building materials and paper industries. Wildlife, similarly, is managed for maximum sustainable yield of game species, not of all wildlife, to provide sport and meat for human hunters. (Yet another reason why "sustainable development" is an unfortunate label for the symbiotic relationship between people and land, envisioned by Leopold, is the inevitable confusion—especially in the minds of traditionally trained foresters and other resource managers—of "sustainable development" with "maximum sustainable yield.") Ecosystem management, on the other hand, aims, first and foremost, to maintain the health and integrity of ecosystems. Commodity production is a secondary and subordinate aim, to be pursued to the extent that it is compatible with maintaining the health and integrity of ecosystems.

This understanding of ecosystem management raises two more questions: What is ecosystem health?; and What is ecosystem integrity? Ecosystem (or "land") health was defined by Leopold (1949, p. 221) as "the capacity of the land for self-renewal." Currently the concept is understood to refer to the capacity of ecosystems to maintain their functions—such as sustaining biomass production, cycling nutrients, holding soil, and modulating stream flow (Costanza et al. 1992). This functional understanding better incorporates orderly ecological change than Leopold's more recursive definition. Let integrity, on the other hand, refer to an ecosystem's historic structure—its complement of component species in their characteristic numbers. Maintaining ecosystem integrity, so understood, is a more exacting norm of ecosystem management, since ecosystem functions may be little impaired by the incidental loss of non-keystone species, by the competitive exclusion of native species by exotics, or by the gradual and orderly change from one type of community to another.

In addition to directly managing ecosystems to maintain their health and integrity—by prescribed burns, afforestation, culling weedy species, excluding or eradicating exotics, protecting or reintroducing natives, and so on—ecosystem management entails managing human economic activities. It entails finding new ways of living on the land. Leopold himself was especially distressed by the increasing industrialization of agriculture during the mid-twentieth century (Leopold 1945) and looked for ways of making agriculture more compatible with ecosystem health and integrity (Leopold (1939b). Finding methods of harvesting timber that do not compromise the health and integrity of old-growth ecosystems is part of the current Clinton plan to resolve the jobs versus old growth conundrum in the Pacific Northwest (Egan 1993). Ecological range management might be achieved by removing all domestic stock and reestablishing native ungulates—bison, deer, antelope, and elk—in their historic numbers. Range "ranching," in such a scenario, might consist of erstwhile cowboys and -girls culling the herds, strictly regulated by the Fish and Wildlife Service or the BLM, and selling the meat on the expanding organic and gourmet foods market (Callicott 1991).

In sum, then, a human-harmony-with-nature conservation philosophy is more consistent with evolutionary and ecological biology than are both preservationism and resourcism. The ideal of this philosophy of conservation is to share the Earth with all our "fellow-voyagers . . . in the odyssey of evolution" (Leopold 1949, p. 109) and to provide all the Earth's species with adequate living space. As things presently stand, however, to do that, to nurture biological diversity at every scale, takes more than setting aside habitat. It requires ecosystem management, that is managing ecosystems primarily for their health and integrity, not for our commodity production. Since we human beings are part of nature, according to this way of thinking, human economic activities are not necessarily and by definition incompatible with ecosystem health and integrity. Complementing wildlands management we must aggressively pursue "sustainable development," that is, the initiation of human productive activities which are limited by ecological feasibility no less than by economic feasibility.

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Aldo Leopold and the Evolution of Ecosystem Management

Susan Flader[1]

Abstract — In his 1992 policy statement on ecosystem management, the chief of the Forest Service stated a principle to "strive for balance, equity, and harmony with land . . . by sustaining what Aldo Leopold called the land community." Aldo Leopold (1887-1948) developed his ecological approach to land management and his concepts of land health and a land ethic through a lifetime of observation, experience, and reflection.[2] His most penetrating observations came as a forest officer in the Southwest in the early 1920s, when he sought to understand the problem of soil erosion and the role of fire on forest watersheds. His forest inspection reports and other writings of the time reveal a strong commitment to ecological analysis and ecosystem-based objectives aimed at restoring the integrity of the landscape. But his experience within the Forest Service as he sought to develop objectives of management and principles of administration that would move the service beyond its traditional bounds suggests that implementing an ecosystem approach to management on national forest lands may not be easy.

In his 1992 directive on ecosystem management, setting forth a new management philosophy to guide the national forest system as it enters its second century, chief F. Dale Robertson declared a principle to " strive for balance, equity, and harmony between people and land . . . by sustaining what Aldo Leopold (1949) called the land community." In this significant document Aldo Leopold is the only person besides Gifford Pinchot to be named and his 1949 classic, *A Sand County Almanac*, is the only publication referenced. Consider the role Leopold occupies in the progression of ideas that have guided the Forest Service, as defined by Chief Robertson: Gifford Pinchot is credited with articulating the conservation philosophy that underpinned national forest management from the inception of the Forest Service in 1905, the conservation approach was augmented by the multiple-use philosophy enshrined in law in 1960, and now Aldo Leopold's enlarged concept of the land community, expressed most clearly in his essay on " The Land Ethic" in his 1949 book, has been identified by the chief as the basis of the ecosystem management philosophy that will take the national forests into the next century. Clearly, Aldo Leopold (1887-1948) was a man ahead of his time. Just how far ahead is apparent when we reflect that he made some of the greatest advances in thinking about system-based land management during the first fifteen years of his forty-year career, especially during the early 1920s.

This paper will examine Leopold's experience as a young forester seeking to understand the dynamics of a landscape subject to everchanging physical, biological, and cultural forces and, in his definition of the problem and his proposed course of action, pushing the still youthful Forest Service farther and faster than it was prepared to go. We will seek to understand something of what may be involved in implementing an ecosystem approach to management within an agency culture that is, if anything, more entrenched now in its traditional ways than it was during Leopold's time. But we may also come to appreciate through this story that the Forest Service long has had among its traditions a refreshing openness to mavericks like Aldo Leopold.

Aldo Leopold graduated from the Yale Forest School in the class of 1909 and left that summer for his first assignment in the Forest Service, as forest assistant on the Apache National Forest in Arizona Territory (Flader, 1974; Meine, 1988). The Apache was one of a series of newly created forests straddling the highlands along the Mogollon Rim that trended from the

[1] *Susan Flader is Professor of American Western and Environmental History, University of Missouri-Columbia.*

[2] *This paper is based in large part on original, unpublished manuscript material from Forest Service records and Leopold's personal papers, most of which may be found in the Aldo Leopold Papers, University of Wisconsin Archives, Madison, Wisconsin.*

Prescott Forest near Flagstaff three hundred miles southeastward to the Gila in New Mexico Territory. It was the region where Leopold would make the most telling observations of his fifteen-year career in the Southwest, and he began that first summer, ravenously absorbing impressions of watersheds and wildlife and history and culture as well as of board feet of ponderosa pine. He led a reconnaissance party that cruised timber along the route of a proposed road from Clifton to Springerville that would have to clamber high over the mountains because an earlier route up the valley of Blue River had been washed out by severe flooding and erosion. It was his first introduction to the realities of erosion in the Southwestern environment that would shape so much of his thinking about system-based management.

Leopold had trouble on that first assignment, enduring a months-long personnel investigation into charges of incompetence and inefficiency in his handling of the reconnaissance crew. But technically trained men were scarce in those days; his superiors accepted half the blame for his missteps and gave him another chance. Having learned that the service expected absolute adherence to administrative procedures and the minutiae of management, he did so well on his second chance that in 1911 he was appointed deputy supervisor and a scant year later supervisor of the Carson National Forest in northern New Mexico. He was age 25.

The course of Leopold's life was changed by an attack of acute nephritis, a kidney disease, resulting from exposure on an arduous trip to settle range disputes in April 1913. He nearly died; and, given the state of medical opinion at the time, he had to give up all hope of resuming the strenuous life of a forest supervisor in roadless mountain terrain. During eighteen long months of convalescence, much of the time back home in Burlington, Iowa, he had ample time to reflect on the meaning of Forest Service work, and he shared some of his thoughts with his compadres back on the Carson in a series of letters published in the *Carson Pine Cone*, a newsletter he had founded:

> *After many days of much riding down among thickets of detail and box canyons of routine, it sometimes profits a man to top out on the high ridge leave without pay, and to take a look around.*

Leopold had already learned that the Forest Service was dedicated to its thickets of detail and its maze of routines. But what was the measure of success in forest management? "My measure," wrote Leopold, "is THE EFFECT ON THE FOREST." Too often, it seemed to him, foresters fell into a rut of routine, following the prescribed procedures without considering the objectives. The rangers on the ground, as he saw it, had the responsibility to apply the stated principles of forest management in detail on particular areas and to monitor and gauge their effect on the forest. To Leopold, the greatest necessity was for "clear, untrammelled, and independent thinking on the part of Forest Officers."

Gifford Pinchot's management philosophy is often termed "scientific management." Knowing that Leopold and other early foresters were trained in botany and silviculture that were just then beginning to incorporate early concepts of ecology such as forest succession and climax, it is perhaps natural to assume that scientific management meant management according to principles of botanical or even ecological science. Yet a careful reading of early Forest Service administrative correspondence makes it clear that the term "scientific management" referred to the principles of *industrial* management just then being articulated by the time-and-motion-study expert Frederick Winslow Taylor (1911). Leopold himself was attracted to Taylor's ideas by the early 1920s, when he became chief of operations for all national forests in the Southwest, and he participated avidly with his operation counterparts in other regions of the country in a round-robin discussion of the application of Taylor's ideas to forest administration.

This is not to say that there was no biological basis for early forest management. There was, but the biological basis—which was in part ecological—was simply *assumed* (Flader, 1976). As such, it was scarcely open to question. The ecological concept of forest succession, the admonition to harvest only individually marked, mature trees, usually of climax species, the absolute control of fire (which set back forest succession)—all these notions were part of the ideology of American forestry—dogmas assumed as givens, with little need of further testing or research. Another given was the doctrine of forest influences, the belief that forest tree cover at the headwaters of streams was crucial in preventing destructive flooding and erosion downstream. These doctrines, and more, would become open to question as Leopold began to think hard about what he saw happening on the ground in his new role as chief of operations and principal inspector for twenty million acres of national forests in the Southwest.

For five years, 1919-1924, Leopold criss-crossed the forests of the Southwest, usually on horseback, observing conditions nowhere more trenchantly than on "that tumbled sea of pale blue hills" along the Mogollon Rim, where he had first encountered the Southwest as a timber "reconnaisseur" a decade earlier. The reports of his earliest inspections were sketchy, though he made it clear that he was still looking for imagination and initiative on the part of forest rangers and was determined to judge their success by the effects of management on the forest. In particular, he began noting the effects of gullying and soil erosion on forest ranges and arguing for actual work on the ground to test and improve techniques of management. By early 1920, during an inspection of the Prescott National Forest, he wrote home to his mother that he was "seriously thinking of specializing in erosion control. The problem is perfectly tremendous here in the Southwest and I seem to be the only one who has any faith in the possibilities of tackling it successfully."

Despite his inclination to deal with real problems on the ground, Leopold in his new operations post again had to overcome doubts about his administrative ability on the part of both subordinates and superiors, who thought of him as highbrow and inattentive to detail, moving along "with his feet somewhat off the ground." As he had several times previously, Leopold declined transfers to other regions or positions in order to prove he could master the job as chief of operations. He proved it by designing and implementing a new, more systematic method of forest inspection, complete with printed, notebook-sized tally sheets for recording a myriad of details on everything from the cleanliness of outhouses to the condition of grass and sod on pastures. In the area of fire control alone, his tally sheets increased the number of observations required of the inspector from twenty points under the old system to 165 points in the new, all minutely classified as to subject and administrative unit and designed to facilitate comparisons year by year and forest by forest (Leopold, 1921).

Leopold's superiors in the Southwest and in Washington were impressed—all those details, so efficiently catalogued. They were particularly impressed with the first field test of the new system, his 1922 inspection of the Gila National Forest. This was the now-famous report in which he recommended a wilderness area policy for the Gila and drew a red line on the map to indicate the limits of motorized accessibility—the first step toward designation of the Gila in 1924 as the prototype of national forest wilderness areas. But it was not the wilderness area proposal that attracted his superiors' attention to the Gila report; rather, it was the painstaking detail and comprehensiveness of the inspection itself.

Leopold considered his contributions to the development of a forest inspection system for the Southwest to be "one of the two or three points" in his own Forest Service career that gave him the greatest satisfaction. But, while his supervisors may have been impressed by his systematic attention to detail—what Leopold called the *machinery* of inspection—to him what still mattered most was the *results on the ground.*

In an address on "Forest Inspection as Developed in the Southwest" presented to the New York Forest Club (1924b) he tried to grapple with the inherent difficulty of expressing what he was trying to accomplish in language that could as readily be applied today to the difficulty of defining ecosystem management:

> *It is always difficult to flatten out upon a printed page a system of thoughts and facts which are concentric to a single idea. Their relationships to that idea and to each other are actually expressible only in three-dimensional space. The flattening process inevitably severs many of these relationships and leaves them at loose ends.*

And then the kicker, whether for inspection or for ecosystem management: "The only way to really see it is to watch it work on the ground." Inspection to Leopold was a technique for diagnosing local problems and monitoring the effectiveness of management solutions.

Even as Leopold was developing his inspection system he was also struggling, especially through his repeated forays into forests along the Mogollon Rim, to understand the dynamics of southwestern watersheds and to consider the implications of his findings for conservation policy and social values. These lines of endeavor came together incrementally during his inspections, but nowhere more so than on the Prescott in 1922. By this time he had already tallied thirty mountain valleys in southwestern forests and had found twenty-seven of them damaged or ruined. Where several years earlier he had thought artificial controls such as check dams, willow plantings, and plugging gullies might be the answer, he was now much more intent on understanding the "virgin state" of the watersheds and the causes of erosion in order to determine the appropriate objectives of management.

When he compared observations in the field with Prescott supervisor Basil Wales and local rangers, he discovered that they had significantly different interpretations of the history of the area and hence different notions of what management should seek to accomplish. Where Wales and his rangers thought the grass cover had always been thin on the granite soils of the Prescott and assumed, like most foresters of the day, that grazing pressure was essential in order to hold down the fire hazard of brush, Leopold saw evidence in the fire scars of ancient juniper stumps to conclude that fire had been a recurring feature of the virgin landscape. The grass cover had been much heavier and the brush much thinner than at present, he surmised, owing to grass fires and grass-root competition. In his view, it was overgrazing and trampling by cattle that had thinned the grass, thus inhibiting the fires and initiating both the destructive erosion and the encroachment of brushfields that were now a severe fire hazard. Leopold's interpretation, it should be noted, flew in the face of virtually the entire corpus of scientific dogma in the Forest Service of his day.

More to the point, differences in interpretation called into question the objectives of management. "If the prime objective is wood products," Leopold wrote, "we may continue to overgraze, letting in the woodland and sacrificing watershed values. If on the other hand the prime objective is watersheds, we should restore the grass, which all the evidence indicates is a better watershed cover than either brush or woodland." So struck was Leopold with the problem of determining the proper objectives of management for particular areas that he sat down to draft a paper on what he now called "Standards of Conservation" (1922c), using examples from the Prescott.

Here Leopold was dealing with the fundamental problem of ecosystem management—the problem of specifying objective standards of conservation. But he never finished the paper. In fact, it ends in mid-sentence, just as he was trying to explain how one might use management plans to set the standards of conservation. Baird Callicott (1991) has suggested that perhaps when Leopold got to this point he may have said to himself "Who are you kidding?" and simply put down his pencil. He realized that as long as Pinchot's utilitarian calculus prevailed in the Forest Service, even the most sophisticated science would not suffice to set objective standards. As Callicott put it, "The paper self-deconstructs, so to speak."

Aldo Leopold would devote the remaining quarter century of his life to working on the scientific and philosophical problem of determining the objectives of conservation. His first significant effort came several months after his inspection of the Prescott in a paper titled "Some Fundamentals of Conservation in the Southwest" (1923), in which he first sought to draw his observations about vegetation change and soil erosion into a cultural and philosophical context. It was here that he first expressed his intuitive sense of a living earth and addressed the implications of conservation as a moral issue. But again he did not publish, whether because of uncertainty about the philosophical argument or, just as likely, because of criticism from colleagues about his analysis of the problem of erosion. Instead, he turned his formidable analytical and writing skills to explaining more clearly the processes at work on southwestern watersheds and the implications for management in a series of papers that drew on his observations of forests along the Mogollon Rim. In a *Journal of Forestry* article (1924a) titled "Grass, Brush, Timber, and Fire in Southern Arizona" he issued a direct challenge to Forest Service dogma: "Fifteen years of forest administration were based on an incorrect interpretation of ecological facts and were, therefore, in part misdirected." In another paper, "The Virgin Southwest and What the White Man Has Done to It" (1927) he drew evidence from the accounts of early explorers along with his own uncanny skill at reading history backward in the land to sketch a vision of what the Southwest had once been and hence what management might aspire to restore.

Then in 1933 in his well known essay "The Conservation Ethic" he returned to the theme of conservation as a moral issue, this time thoroughly grounded in an understanding of the dynamic functioning of interrelated elements of the system, physical and biological, natural and cultural, through time. It was this essay, significantly enhanced by a clearer statement of the concepts of land health and the biotic community, that became his celebrated essay "The Land Ethic," first published in *Sand County Almanac* a year after his untimely death at age 61. And now it is his "land ethic" philosophy that is presumably pointing the way to the future in Chief Robertson's directive on ecosystem management.

If we would look for guidance as to the fundamental objectives for ecosystem management, we could do no better than to start with Aldo Leopold's famous dictum in "The Land Ethic": "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise."

But if we would look for insight as to what might be involved in actually *implementing* an ecosystem approach to management on national forest lands, we might rather recall Leopold's insistence on setting specific standards of conservation for each area through careful observation, historical study, and scientific research and then monitoring and evaluating the effect on the forest. Leopold's own experience as a forest inspector in the 1920s striving to comprehend processes of ecosystem change along the Mogollon Rim in Arizona and New Mexico might be our guide, informed by his even earlier call for "clear, untrammelled, and independent thinking."

For reasons that have never been satisfactorily explained, Leopold left the Southwest in 1924 to assume a new position as associate director of the Forest Products Laboratory in Madison, Wisconsin, then the principal research arm of the Forest Service. Never particularly happy in an institution devoted to utilization of the tree after it was cut when everything about him made him interested in the forest as a living community, he elected to leave the service in 1928 to devote himself more fully to his consuming interests in wildlife and conservation.

By any standard, Leopold had enjoyed an extraordinarily successful career in the Forest Service. Despite some challenges along the way, he had won the respect of colleagues up and down the line for his unswerving loyalty to the agency, his dedication to its mission of conservation, his obvious administrative skills, his open, ever-questioning mind, and his vision for the future. Because he never gave up trying to move the service farther and faster than it was prepared to go, he actually moved it farther than his colleagues at the time would have thought possible. But he was under no illusions as to the distance yet to be traveled.

The day before he retired from the Forest Service in June 1928, the *Service Bulletin* (a house organ) published a response by Leopold to a critic of his wilderness proposal that may stand as his valedictory challenge to the agency:

> *The issue is whether any human undertaking as vast as the National Forests can be run on a single objective idea, executed by an invariable formula. The formula in question is: Land + forestry = boards. . . .*
>
> *Whether we like it or no, National Forest policy is outgrowing the question of boards. We are confronted by issues in sociology as well as silviculture,- we are asked to show by our deeds whether we think human minorities are worth bothering about; whether we regard the current ideals of the majority as ultimate truth or as a phase of social evolution; whether we weigh the value of any human need . . . wholly by quantitative measurements; whether we have forgotten that economic prosperity is a means, not an end.*

To Aldo Leopold, the decision—at that time regarding wilderness, today concerning ecosystem management—would indicate whether the Forest Service was simply a bureau that executed the laws, or "a national enterprise which makes history." Naturally, Leopold challenges us to make history.

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Normative Conflicts and Obscurantism in the Definition of Ecosystem Management

Michael Soulé[1]

Both conservation biology and the natural resource disciplines are normative, and both are concerned with the conservation of biological diversity, but there are differences in their fundamental values. Conservation biologists are generally more concerned with protecting the entire range of biotic diversity, whereas natural resource professionals are committed to providing resources, including commodities. These different missions are not always reconcilable, notwithstanding attempts to define "sustainability" and the "ecosystem approach" in ways that imply that human needs and biodiversity can be harmonized. One of the reasons for the current popularity of "ecosystems management" among politicians is that it is proactive. But another reason is that its objectives are vague and not usually tied to conflict-generating endangered species. One explanation for the vagueness is that there are at least five, distinct, definitions of ecosystem management: (1) description and classification of plant/animal associations, (2) providing ecosystem services, (3) maintaining ecosystem integrity, (4) ensuring the continuation of ecosystem processes and natural disturbance regimes, and (5) balancing human needs and conservation, per se. A critical analysis will show that none of these approaches to ecosystem management, alone, is realistic unless it is based on the management of single species. With the exception of prescriptive burning, most management interventions are based on the ecological requirements of single species. Constant repetition of the mantras of "sustainability" and "holistic ecosystem approach" will not, alone, lead to a truly synthetic, ecological approach to management. In fact, management will always be site specific, and based on single species. The current fashion of species bashing is anti-scientific and provincial, especially in view of the environmental conditions in many tropical nations and the high probability that many large animals will not persist in nature in large regions of the world during the coming "demographic winter."

[1] *Board of Environmental Studies, University of California, Santa Cruz, USA.*

Biodiversity and Land Use

Neil E. West[1]

Abstract — Biodiversity is a multifaceted phenomenon that is increasingly incorporated into the inventory, planning, management and monitoring of wildlands throughout the world. A view of all facets of biodiversity, at multiple scales in time and space, is required to understand the tradeoffs that come from either a manager's action or inaction. It is impossible to simultaneously optimize for all aspects of biodiversity. All biotic and environmental variables are dynamic, preventing us from ever bringing biodiversity into stasis. We are thus being forced into prioritizing which features of biodiversity take precedence at particular places and times. Earlier choices influence later options possible, especially if extinction ensues. Since these are ultimately moral choices, far more than scientific understanding is involved. Conserving and enhancing biodiversity must become an integral part of all land management, not just on passively mismanaged reserves. Both public and private lands hold and benefit from biodiversity. Management with sensitivity to biodiversity will require partnerships, cooperation and integration beyond any past experience. The California Council of Biological Diversity is a leading example of how this might be done.

INTRODUCTION

Biodiversity is currently one of the most frequently used terms in both popular and scientific discussion. Concerns for biodiversity started out in the 1970's with focus on threatened and endangered species, but has been progressively broadened until all facets of the variety of life on earth have been included. The burgeoning of public interest has far out paced the abilities of both scientists and land managers to define, evaluate, and manage for biodiversity. The publicly perceived needs are so great that we scientists and managers have not been given much time to carefully think through the issues. Consequently, there has been some confoundment of definition with application (Landres 1992). It is my first purpose here to try and separate definitions from applications, particularly with regard to land management.

While humans have been using lands for a long time, the degree of use and extent of transformation has been accelerating. Today, there is no part of the earth that escapes at least secondary human impacts. The regions of earth differ only in the degree of alteration. Today, about 11% of the earth's land surface is in intensive agricultural use, about 24% is grazed by domestic animals, and about 3% is occupied by urban and industrial developments. The world's forests occupy about 31% of the land. The remaining 31% is occupied by deserts, tundras, rocky barrens or ice or snow where hunting and mineral extraction can still occur (Solbrig 1993).

With the development of more efficient growing and harvesting techniques constantly emerging, combined with exponential growth of the human population and both their real and perceived needs, the rates of land alteration and loss of biodiversity have become magnified and are leading well beyond the largely localized impacts of the past. We have recently come to realize that establishing and managing conservation and preservation type reserves will never be sufficient to maintain biodiversity. There is also an upper limit to how much of a landscape people will tolerate being put in reserves. We thus have to learn how to accommodate biodiversity in the management of multiple use and agricultural lands (Franklin 1993), both publically and privately owned.

DEFINITIONS

Before we can begin to adjust land management to accommodate biodiversity on the vast majority of our landscapes, we need to realize that biodiversity is a concept cluster (Figure 1). That is, many separate and yet interlinked

[1] *Neil E. West is Professor, Department of Range Science, Utah State University, Logan, UT.*

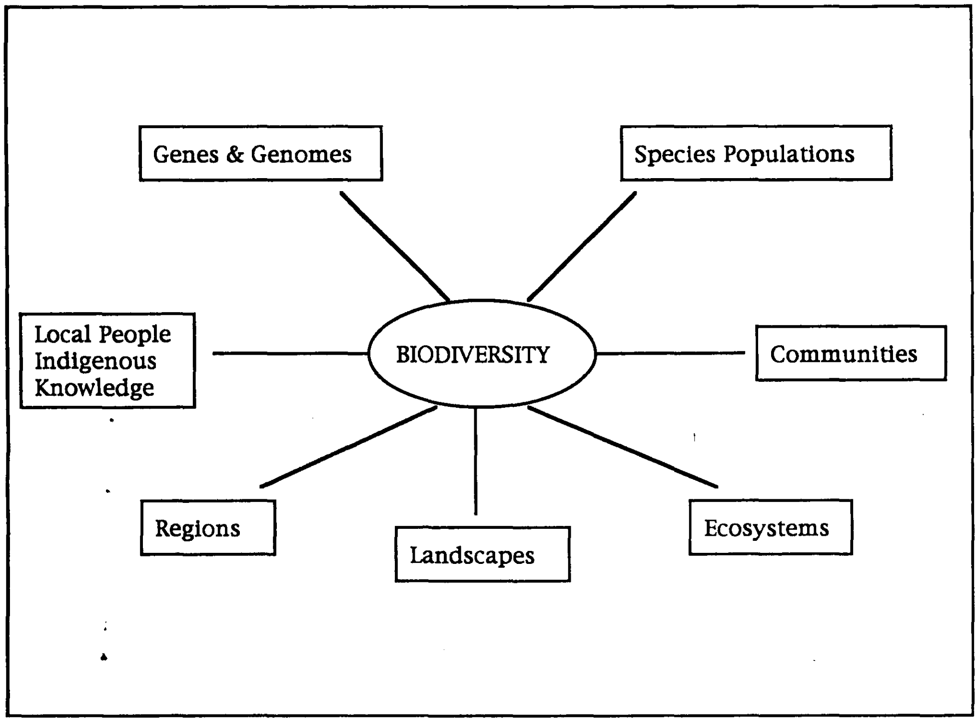

Figure 1. — Components of biodiversity.

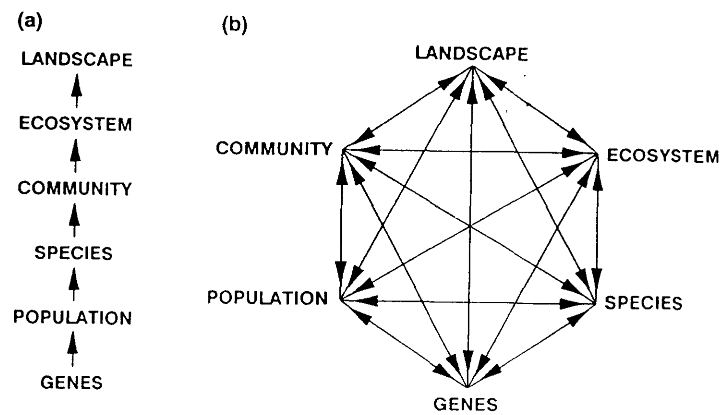

Figure 2. — Comparison of (a) linear, hierarchial view of the elements of biodiversity, and (b) interactive view of the elements of biodiversity, showing all possible pair-wise combinations of interactions (from Landres 1992).

phenomena are involved, from gene flow between individuals to ecosystem processes. These phenomena all operate on widely different but covarying scales in space and time.

Phenomena that occur at larger spatial scales tend to occur at longer temporal scales, although important exceptions arise. For instance, ecosystem phenomena can occur within organisms as well as over large expanses of land (Allen and Hoekstra 1992). Mankind has forced many processes off the natural tendencies expressed by the slope of usual temporal-spatial expectations. For instance, extinction has usually been a slow process occurring over geological eras. Our activities have now greatly accelerated permanent loss of species.

Human size, visual acuity, and life span bias us toward certain observables which are not necessarily the most important features or processes. Our past excessive focus on charismatic megafauna is an example of this. Detection of gradual changes at larger spatial (e.g., regions) and longer temporal scales (e.g., several decades) is inherently difficult because we can't directly sense them.

We have also begun to realize that the linear, hierarchial view of biodiversity (Figure 2a) is flawed. Considerations of larger scaled phenomena need not always "bubble up" from lower levels. Important interactions frequently transcend the simpler hierarchy visualized by Figure 2a, allowing acknowledgement of important feedbacks such as species introductions on ecosystem functions, predation on gene flow of prey populations, etc. (Figure 2b).

Another flaw in the usual hierarchial view of biodiversity (Fig. 2a) is that it implies a mechanistic view of ecosystems (Botkin 1990). Ecosystems are more than simply a sum of their parts. The interactions and net activity are the important consequences. Managing for species misses this point. For instance, while managing for a long-lived "umbrella" species will likely maintain the appearance of fully functioning ecosystems over a single human lifetime, processes such as nutrient cycling and evolution may be impeded if crucial influences such as fire and hydrologic regimes are altered. Thus, managing with biodiversity in mind involves much more than simply maintaining native species or ecosystem processes. All levels and all interactions must be considered kinetically. That is, nature is dynamic with only tendencies toward equilibria that are usually never reached (Kaufmann 1993). Changing climates, genetic pools and mixes of species in communities ensure that stability is wishful thinking.

The discounted notion of the balance of nature as a single static point of ecosystem development is very recalcitrant (Pimm 1991). Renewable natural resource scientists have much educating to do in explaining the more complex notions of modern ecology to both resource managers and the public (Kaufmann 1993).

INTERACTIONS OF BIODIVERSITY AND LAND USE

Trying to globally generalize about how biodiversity is related to land use is overwhelming. Each situation draws a unique combination of biota, environment and sociological economical and political circumstances. Unique combinations of biota, and environments, are juxtaposed against sets of specialists speaking their own jargon, preferring their own familiar measurements and pushing their own agendas, hidden or open.

In general, there has been a trend toward biological simplification and cosmopolitization. Accelerated erosion, salinization and pollution of soils, have generally had negative down stream impacts on water bodies, both at and below the soil surface. Mankind has appropriated up to 70% of the world's net primary productivity (Vitousek et al. 1986) and is placing increasing emphasis on fewer and less genetically diverse primary producers.

Diminishment in vegetation richness, structure and production usually leads to diminishment of animal and microbial contributions as well. This is because plants are both food resources and habitat for heterotrophic organisms. The relationships are, however, far from simple and linear. Some treasured species are dependent on disturbances caused by others. For instance, the blackfooted ferret is a carnivore dependent on prairie dogs. Prairie dogs thrive only where prairie is heavily grazed by either bison or cattle (Archer et al. 1987). Because of perceived competition between cattle and prairie dogs for forage, the rodents have been reduced to the point that now occupy only about 2% of the area they covered prior to the coming of European man to North America. Hence, the endangerment of the ferret. This is a good example of the principle that not all facets of biodiversity can be simultaneously optimized with economically viable human use of the land. The challenge is to find the levels of compromise that will accommodate both some retained biodiversity and human needs now and into the future. The details of how this is done will vary enormously across the globe.

ACCOMMODATING BIODIVERSITY IN LAND MANAGEMENT

Few care to consider triage in dealing with biodiversity because it admits being party to some loss of human control. The inertia associated with the human population already here and the unlikelihood that it will stabilize anytime soon makes inevitable much loss of biodiversity especially in developing countries (West 1993). Americans should not be telling the rest of the world what to do if they can't lead by example. Accordingly, let's turn to how we in the U.S. can cope with diminishing biodiversity while simultaneously managing land for more direct values.

John Wesley Powell warned us in the last century that we would be wise to make boundaries of political subdivisions congruent with natural ones. This advice wasn't taken and we're now paying the price for some expediency taken by our predecessors. Biodiversity issues are forcing us to forge new institutions to deal with the reality of natural boundaries. Development of these institutions is most urgent in the most rapidly developing parts of our country because the results of continued fragmentation and other alterations of landscapes are most apparent there. I feel that it behooves those in the relatively less impacted Intermountain West and Great Plains to become aware of how biodiversity is being dealt with elsewhere. Learning from both successes and failures could enhance our ability to deal with these emerging issues. California Governor Pete Wilson's style of "preventive government" is worth observing (Wheeler, in press).

The California Example

A political majority in California finally came to realize that sustaining in acceptable condition its enormous biodiversity was a prerequisite for maintaining its economic prosperity. Rather than continue focusing protection efforts on particular species at specific sites, California has found means to identify and deal with whole biogeographical regions involving many ownerships and political jurisdictions. This action has been taken after several decades of tortuous, expensive, piece-meal activities focused on individual species, sites, and resources. A more effective approach was conceived as the bioregional strategy.

Statewide

The bioregional strategy involves a hierarchial approach, allowing co-ordination, information exchange, conflict resolution, and collaboration at state to local levels. The top most group is the statewide Executive Council on Biodiversity. This council is chaired by the Secretary of the Resources Agency of California and is made up of the highest officials of the California State Departments of Fish and Game, Forestry and Fire Protection, Parks and Recreation, State Lands Commission, the University of California's Division of Agriculture and Natural Resources, the U.S. Forest Service, U.S. Fish and Wildlife Service, National Park Service, and the Bureau of Land Management.

The executive council sets statewide goals for the protection of biological diversity, recommends consistent statewide standards and guidelines, encourages cooperative projects and sharing of resources and cooperation in developing biodiversity related policies and regulations, land management, land use planning, land reserve acquisition and exchange, private landowner assistance, educational outreach, public relations, and staff training, monitoring, inventory and assessment, restoration and research and technological development. The council meets quarterly to review progress in accomplishing its mission. Representatives of other state and federal agencies and special interest groups are frequently invited to participate in these meetings to help enhance consensus and participation in the adoption of bioregional strategies.

Bioregional

One of the earliest outputs of the statewide council was the establishment of bioregional boundaries (Figure 3) and associated bioregional councils. The bioregional councils are composed of regional administrators of the signatory agencies. These ten regional councils develop regional biodiversity

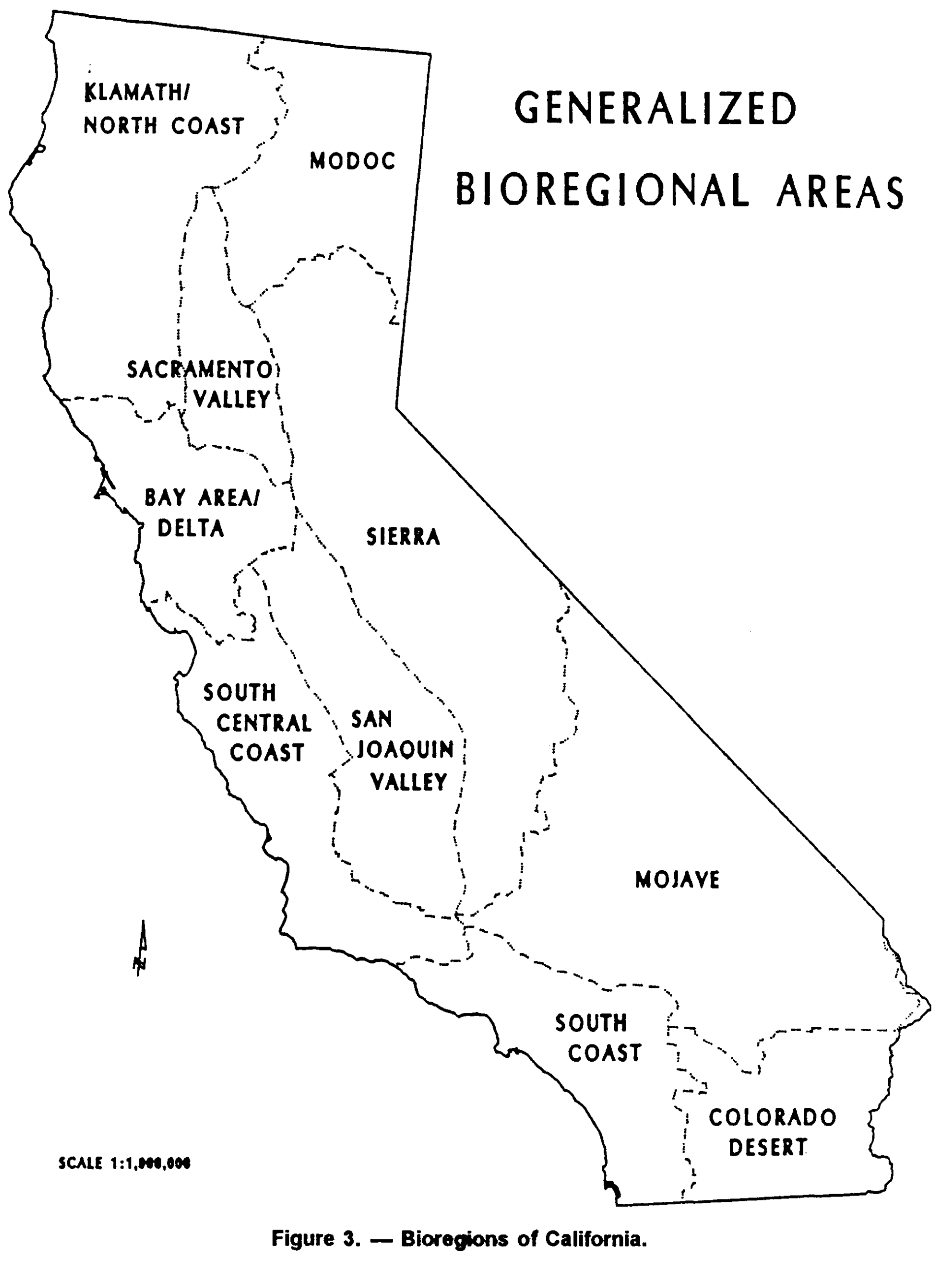

Figure 3. — Bioregions of California.

strategies that incorporate the policies, principles and activities of the state council. Regional solutions to regional issues and needs are encouraged, consistent with statewide goals and standards. The regional councils work with regional and local (mostly county) authorities to implement biodiversity policies. The regional councils, in turn actively encourage the development of watershed or landscape associations to assist in implementing regional strategies applying to part of each region.

Local

Local staffs of signatory agencies assist in formation of watershed or landscape associations. Along with local public, landowner and private organizations, specific cooperative projects are devised to achieve objectives that translate upward to the region and state.

There has already been a relatively long history of locally coordinated land or resource management planning going on in the western U.S. where mingled ownership and disproportionate use or impacts has provided incentives to cooperate (Anderson 1977a&b). What is new is the addition of concern for biodiversity. Since local biodiversity is inevitably tied into regional, state, national and even global concerns, California is showing us a way of expanding coordinated resource management planning upwards.

Most people first learn of biodiversity when a local controversy emerges. The usual scenario has been when a listed species impedes economic development. Rather than continuing these costly and exhausting species by species battles, it is time to consolidate and coordinate information and plan more general, longer-lasting solutions. Better public education, dialogue, and participation could minimize the disruption of human communities and expectations.

Guidance from regional and state councils is helpful in setting standards for defining and measuring baselines of biodiversity and providing experience in negotiating solutions. The tools include mitigation, development banks, planning and zoning authorities, land and reserve acquisition, incentives to private landowners (e.g. purchase of conservation easements), alternative land management practices, restoration and fees and regulation.

Coachella Valley

There have already been several successfully resolved local biodiversity situations in California. One example is the Coachella Valley Preserve System near Palm Springs. This solution was provoked by need to preserve habitat for the fringe-toed lizard (*Uma notata*). A 13,000 acre sanctuary was created while allowing for managed development of human use in part of the lizard's habitat. This cooperative effort involved federal, state and local government as well as citizen groups and private developers. The Preserve is jointly owned and managed under a Memorandum of Understanding among the Bureau of Land Management, US Fish and Wildlife Service, California Department of Fish and Game, Department of Parks and Recreation, and the Nature Conservancy, the latter which acts as the coordinating agency.

Many other completed and on-going local efforts could be show-cased, but space is limited here. We can conclude from the California experience that maintaining both biodiversity and economic viability involves a landscape approach. Actively managed reserves retain as much biodiversity as possible while more sensitive management of the remaining, much higher fraction of the landscape in the multiple use or agricultural categories provides buffers to the reserves and integrity for the entire wildland portion. Part of the landscape must continue in intensive use for food production and space for human occupancy, travel and transportation corridors (Figure 4). While little biodiversity remains in the tamed areas, the biodiversity that they once contained is now largely present on the wildlands and not lost entirely. While the resulting mix of land use categories is not like that prior to when European man entered the scene, the strategy allows the current human population to live while considering what will be around for future generations.

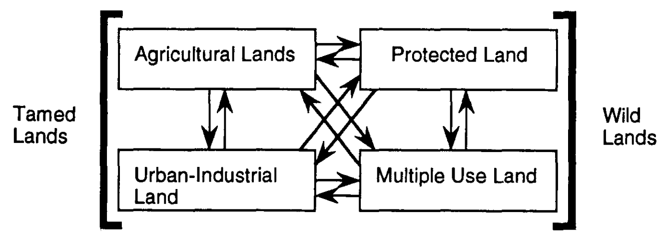

Figure 4. — A compartment model of land use categories for planning and land management based on ecological theory. Modified from Odum (1969).

The Future

I wish to complete my overview of this topic by speculating on how I think biodiversity will be accommodated in the U.S. in the future. Mainly because actions aren't usually taken until species are on the brink of extinction, dissatisfaction with the Endangered Species Act is building. It seems that an Endangered Ecosystems and/or Landscapes law will eventually replace it (Orians 1993). Major land management agencies such as the Forest Service and Bureau of Land Management are already rapidly moving toward a focus on ecosystems as a basis for management, making this possibility somewhat easier.

It seems only logical that a California-like approach to dealing with biodiversity will come into use elsewhere. California's great inherent biodiversity, plus large and rapidly growing human population has forced them into earlier action. The severe loss of total area of some ecosystem types and rapid fragmentation of others, means that there is little time to waste in preserving biodiversity in California. It is cheaper, but not necessarily easier, to take a proactive stance, such as California has done, rather than wait until most natural systems are lost and then expensively try to restore some semblance of a natural system later. Extinction of critical species could make restoration or even rehabilitation impossible.

The recent Wildlands Project for the Coast Range of western Oregon is an even bolder attempt than California has taken to be proactive concerning biodiversity (Mann and Plummer 1993). This proposed zoning into core areas, buffer zones and corridors would displace humans now living on some of that land. The fate of this proposal will reflect both the strength of our science and the will of the American public on this topic.

It also seems inevitable that some national leadership is needed to deal with biodiversity issues that cross state boundaries. This seems to be a natural role for the proposed new Bureau of Biological or Ecological Survey within the U.S. Department of Interior. Along with other agencies such as the Forest Service, the Environmental Protection Agency, and the Department of Defense, they could co-ordinate on multi-state scales and complete the hierarchy that state and federal governments have established in California. The Nature Conservancy is a natural for an expanded role in meditating interstate disputes.

The data generated by the GAP analysis (Scott, in press) gives us a start in identifying and ranking land areas for closer management of biodiversity values. The Environmental Protection Agency's EMAP (Environmental Monitoring and Assessment Program, Messer et al. 1991) should begin to bring us nationwide feedback on biodiversity, as well as environmental conditions nationwide.

These are the kinds of efforts that we can begin to showcase worldwide. When we can concretely demonstrate our willingness to adjust American land use practices in the interests of biodiversity, then we can legitimately begin to offer assistance to developing nations to begin taking similar actions.

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Sustaining and Restoring Western Wetland and Riparian Ecosystems Threatened by or Affected by Water Development Projects

David J. Cooper[1]

Abstract — The most abundant wetland and riparian ecosystems in the West are marshes in intermountain basins and floodplain forests and shrublands along the major rivers. These ecosystems are impacted by existing dams, water diversions and ground water pumping. This paper reviews issues related to the affect of water development projects on wetlands. Marsh wetlands in Colorado's San Luis Valley occur where stream flow from adjacent mountain ranges builds ground water mounds on a regionally high water table bringing the water table to the ground surface. Wetlands occur where shallow flooding or seasonally high water tables occur. Strong plant species zonation occurs along the depth to water table gradient and the entire diversity of wetlands occurs in a range of 5 vertical feet. Direct gradient analysis indicates that a long-term average water table drop of as little as 0.66 feet will cause a shift in wetland plant species and a water table drop of more than 5 feet could result in the loss of entire wetland complexes. Along Colorado River floodplains Fremont cottonwood dominated forests are the most characteristic vegetation type. These tree rely on high water tables for vigorous growth and periodic flooding for recruitment of new individuals. Western wetland ecosystems do not appear sustainable where hydrologic changes decouple marshes from their water source, or where insufficient stream flows for seedling establishment and vigorous growth occur.

INTRODUCTION

Humans have interacted with landscapes in the Rocky Mountain and intermountain West for thousands of years. The use of fire, small scale irrigated agriculture and hunting were common. When Caucasians came to the West in the late 1800's the scale of landscape use changed abruptly. The quest for gold, beaver and timber came first and did not sustain a large or stable population. The pervasive aridity of this vast region kept the human population small as well until they learned that the most precious resource is water and this resource could be exploited to allow development as no other resource had previously.

[1] *David J. Cooper is a Research Scientist, Department of Fishery and Wildlife Biology, Colorado State University, Fort Collins, CO 80523.*

Caucasion settlers to the West met landscapes that were either too dry, or seasonally too wet, for the European style agriculture that they wished to develop. With the assistance of federal, state and local governments settlers labored continuously over the past 100 years to make this land of extremes more mesic and suitable for their industry. The original concept of "reclamation" used in the United States was reclaiming these unusable lands from nature and putting them to practicable use. In general, this required massive hydrologic changes to the rivers, groundwater systems and even drylands across this region.

To make the arid deserts and semi-deserts more mesic an extensive system of dams, reservoirs, water diversion structures, canals, ditches and irrigation systems was constructed. To make seasonally flooded and saturated lands more mesic flood control dams, drainage ditches and drain tile systems were built and installed. These systems survive intact today and more are built each year.

While these changes have made modern agriculture and settlement possible riparian and wetland ecosystems have suffered dewatering and drainage. The Colorado River has been called the world's largest plumbing system (Graf 1985) and today is controlled by numerous mainstem dams, diversions and reservoirs. Even small tributaries have been modified for local and regional agricultural and municipal use and few tributaries retain their natural hydrologic regime. As a result few riparian and wetland ecosystems below 8,000 feet elevation still function as they did prior to settlement.

The hydrologic regime that drives these riparian ecosytems begins in high mountain watersheds that naturally store water as snow from October through April which melts rapidly during May and June producing the flood pulse that characterized western rivers. Today impoundments store this snowmelt water for use later in the summer. The highest annual flows on regulated rivers now occurs in July or August as demand for irrigation water or electricity drive river flow patterns.

To claim a water right in many western states the law requires the user to divert water from streams and put the water to "beneficial use." A use like irrigation may return a portion of this water to the river where it may be reused several more times before it leaves a state's borders. Many western states use the prior appropriation doctrine which has as its tenet "first in time, first in right." Those claiming and adjudicating the earliest (by date) water rights own the volume of water they put to beneficial use. Thus, water is a commodity, similar to land ownership, and can be sold. Senior water right holders must receive the water they have rights to before any junior water rights holders can use water. In general, all streams are over-appropriated and only in the wettest years can all water-rights holders receive the water they "own." In dry years, only senior water rights holders receive water. Thus, there is always a demand for more water.

Even though numerous dams have been built to hold snowmelt runoff there is a continual demand for additional surface water. This has led to interest by municipalities, agriculturalists and others on the availability of renewable and non-renewable ground water resources. At the same time considerable interest has focused on the consumption, and in some opinions, the waste, of ground water by phreatophytes such as cottonwood trees, willows, tamarisk and other species (Robinson 1952, Blaney 1954). For many decades the Ogallala Aquifer of the Great Plains has been tapped for agricultural use, and the City of Los Angeles has tapped the Owens Valley of eastern California for municipal use (Sorenson et al. 1989). Other aquifers are now being explored for large scale ground water development, including Colorado's San Luis Valley by American Water Development, Inc. (AWDI), and central Nevada by the Las Vegas Valley Water District.

There have been several studies on the effect of hydrologic modifications on riparian ecosystems in the West (eg. Bradley and Smith 1986, Stromberg and Patten 1991), however few studies focus on the Colorado River and its tributaries. There have been very few studies on the effects of large scale ground water development of wetlands.

In this paper I present the results of an original study of the potential effects of ground water pumping on wetlands in Colorado's San Luis Valley, a large intermountain basin. I also provide a review of existing data and contribute ideas from ongoing research on the Green and Yampa Rivers regarding the effects of hydrologic control on riparian ecosystems. The goal of these case studies is to discuss the long-term sustainability of these ecosystems and to provide ideas for the restoration of impacted ecosystems.

METHODS

The Study Areas

San Luis Valley. The San Luis Valley is the largest intermountain basin in the Rocky Mountains being approximately 100 miles long and up to 60 miles wide. It is surrounded by mountains and receives the lowest average annual precipitation of any area in Colorado, 6.9 inches at the town of Alamosa. The principal land uses are farming and ranching. The northern San Luis Valley is a closed basin with streams from the Sangre de Cristo and San Juan Mountains draining in and no surface streams exiting. The area supported large Pleistocene glaciopluvial lakes and today water tables are within 5-10 feet of the ground surface. The ground water is brackish to saline and where the capillary fringe reaches the soil surface soils are too saline for crops.

Fresh water reaches the valley as high mountain snowmelt fed rivers flow into the arid basin. Approximately 12 named streams reach the valley floor and several sumps with extensive wetland complexes occur. Waterfowl use of this region is heavy and five federal or state wildlife areas occur. American Water Development Inc. has proposed to pump up to 200,000 acre feet of ground water from the San Luis Valley each year.

The Upper Colorado River. The upper Colorado River has its headwaters in New Mexico, Wyoming and Colorado's Rocky Mountains. It is a vast region of rugged mountains that rise above relatively arid plains and basins. Precipitation increases from less than 10 inches in the lowlands to over 40 inches above 10,000 feet elevation. In general precipitation accumulates as snow in the mountains from October through April, most of which is then released in a period of 1 to 2 months from mid-May through June. This flood pulse characterizes the riparian ecosystems of the region and the life history of many species, such as Fremont cottonwood (*Populus fremontii*), is closely tied to flooding.

Observations are provided from an ongoing study of the Green and its tributary the Yampa River in the vicinity of Dinosaur National Monument. The Yampa is the largest river in the Colorado River system with a near-natural hydrograph. The Green has two large mainstem dams, Flaming Gorge and Fontanelle, and its flow is regulated for agriculture and electricity generation.

On Site Study

San Luis Valley. The goal of the San Luis Valley study was to develop a model of vegetation distribution, vegetation standing crop, and soil salinity in relation to ground water levels and to use this information to predict the potential impact of a proposed ground water table drawdown on the wetlands. Ground water modelers employed by the Rio Grande Water Conservation District and the State of Colorado Engineers Office provided assistance for determining the spatial and temporal extent of ground water drawdowns that would result from AWDI's activities. In addition, AWDI hired a ground water hydrologist to model the potential ground water changes. Not surprisingly, each model showed somewhat different potential impacts. The models all indicated water table drops over 100 years ranging from more than 200 feet to near 0-5 feet over the entire valley.

My study involved establishing 200 shallow (6 to 14 feet deep) ground water monitoring wells along topographic and vegetation gradients that extended from drylands to the centers of wetland basins. The goal was to determine the range of water table conditions that each plant community occupied and also to determine the source of water supporting each community. Wells were monitored weekly from April through October 1991. There is very pronounced zonation of vegetation and the plant community patterns are repeated and easily recognized throughout the area. Wells were located on community boundaries so that for each community there would be one well on the upper and one on the lower boundary allowing a true mean water table depth for the entire community to be determined for each sample date. Within each community being monitored a releve $25m^2$ in size was analyzed to document the floristic composition and coverage by plant species. In addition, a 0.2 m^2 quadrat was clipped to determine standing crop at the peak of the growing season. A soil sample from each stand was collected, air dried and weighted, mixed with five times its weight of distilled water, shaken vigorously and the electrical conductance determined as a relative measure of salinity.

Direct gradient analysis was used to develop models of species, community, and standing crop relationships to the water table and to soil salinity. These models are then used to develop an understanding of species distributions along the water table gradient and to predict the potential impacts of a water table drawdown. Plant species nomenclature follows Weber (1989).

Upper Colorado River. Existing literature is reviewed on the effects of hydrologic changes to riparian ecosystems on the Colorado River and similar rivers in the West. In addition, ongoing studies on the Yampa and Green Rivers in Colorado and Utah are used to provide background for assessing the status of our knowledge. One hundred ground water monitoring wells along with staff gauges are installed to monitor ground water recharge and discharge patterns in floodplain soils. We are determining the ages of trees along the river and developing a model of flow requirements for cottonwood recruitment as well as studying the effects of competition from salt cedar (*Tamarisk* spp.) on cottonwood recruitment.

RESULTS

San Luis Valley. Water table profiles perpendicular to streams flowing into the San Luis Valley, such as Sand Creek, show that water flowing from the mountains in early summer raises the ground water table to the soil surface along its course. The stream builds a linear ground water mound on top of the regional water table that extends from the mountain front to sumps in the interior of the valley. A cross section of Sand Creek showing the water levels at several times during 1991 is shown in Figure 1. On this figure it can be seen that the water table at its lowest in October has a concave shape, while at its height in June it has a convex shape. San Luis Valley wetlands are large marsh complexes that occur in sumps where ground water mounds are built by streams, such as Sand Creek. The fresh water sits on top of and mixes somewhat with the regional brackish water table and disperses laterally during the summer and by late summer most wetlands are dry.

Soil salinities are highest in areas where the maximum water table during 1991 was 1.0 to 4.0 feet below the soil surface. Soils more than 4 feet above the water table are leached free of salts and areas that are seasonally flooded by snowmelt are also relatively salt-free. The relationship of soil salinity to the water table is shown in Figure 2.

Wetland vegetation has strong zonation in the study area. The deepest water is dominated by true aquatic plants, particularly *Persicaria amphibia*, *Potamogeton pectinatus*, *P. pusillus*, *Zanichellia palustris*, and *Ruppia maritima*. Sites with seasonal standing water 1.0 to 2.5 feet deep are dominated by *Schoenoplectus acutus*. More shallow waters are dominated by *Eleocharis palustris* and/or *Scirpus pungens*. Marsh edges that are flooded with only shallow water or never flooded are dominated by *Juncus arcticus*. Higher along the gradient where the seasonal high water table is from 1.5 to 4 feet below the soil surface are dominated by the halophytes *Spartina gracilis*, *Amphiscirpus nevadensis* and *Distichlis stricta*. Above the salt accumulation zone non-halophytes tolerant of arid conditions characterize the vegetation including *Bouteloua gracilis* and *Oryzopsis hymenoides*.

Regression analysis was used to analyze each plant species' distribution along the water table gradient. Some but not all regressions were statistically significant because at different water levels a species may be more or less abundant based on land use patterns. A two dimensional direct gradient analysis model was used to illustrate the distribution of the major plant species along the gradients of maximum water table height during 1991 and soil salinity. This model, shown in Figure 3, was used to predict vegetation changes for site if permanent water table drawdowns were to occur. This model indicates that

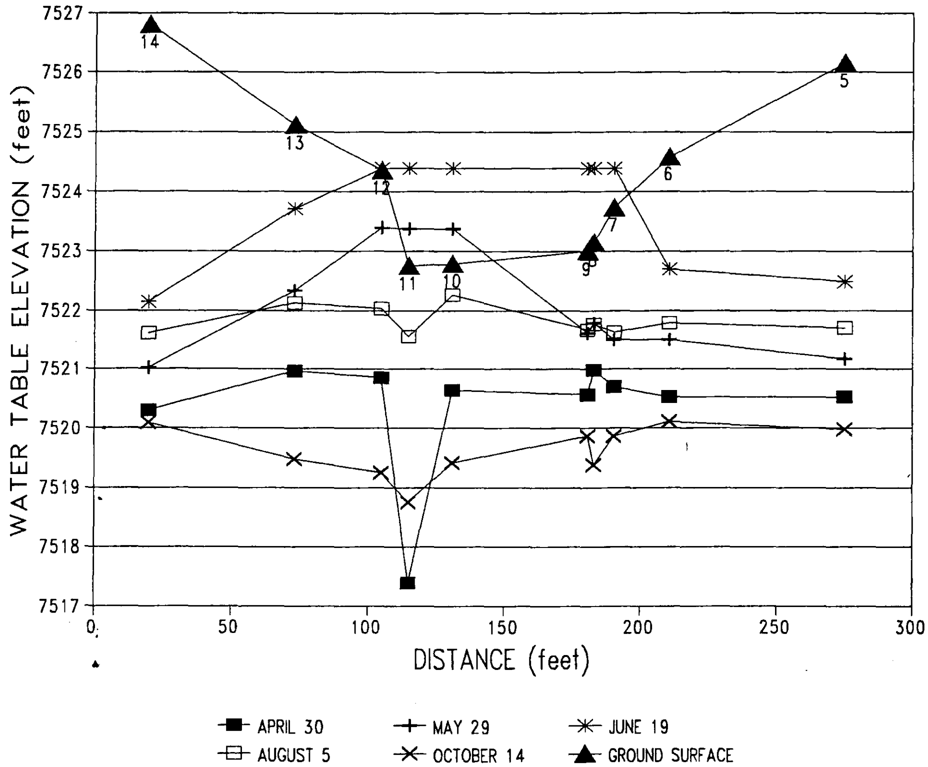

Figure 1. — Water table profiles across a San Luis Valley wetland during 1991. The water table is relatively flat in profile in April, becomes a convex ground water mound during the period of water table high during June, and becomes a concave ground water depression during the water table low in October.

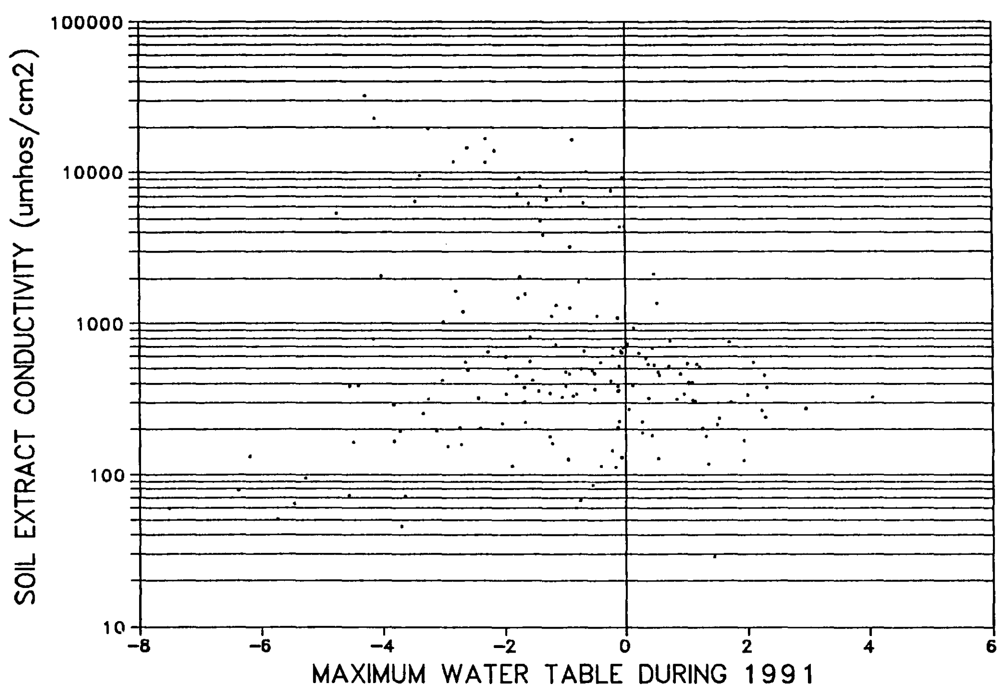

Figure 2. — Scattergram of soil extract conductivity (umhos/cm^2) vs. maximum water table depth for San Luis Valley study stands during 1991.

species maximums are an average of approximately 0.66 feet apart, thus long term shifts in species composition could be expected from a relatively small drawdown.

The entire diversity of wetland vegetation along the gradient from open water dominated by aquatic vegetation to salt flats and drylands dominated by *Spartina gracilis*, and *Bouteloua*

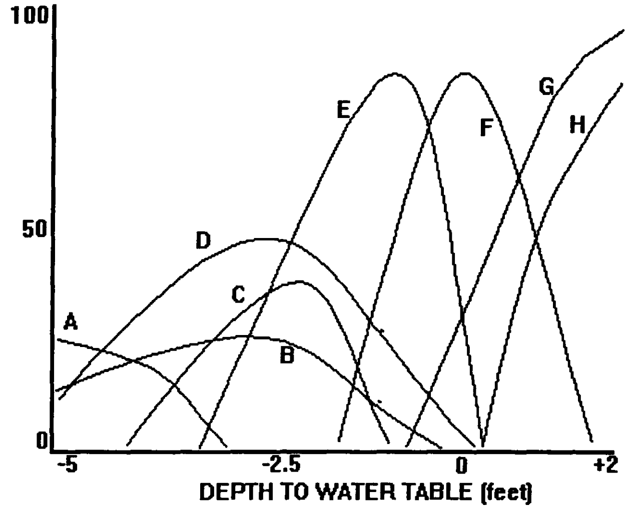

Figure 3. — Percent cover of plant species along the depth to water table gradient. Each species diagramed occupies a different niche. Species A is *Bouteloua gracilis*, B is *Sarcobatus vermiculatus*, C is *Spartina gracilis*, D is *Distichlis spicata*, E, is *Juncus arcticus*, F is *Eleocharis palustris*, G is *Scirpus acutus*, and H is all aquatic vegetation.

gracilis occurs within approximately 5 vertical feet. Thus, water table drawdowns of more than 5 feet could eliminate all wetlands in large regions of the study area. As stated previously the water table also controls the distribution of salt in soils and water table drawdowns of less than 5 feet could result in salt accumulating to very high concentrations lower in wetland basins than they had previously concentrated. Plants not killed by a declining water table could be killed by excessive salt accumulations.

A major function of these wetlands is bird habitat, particularly shorebirds, ducks, fishing birds, and sandhill cranes. These birds all use the open water, bulrush and wetland edge habitats. Thus, the ecosystems they utilize are threatened.

Concurrent with this field project the location of major wetland complexes were identified using Landsat imagery by the State of Colorado Division of Wildlife. The resulting wetland maps were compared with potential water table drawdown maps to determine if water table changes would occur in the wetland areas. Our analyses indicated that most of the large wetland complexes in the northern San Luis Valley occur in areas with greater than 5 feet of anticipated drawdown. Therefore, we predict significant impacts to the nearly 100,000 acres of San Luis Valley wetlands. Impacts would occur from three types of impacts; 1) direct ground water lowering under wetland basins, 2) disconnecting these basins from their water sources which are distant mountain ranges, and 3) increased salinity in the wetland basins as the water table declines.

Colorado River. Floodplains below 6,500 feet elevation on the Colorado River system naturally are populated by *Populus fremontii* which forms forests. There are three critical issues regarding the persistence of these floodplains forests; 1) flooding must periodically erode high terraces and deposit fresh sediment as point bars where new individuals can be established, 2) floodplain ground water levels must be high enough to support existing trees, 3) competition from tamarisk (*Tamarix* spp.) and other non-native plant species may limit cottonwood seedling survival.

Cottonwoods produce seeds for approximately 3 to 4 weeks each year, during early June through early July, which coincides with the time that flood waters are receding exposing bare silt and sand bars. Seeds are aerially dispersed, can float on water and have a short period of viability, reported to be less than 2 weeks. Seeds must contact bare, wet mineral soil to germinate and become established. Erosive floods are required to create suitable seed beds close enough to the summer water table that seedlings can grow tap roots to keep pace with the water table as it declines. High spring flows must be prolonged and the descending hydrograph limb must be gradual without sudden declines. In addition, seedlings must get established far enough from a stream that they will not drown or be eroded the following spring (Bradley and Smith 1985). The safest sites are as far from the channel as possible where wet bare sediments occur that will retain a high water table into July and August.

On the Animas River in southwestern Colorado, Baker (1990) determined that *Populus angustifolia* established only every 10 to 15 years. Elsewhere in the Colorado River system forests are even aged and patchy indicating that successful recruitment occurs in few years. The last year of seedling establishment on the Green and Yampa Rivers in northwestern Colorado and eastern Utah was in 1984 when large and prolonged flows occurred (Cooper, unpublished).

Impoundments on main stems have been built to detain the high spring flows for flood control; to provide agricultural water for the low flow months of July, August and September, and for electricity generation, particularly during the summer. The effect of this is a greatly modified spring and summer hydrograph, such as occurs on the Salt River, Arizona (Fenner et al. 1985). Flooding is prevented, although during 1984 extremely large flows caught dam operators unprepared and water spilled over the tops of several dams on the Colorado River system causing flooding. Water released from dams also lacks sediment and bank erosion is common. Point bars in general have degraded and germination sites occur only on eroding banks that will be reflooded or scoured the next summer. Thus, safe sites for seedlings do not occur and few seedlings are being establishing on controlled rivers.

Mature cottonwood trees can live more than 60 years, although in areas with low summer water tables, such as the Green River below Flaming Gorge Reservoir, crown dieback is common and trees are in poor health. The fate of established Fremont cottonwood stands on regulated rivers is uncertain, and Howe and Knopf (1991) predict a great decline in Rio Grande River cottonwoods within 50 years. The same story could be told for most rivers in the Colorado River system. The question then becomes, is it possible to sustain the ecosystems that Fremont cottonwood dominates?

An important concept for floodplain managers is that there is no climax wetland ecosystem on upper Colorado River floodplains. Young cottonwood stands accumulate sediment from floods and build terraces that can be 2 to 5 meters above the mid-summer river level. As the stands mature an understory of sagebrush, rabbitbrush and other semi-desert plant species become the dominant understory for cottonwoods foretelling the stands future, a semi-desert shrubland climax. Flood disturbance including erosion and sediment deposition is essential for cottonwood recruitment and the perpetuation of wetland ecosystems.

Ground water levels in floodplain terraces are recharged by high stream flows. Reduced stream flows caused by water diversions, impoundments can result in lowered ground water levels which can reduce cottonwood canopy vigor and stem growth. Stromberg and Patten (1991) have shown that reductions in flows on Bishop Creek in the eastern Sierra Nevada of California have reduced cottonwood growth. They estimate that 40-60% of the estimated natural total annual flow volumes would be needed to maintain healthy tree growth rates. Thus, streams that lack large enough flows to recharge ground waters may not be able to sustain healthy floodplain forests.

DISCUSSION

The two major wetland types in the Rocky Mountain West, marshes in intermountain basins and cottonwood forests on the larger rivers have been affected by water development projects for decades. In addition, new projects are being planned including water diversions and massive ground water pumping. In considering these impacts several characteristics of each ecosystem must be considered, including linkages at the landscape level that provide the hydrologic driving force, the life history characteristics of species that rely on these driving forces, and thresholds of change that ecosystems can sustain before changing to another ecosystem.

However, all ecological perspectives and restoration concepts must be viewed through the eyes of the western urban or rural citizen — water development projects are integral to the West's economy as we know it today. Water conservation measures could certainly make great strides toward reducing the amount of water utilized, particularly for agricultural purposes. But due to the prior appropriation doctrine, water made available through conservation would be utilized by junior water rights holders or new water rights appropriated. There is not enough water in any river basin for both the West's growing urban and agricultural population and the native ecosystems that depend upon large amounts of water. Maintaining ecosystems as they occur today will be difficult enough and restoration presents unique problems.

Marsh ecosystems, such as Colorado's San Luis Valley are dependant upon a seasonal connection between the mountain water source and the interior basins where water accumulates. Any decoupling of this flow system, or lowering of the regional water table under wetland basins which would make the amount of recharge water delivered from the mountains less effective, will result in wetland destruction. Even long term average water table drops of 0.66 feet could most likely produce shifts and diebacks in wetland vegetation. Where water tables drop and/or surface flooding no longer occurs plants may survive until salt, from the brackish regional ground water table, accumulates in surface soils killing the plants.

The greatest values associated with these marsh wetlands is shorebird and waterfowl feeding and nesting. These species require seasonally or permanently inundated or saturated wetlands. Sustaining these ecosystems will require that little change to the hydrologic system occur, especially during the spring and early summer months. It may be possible for small winter drawdowns to occur, but only in years when there is sufficient snowpack to allow streams to fully recharge the ground water system and build ground water mounds that will couple the mountains with the interior basins.

For basins that are already impacted, such as Nevada's Stillwater and Carson Basins, restoration will require the purchase of water rights from agricultural users that will facilitate spring flooding. Perennial water is not required and marsh levels should be highest in the early spring and drop steadily through the summer. Most marshes naturally dry up by September.

Many state governments embrace the concept of minimum stream flows to protect fisheries. However, riparian ecosystems have evolved with maximum stream flows that create very dynamic floodplains and carry large sediment loads. For sustaining riparian ecosystems, periodic maximum stream flows are required to create new sediment bars for tree recruitment and stand replacement. Floodplain forests require annual flows high enough to recharge water tables under floodplain terraces. Stromberg and Patten (1991) indicate that water diversions of greater than 40-60% reduce tree vigor and growth. Minimum stream flows established for fishes, for example 800 cubic feet per second that leaves Flaming Gorge Reservoir on the Green River, do not appear to be large enough for forest trees to retain their vigor. In addition, the critical question of new plant recruitment and floodplain evolution will require large springtime flows that can significantly erode and deposit sediments creating safe sites for seedling establishment.

The possibility of much larger flows being released from existing reservoirs is not likely at present because few reservoirs have the ability to pass the needed volumes of water and because water rights holders may not be sympathetic to this use of water, unless the water rights are purchased and dedicated as instream flows which are a beneficial use in some states. The issue of whether large releases of sediment-free water would help or harm floodplain systems has not been adequately addressed. This erosive water can remove existing fines and erode beaches as has occurred on the lower Colorado River below Glen Canyon Dam (NAS 1991). I suggest that dam operators consider the construction of a sediment-water slurry pipeline running from the inlet of every reservoir to the dam for sediment reintroduction into the river below the dam.

Several potential solutions present themselves as important for sustaining and/or restoring these ecosystems; 1) dam removal, 2) purchase of water rights to dedicate as in-stream flows that will resurrect the natural flood pulse hydrograph, 3) the development of a system for moving sediments from reservoir inlets in a slurry pipeline to the dam face where they can be placed back into the river with releases to restore the sediment transport function of the river.

The sustainability of wetland ecosystems must be considered at three scales, landscape, ecosystem and plant population. At the landscape scale, the delivery of water is essential to consider. Any changes in the timing, volume; sediment characteristics and stream power of the water will manifest through the entire watershed below. At the ecosystem scale the presence of dikes, drainage systems, and basin characteriscs are important to consider. Wetlands along a stream with its natural hydrologic regime would not function properly if they were diked off from the flow. Finally the life history characteristics of the key plant species, such as cottonwood trees, are vital to consider for understanding the types of impacts that flow modifications would have on ecosystem function.

Most low elevation riparian ecosystems do not appear sustainable given the legal and structural constraints on water in the system today. Restoration is possible, but it would require large changes in the way water is used in the West.

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Potential Effects of Timber Harvest and Water Management on Streamflow Dynamics and Sediment Transport

C.A. Troendle and W. K. Olsen[1]

Abstract — The sustainability of aquatic and riparian ecological systems is strongly tied to the dynamics of the streamflow regime. Timber harvest can influence the flow regime by increasing total flow, altering peak discharge rate, and changing the duration of flows of differing frequency of occurrence. These changes in the energy and sediment transporting capability of the fluvial system can cause an alteration in both channel morphology and aquatic habitat. Depending on practices used, timber harvest can increase the rate of sediment introduction to the channel system, thus further confounding the energy/transport relationship.

Diversion and augmentation also alter the natural flow regime and disrupt the energy distribution in the system. Diversion decreases the energy regime available to transport the sediment load and may cause aggradation and vegetation encroachment. Elevated flow regimes from augmentation may result in extensive scour, loss of aquatic habitat, and ultimately a change in the relationship between the aquatic and terrestrial components.

This paper addresses the flow parameters which influence sediment transport and the implications of changing flow dynamics, whether from flow or forest management, and the effect it has on the transport process.

INTRODUCTION

Riparian and wetland areas provide productive fisheries and wildlife habitat, diversity of aesthetic scenery and recreation sites, sediment filtering and flood reduction, high quality water, points of recharge for ground water, commercial timber, and sustainable forage for domestic livestock and wildlife. Riparian and aquatic conditions provide a good index to overall watershed condition.

The Organic Act of 1897 identifies two key goals in establishment of the National Forests: maintaining a continuous supply of timber and securing favorable conditions of streamflow. In essence, the latter charge, as expanded by the 1964 Multiple-Use Sustained Yield Act, implies upland watershed conditions be maintained such that ecosystem diversity and integrity be managed to sustain beneficial use of the aquatic ecosystem both on national forest lands and to downstream users. One may also assume a charge to maintain and protect the physical and biological continuity in the aquatic system. The implications are the same to all of us whether the land is public or private, and if public, regardless of the administering agency.

Although only 6 to 8 percent of the lands in the West are classified as riparian, the vast majority of biological diversity is found in these areas. The riparian ecosystem, both its terrestrial and aquatic components, is an extremely important part of the landscape. Maintaining the condition, productivity, and integrity of these systems is extremely critical to sustaining productivity and biological diversity at the landscape level. One of the most important parameters in developing, maintaining, and sustaining the viability of these riparian ecosystems is the streamflow regime.

The flow regime, so critical to maintaining viability, varies naturally, as the result of land management practices, and through water management. We have little control over natural variability, but we must recognize its existence. This natural variability may be further modified by land management practices and streamflow manipulation.

[1] Troendle is a research hydrologist and Olsen is an operations research analyst at the Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Timber harvest can influence the flow regime by increasing total flow, altering peak discharge rates, and changing the duration of flows of differing frequency of occurrence. These changes in flow alter the energy and the within-channel sediment transporting capability of the fluvial system and can result in an alteration in both channel morphology and aquatic habitat. Depending on practices used, soil disturbance associated with timber harvest can increase the rate of sediment introduction to the channel system, and further confound the energy/transport relationship.

Water management (diversion/augmentation) also alters the natural flow regime and disrupts energy distribution in the system. Diversion decreases flow and the attendant energy available to transport sediment and may cause aggradation and subsequent vegetation encroachment into the channel system. Elevated or extended flow regimes, due to augmentation, may result in extensive scour, loss of aquatic habitat, and ultimately a change in the gradient between the aquatic and terrestrial components.

This paper will first address the relationship between flow dynamics and sediment transport, then characterize the effect of forest disturbance and water management on the flow and energy regime, and lastly draw inference about the effect of flow change activities on sediment transport.

INSTREAM FLOW REQUIREMENTS

In order to maintain the physical and biological integrity of the aquatic system, one might argue the entire or natural "run of the river" is needed. After all, this regime produced what one sees in an undisturbed system. However, given the competing demands for water, and the need to manage other resources, this is not often a feasible alternative; so one must consider flow in terms of key components necessary to ensure some desired future condition. *Overbank flows* are needed to help sustain the terrestrial component and deliver nutrient laden sediments and export detritus or organic material. *Low or base flows* are needed to sustain and ensure survival of the biological component of the aquatic system. Other *effective discharges* are also needed to physically maintain the channel system or conduit. In general, instream flow needs require some frequency of occurrence and duration of a wide range of flow levels.

This paper deals primarily with the effect of flow dynamics on sediment transport and subsequent implications of impact on channel morphology and aquatic habitat. Examples drawn from the long-term data sets, collected in the subalpine environment of the Fraser Experimental Forest, CO, demonstrate the relationship between flow and sediment movement. In general the streams in this and most upland forested environments are either step-pool or riffle-pool systems and for the most part, suspended sediment transport would tend towards being supply limited while bedload transport, at times, would be energy limited (Knighton 1984).

Beginning with the Fool Creek Watershed, streamflow from watersheds on the Fraser Experimental Forest has been monitored since 1941. In total, nine watersheds are currently gaged and six of those nine have stilling ponds associated with the V-notch or Cypoletti weir being used. The hydrographs from these watersheds are snowmelt dominated and peak annual flow has never been rain fall dominated during the 50 years of observation (see Troendle and Kaufmann 1987, Troendle 1991). The record is such that instantaneous flow values (15 minute intervals) are available for all watershed years of record. Instantaneous flows are integrated into hourly, daily, monthly, and seasonal (April 15 - October 1) streamflow estimates.

Leaf (1970) demonstrated that weir ponds, or stilling ponds, are effective sediment traps. Virtually all of the suspended sediment and bedload drops out in the ponds. Each year the material accumulated must be removed. Prior to removal, the ponds are drained and an intensive survey made of the surface elevation of the material in the pond. The accumulated material is then removed, and the survey repeated. The difference in mean elevation of the two surveys is used to calculate the volume of material removed. The organic constituent is variable so density of the material removed varies from a specific weight of 1.4 to 1.7, for differing watersheds. The material removed from the ponds is an index of total sediment export for the runoff period (April - September). For some watersheds, records of sediment export have been kept since 1956. The hydrologic record indicates that the wettest (1957 or 1983) and driest (1977) years in the last 50 occurred during the period when sediment data was being collected.

East St. Louis Creek is an 803 ha control watershed. Draining to the north, it ranges in elevation from 2895 m to 4002 m, and has been gaged since 1943 with sediment data collected since 1965. Lexen Creek, a 124 ha control watershed, ranges in elevation from 3002 m to 3536 m and flow and sediment records date to 1955. Main Deadhorse Creek, a 270 ha watershed, ranging in elevation from 2880 m to 3536 m, has also been monitored since 1955. Main Deadhorse Creek also contains two subdrainages; the North Fork, a 40 ha south facing drainage gaged since 1970, and Upper Basin, a 78 ha subbasin gaged since 1975. Sediment data from Deadhorse Creek and its two subbasins coincides with the start of the hydrologic record.

Deadhorse Creek is also a treatment watershed. The main basin was undisturbed from 1955-1970 at which time an access road to the North Fork gaging site was built and the streamgage, a 90^{o} V-notch weir, installed. In 1975 the access system was extended to the Upper Basin gaging site and the stream gage, also a 90^{o} V-notch weir, installed. In 1977 and 1978, a road system was built into the North Fork subdrainage and 36 percent of the North Fork was harvested in small clearcuts (Troendle, 1983a). The north slope of Main Deadhorse Creek (an area outside the two gaged subbasins) was harvested using a shelterwood harvest in 1981 (Troendle and King, 1987). Twenty-six percent of the Upper Basin was harvested in 1983 and 1984 using small irregularly shaped clearcuts. Timber harvest, and its attendant impact, has caused a significant increase in sediment production from the North Fork of Deadhorse Creek only (Troendle, 1983a). Changes in sediment production have not been detectable at the main gage nor from the Upper Basin (Troendle 1983a, Troendle and King 1987).

The objective of the analysis reported in this section was to determine which, if any, descriptors of flow, measured at the gage, correlated with the accumulation of sediment in the pond. The intent was to determine, over time (years), if one expression of flow or another might be better correlated with the accumulation of sediment.

The expressions of flow used in the analysis included volumetric (total seasonal), instantaneous (peak discharge) and duration (i.e., duration of bankfull).

The total seasonal flow, in acre feet per square mile, was used as the expression for total flow volume while instantaneous flow was expressed as peak daily discharge in cubic feet per square mile. Expressions of flow duration had to be calculated in a more arbitrary manner. Andrews (1980) noted discharges approximating "bankfull" were most effective in moving sediment, over time. As a surrogate for the estimate of "bankfull", we assumed daily flows having a 1.5 year (Weibull distribution annual series) reoccurrence interval approximated the bankfull or effective discharge rate for the systems at Fraser.

Once the 1.5 year return interval daily flow was determined for each of the watersheds, the following flow durations were determined for each year of record. The duration, in days, in which 20, 40, 60, 80, 100, 120, 140, and 160 percent of bankfull flow (1.5 yr.) was equaled or exceeded. Figure 1 represents all the instantaneous peak daily flows observed on East St. Louis Creek for the period of the record, as well as the estimated bankfull discharge and some of its various percentages. For each watershed the data set constructed consisted of a dependent variable, accumulated sediment (a volume) for the year, and several independent flow parameters that included the expression of total volume (total seasonal flow), maximum rate or peak discharge, and the various expressions for duration.

A correlation matrix was then developed identifying the relationship between sediment accumulation and the various flow parameters.

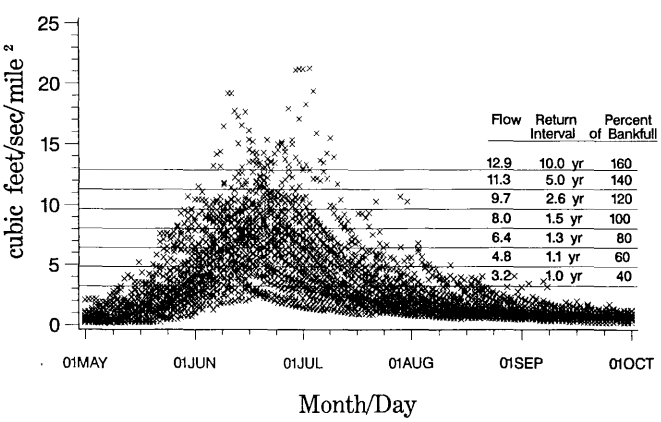

Figure 1. — Instantaneous peak daily flows on East St. Louis Creek for the period of record.

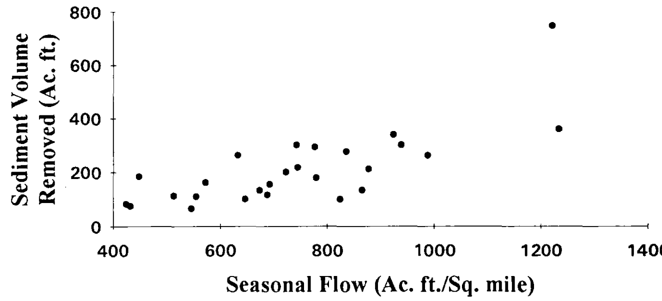

Figure 2. — Sediment volume removed vs. seasonal flow at East St. Louis Creek 1965 - 1990.

Figure 2 represents the relationship between sediment export from East St. Louis Creek (Y) and total seasonal flow. Although the variability is great, as flow increases so does sediment export. The two highest values represent the years 1983 and 1984 with 1983 being the wettest year on record (1943-1992) for this watershed. The R^2 for the relationship shown on Figure 2 is 0.49 (see also Table 1). Export from East St. Louis Creek is better correlated with the duration (number of days) of flow at or exceeding 60 percent of bankfull ("bankfull" being the 1.5 year annual daily flow value) than with total flow alone. Sediment export from St. Louis Creek is also well correlated with peak flow (Table 1).

Table 1. — Correlation of sediment accumulation and various flow parameters for East St. Louis Creek, Fraser, Colorado.

Parameter	R^2
Total Flow	.49
Peak Flow	.61
60% Bankfull*	.62
80% Bankfull	.52
Bankfull	.62
120% Bankfull	.53

**Bankfull is estimated as the 1.5 year return interval mean daily flow value.*

Five watersheds have similar data of varying length (approximately 80 years of total station record). All data were pooled for a composite analysis. In the process, all flow and sediment parameters were normalized to the mean for their respective watersheds to eliminate individual watershed effect. The duration of flow at or exceeding 80 and 100 percent of bankfull appears to be the most strongly correlated with total accumulation. The variability is, of course, quite large both within and between watersheds (fig. 2), and sediment production from the individual watersheds may correlate with one flow parameter better than another (i.e., compare Table 1 vs. Table 2). It should be noted that all R^2s presented are significant ($P<.05$) but tests were not made to determine significance between R^2s.

Table 2. Correlation of normalized sediment accumulation and various flow parameters for all five watersheds, Fraser Experimental Forest, Fraser, Colorado.

Parameter	R^2
Total Flow	.44
Peak Flow	.48
60% Bankfull*	.44
80% Bankfull	.53
Bankfull	.53
120% Bankfull	.48

**Bankfull is estimated as the 1.5 year return interval mean daily flow.*

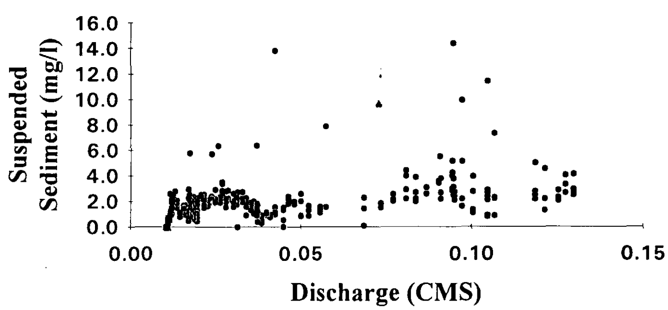

Figure 3. — Suspended sediment vs. discharge at Lexen Creek (5/22/79 to 9/6/79).

The suspended component of total sediment load is more chaotic and seems less predictable. Figure 3 represents the suspended sediment load for Lexen Creek (data not available for East St. Louis Creek) relative to the hydrograph for the one year when automated sampling was done throughout the runoff period. Although the suspended sediment concentration is significantly ($P<.05$) correlated with flow (fig. 3) as is total accumulation (fig. 2 total load), the extreme or highest concentrations appear to be more sporadic and less flow dependent. Approximately one metric ton of suspended sediment was exported in 1979 from Lexen Creek. This compared with approximately 0.75 metric tons of bedload. As a point of reference, the expected dissolved load in 1979 would average 30 mg.l (Stottlemyer 1987)and represent a seven or eight metric ton load over the course of the same runoff season.

MANAGEMENT IMPACTS ON STREAMFLOW DYNAMICS

The flow regime can be impacted by either land use or water management practices. For purposes of this paper, the only land-use practice considered is forest disturbance due to timber harvest. Other disturbance such as insect and disease attack, fire, and to some extent land-use change, also influence flow but will not be specifically considered here as the direction of change is similar to that for timber harvest. Water management can consist of diversion, augmentation, or both. Consideration will be restricted to diversion as it has the potential to reduce flow and alter the availability of water for instream flow needs. The implications associated with flow augmentation are aligned with the increasing energy associated with the response to forest disturbance.

Forest Disturbance

Effect on Water Yield

Bosch and Hewlett (1982) summarized the results of nearly 100 experiments worldwide on the effect of timber harvest on water yield. More specific regional summaries have been presented by Douglass (1983), Harr (1983), Kattleman, et al. (1983), and Troendle (1983b). The basic nature of process response to timber harvest, whether worldwide, regional, or local is conceptually similar. Timber harvest reduces the transpirational draft of water, thus reducing soil water depletion. In addition, canopy interception and subsequent vapor loss of precipitation (rain and snow) can also be significantly reduced, thus delivering a greater percentage of precipitation to the forest floor. The combination of transpiration and evaporation changes represents the net evapotranspiration (ET) change following disturbance. Depending on soil moisture levels, more water (ET savings) may be available to drain from the soil toward the channel. Soils are generally as wet or wetter following harvest as they were before harvest. Because of the wetter soil, a higher percentage of precipitation entering the disturbed site is often available for streamflow due to the reduced storage requirement in the soil.

The largest differences in the nature and timing in flow response, observed either regionally or nationally, reflect differences in the climatic regimes that drive the respective systems. In humid areas, with frequent large rainfall events, treatment response is demonstrated as increases in individual, and frequent storm flows, as well as reflected in elevated base flows between storms or in non-storm periods — a reflection of frequent wetting and draining of the wetter soil. In arid areas, the rainfall may be intercepted and vaporized in a different manner or pathway following harvest; thereby, greatly reducing the opportunity for flow change in all but the wettest seasons or years. In the subalpine, for example, the summers are often arid (precipitation limited) and demonstrate little response while the winters accumulate snowpack (energy limited) that upon melting and entering the wetter soil, causes a large, concentrated increase in flow early in the runoff period. The timing and magnitude of hydrologic response to forest disturbance varies from region to region but there are more similarities in the processes being altered and the response that occurs than there are differences.

The original paired watershed experiment (water balance study) in the United States was done at Wagon Wheel Gap at the head of the Rio Grande River in southwestern Colorado (Bates and Henry, 1928). Streamflow from two watersheds was monitored from 1911 to 1919 and then one of them was cut. Of the 530 mm of precipitation falling on the watersheds, approximately 150 mm was returned as streamflow prior to harvest with the remaining 380 mm lost to evapotranspiration. Following harvest, flow increased on the disturbed watershed by an average 25 mm with as much as 50 mm occurring in the wet years.

The more classic experiment, because of duration of both record and response, has been the Fool Creek Watershed on the Fraser Experimental Forest , CO (Troendle and King, 1985). Forty percent of the 290 ha drainage was harvested in alternating clearcut and leave strips during 1954-1956. The average hydrograph for before and after treatment is depicted in figure 4. On average, total seasonal flow increased by 40 percent, peak flow increased by 20 percent, and most all the detectable change occurred in the month of May. (Troendle and King, 1985). The response at Fool Creek depicts the nature of the change that occurs, either annually or on an event basis, when we disturb the forest in the subalpine environment. First, total flow tends to increase (in this case by almost 90 mm). Oftentimes the peak flow increases (in this case by 20 percent) and usually the duration, or period of time the higher flows occur, increases. The same type of response occurs almost everywhere else, but reflected on either an annual or event basis, depending on the climatic regime. However, we do not usually see the strong relation or positive correlation between annual precipitation and flow changes in humid areas that we observe in more arid regions. Where precipitation is limiting we tend to observe a precipitation dependent response. Where energy is limiting (humid areas) we tend to observe a more consistent response dependent on degree of disturbance. A similar relationship holds with peak flows; generally smaller peaks (precipitation limited) are influenced proportionally more than large peaks with the largest, or extreme events probably not effected by treatment.

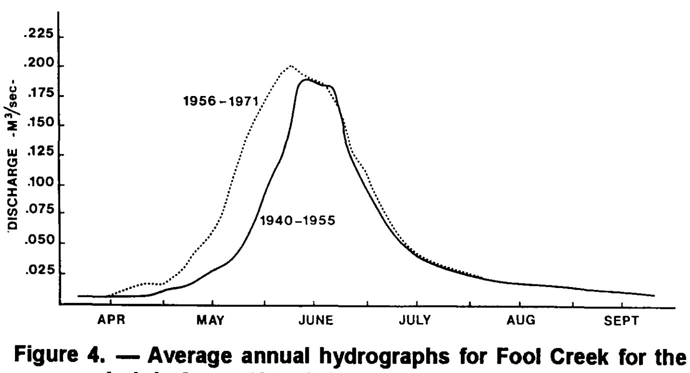

Figure 4. — Average annual hydrographs for Fool Creek for the period before (1940-1955) and after (1956-1971) timber harvest.

To better demonstrate the nature of the change in flow regime or flow duration that occurs following harvest, Fool Creek can also be used. Equations developed between Fool Creek and its control, East St. Louis Creek, for the calibration period (1943-1954) allow the prediction of the expected duration, or frequency of occurrence, of flows of various magnitudes. The expected frequency can then be compared with the observed frequency to determine effect of treatment. Figure 5 represents the percentage of time flows ranging from 40 to 180 percent of bankfull (1.5 year return interval flow is the surrogate for bankfull) would be expected to occur based on pre-harvest conditions, and the percentage of time they occur following harvest. At approximately "bankfull" (or Q_n/Q_b = 1.0) the duration of flow went from 3.5 days before to over 7 days following or was more than doubled. The highest flow durations were unaffected, lower flow durations were less affected. The duration of flows in the range from 80 to 120 percent of bankfull appear to have been most affected.

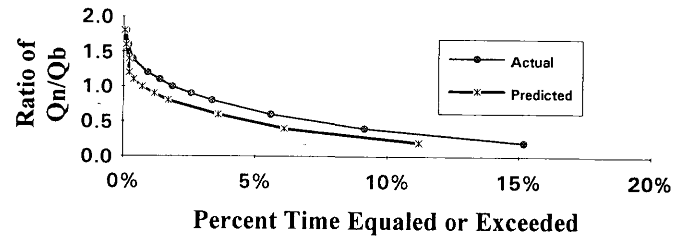

Figure 5. — Flow duration curve for Fool Creek (1957-1991) comparing the mean duration predicted from East St. Louis Creek vs. the actual mean.

Effect on Sediment Transport

Numerous studies have shown forest disturbance can increase the amount of introduced sediment to channel systems. Any increased introduction of fine material to the channel system would probably result in an increase in suspended sediment export. This has been demonstrated to have happened on Fool Creek (i.e., Leaf, 1970), and elsewhere. Following partial clearcutting of the North Fork of Deadhorse Creek in 1977-1978, a significant increase in both sediment export and flow was observed (Troendle, 1983a). Peak flow rate significantly increased by 50 percent (Troendle and King, 1987).

A covariance analysis was conducted to evaluate the interaction of flow and sediment on the North Fork and Lexen Creek (control) with a dummy variable for treatment. Both 1983 and 1984 were wet years, and dominated the post treatment period as the remaining post treatment years were quite low in both flow and sediment production. However, the analysis indicated sediment accumulation is strongly flow related and not significantly related to disturbance, at least for the two largest values. The adjusted slope of the flow/sediment relation was the same for both the pre- and post-harvest periods on the North Fork of the Deadhorse Creek. The intercept of both lines was also the same. The increase in sediment was from within channel and not the result of increased sediment introduction following road building and harvest. The total volume of sediment removed from the weir pond is strongly correlated with flow, whether before or after treatment (fig. 6).

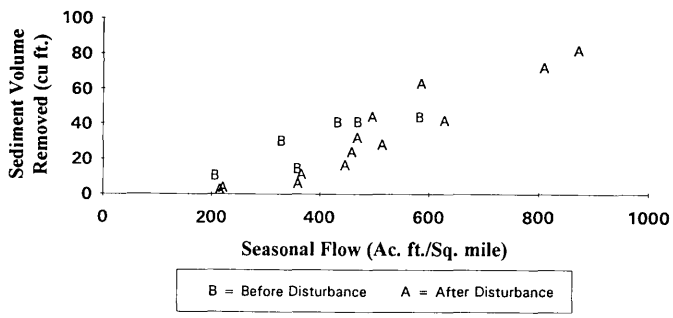

Figure 6. — Sediment volume removed vs. seasonal flow at Deadhorse (north) Creek.

Figure 7 represents the relationship between sediment accumulation and flow at Main Deadhorse Creek. Main Deadhorse Creek contains the North Fork, as well as the Upper Basin and the North Slope harvest sites. In total, approximately 18 percent of the basal area of the Deadhorse watershed has been harvested with no detectable impact on flow (peak or volume) at the gage (Troendle and King, 1987). Analysis of the sediment data does not indicate any significant change occurred; supporting the observation on the North Fork that there is no increase in introduced material due to disturbance alone. In the case of Main Deadhorse Creek there is no detectable flow change so one would not expect a flow driven sediment increase.

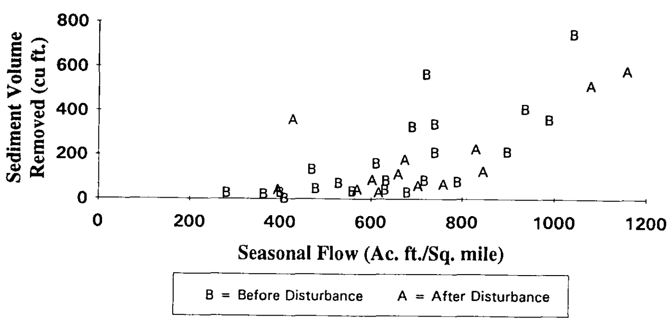

Figure 7. — Sediment volume removed vs. seasonal flow at Main Deadhorse Creek.

Water Management

Because 70-80 percent of the water supply in the Western United States is generated from 15-20 percent of the land base, a significant portion of that water is diverted, transferred from one basin to another, stored on or off site, and in effect manipulated so it can be delivered when and where it is needed for other purposes.

The net effect in either the diversion or augmentation process is to alter the natural flow regime. The watersheds on the Fraser Experimental Forest also lend themselves well as an example of the effect of diversion on the flow regime. The U.S.G.S. maintains a gage on Main St. Louis Creek, the 9300 ha drainage containing East St. Louis Creek, that dates to the mid-1930's. In 1956, the Water Department of the City of Denver, CO started diverting a significant portion of the flow from various tributaries on St. Louis Creek. For the period 1943-1955 the flow from both East St. Louis Creek (the control watershed, described earlier) and Main St. Louis Creek was unaltered. Both drainage's have similar aspect, relief, and vegetative cover with East St. Louis Creek being a small tributary of St. Louis Creek; and one-tenth its size. The flow from Main St. Louis is highly correlated with that from East St. Louis such that East St. Louis Creek can be used to estimate "expected" flow for Main St. Louis Creek for the period of diversion (1956 to present).

A fairly significant proportion of the expected seasonal flow on Main St. Louis Creek (approximately 50 percent) is diverted (fig. 8). The consistent pattern has been to take a high percentage of the flow in average and drier years and little flow in the wetter years. (Please note that the scale on Figure 8 does not clearly reflect the fact that there is a significant bypass of at least 280 to 480 l.s base flow at all times.)

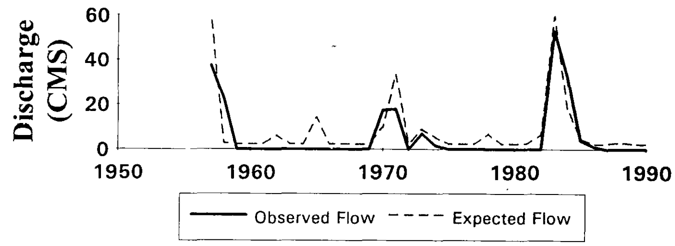

Figure 8. — Comparison of observed and expected flow equal to or exceeding bankfull at St. Louis Creek for the post diversion period 1957 - 1990.

In a manner similar to the one used for Fool Creek, we estimated the expected frequency of occurrence of flow near bankfull discharge (1.5 year return interval) and compared it with observed occurrence (fig. 9). The greatest impact of diversion appears to be at flow levels less than bankfull. Although there is significant reduction in the duration of flows at 80 to 100 percent of bankfull, the greatest impact is on the duration of lower flows.

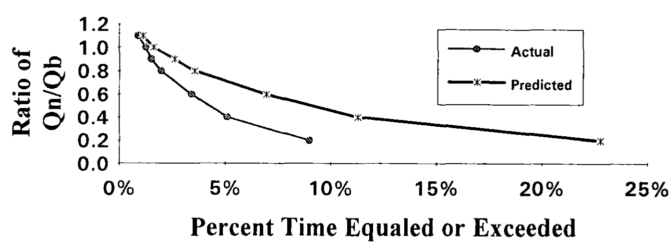

Figure 9. — Flow duration curve for St. Louis Creek (1957-1990) comparing the mean duration predicted from East St. Louis Creek vs. the actual mean.

The climatic regime at Fraser is such that there is an 11 year cycle from wet to dry to wet, as is evident in figure 8. In the wettest years, most of the high flows are by-passed, but in the near average and lower years (perhaps when bankfull does not occur) nearly all flow is taken. As noted earlier, the degree to which diversion alters flow depends on many factors. St. Louis Creek is used only as an example of the nature of the change in flow resulting from diversion.

DISCUSSION AND SUMMARY

It has been well demonstrated that forest disturbance alters the flow regime (see fig. 4). Most reports on response to disturbance have dealt primarily with impact on total yield, some with effect on peak, and few with effect on flow duration (see Troendle and Leaf, 1980, Troendle and King, 1985). However, forest disturbance, particularly timber harvest, has the potential to increase total flow, increase peak discharge, and lengthen the duration of the larger flows apparently also critical to sediment (bedload) movement. Flow augmentation has the same potential. The analysis of flow parameters associated with sediment transport implies increasing any of the three expressions of flow increases sediment transport. In the case of the North Fork of Deadhorse Creek the significant increase in sediment, demonstrated to occur following harvest, appears to be largely a reflection of increased export associated with prolonged duration of higher flows.

Flow diversion, on the other hand, reduces energy as demonstrated by St. Louis Creek. In most years, the critical occurrence and duration of sediment- transporting flows (be it total, peak, or effective flow) are less available. Given the sediment transport relations demonstrated in figures 2, 3, 6, and 7, less of the sediment carried by what would be the "expected" flow would be carried by the "observed" flow. The difference would be deposited in the channel, in bars, spawning beds, etc. Although figure 5 depicts an example of the effect of forest disturbance on flow duration while figure 9 depicts an example of the effect of flow diversion, a direct comparison of the two figures or impacts is not necessarily valid. The forest harvest effects are monitored "on-site" or at the mouth of the first order watershed in which the treatment was imposed. The flow diversion effect is monitored "off-site", or some distance downstream from where diversion is occurring, thus allowing for some recovery. At some of the diversion points, the on-site impact is to totally dry the surface stream. In evaluating these impacts, one has to consider temporal and spacial differences.

Overall, one can conclude that the duration of the higher levels of flow or "bankfull" and above, are needed to ensure sediment movement. Forest disturbance can increase the occurrence of those flows significantly (at least on-site). Lower flows resulting from diversion, reduce the frequency of sediment transporting flows, and may result in aggradation. It is not the purpose of this paper to judge the significance of any possible changes, only that they can occur.

Aquatic systems can best be preserved under natural, pristine conditions. However, competing uses of water and other terrestrial resources almost dictate some modification in flow will occur. The need is to preserve a base flow capable of keeping the thalweg free of vegetation and sustaining the flora and fauna through various life stages. Adequate high flows are needed to transport sediment, keep the channel open, maintain spawning gravels, and prevent undue aggradation. Overbank flows are needed to preserve the riparian environment and for the energy the deeper stage adds to the channel transport processes. Both forest disturbance and water management have the opportunity to adversely alter the needed regime.

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THE BIOLOGY OF RARE AND DECLINING SPECIES AND HABITATS:

Session Summary

William M. Block[1], Chair

Abstract — Rarity, declining populations, and extinctions are natural phenomena. Numerous factors contribute to these phenomena including biotic factors, isolation, and habitat change. Human activity exerts external energy to the environment that accelerates the decline of species and their habitats at rates exceeding natural phenomena. Papers presented in this session provide examples of the effects of human activities on declining species and habitats.

Rarity, declining populations, and extinctions are natural phenomena. Some species are naturally rare because of limited distributions or intrinsic life-history attributes that limit their population size. Limited distributions may occur when a species is endemic to a small isolated area of habitat such as an island. Islands need not be surrounded by water in the traditional sense, but could exist in terrestrial systems as a patch of habitat, terrestrial or aquatic, surrounded by inhospitable areas. Life history attributes that limit population size can be large territories as evident in many large predators, low rates of fecundity, and limited resources spread across a wide geographic area. Further, evolutionary processes and the inability of species to adapt to changing environments may underlie natural population declines of species leading to extirpations and extinctions (Allendorf and Leary 1986).

Numerous factors contribute to these phenomena including biotic factors, isolation, and habitat change (Frankel and Soule 1981). With the exception of island situations, biotic factors—namely, competition, predation (exclusive of humans), and disease—are unlikely causal factors that lead to extinctions. Their primary influence may be to limit populations in size and distribution to the point where additional factors push species towards extinction. Ziswiler (1967) noted that 53 of the 77 species of birds or mammals which have gone extinct in recent history occurred in isolated situations. Two underlying reasons for these extinctions were limited habitat and a deterioration of competitive edge and predatory defense. Habitat alteration is brought about by slow geologic change, climate, catastrophe, and humans. The first three categories of habitat change are largely natural, whereas the fourth is not. Generally, these major factors do not act in isolation, but work simultaneously and result in population and habitat declines.

Human activity exerts external energy to the environment that accelerates the decline of species and their habitats at rates exceeding natural phenomena. Understanding how humans impact ecological systems is essential for developing proactive approaches to conserve species and habitats, and to allow natural events to act on species' populations and their environments. Papers presented in this session provide examples of the effects of human activities on declining species and habitats. These papers provide prime examples of anthropogenic processes that lead to species' declines. The central theme of these papers centers on impacts to habitat, resulting in habitat loss and fragmentation.

Simberloff (1993) echoed this theme in his review of the effects of habitat fragmentation, noting that effects of fragmentation vary with size, shape, and juxtaposition of patches. Some effects include increased dispersal distances, increased vulnerability to predation, and disruption of natural processes such as fire. Fragmentation certainly has been a factor in declines of freshwater molluscs (Mehlhop and Vaughn 1993), endangered butterflies (Schaeffer and Kiser 1993), and numerous other species (Wilcove et al. 1986).

Declining populations resulting from anthropogenic impacts are evident in numerous taxa ranging from common species such

[1] *William M. Block is Acting Project Leader and Research Wildlife Biologist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, located at Flagstaff, AZ. Headquarters is in Fort Collins, in cooperation with Colorado State University.*

as quail (Brennan 1993) to threatened, endangered, and rare species such the black-footed ferret (*Mustela nigripes*). Land uses that cause population declines vary. For example, combinations of agriculture and silviculture underlie declining northern bobwhite (*Colinus virginianus*) populations (Brennan 1993). Water and electric power development have impacted populations of mountain quail (*Oreortyx pictus*) in Idaho and numerous desert fishes in the Southwest (Brennan 1993, Rinne 1993). Grazing has affected populations of numerous native desert fish (Rinne 1993) and assemblages of native grassland species. The negative effects of timber harvest on spotted owls (*Strix occidentalis*) are well known, leading to the listing of two subspecies, the Mexican (*S. o. lucida*) and northern (*S. o. caurina*) spotted owls, as federally threatened (Gutierrez 1993). Corn (1993) noted that reasons for many declining amphibian populations are not clear, and stressed the importance of understanding natural population fluctuations to evaluate whether current trends are indeed caused by human activities.

Obviously, historic patterns of land use, particularly following European settlement of North America, have had pronounced, and frequently negative, effects on native flora and fauna. Many cause-effect relationships are understood, far more are not. Further, ramifications of past and present land-use practices on future populations is certainly unknown, but if current practices continue without change, the outlook is bleak for many species.

Papers in this session and throughout the conference shared a common message. That message is that resource conservation must embark on a new, proactive approach. Sustaining current ecosystem conditions will doom numerous additional species to extirpation and possibly extinction. The case histories provided in this session certainly verify this as the case. Change cannot be simply a new vocabulary or set of jargon to allow "business as usual" to occur under a newly articulated management vision. It must be a complete change in focus at all levels of resource management. For the conservation of natural resources to be possible, functional disciplines must break down the barriers that impede communication (Gutiérrez 1993). Likely, shifts in social and economic systems will be needed for ecological approaches to resource management to be successful. Reactive management for TES species must move from providing conditions just for those species to considering ecological systems in their entirety. Certainly, the challenge to resource professionals is great. If we do not meet this challenge, natural systems will continue to erode and the very health of this planet will be in jeopardy.

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Broad-Scale Population Declines in Four Species of North American Quail: An Examination of Possible Causes

Leonard A. Brennan[1]

Abstract — Christmas Bird Count (CBC) data from 1960-1989 indicate that California quail (*Callipepla californica*), northern bobwhite (*Colinus virginianus*), and scaled quail (*Callipepla squamata*) populations have experienced significant declines in major portions of their geographic ranges. Additionally, surveys and hunter bag returns during the past 50 years indicate that mountain quail (*Oreortyx pictus*) populations have experienced a series of local extinctions across broad areas (several thousand km^2) in Idaho and Nevada. Although changing land uses can be related to these declines, no single factor can be linked to all species. For northern bobwhites, clean farming methods in agricultural environments and intensive, high-density pine-dominated silviculture seem to be the two major reasons for broad-scale population declines, especially in the southeastern states.

For mountain quail, regional extinctions in Idaho and Nevada are apparently related to two factors: (1) intensive agriculture and associated hydro-power reservoir impoundments along the Snake River corridor, and (2) disruption of key habitat resources along secondary riparian corridors by excessive cattle grazing. Factors responsible for declines in California quail and scaled quail populations are at present unknown, but are apparently related to abuses associated with excessive grazing of western rangelands. Management strategies that can be used to sustain quail populations in wildland environments are summarized in an ecological context.

INTRODUCTION

Historically, populations of New World quail (Odontophorinae) have been considered a sustainable by-product of many agricultural and silvicultural activities (Stoddard 1931, Leopold 1933, Rosene 1969, Leopold et al. 1981).

Abundant quail populations in rural and wildland environments improved the quality of life for people by providing recreational opportunities, economic returns from leasing lands for hunting, and other positive social values that resulted from a consumptive connection with wild vertebrate resources (Leopold 1933). Changing patterns of land use in agriculture and forestry have, however, called into question what were once symbiotic relationships between people, quail, farming and forestry.

During the past decade, reports indicated that northern bobwhite populations (*Colinus virginianus*) had declined at many locations (Roseberry and Klimstra 1984, Droege and Sauer 1990). This downward trend of one of the most common and widely distributed game birds in North America surprised many people. Further analyses revealed that northern bobwhites had indeed declined on both continental, regional and statewide scales (Brennan 1991, Brennan and Jacobson 1992).

The extent and magnitude of the bobwhite decline resulted in a Strategic Planning Workshop for Quail Management and Research in the United States that was held at the Third National Quail Symposium in 1992 (Brennan 1993a, 1993b). This workshop was the first attempt to develop a comprehensive

[1] *Leonard A. Brennan is Director of Research, Tall Timbers Research Station, located in Tallahassee, FL.*

strategy for quail management and research in the United States. It followed a regional strategic planning effort for upland game birds that was developed for western states by the Bureau of Land Management (Sands and Smurthwaite 1992).

My objectives in this paper are to (1) summarize long-term trends of quail populations at the continental scale in the United States and evaluate evidence of declines, (2) identify real and possible causes for observed declines and geographic range contractions, and (3) summarize strategies for management and research that might be used to sustain quail populations in an ecological context. My overall purpose is to use quail populations as an example of what happens when relationships between seemingly abundant vertebrate populations and land use practices are taken for granted.

Hopefully, these case histories will raise awareness of problems facing this unique, and often overlooked group of native avifauna.

EVIDENCE OF DECLINES

Brennan (1993a) summarized population trends for 6 species of quail in the United States from 1960-1989 based on Christmas Bird Count (CBC) data.

Three of the 6 species of quail in this study (California quail, *Callipepla squamata*; northern bobwhite; and scaled quail, *C. squamata*) showed statistically significant evidence of declines (Figure 1). None of the species in this study showed evidence of increasing populations (Figure 1).

Independent analyses of Breeding Bird Survey (BBS) data collected by the U.S. Fish and Wildlife Service (Sauer et al. 1993) corroborated the patterns shown by the CBC data. Although the mountain quail (*Oreortyx pictus*) showed no evidence of decline based on CBC data (Figure 1), game biologist surveys, hunter bag returns, and comprehensive field surveys have indicated that this species has undergone nearly a statewide, regional extinction in Idaho (Figure 2) and Nevada (Brennan 1993a).

POSSIBLE CAUSES OF POPULATION DECLINES

Characteristics of Declining Quail Populations

Populations decline when rates of birth and/or immigration are less than rates of death and/or emigration (Begon and Mortimer 1986). With respect to species such as quail, normal annual mortality rates can be as high as 80-90% (Rosene 1969, Leopold 1977, Roseberry and Klimstra 1984). Throughout evolutionary time scales, quail have evolved characteristics such as large clutch sizes (Leopold et al. 1981) and indeterminate egg-laying (Welty 1975) which serve as reproductive strategies that can potentially offset such high mortality rates. However, the high percentage of annual turnover that most quail populations experience means that when habitat components, or other key resources needed for survival are eliminated, populations can decline and disappear at an extremely rapid rate.

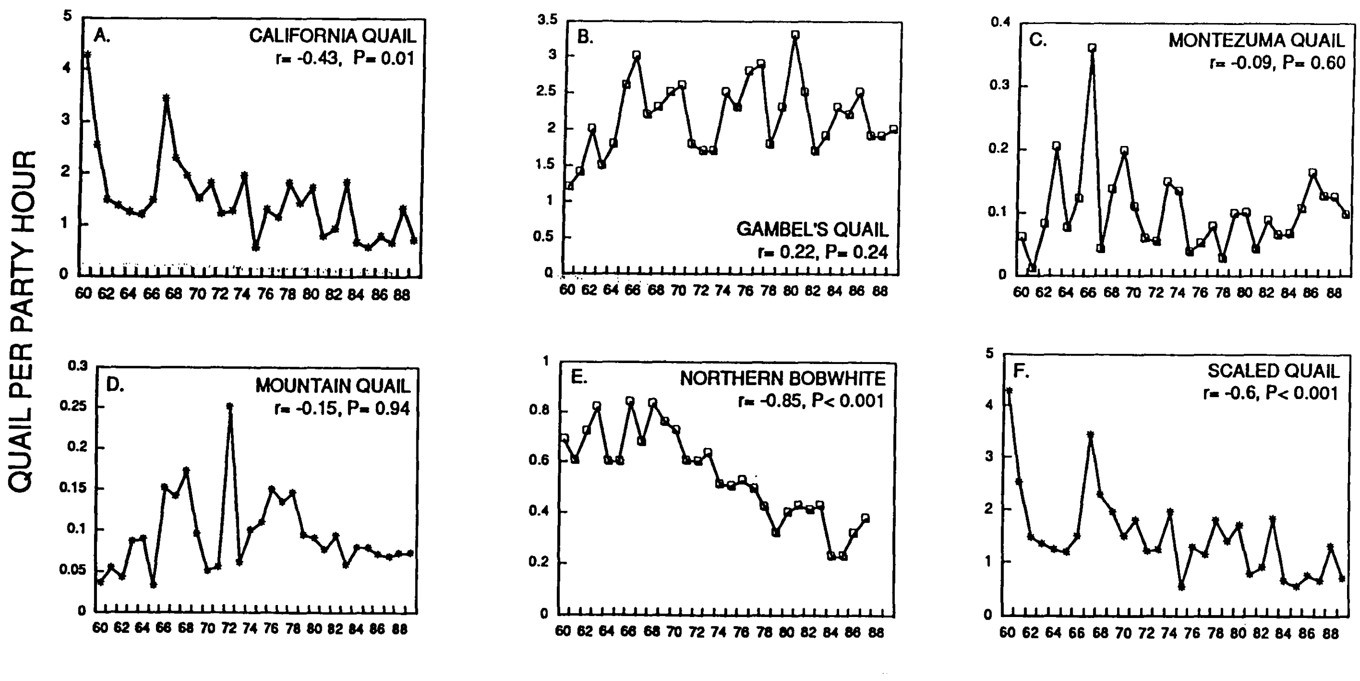

Figure 1. — Quail population trends in the United States based on 31 years of Christmas Bird Count data. Statistics are correlation coefficients (r) and probability that the slope of the regression line is significantly different from zero. Data from Brennan (1993a).

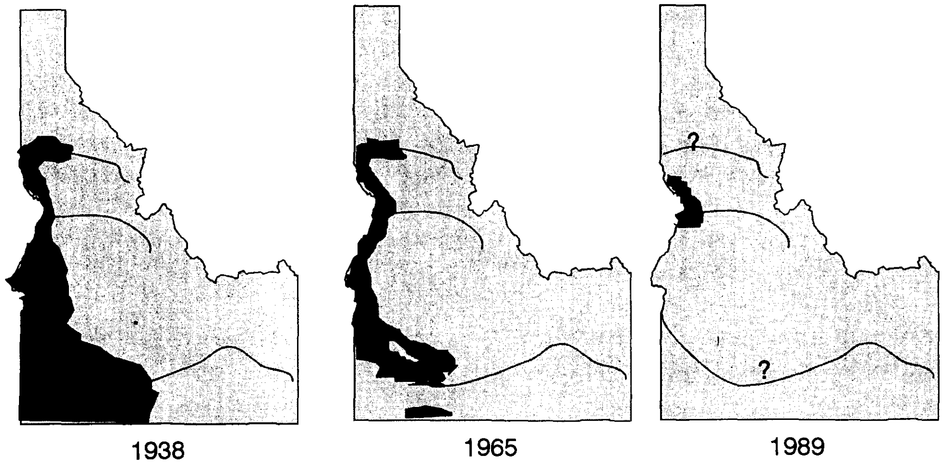

Figure 2. — Changes in the geographic distribution of mountain quail in Idaho during the past 50 years. 1938 map based on data from Murray (1938). 1965 map based on Ormiston (1966). 1989 map compiled by Idaho Fish and Game Department and other surveys.

If habitat or other limiting conditions do not become favorable within a relatively short time period, local and regional extinctions can occur. Where a limiting factor is abiotic (such as water from sporadic annual rainfall in the Rio Grande Valley of Texas), annual bobwhite population fluctuations can be dramatic (Lehmann 1984). In this situation, biotic habitat components remain relatively stable, and alternating wet and dry periods, which often persist across multiple years, are the primary cause of the fluctuations. However, when biotic habitat components are degraded through changing land use, application of agrochemicals, or other factors, populations of small galliformes such as quail or partridge (*Perdix perdix*) often decline and disappear quickly and thus undergo local or regional extinctions. With quail, such extinction processes may occur so quickly and at such a broad scale that recovery in a metapopulation context (Hanski 1991, Rolstad 1991) may not be possible.

Northern Bobwhite

Numerous factors have been attributed as being responsible for the broad-scale declines that northern bobwhites have experienced during the past 30 years. These factors range from the geographic expansion of the coyote (*Canis latrans*) in the south, to broad-scale increases in hawk and owl populations, to the invasion of the imported fire ant (*Solenopsis* spp.).

Experimental evidence linking factors such as these to bobwhite declines does not exist. In some situations, circumstantial evidence of raptor predation may be compelling in the absence of changing land use and lack of agrochemicals. However, linking factors such as coyotes and fire ants to the broad-scale bobwhite decline are myths that must be eliminated through education (Brennan 1991). Study of coyote foods in the southeast indicated that bobwhites are the least-common dietary item of coyotes (Wagner 1993). Although Allen et al. (1993) challenged interpretations made about the lack of widespread, direct antagonistic relationships between fire ants and quail (Brennan 1991), they have yet to present experimental or circumstantial evidence that the presence of fire ants limits quail population productivity.

Issues such as fire ants, coyotes, global warming, and other such potential epiphenomena are, in many ways, red herrings that threaten to steer us off the track of the real problems that are at the root of the bobwhite decline (Brennan 1993c). These problems relate to changing land use in agriculture and forestry, and in the ever-increasing urbanization that eliminates bobwhite habitat, and/or erodes its quality on a broad scale.

With bobwhites, changing land uses have clearly had a broad and largely negative impact on populations (Klimstra 1982, Brennan 1991). In agriculture, the herbicides may indirectly reduce or eliminate arthropod resources needed by growing chicks. Elimination of native weedy plants which provide substrates that produce abundant insects has broad, negative impacts on partridge (Potts 1986). This relationship may very well hold true for bobwhites and other quail, but it needs to be tested. In forestry, the widespread proliferation of high-density pine plantations, and reduction in use of prescribed fire has eliminated hundreds of thousands of acres of old-field habitats that once produced quail (Brennan 1991). In rural social contexts, the collapse of the tenant farming system in the southern U.S. and a broad-scale move from an agrarian to a service-based economy (Winter 1988, Bradshaw and Blakeley 1982) has apparently had devastating effects on quail (Brennan 1991).

The linkage between declining bobwhite populations and changing land use becomes clear when local case histories are examined in light of good quail management and habitat is either improved or maintained. For example, case histories in Mississippi (Brennan et al. 1991, Brennan 1992a, Brennan 1993c, Brennan 1993d) point to a dramatic increase in bobwhite numbers when habitat conditions are improved, but other effects (such as predators and fire ants) are kept constant. Conversely, when habitat conditions are allowed to erode, bobwhite numbers will decline concomitantly (Dimmick 1992). Furthermore, the vast area (200,000 ha) of private lands managed for bobwhites in the Red Hills region of southern Georgia and northern Florida continues to produce abundant quail populations at the same time bobwhite numbers continue to decline elsewhere in the southeastern coastal plain. The linkage between land use and bobwhites is an issue that has been raised on a regular basis for over 60 years (Stoddard 1931, Rosene 1969, Roseberry and Klimstra 1984), yet, often seems to be neglected in favor of some other more easily identifiable villain such as predators (Mueller 1989) or fire ants (Allen et al. 1993).

Mountain Quail

Mountain quail clearly represent a classic example of how quail populations can be sustained or eliminated as a function of land use. In the montaine areas of northern California, mountain quail populations are apparently stable, and can persist at densities of up to 30 birds per 100 ha (Brennan and Block 1986). Conversely, populations have undergone broad regional and local extinctions in Idaho as a result of key wintering and breeding habitats being eliminated as a result of anthropogenic changes to key aspect of their habitat.

In contrast to California where extensive areas of chaparral vegetation provide good-quality mountain quail habitat across large regions (Brennan et al. 1987), the local restriction of mountain quail to linear arrangements of creekside and riparian brush communities in Idaho has apparently made them vulnerable to elimination of wintering habitat from hydro-electric dams along the Snake River corridor and its tributaries. When mountain quail migrate from high elevation breeding habitats to low elevation wintering habitats, they can encounter a variety of risks, not the least of which are reservoirs that eliminate vast areas of wintering habitat. Furthermore, excessive grazing simplifies the floristic composition of the creekside brush communities on which these birds rely, and decreases their suitability as mountain quail habitat (Brennan 1992a).

We can also gain insight into factors that limit mountain quail in Idaho by looking at the characteristics of the places where they continue to persist. The remnant populations that are apparently self-sustaining are located in steep, isolated portions of the Snake and Salmon River Canyons in areas inaccessible to cattle. Although this information is circumstantial and not experimental, it provides strong inferential evidence that rangeland abuses from grazing may be responsible, at least in part, for the declines mountain quail have experienced in Idaho. It also offers evidence of an opportunity for improving habitat for this species by practicing good rangeland stewardship. The hydroelectric impoundments and intensive agriculture are clearly established fixtures along the Snake River corridor, and are not likely to change. Modifying the way cattle are managed is clearly the most significant opportunity for restoration of this quail in portions of its former range in Idaho.

Scaled Quail

Evidence of the scaled quail decline surprised many of the participants at the Third National Quail Symposium last year. Mechanisms responsible for the decline in scaled quail populations are not as well understood as the factors behind the northern bobwhite and mountain quail declines. There are, however, some potential relationships between excessive grazing and this decline that should be explored. Scaled quail clearly have an affinity for desert grasslands with sparsely scattered shrubs (Schemnitz 1961, Brown 1989). Homogenous grasslands without a shrub component are usually unsuitable for scaled quail (Schemnitz 1961). Excessive grazing by cattle removes or reduces grasses and forbs and tends to result in an increase in woody and shrub vegetation. Good range stewardship that allows residual grasses and forbs to persist through the winter results in lower scaled quail mortality and an increase in local populations (Brown 1989). Whether this is the case across broad portions of this birds' range and whether such a management strategy can be used to sustain scaled quail populations remains to be tested.

California Quail

Leopold (1977) provides a comprehensive overview of California quail biology and ecology in the context of land use. Although grazing can be used to improve the conditions of some environments for California quail, abuses of this practice can seriously degrade the quality of the habitat for this species. Leopold (1977:158) states "To increase usefulness of brush stands for quail, there must be cover at the ground level as well as overhead...often the most effective way to achieve this end is to exclude livestock from portions of the brush." Brush conversion projects and other so-called rangeland improvements have been known to have deleterious effects on California quail for years, and caused Leopold (1977:160) to state, "I am increasingly distressed at the progressive 'cleaning up' of field borders...[in] modern, slick, mechanical farming." Fifteen years later, these trends continue, and so too does the erosion of California quail numbers.

The widespread human population increase in California during recent years is not apparently directly responsible for broad-scale declines in California quail. CBC count circles from urban and suburban locations did not show a significantly greater than expected number of declines in California quail.

STRATEGIES FOR SUSTAINING QUAIL IN AN ECOSYSTEM CONTEXT

We need to begin with the modest assumptions that (1) quail are renewable resources, and (2) they can be sustained in the context of contemporary land use practices. Emerging trends in agriculture (Robinson 1990) and forest management (Sharitz et al. 1992) indicate that there is some promise and hope for stopping the broad-scale declines that many quail populations have been experiencing. However, whether the mainstream managers in forestry and agriculture adopt these philosophical changes remains to be seen.

In agriculture, the direct and indirect roles of agrochemicals with respect to quail (especially northern bobwhite) need to be assessed. The Conservation Headlands approach to partridge management in agricultural environments in England (Potts 1986) appears to have profound implications for integrating northern bobwhites in modern, production agriculture. This approach entails reduction of herbicide application around field perimeters so that weedy forbs and phytophagus insects can grow and provide food resources for growing partridge chicks.

In forestry, consideration needs to be given to uneven-aged management strategies that emphasize long rotation and single tree selection. Such forestry practices, when combined with frequent, annual burning, have sustained abundant huntable populations of northern bobwhites in the Red Hills plantation country of southern Georgia and northern Florida for over 60 years. Such land use practices can clearly serve as a model for habitat management in other parts of the northern bobwhite's range, especially on public lands where multiple uses are mandated. Another such model is the relationship between the endangered red-cockaded woodpecker (*Picoides borealis*) and northern bobwhites in pine forests of the southeastern coastal plain (Brennan and Fuller 1993). Brennan (1991) and Brennan et al. (1993) observed a significant, positive response of northern bobwhites to habitat management for the red-cockaded woodpecker at Noxubee National Wildlife Refuge in east-central Mississippi. Conversely, the private plantations that have been managed for bobwhites in the Red Hills region of Georgia and Florida support the largest extant population of red-cockaded woodpeckers on private lands (T. Engstrom, personal communication, Tall Timbers Research Station). In northern California, Block et al. (1991) observed that mountain quail were loosely affiliated with a guild of approximately 8 species of birds that shared an affinity for brushy and chaparral-dominated vegetation. Identifying similar linkages that establish positive relationships between management for species of quail and other terrestrial vertebrates (or vice versa) is clearly needed.

In contrast to some of the recent potentially positive conceptual developments for integrating quail with other wildlife resources in forest and agricultural environments, similar relationships in rangeland environments have apparently not been established. Range managers seem to be uncooperative when it comes to implementing comprehensive stewardship and adopting a pay-as-you-go philosophy (Ferguson and Ferguson 1983). The recent uproar at the proposal to lower subsidies for public land grazing fees that encourage overgrazing and associated abuses and link grazing fees on public lands with fair market values is a classic example of this recalcitrant attitude. Whether these complex, wicked problems (Allen and Gould 1986) of public land management can be solved remains to be seen.

The fate of all quail, and many other vertebrates as well, are clearly linked to the ways that we farm our land, graze our grass, and manage our forests. Focusing on strategies that maintain the integrity and functional processes of ecosystems (Regier 1993) would clearly be the most effective way to sustain populations of wild quail. Maintaining system integrity with an ecosystem approach allows managers the opportunity to provide for the annual cycle needs of the birds.

Consider the alternatives. There have been vast amounts of resources poured into recovery efforts aimed at the endangered masked bobwhite (*Colinus virginianus ridgwayi*). Recovery efforts were continually met with failure until a large tract of land (Buenos Aires Ranch) was purchased and managed as a refuge (Brown 1989). Even today, quantitative descriptions of masked bobwhite habitat components are not available, and habitat management on Buenos Aires is largely based on the "best guess" approach, because reliable information has not been compiled (W. Kuvlesky, personal communication, Buenos Aires National Wildlife Refuge).

The masked bobwhite, lesser prairie chicken (*Tympanuchus pallidicinctus*) and other once common game birds have been driven to the brink of extinction by changing land use practices. If contemporary trends in land use continue, and an ecosystem approach to sustaining quail and other wildlife resources is ignored, then we will most likely add other species of once common galliformes to this list. To the naysayers who doubt that birds as common as quail can be potential candidates for extinction, I offer the example of the passenger pigeon (*Ectopistes migratorious*).

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Conservation Planning: Lessons from the Spotted Owl

R. J. Gutiérrez[1]

Abstract —The spotted owl (*Strix occidentalis*) has helped natural resource managers focus on the appropriate scale of management (single species versus larger systems). While larger scale management systems are necessary, endangered species management is relevant. The scientific process and the presence of empirical knowledge are necessary to establish scientific credibility. Scientific credibility is the key to acceptance of conservation plans by the courts, Congress, and the public. Science as a process includes not only gathering knowledge but also communication, appropriate use of models, appropriate inferences, peer review, and other features. The relationship between wildlife biologists and forest managers has deteriorated as a result of this conflict. Perhaps this is due to the differences in operational and philosophical paradigms that govern each group. Another reason could be the education process prevalent within these different paradigms. The educational system could play a greater role in preparing future resource specialists for the challenge of managing complex ecological systems. The lessons learned from the spotted owl controversy are relevant not only to endangered species but also to ecosystem management and new forestry initiatives.

INTRODUCTION

More has been said or written about the spotted owl (*Strix occidentalis*) than any other threatened or endangered species because of its association with ancient forests. The bird has declined in many places primarily because of logging its primary habitat (Gutiérrez 1993b). As a result of past declines and future projected declines in its habitat both the northern (*S. o. caurina*) and Mexican (*S. o. lucida*) subspecies have been declared threatened (USDI 1990, USDI 1993b). At least 9 management plans have been proposed to maintain viable populations of various spotted owl subspecies (USDA 1988, Thomas et al. 1990, USDA 1991, Johnson et al. 1991, SOWS 1991, USDA 1992, Verner et al. 1992, Thomas et al. 1993, USDI 1993a). Status and literature reviews have become commonplace because of the explosion of knowledge about this owl (see reviews in Gutiérrez 1993a).

We probably know more about the status and ecology of the spotted owl than of any other threatened or endangered species (Gutiérrez 1993a). Most of this information stems from research culminating in the past decade. This information base has allowed science-based management plans. Since my research group and I are associated with a university, our work has an air of independence because we presumably have no vested interest in the outcome of the research and political pressures may be less than for government scientists. Finally, I am a member of both the northern spotted owl recovery team and the California spotted owl technical team. In view of my independent research and management involvement, I wish to share my perspectives on conservation planning for the spotted owl and the lessons I have learned.

I present some insights I have gained in this process as they may affect future conservation planning, particularly "New Forestry" initiatives. Although many professionals believe the spotted owl is a major obstacle to forest management, I believe the bird galvanized our recognition that ecological systems cannot be managed piecemeal.

[1] *R. J. Gutiérrez is Professor of Wildlife Management, Department of Wildlife, Humboldt State University, Arcata, California 95521.*

LESSONS FROM THE SPOTTED OWL: THE ROLE OF SCIENCE IN RESOLVING CONFLICT

Science as a Process

Throughout the spotted owl conflict, the role of science as a process has been shown to be decisive in litigation, judicial review, congressional support, and public acceptance of management plans or inferences about the fate of the owl. Key features of this scientific process have been repeatability and generality of results; full disclosure of methods, experimental design, analytical process and results; and willingness of investigators to present inferences drawn from the results. The objectivity and openness with which individual scientists have operated have been critical to the fate of plans. Even when a plan such as USDA (1988) has been rejected, it was rejected because the preferred alternative carried an unacceptable risk of failure in the eyes of the court and concerned citizens. This level of risk was noted by the scientists themselves (see also Marcot and Holthausen 1987, USDA 1988).

Murphy and Noon (1992) described the development of the Interagency Scientific Committee's (ISC) spotted owl management plan. This plan has been challenged repeatedly by the timber industry, individual foresters, advocate scientists (those scientists hired specifically to challenge the plan), and lawyers. The plan survived because of 1) the scientific process used to develop the conservation strategy; 2) the scientists' willingness to articulate the process leading to the inferences about the potential response of the owl to a reserve design; and 3) the empirical knowledge about the owl that supported the plan.

It is evident that courts, politicians, and the public are becoming more sophisticated in forestry related issues. Thus, those advocating more holistic resource management should be aware that the process they follow as well as the knowledge they use to support novel approaches to sustainable resource exploitation will be heavily scrutinized. The days of pronouncing change without science-driven plans are over.

Facts

Theory can be a powerful tool in the development of endangered species conservation plans. However, it pales in comparison to empirical knowledge about the life history of a species. A theory used to develop a plan without an empirical base will not be scientifically credible, if an alternative plan is supported by natural or life history information.

For example, the first U. S. Forest Service northern spotted owl management plan called for 405 ha spotted owl management areas (SOMA) to protect suitable habitat (see USDA 1988). In addition, this plan called for as few as 500 SOMAs. The number 500 was chosen on the basis of a widely misunderstood (i.e., by wildlife biologists and forest managers) theory of maintenance of genetic diversity (Franklin 1980). Since loss of genetic diversity was receiving much attention in the budding "conservation biology" field (Soulé and Wilcox 1980), it was assumed that genetic diversity was a key to maintaining spotted owls. The SOMA plan lacked credibility because simple facts of natural history (e.g., spotted owl home ranges are on average much larger than 405 ha; owls usually exist as adjacent rather than isolated pairs) suggested that this plan neither accounted for home range needs nor considered social and population dynamics. Further, it was quickly shown that genetic considerations were unimportant relative to short-term demographic change (Barrowclough and Coats 1985). Thus, not only was the theory irrelevant to the problem, but also the structure of the SOMA system was inappropriate in light of natural history.

Empirical knowledge has repeatedly proven to be a powerful force in developing spotted owl plans. The northern subspecies was listed, in part, on the basis of declining trends in many owl populations, the absence of owls in areas where they were expected (e.g., Western Washington Lowlands Province), and declining habitat trends (USDI 1990). The data showing decline were birth and death rates estimated from capture-recapture studies. The absence of owls from certain areas was noted from detailed surveys throughout the bird's range. The inference of declining habitat was based on owl habitat selection studies (e.g., Forsman et al., 1984) and rates of logging. In fact, no empirical study has not demonstrated that spotted owls show strong selection for either primary (old growth or mature) forests or forests with complex structure and relatively large trees (see Bart and Earnst 1993 for a summary).

Knowledge is strength because it supports a planning endeavor. It provides the mechanism for articulating and defending the plan to decision-makers. It also allows decision-makers to make informed choices if scientists clearly interpret the inferences taken from data (see below).

Although many wildlife managers are concerned over the amount of money spent on spotted owl research, it is justified for the following reasons. The spotted owl is the first species declared threatened when populations were still relatively widespread. Thus, the question seems to have been "how many owls do we need?" This question is often posed as "minimalist" theory (i.e., minimum viable populations, e.g., Shaffer 1987). In order to answer the question, basic ecological information is needed. Following the acquisition of this information, a theoretical framework is needed to predict the consequence of various impacts on owl populations. This means that we must be able to model population dynamics and habitat trends over time. Finally, populations must be monitored to assess the efficacy of the management plan. While a huge amount of money has been spent on spotted owls, the original questions were beyond what wildlife biologists had confronted previously and this warranted the expenditures. In any event, all of this money is equivalent to old growth timber value within 1-2 owl home ranges at today's prices! Further, nearly all scientists agree that a pro-active approach to conservation (i.e., when a species is still relatively abundant) is much less costly than "emergency room" approaches.

Models

Models are important to assess potential affects of either population trends or management impacts on long-term viability of the population. However, models cannot replace empirical knowledge because models should be based on empirical knowledge. From my non-model builder's viewpoint, models are tremendously influential. Models themselves have formed the basis for arguments about the reliability of a plan in political and legal settings. We as scientists and resource managers must stress the fact that models are only tools and are only as good as their foundation. They should be used in an exploratory fashion and given credence only after empirical testing. If models are allowed to become the focus of conservation planning in lieu of empirical information, we will see a chaotic future in conservation as advocate scientists produce a battery of dueling models (e.g., see Boyce 1987).

In summary, appropriate management plans are based on knowledge and basic research is needed to achieve this knowledge. Models are tools that can be used to explore patterns suggested by knowledge.

Communication

Presentation of information is as important as its acquisition. Decisions-makers, faced with economically consequential choices, naturally tend to favor economic considerations. This is true of the spotted owl issue (Thomas and Verner 1992). Therefore, it is important that the manner in which information is presented to decision-makers be unambiguous and contain information that not only they can use but also those who challenge those decisions can use. As scientists we have an obligation to present the entire set of facts. One example is the presentation of point estimates and measures of their variability. This simple information can make the difference between the success and failure of a plan (e.g., the SOMA plan).

Appropriate Inferences

Appropriate inferences drawn from empirical or theoretical work have been critical to the acceptance of conservation plans. The adoption of a 405 ha SOMA was an inappropriate inference, given available information on owl home range sizes. This figure actually fell below the observed range of owl home range sizes: 408 ha of old forest (Forsman et al. 1984). The recommendation of 405 ha SOMAs was made not on scientific grounds but on economic ones. This and all other spotted owl conservation plans have been driven in part by economic considerations (Thomas and Verner 1992).

The ISC developed a habitat reserve system only after a series of hypotheses were tested regarding population stability and habitat selection (Thomas et al. 1990). Empirical knowledge about the birds supported those tests. Finally, population dynamics modeling was used to structure the reserve design (specifically, the number of reserves and their sizes). Thus facts and theory led to a reserve design that was more appropriate to the conservation of owls than was the SOMA plan. However, the reserve design itself is still a hypothesis that needs to be tested (Thomas et al. 1990).

To develop a management plan for the California spotted owl (*S. o. occidentalis*), a technical team was formed by the U. S. Forest Service (Verner et al. 1992). In contrast to northern spotted owls, California spotted owls were relatively uniformly distributed and it was not certain that populations were declining (see Verner et al. 1992). However, it was evident that California spotted owls were habitat specialists and that their habitat was declining under proposed forest harvest strategies (Verner et al. 1992). A reserve design like the one proposed by the ISC was rejected on the basis of empirical information. Verner et al. (1992) developed an interim plan that protected limited habitat around individual owl sites and also protected the large (> 76 cm) trees throughout the Sierra Nevada that characterized California spotted owl habitat. Thus, a plan was conceived that was appropriate to the data. The California spotted owl plan has withstood withering attacks by foresters and the timber industry partly because inferences drawn were based on empirical information.

Much has been said regarding the role of scientists as advocates. Wildlife management evolved in a conservative arena. Thus, many wildlife scientists are reluctant to advocate a position on conservation decisions. Each scientist must choose whether to be an advocate or to remain neutral regarding his or her own information. However, I believe that scientists have an obligation to explain their work as well as the inferences allowed by their work because they know the data and its limitations best. If a scientist does not advocate action based on his or her work, many others will do the same.

Peer Review

In peer review, qualified scientists review the work of other scientists. In order to be effective, referees must be chosen by a neutral third party to avoid a conflict of interest. Ideally, referees are unbiased, honest, and knowledgeable. Referees review the experimental design, methods, analysis, logic, and inferences in a rigorous manner. Peer review is absolutely necessary for the progress and integrity of science and scientists alike. It is also important for management plans to be subjected to peer review not only for their scientific content but also for the inferences that directed each decision within the plan (e.g., a reserve plan vs. a non-reserve plan). Scientific work receiving the benefit of such scrutiny usually is held in higher regard than non-peer reviewed work. It has been the *modus operandi* of most spotted owl scientists to seek peer review. In addition, all recent conservation plans have been peer reviewed by appropriate scientific societies. These evaluations have given current spotted owl management plans scientific credibility because the scientists and planners have invited criticism and evaluation. In contrast, many documents propose alternative management plans that have been publicized without peer review (e.g., Craig 1986, SOWS 1991, Anon. 1991). These documents almost uniformly lack scientific credibility (e.g., see Simberloff 1989 for one review). A corollary of peer review is that authors of plans, reviews, or scientific documents should be identified. Recently, anonymous reports have appeared that claim no specific author (e.g., Anon 1991). These anonymous reports lack credibility because authors are not identified.

Delphi Approach

The "Delphi Approach" is a form of professional judgment. If a rigorous, quantitative analysis cannot be made, a consensus among experts can be used to draw inferences or to develop tentative aspects of conservation plans. This approach is well recognized as the basis for many of the Habitat Evaluation Procedures of the U. S. Fish and Wildlife Service. One of the few "weak" links in the ISC strategy (Thomas et al. 1990) is the application of the "50-11-40 rule". This construct states that at least 50% of the landscape not explicitly protected within owl reserves should be covered by trees of at least 11 inches diameter at breast height and also be covered by 40% canopy closure. This rule presumably facilitates juvenile dispersal (Thomas et al. 1990). Yet there is a paucity of information regarding habitat selection by juvenile spotted owls. Thus, there is little empirical support for this "rule", but its acceptability lay in the professional experience of the ISC as well as their honesty and openness following their recommendation. In addition, science as a process governed their deliberations. The delphi approach under some circumstances is acceptable if there is no other alternative. In fact, it appears that the courts are willing to accept professional judgment in these matters if there is no other alternative. I propose a "litmus test" for a delphi procedure. That "test" is the willingness of the individual scientists to defend the plan. Often a weak link in a plan such as the 50-11-40 rule is considered a fatal flaw, but Murphy and Noon (1991:776) argue that a plan is a strong as its strongest link rather than as weak as its weakest link.

Single Species Management

One of the most controversial aspects of endangered species management is the high cost of managing species like the spotted owl. If every species received the same attention as the spotted owl the cost of management would be prohibitive. Thus, the owl controversy sharpened the focus of conservation planners and politicians alike on the issue of single species management as a conservation strategy. It was clear when I worked with the northern spotted owl recovery team that the owl was the tip of the "species iceberg." Anthony et al. (1993) demonstrated that while the recovery plan would help conserve old forest wildlife it would not assure their future viability. Concurrently, the court was forcing the U. S. Forest Service to expand the spotted owl analysis to include other species associated with late seral stage forests (Thomas et al. 1993). Conservation planners and land managers have been expanding their horizons as exemplified by the U. S. Forest Service's new scientific analysis team report on the spotted owl and other species (Thomas et al. 1993), New Perspectives Initiative by the U. S. Forest Service (Kessler et al. 1992) and ecosystem management ideas (e.g., Swanson and Franklin 1992, Franklin 1993). Despite our willingness to invoke new paradigms of forest management, our knowledge of the impact of forestry and other extractive resource uses on wildlife has been rudimentary at best.

Franklin (1993) argues that ecosystem management is the only course of action. However, the spotted owl and other endangered species are still relevant issues even in an expanded philosophy of management. For one thing they help focus our attention on more broadly defined problems like habitat conservation. In addition, some endangered species have such narrow habitat requirements and limited distribution that they must receive individual protection and attention. Thus, both approaches must be wed together. Contrary to popular belief, endangered species management, in general, is not disruptive to other activities, nor need it be a piecemeal protection program (Wilcove et al. 1993).

RELATIONSHIPS AMONG RESOURCE PROFESSIONALS

Professional relationships deteriorated between foresters and wildlife managers (and conservation biologists) during the spotted owl controversy. Part of this deterioration is the result of a changing emphasis in wildlife management from game management (where most game species are edge species and therefore often "benefit" from timber harvest) to general wildlife management (where timber harvest may be detrimental to some species). The other part, in my opinion, has to do with the educational framework of each discipline. In the past two decades I have never heard a forester say that he/she could not achieve some objective or "desired future condition" (with the exception of "recreating" old growth systems). This is not meant as a slanderous remark but merely illustrates the fundamental difference and philosophy upon which our educations are based. I present a few of these differences below.

The Agricultural Paradigm

Forestry operates primarily under an agricultural paradigm while wildlife management operates under an ecological paradigm. The agricultural paradigm is driven by the fact that one can cut trees (as a crop), prepare the soil, plant seedlings or seeds, fertilize the new crop or allow it to grow on stored nutrients, and then harvest the mature crop. This cycle can be repeated until the soil is exhausted or the system is ecologically disrupted (Maser 1988). The ability to achieve a desired result like growing a tree crop (at least in the short term) is not equivalent to achieving the replication of a myriad of complex forest ecological interactions (Maser 1988). Under the ecological paradigm, forests are viewed as complex systems with many structures and functions. Conversely, under the agricultural paradigm forests are viewed as areas that contain economically valuable trees. It is generally believed by foresters that they can "recreate" spotted owl habitat. In fact, one well known forest ecologist, speaking at President Clinton's forest summit, stated that foresters could recreate spotted owl habitat but not old growth. A wildlife biologist's question is "how can we recreate spotted owl habitat when we do not know what it is?" In order to understand truly a species' habitat we should know not only the structure of its habitat but what regulates the population and how regulatory mechanisms are related to habitat conditions. For the spotted owl, we do not understand population regulation nor is the information on habitat structure well developed (e.g., there is not a published paper on habitat characteristics of northern spotted owls north of the California border!). Thus, the ability to foster tree growth through silvicultural practices (agricultural paradigm) is not equivalent to "recreating" owl habitat (ecological paradigm). The current occupation of previously logged forests by spotted owls is not evidence that we can "recreate" owl habitat. These birds nearly always occupy regenerating forests with residual old forest components. We still do not know if these owls constitute viable populations nor if foresters can consistently derive these conditions in a timely manner through silviculture. Of course, the fact that we do not know the critical features of owl habitat does not mean that we should not attempt forestry experiments designed to maintain owl habitat. But these experiments should be done in a replicated, rigorous fashion that lends scientific credibility to the results (Murphy and Noon 1992, USDI 1993). For that matter, all new forestry initiatives should be considered experiments because of the uncertainty in their outcomes (see below).

The Uncertainty Principle

Uncertainty is the nature of science. Murphy and Noon (1991) discuss this fact in relation to their experience while working on the ISC strategy. Even some politicians recognize this feature of science (e.g., Gore 1992:38). Yet when discussing forestry and its implications for spotted owl habitat management, I am left with the impression that many foresters believe silviculture is predictable (i.e., there is little or no uncertainty). Some possible reasons for the certainty expressed by foresters are the agricultural paradigm discussed above, their view of ecological succession, and the educational process itself.

A debate raged among ecologists during the early part of this century on the nature of ecological succession. One school of thought proposed that succession was probabilistic (Gleason 1926) and the other proposed that it was deterministic (Clements 1916). Ecologists now believe that succession is generally probabilistic for a variety of reasons (Drury and Nisbet 1973). However, foresters apparently harbor a notion that succession is deterministic because, if it really were probabilistic, there would be a higher level of uncertainty in the outcome of silvicultural prescriptions. To be sure, one aspect of silviculture is to reduce the probabilistic nature of succession through manipulation (e.g., site preparation, fertilization, herbicide application). Nevertheless, the confidence expressed by foresters must be a function of their belief that succession will follow a predictable path. In contrast, wildlife biologists generally acknowledge uncertainty about the outcomes of habitat manipulation for non-game and endangered species (e.g., Shaffer 1987, Murphy and Noon 1991).

I cannot speak with much authority on the educational process of foresters, but I can about wildlife biologists. Humboldt State University has one of the largest wildlife programs in the United States. Thus, my colleagues and I teach many students each year. It is a cornerstone of our philosophy to express and illustrate the uncertainty of biological prediction and management planning. Rather than teaching that uncertainty is a weakness we teach that it is a strength because it forces one to work within one's limitations and to consider the consequences of one's actions. Thus, it serves to temper predictions and to err on the conservative side of actions that may affect species, particularly endangered ones. From my limited interactions with forestry students at four universities I have an impression that scientific uncertainty in forestry is not expressed as a philosophical component of their curricula.

As long as disciplines operate within different paradigms they have a lower chance of meaningful interdisciplinary work. Rather than being willing partners in creating novel solutions to problems, one seems to be dragged along by the other. My experience with spotted owls and forestry issues has taught me that there is little trust between forest managers and wildlife biologists. This lack of trust was explicitly addressed in the ISC report (Thomas et al. 1990). It was a topic of discussion many times within the recovery team. More recently, Interior Secretary Babbitt openly expressed this lack of trust in public forum. Finally, even foresters themselves recognize that there are fundamental problems with traditional forest management, particularly within the U. S. Forest Service (e..g, see the newspaper "Inner Voice").

How do we restore trust and regain our working relationship as resource managers? We have to re-assess the philosophical and scientific basis upon which our disciplines operate. The willingness of wildlife biologists working with spotted owls to expose their own uncertainty even when subjected to professional as well as public criticism has resulted in their being viewed with favorable public and judicial sentiment. The expression of uncertainty is viewed as honesty and humility by the general public. It is also necessary for scientific advancement. The public image of forestry and foresters, unfortunately, has suffered as a result of the controversy surrounding the spotted owl. One only has to recount mass media advertisement campaigns, which attempted to project a positive forestry image, as evidence of an unfavorable public image. On the other hand, wildlife biologists/managers suffer from being one of the few fields in which the lay person thinks he or she knows more than the professionals.

This issue of trust is not trivial because the success of new forestry initiatives must be executed in an open and honest manner with the sustainability of resources clearly the primary objective. I worry about these new initiatives because I already hear grumbling that it is euphemism for another round of resource exploitation.

My comments should not be construed to be forester bashing but rather to point to a starting place where this destructive cycle of mistrust and non-cooperation can be broken. Educators of foresters and wildlifers, myself included, have not done enough to bridge the disciplinary gap. Indeed, competition and antagonism is prevalent, if not encouraged by some faculty, among students toward their sister profession. This is not only counter productive but unprofessional. The rift between disciplines can best be bridged by young minds who still have not been exposed to the prejudicial legacy that we have built for them. I also believe that forestry, at least as practiced on public lands, should be governed under an ecological rather than an agricultural paradigm. New forest management initiatives imply a shift from an agricultural to an ecological paradigm. This may necessitate fundamental changes in the foundation of forestry school curricula.

DISCUSSION

Relevance of Spotted Owls to Ecosystem Management

Knowledge about spotted owls, and about other species (including plants), is gained from extensive and intensive research. This information can be the foundation for models to explore future scenarios in resource management. The call for changing forest management and policy is an event that ecologists welcome. Yet despite the eloquent rhetoric about new initiatives in forest management (Kessler et al. 1992, Swanson and Franklin 1992, Franklin 1993), the success of these initiatives will depend upon knowledge and action, not words and policy directives.

As part of the work of the northern spotted owl recovery team and the California spotted owl technical team, each team wished to know how much owl habitat remained, where it was located spatially, what were the past cutting patterns and history, where the future harvest was going to occur, etc. We were unable to answer these questions. Often the data were not computerized nor cataloged for easy retrieval. Process, procedures, and data bases on forest resources varied by national forest and sometimes by forest district. Our attempt to answer some fundamental questions necessary for future prediction was often bewildering because the information available was chaotic in organization, distribution and similarity. Often basic knowledge was unavailable, and if it was available it often was neither easily accessible nor in an usable form. If "adaptive" management will be the guide for ecosystem management, it should be based on knowledge derived from research (preferably experiments). Ecosystem management also implies that management will occur in a spatially explicit environment. The U. S. Forest Service, which manages the majority of owl habitat and which will serve as a primary initiator of new forestry initiatives (Kessler et al. 1993), does not have a physical infrastructure and centralized philosophy to accomplish ecosystem management, if I am to judge from my experience working on the owl teams.

I do not intend to criticize individuals who work in the U.S. Forest Service. Rather, I am stating a fact about the structure of the organization. Record keeping and retrieval are not spatially explicit or consistent throughout the Service. This will inhibit future planning, monitoring, and adaptation to a spatially explicit philosophy of landscape or ecosystem management. The following actions would help rectify this problem within the U.S. Forest Service. First, a substantial amount of time should be spent on developing an infrastructure to meet the needs of the organization (primary objectives being the acquisition of an agency consistent, spatially explicit Geographic Information System, a data base system, and baseline resource reference system. It would be effective to acquire a system that will be accessible to non-agency resource managers and scientists. Second, those who are statespersons of change in ecosystem management or other large scale initiatives should spend as much time advocating the development of infrastructure and science as a process as they do articulating philosophy. I serve on the boards of The Nature Conservancy and Tall Timbers Research Station and I have found that it is much easier to sell an idea than a reality. The reality of an infrastructure to support ecosystem management will be more expensive to achieve than the acceptance of new policy. Third, a balance needs to be found between individual initiative and creativity (i.e., maintaining the ability of national forests and districts to respond to local conditions) and consistency of record keeping (i.e., a centralized and internally consistent management system). Fourth, a philosophy of science can be introduced to new forestry initiatives by considering every action that affects the environment as a test of a hypothesis with a set of predictions and a course of monitoring.

The spotted owl helped change the way we view forests as well as resource and habitat sustainability. The process of science followed by spotted owl scientists and the acquisition of knowledge about the bird that supported management plans served as a model for endangered species conservation and, one hopes, for more comprehensive forest management strategies. In my view, rather than the owl being a roadblock to management, it served as a stepping stone to progress. Whether science as a process will continue to be a major influence on conservation decisions or whether the political considerations will supersede science will be a test of the importance of science to society.

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Despite occasional differences of opinion and disagreements, the line staff and administrators with whom my students, colleagues, and I have worked over the past 14 years have been extremely helpful to our research efforts. In addition, I never have had Forest Service administrators or wildlife scientists try to suppress or change the results of our research. Thus, I wish to acknowledge this academic freedom and to thank the multitude of agency scientists and staff who are trying to expand their role and that of their agency to meet the demands of 21st century resource management.

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What We Know and Don't Know About Amphibian Declines in the West

Paul Stephen Corn[1]

Abstract — The problem of declining amphibian species is thought to be particularly acute in western North America, but there are many gaps in our knowledge. Although several declines have been well-documented, other declines are anecdotal or hypothesized. Most documented declines are of ranid frogs or toads (*Bufo*). Species from montane habitats and those occurring in California have been best studied. Status of many desert species is unknown. Habitat destruction and introduced predators are the most common threats to amphibian populations. Some declines may represent natural variation in population size. Causes have not been determined for several cases where common species have declined over large areas. There are important considerations for ecosystem management, whether changes in amphibian populations are natural or caused by human activities. Causes for declines must be known so that management can be prescribed (or proscribed) to eliminate or minimize these causes. The natural variability of amphibian population numbers and the complexity of metapopulation structure emphasize the necessity of considering multiple temporal and spatial scales in ecosystem management. The decline of amphibian species throughout the world has received considerable recent attention (e.g., Blaustein and Wake 1990, Griffiths and Beebee 1992, Yoffe 1992). Much of this attention derives from a workshop held in February, 1990 on declining amphibians sponsored by the National Research Council Board (NRC) on Biology in Irvine, California (Barinaga 1990, Borchelt 1990). Because of media attention in the aftermath of this conference, it is a popular perception that amphibian declines are a new phenomenon that herpetologists have been slow to recognize (Griffiths and Beebee 1992, Quammen 1993). However, concern about amphibian populations in the United States dates back over 20 years. Beginning in the 1960s, a large, well-documented decline of northern leopard frogs (*Rana pipiens*) occurred in the upper Midwest (Gibbs et al. 1971, Hine, 1981, Rittshof 1975).

Participants in the NRC workshop concluded that declines of amphibians were a global problem. Unfortunately, another conclusion was that much of the information on declines was anecdotal and few causes of declines had been discovered. In the past three years, some progress has been made in documenting the extent of amphibian declines, but many of the causes remain undiscovered.

In North America, amphibian declines have been most numerous in the West and have occurred among species occupying both montane and low-elevation desert habitats. Several species have declined or disappeared from relatively undisturbed habitats, including National Parks and Wilderness Areas (Bradford 1989; Bradford et al., in press; Carey 1993; Corn et al. 1989; Fellers and Drost, in press). Amphibians are key components of many ecosystems, both on small (Seale 1982) and large (Hairston 1987, Petranka et al. 1993) scales, so their disappearance may complicate efforts to manage ecosystems on a sustainable basis.

[1] *Paul Stephen Corn is Zoologist, USDI Fish and Wildlife Service, National Ecology Research Center, 4512 McMurry Avenue, Fort Collins, Colorado 80525-3400*

My objective for this review is to summarize the current knowledge of amphibian declines in the western United States (west of the Great Plains). Many declines are now well documented, and the causes for many of these have also been determined. I will describe the unknown–suspected declines that have not been documented and declines for which causes have not been determined. Finally, I will discuss the problems amphibian declines create for ecosystem management.

KNOWN DECLINES

At least 79 species of amphibians inhabit the western United States, including 38 salamanders and 41 anurans (frogs and toads) (Stebbins 1985). Only three of these taxa are listed by the U. S. Fish and Wildlife Service (USFWS) as threatened or endangered. The desert slender salamander (*Batrachoseps aridus*) is endangered in southern California because it is an endemic species restricted to one moist desert canyon (Bury et al. 1980). The Santa Cruz long-toed salamander (*Ambystoma macrodactylum croceum*) is endangered in the Monterey Bay area of central California because most of its breeding ponds have been drained for urban development or agriculture (Ruth 1988). The Wyoming toad (*Bufo hemiophrys baxteri*) is endangered in the Laramie Basin of southern Wyoming (Baxter et al. 1982). The cause or causes of the decline of this species are still unknown. Besides these listed species, moderate to serious declines of several other amphibians have been described in recent years.

No widespread declines of salamanders have been documented. Collins et al. (1988) described the loss of at least 4 of the 17 known populations of the Huachuca tiger salamander (*Ambystoma tigrinum stebbinsi*) in southern Arizona. The size of a population of Arizona tiger salamanders (*A. t. nebulosum*) inhabiting a small group of high-elevation ponds in central Colorado declined from about 1700 in 1982 to 200 in 1987 (Harte and Hoffman 1989), but the same population fluctuated from 2900-3500 salamanders in 1988-1991 (Wissinger and Whiteman 1992).

Most declines have been of anurans. Declines have been verified for several species of toads (genus *Bufo*). Baxter et al. (1982) noted that declines of the Wyoming toad began in the mid-1970s, and within a decade it was considered probably extinct (Lewis et al. 1985). A single population was discovered in 1987 (Anon. 1987). The persistence of the Wyoming toad now relies on a small captive population, because no egg masses have been observed in the wild population since 1991 (Wyoming Dept. Game and Fish, unpublished data). The Amargosa toad (*B. nelsoni*) is also an endemic species, restricted to a few springs in southern Nevada and eastern California. Altig and Dodd (1987) found this species to be absent from most known localities in Nevada.

The Yosemite toad (*B. canorus*), more widely distributed than the previous species, occurs at high elevations in the Sierra Nevada in California. This species apparently has declined over much of its range. At seven locations in the eastern Sierra Nevada, Kagarise Sherman and Morton (1993) observed averages of 6-70 toads per site per day in 1976 but only 0-5 toads per site per day in 1990. Bradford and Gordon (1992) found this species present at only 17 of 235 sites within 30 randomly selected 15 km^2 (5.8 mi^2) study areas above 2,440 m (8,000 ft). The boreal toad (*B. boreas boreas*) was widely distributed and abundant in the southern Rocky Mountains in Colorado, southeast Wyoming, and northern New Mexico. Corn et al. (1989) failed to find the boreal toad at 49 of 59 (83%) known localities in the Front and Park Ranges of Colorado and the Medicine Bow Mountains in Wyoming. Carey (1993) documented the extinction of several populations of this species in central Colorado. Continued surveys in Colorado have located only a few scattered small populations (Corn, unpublished data). Olson (1992) observed declines of two populations of boreal toads in the Oregon Cascades in 1991 and Blaustein and Olson (1991) described mortality of thousands of boreal toad eggs at a third site nearby in 1990.

All other documented amphibian declines in the West are of frogs of the genus *Rana*. Hayes and Jennings (1986) compiled references, both anecdotal and well-documented, that indicated every ranid frog in the West had undergone either local or regional declines. Leopard frogs (*R. pipiens* Complex) have been hit particularly hard. The Vegas Valley leopard frog (*R. fisheri*) occurred only in the warm springs of the Las Vegas Valley. This species is now extinct (M. R. Jennings 1988b). The relict leopard frog (*R. onca*) occurred along the Virgin River in southern Nevada and southwestern Utah. This species has recently been considered conspecific with the Vegas Valley leopard frog (Pace 1974) and also extinct (M. R. Jennings 1988b). However, R. D. Jennings (1993) found three small populations of relict leopard frogs in Nevada, and he presented a morphological analysis indicating that relict and Vegas Valley leopard frogs were distinct species. Corn and Fogleman (1984) observed extinction of northern leopard frogs from a few ponds in northern Colorado, and more extensive surveys found this species absent from 29 of 33 (88%) known localities in northern Colorado and southern Wyoming (Corn et al. 1989). Northern leopard frogs were also absent from 13 of 28 (46%) known localities in Arizona (Clarkson and Rorabaugh 1989). Chiricahua leopard frogs (*R. chiricahuaensis*) were absent from 34 of 36 (94%) of known localities in Arizona (Clarkson and Rorabaugh 1989), and lowland leopard frogs (*R. yavapaiensis*) have been extirpated from the lower Colorado River in Arizona and California and the Imperial Valley in California (Clarkson and Rorabaugh 1989; Jennings and Hayes, in press).

The U. S. Fish and Wildlife Service (USFWS) was petitioned to list the spotted frog (*R. pretiosa*) as threatened in 1989, and USFWS found that listing was warranted but precluded for populations of spotted frogs in western Oregon, western Washington, northern Nevada, southern Idaho, and the Wasatch Front in Utah (USFWS 1993b). Spotted frogs have been rare in western Oregon and Washington for several decades (Dumas 1966, Nussbaum et al. 1983). McAllister et al. (1993) searched

60 locations in western Washington from 1989-1991 and found only a single individual of this species at one site. Turner (1962) failed to find spotted frogs at several known localities in northern Nevada. Spotted frogs are currently restricted to a few disjunct areas along the Wasatch Front in Utah, but Ross et al. (1993) observed 126 adult frogs and 162 egg masses at 19 sites in 1991 and 124 adult frogs and 478 egg masses at 54 sites in 1992. In western Utah, spotted frogs are distributed among a few isolated marshes but are more numerous than in central Utah. Hovingh (1993) observed several hundred egg masses in seven populations in the Tule Valley in periodic observations from 1981-1991, and O. Cuellar (Dept. Biology, Univ. Utah, unpubl. manuscr.) found 354 egg masses at Gandy Salt Marsh in Snake Valley in 1992.

The Tarahumara frog (*R. tarahumarae*) occurs mostly in the Sierra Madre Occidental of Mexico, but all five known populations in southeastern Arizona and several other populations in northern Sonora have disappeared (Hale and Jarchow 1987). The California red-legged frog (*R. aurora draytonii*) was once perhaps the most common ranid frog in California, but it has undergone a long-term and severe decline in the San Joaquin Valley (Moyle 1973), Central Valley (Hayes and Jennings 1986), and apparently has been extirpated from drainages in the Mojave and Sonoran Deserts in southern California (Jennings and Hayes, in press). The USFWS was petitioned to list the California red-legged frog as threatened or endangered, and the agency recently determined that such action was warranted (USFWS 1993a).

Mountain yellow-legged frogs (*R. muscosa*) were common in the Sierra Nevada in California, but have been absent from aquatic habitats at middle and lower elevations for several decades (Bradford 1989). This species has recently declined also in remaining high elevation sites. In 1989-1990, Bradford et al. (unpubl. manuscr.) resurveyed 27 sites in Sequoia and Kings Canyon National Parks where mountain yellow-legged frogs were observed in 1978-1979, and the species was absent from all but one site. Bradford and Gordon's (1992) survey of 30 randomly selected study areas found this species present at only 12 of 235 sites. Fellers and Drost (in press) searched 16 known localities and 34 other areas of Lassen Volcanic National Park, California in 1991 for Cascades frogs (*R. cascadae*). They found only two individuals at one site. Nussbaum et al. (1983) mentioned a decline of this species in Oregon, but quantitative data for northern populations have not been published.

Data Limitations

Several studies have documented amphibian declines by surveying for presence or absence of a species at known localities where previous occurrence was recorded from museum specimens or from the literature (Altig and Dodd 1987; Bradford et al., unpubl. manuscr.; Clarkson and Rorabaugh 1989; Corn et al. 1989; Fellers and Drost, in press; Jennings and Hayes, in press). Surveys for presence or absence have some problems. Some species, particularly leopard frogs, may be difficult to detect, because adults disperse after the breeding season. Failure to detect a species that is present, of course, overestimates any decline. There are several other caveats that apply to presence/absence data, all of which may overestimate the number of known localities which, in turn, can lead to an overestimate of decline. First, museum records do not always represent breeding populations. Second, some records are from marginal habitat, where breeding may occur but is rarely successful. This is a particular concern for localities at high elevations, but it is extremely difficult in practice to judge which localities are marginal. Finally, museum and literature records are usually combined over several decades, and this ignores natural processes of extinction and recolonization.

There are solutions to these problems. Determination of presence or absence should not be based on single surveys. Multiple surveys of single sites in different seasons are necessary to verify that a species is absent. One method to alleviate the inflation of known localities is to conduct presence/absence surveys for more than one species. The total number of localities searched will usually be greater than the number of known localities for any one species. Presence of a species at many previously unrecorded localities suggests that a widespread decline is unlikely, even though the species may be absent from several known localities. For example, Corn et al. (1989) found tiger salamanders absent from 12 of 22 (55%) known localities but present at 11 new localities, chorus frogs (*Pseudacris triseriata*) absent from 20 of 56 (36%) known localities but present at 19 new localities, and wood frogs (*Rana sylvatica*) absent from 9 of 29 (31%) known localities but present at 9 new localities. I conclude that these species have not declined appreciably in the Rocky Mountains. Conversely, a species that is absent from most known localities and which is not found at many new sites probably has undergone a decline. Corn et al. (1989) found boreal toads absent from 83% of known localities and at only 2 new localities and northern leopard frogs absent from 88% of known localities and at no new localities. I conclude that both of these species have undergone serious declines.

SUGGESTED DECLINES

The foothill yellow-legged frog (*Rana boylii*) inhabits rocky streams at middle elevations in California and Oregon. Moyle (1973) felt that this species had declined in the San Joaquin Valley and the adjacent foothills of the Sierra Nevada, and Hayes and Jennings (1988) and Jennings (1988a) described a general decline in California. No data have yet been published, however, on numbers of populations that have disappeared or rates of decline.

Several species of amphibians inhabit or are associated with small streams in the forests of the Pacific Northwest (Bury 1988): tailed frog (*Ascaphus truei*), giant salamanders (four species of *Dicamptodon*), torrent salamanders (four species of

Rhyacotriton), and woodland salamanders (at least three species of *Plethodon*). Logging and associated road building destroy or alter amphibian habitat, especially through sedimentation in low-gradient streams (Bury and Corn 1988). Amphibian populations may be eliminated or severely depressed for several decades (Corn and Bury 1989). Because approximately 90% of the low- and mid-elevation forests west of the Cascades have been logged (Morrison 1989), and much of that in the last 40 years (Harris 1984), it is a reasonable hypothesis that populations of stream-dwelling amphibians have declined over much of the landscape (Welsh 1990). Raphael (1988) predicted declines of three species of terrestrial salamanders in northern California, based on continued harvest of old-growth Douglas-fir (*Pseudotsuga menziesii*) forest: Petranka et al. (1993) recently made a similar prediction for terrestrial salamanders in the Appalachian Mountains of the southeastern United States. There have been no studies, however, documenting changes in the regional distribution and abundance of stream amphibians.

CAUSES OF DECLINES

A variety of explanations have been offered for amphibian declines in the West, but few of these have been tested rigorously. One certain reason for the lack of experimentation is the lack of experimental subjects. Most amphibian declines have been observed after the fact, so causes for declines have been based more on correlative than experimental evidence. Causes for declines fall into two broad categories: human-induced (anthropogenic) or natural (usually climatic) factors. Most anthropogenic causes are attributable to habitat destruction or alteration.

Habitat Destruction and Alteration

Several amphibian declines are clearly attributable to conversion of wetland habitat to urban or agricultural use or by water development projects. The transformation of the Las Vegas Valley from a spring-fed wetland to a large city and the Colorado and Virgin Rivers to Lake Mead have caused the extinction of the Vegas Valley leopard frog and the near-extinction of the relict leopard frog (Jennings 1988b; Jennings and Hayes, in press). The large reservoirs and channelization of the lower Colorado River in Arizona and California and agricultural development of the Imperial Valley in California have eliminated appropriate habitat for the lowland leopard frog (Jennings and Hayes, in press). Similar changes are blamed for much of the disappearance of California red-legged frogs from desert drainages in southern California (Jennings and Hayes, in press). Moyle (1973) considered habitat alteration to be a factor in declines of California red-legged frogs and foothill yellow-legged frogs in the San Joaquin Valley. Jennings (1988a) considered alteration of riparian vegetation by livestock grazing to be an important factor in the decline of ranid frogs in California. The role of logging in the possible decline of amphibians in the Pacific Northwest was discussed previously.

Introduced Predators

Introduction of exotic species of predators, for which native species may have poor defenses, is a special case of habitat alteration. The bullfrog (*R. catesbeiana*), a large ranid from eastern North America, has become established throughout the West (Bury and Whelan 1984). Predation or competition by bullfrogs has been blamed for declines of relict leopard frogs (Cowles and Bogert 1936), spotted frogs (Dumas 1966), northern leopard frogs (Hammerson 1982), and California red-legged and foothill yellow-legged frogs (Moyle 1973). Hayes and Jennings (1986) pointed out that there was little experimental evidence to support this hypothesis, and suggested that predation by introduced warm water fish, mainly centrarchids (*Micropterus* spp. and *Lepomis* spp.) and catfish (*Ictalurus* spp.), and habitat alteration were equally likely to explain declines of ranids in California. Jennings and Hayes (in press) observed that other introduced potential predators, including mosquitofish (*Gambusia affinis*) and red swamp crayfish (*Procambarus clarkii*), as well as bullfrogs, were present at most historical native frog localities in southern California.

Introduction of salmonid fish (*Oncorhynchus* spp., *Salmo* spp., *Salvelinus*, spp.) into historically fishless waters is thought to be responsible for the decline of the mountain yellow-legged frog in the Sierra Nevada. Bradford (1989) sampled 67 lakes and documented that frogs and fish did not coexist in any of them. Bradford et al. (in press) found that presence of fish has fragmented remaining populations of mountain yellow-legged frogs in Sequoia and Kings Canyon National Parks. This fragmentation may be contributing to the continuing decline of this species (Bradford et al., unpubl. manuscr.).

Human exploitation might also be considered to be a form of predation. Although bullfrogs are a game species in most states, there is no evidence that other native frog species are sought in large numbers. There were large harvests of California red-legged frogs for food from 1888-1935 (Jennings and Hayes 1985), but there is no real evidence that this contributed to the current scarcity of this species.

Pollutants

Hale and Jarchow (1987) speculated that deposition of heavy metals from copper smelters in Arizona and Mexico was responsible for the disappearance of the Tarahumara frog, but evidence to support this hypothesis was lacking. Harte and Hoffman (1989) concluded that episodic acidification during snowmelt in Colorado may have caused mortality of tiger salamander embryos. Wissinger and Whiteman (1992) did not observe acid conditions in the same populations of salamanders, and found that salamanders did not breed during the initial stages of snowmelt. Anthropogenic episodic acidification occurs during initial snowmelt when acid anions (sulfate and nitrate deposited throughout the winter) are flushed out, lowering the buffering capacity of surface waters (Vertucci 1988). This is usually before there is open water in breeding ponds and before breeding begins for most amphibians (Vertucci and Corn 1993).

Chronic acidification (the permanent lowering of buffering capacity) is probably a minor occurrence in the Rocky Mountains that is not responsible for amphibian declines. Acid deposition is relatively low, and no amphibian species breed exclusively in habitats with the lowest buffering capacity (Corn and Vertucci 1992). Similarly, Bradford et al. (1992) argued that acid precipitation was not responsible for declines of mountain yellow-legged frogs and Yosemite toads in California.

Disease

Mass mortality of amphibians from disease is not uncommon and may be a natural feature of the biology of a species, or it may be induced by an anthropogenic agent. Redleg disease or other bacterial infections have killed larval tiger salamanders in Arizona (Collins et al. 1988) and Utah (Worthylake and Hovingh 1989), mountain yellow-legged frogs (Bradford 1991) and Yosemite toads (Kagarise Sherman and Morton 1993) in California, boreal toads in Colorado (Carey 1993), and Wyoming toads (Wyoming Dept. Game and Fish, unpublished data). Redleg affects amphibians whose immune systems have been weakened by stress, and Carey (1993) hypothesized that a regional anthropogenic stress was responsible for the declines of boreal toads in Colorado. This hypothesis explains the apparent synchronous decline over a large area, but the stressor has yet to be identified. The only potential anthropogenic stressor discussed by Carey (1993) was acid precipitation, but there are no data to support this hypothesis.

Kagarise Sherman and Morton (1993) suggested that stress from handling and observation may have contributed to redleg disease in Yosemite toads. They observed the greatest mortality in 1978-1979 when observations of breeding toads were most intense, including the use of drift fences and pitfall traps.

Climate

Weather is one of the most significant natural killers of amphibians. Many amphibians breed in temporary ponds that may or may not persist long enough for tadpoles to transform. Temperature extremes or fluctuations in water level during breeding may kill large numbers of embryos. Such short-term events are unlikely to have caused large declines of western amphibians. Many species are long-lived and occasional mass mortality of embryos or tadpoles can be tolerated (Olson 1992). However, several species of southwestern anurans inhabit streams in canyons, and short-term events such as flash floods (spates) can cause catastrophic mortality of adults. Flooding has been suggested as contributing to declines of California red-legged frogs (Hayes and Jennings, in press) and foothill yellow-legged frogs (Sweet 1983) in southern California, and Chiricahua leopard frogs in New Mexico (R. D. Jennings, Western New Mexico State Univ., Silver City, NM, pers. comm.). Metter (1968) described catastrophic mortality in tailed frog populations after flash floods in Idaho and Oregon. The streams were altered substantially, most tadpoles were washed away, and many adult frogs that survived suffered amputated limbs.

Excessive precipitation does not have to be concentrated in a brief period. Bradford (1983) documented mass mortality of mountain yellow-legged frogs in the Sierra Nevada. Overwintering frogs died from oxygen depletion in shallow lakes when heavy precipitation resulted in ice cover that was thicker and more persistent than normal.

Drought may also cause amphibian declines over large areas. Corn and Fogleman (1984) described extinction of several populations of northern leopard frogs in Colorado when breeding ponds dried after a severe winter drought in 1976-1977. Kagarise Sherman and Morton (1993) felt that low snowfall in several years since 1971 has contributed to the decline of the Yosemite toad by causing ponds to dry before tadpoles complete metamorphosis.

Carey (1993) listed cold weather as a potential stressor that could cause suppression of the immune system leading to redleg disease. Cold weather was associated with mortality of Tarahumara frogs (Hale and Jarchow 1988), mountain yellow-legged frogs (Bradford 1991), Yosemite toads (Kagarise Sherman and Morton 1993), boreal toads (Carey 1993), and Wyoming toads (Wyoming Dept. Game and Fish, unpublished).

Population Dynamics

It has been suggested that many changes in amphibian abundance that have been termed declines may be fluctuations that are within the natural range of variation in population size. Pechmann et al. (1991) monitored numbers of amphibians breeding at a pond in South Carolina and found that some species could be rare or absent for several years and then reach high abundance in one or two good years. Harte and Hoffman (1989) and Wissinger and Whiteman (1992) may have observed a similar phenomenon in the decline and recovery of tiger salamanders in Colorado. Amphibian populations can be extremely variable from year to year (Berven and Grudzien 1990, Pechmann et al. 1991), but stochastic variation is an unlikely cause when most populations in a large area decline or go extinct at the same time (for example, boreal and Wyoming toads).

Unknown

As yet, there are no satisfactory hypotheses to explain declines of boreal toads in Colorado, Wyoming toads, and mountain yellow-legged frogs in the Sierra Nevada. Boreal toads and mountain yellow-legged frogs were widely distributed and abundant, and the lack of a ready answer for their decline is alarming, even if the declines represent natural fluctuations in the populations of both species. Variable environments periodically turn hostile and produce periods of intense selection pressure, or "ecological crunches" (Wiens 1977), with associated declines in population size. If populations are too small, random demographic variation can create an extinction vortex (Gilpin and Soulé 1986) that makes extinction a common event.

IMPLICATIONS FOR ECOSYSTEM MANAGEMENT

Bury et al. (1980) reviewed the status of amphibians in the United States thought to be declining or in danger of declining. They listed 15 species from the West, including 9 salamanders of the family Plethodontidae, all of which were endemic or isolated species with small ranges. Most species were listed because they were highly susceptible to habitat destruction, and few actual declines were documented. There are now at least 15 species of amphibians with documented declines, including 7 species of anurans with large distributions (boreal toad, Yosemite toad, northern leopard frog, lowland leopard frog, Chiricahua leopard frog, California red-legged frog, and mountain yellow-legged frog). Declines of common, widely distributed species are much more of a problem for ecosystem management than is conservation of narrowly distributed endemics, for which complete protection of small areas of suitable habitat is the most appropriate action.

Any attempt at ecosystem management must take into account temporal and spatial variation in ecosystem processes (Landres 1992). This is especially true of amphibians, not only because of temporal variation in population size, but because amphibian populations are structured in a variety of ways. The term metapopulation describes a group of populations, linked by migration, that undergo a dynamic process of extinction and recolonization (Hanski and Gilpin 1991). A variety of metapopulation models have been developed that describe real populations with varying degrees of success (Harrison 1991). Different amphibian species may have very different metapopulation structures. In Virginia, wood frogs inhabiting small ponds within a radius of 1 km are essentially a single genetic population (Berven and Grudzien 1990). Ponds outside this radius are connected by occasional migration by juvenile frogs. This is very similar to the situation for pool frogs (*R. lessonae*) in Sweden, where populations more than 1 km from other populations have a higher probability of extinction (Sjögren 1991). However, red-spotted newts (*Notophthalmus viridescens*) that share the same ponds in Virginia with wood frogs are dominated by a single large, stable population that sends migrants to peripheral sites that suffer high rates of extinction (Gill 1978). Active management would be very different for wood frogs and newts, and specific recommendations for each species might be different. This is a potential conflict because both species occupy the same ponds.

Predation by introduced fish is a significant problem for amphibians and also native fish (Rinne and Minckley 1991) in the West. However, sport fishing is a huge industry supported by the public and administered by Federal land management and State game and fish agencies. If ecosystem management includes preserving native species (Samson 1992), non-native fish must be removed from large areas of the West. Such an action, although probably feasible, would generate much controversy and strong private and public opposition. To implement ecosystem management fully, there are many other equally difficult challenges and hard choices to be made.

In summary, there is much we know and don't know about amphibian declines in the West. We know that ranid frogs in the Southwest and California have suffered large declines, and several species of toads have declined throughout the West. We don't know the extent of declines of stream-dwelling amphibians in the Pacific Northwest, and we have little knowledge of the status of most desert species, such as spadefoot toads (*Spea* spp.). Habitat destruction and introduction of alien predators have probably caused most declines of ranid frogs. We do not know the causes of other declines, including boreal and Wyoming toads. Including amphibians in plans for ecosystem management may cause conflicts with other management objectives and desired conditions.

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Threats to and Sustainability of Ecosystems for Freshwater Mollusks

Patricia Mehlhop[1] and Caryn C. Vaughn[2]

Abstract — In North America, two groups of freshwater molluscs are most threatened by human activities and require ecosystem approaches to their sustainability. Prosobranch snails in the family Hydrobiidae are restricted to small spring systems and are limited by their relative immobility, dependence on highly oxygenated waters and use of gills. Many are narrow endemics of localized springs, which are altered by ground water depletion and surface water diversion and by changes in water quality such as nutrification and chemical pollution from non-point sources. Spring alteration can result in direct species extirpation. Conservation through threat assessment and abatement is recommended. Most rare and declining native mussels are Unionidae in riverine ecosystems. Their relative immobility, long lifespan, filter-feeding habits, and parasitic larval stage make them highly vulnerable to habitat disturbance. The major cause of their declines has been the fragmentation of river ecosystems through impoundments, channelization and other activities such as timber harvesting, which alter flow and sedimentation patterns. Fragmentation acts to increase the distance between mussel subpopulations and may have major consequences of the metapopulation structure of species, particularly rare species and those with narrow fish host requirements. As some populations are eliminated and dispersal distances are increased, demographic and genetic constraints will diminish the ability of local populations to respond to natural environmental disturbance as well as human-induced changes. Sustainable ecosystem management in river systems will require devising strategies to conserve mussel metapopulations.

INTRODUCTION

Lotic systems harbor a diverse array of species, including some of the most threatened (Allan and Flecker 1993). Those in the United States have been altered by humans in ways that often are detrimental to their native inhabitants. One consequence of this is that the native molluscan fauna in those systems has declined. We examine here ecological and life history characteristics of two groups of molluscs, prosobranch snails in the family Hydrobiidae and riverine bivalves in the family Unionidae, that have suffered declines due to human activities or appear to be threatened with declines in the future. Their distribution and life history characteristics render them vulnerable to human alteration of their habitats.

[1] *Research Zoologist and Director, New Mexico Natural Heritage Program, University of New Mexico, Albuquerque, New Mexico USA.*

[2] *Aquatic Ecologist, Oklahoma Natural Heritage Inventory, Oklahoma Biological Survey, University of Oklahoma, Norman, Oklahoma USA.*

HYDROBIIDAE

The aquatic snail family Hydrobiidae is species rich and ranges worldwide. Many of the North American species occur as narrow endemics in one or a few small spring systems as living "fossils" that flourished during the Pleistocene (Deixis 1992, Taylor 1987). The systematic relationships of most North American species have only recently been addressed (Hershler 1984, 1985, 1989; Hershler and Landye 1988; Hershler and Longley 1986; Hershler and Sada 1987; Hershler and Thompson 1987; Taylor 1987; Thompson 1968, 1969), and many species remain undiscovered and undescribed (T. Frest, personal communication, R. Hershler, personal communication). Currently, 5 species have been listed as endangered (Federal Register 1991a, 1992), 10 are considered to merit listing as endangered or threatened, and 84 are under review for listing (Federal Register 1991b) (fig. 1).

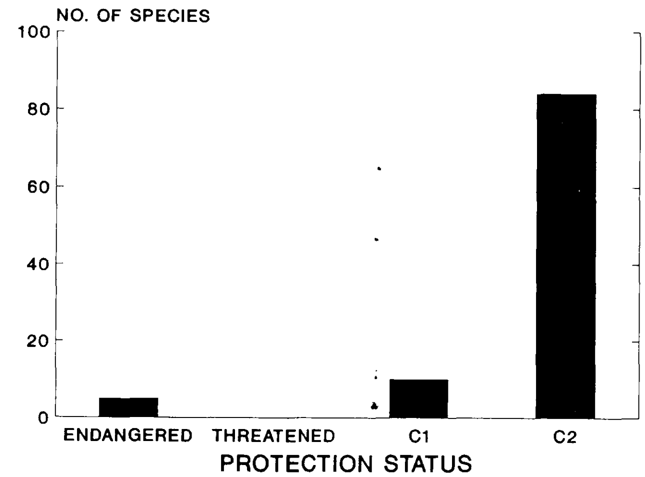

Figure 1.—Federal status toward listing of rare or declining snails of the family Hydrobiidae in the United States. Histogram shows number of species listed as endangered or threatened, number of candidate 1 species (species that merit listing) and number of candidate 2 species (species requiring further study to determine status).

Freshwater hydrobiids are indicators of artesian spring ecosystems with permanent, flowing, highly oxygenated waters (Ponder et al 1989). The waters may be highly mineralized, but must be relatively unpolluted. When hydrobiids occupy a significant portion of a spring system, it is an indication that the system is functioning and intact.

Life History and Ecological Characteristics

Hydrobiids are gill breathing and thus intolerant of drying or anaerobic conditions. Reproduction occurs annually or more often depending on water temperature (Deixis 1992, Hershler 1984, Mladenka 1992, Taylor 1987), and survivorship is estimated to be approximately one year (Mladenka 1992, T. Frest personal communication). They are found in flowing waters, often in thermal springs. The ecology of these snails in North America has received little study until recently (eg., Deixis 1992, Hershler 1984, Mladenka 1992, Reiter 1992). Here we examine ecological data for 59 species in the subfamilies Hydrobiinae and Littoridininae that have been reported as rare or threatened, or which occur in a narrow range in springs and their associated outflows. The sources of information consulted for each species are given in Appendix 1.

Of 59 species, most occur at only a single site and most of the remaining occur at only two or three sites (fig. 2). Occurrences represent single springs with no surface connection to other inhabited springs or parts of spring systems separated by more than 500 m of uninhabited waters. Because studies have not been conducted on gene flow among occurrences, it is not known whether an occurrence is the equivalent of a population.

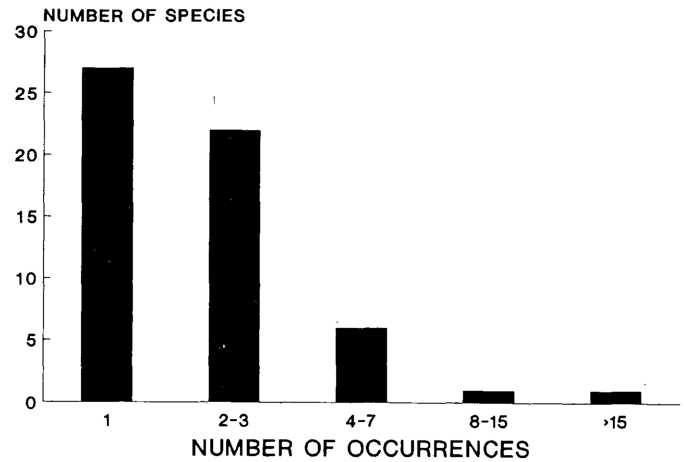

Figure 2.—Number of known occurrences per species of hydrobiid snails that are rare or threatened or have a narrow range of distribution.

Maximum occupied range was estimated in miles for 58 species as the greatest linear distance between two occupied points. Of those, 43% are known to occupy a range less than 0.1 mile, and less than 9% have a range greater than 10 linear miles (fig. 3).

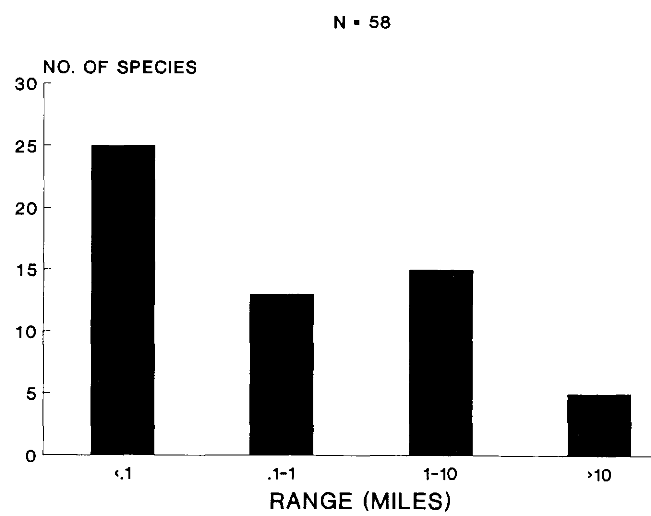

Figure 3.—Maximum occupied range per species (linear miles) of hydrobiids in the subfamilies Hydrobiinae and Littoridininae that are rare, threatened or have a narrow range of distribution.

Substrates occupied by each of 50 species were grouped into seven substrate types. Species in the Littoridininae were most often reported on vegetation, including algal mats and on soft substrates, such as mud and flocculent, but they were reported also on fine substrates such as silt and sand and on tufa (fig. 4). Species in the Hydrobiinae were reported from the same substrates as Littoridininae and also from wood, from stones, including pebbles and cobble, and from boulders and bedrock. It is not clear whether substrate associations reflect particular substrate preferences or hydrologic regimes of the occupied springs and spring runs, which in turn influence substrate availability. Mladenka (1992) showed experimentally that *Pyrgulopsis bruneauensis* (subfamily Hydrobiinae) preferred gravel and sand to silt.

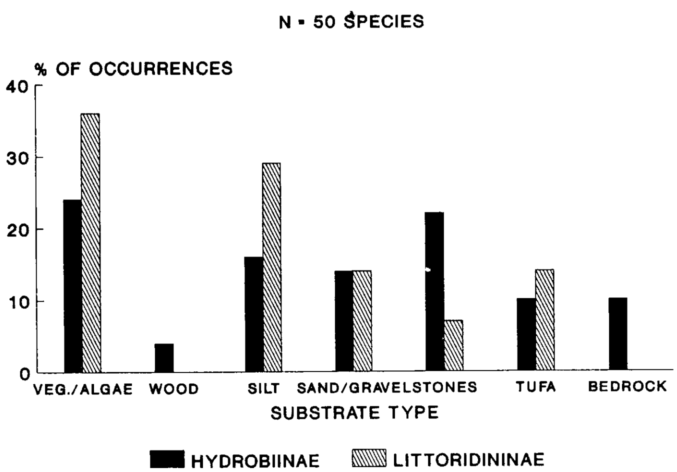

Figure 4.—Reported substrates at occurrences of hydrobiid snails in the subfamilies Hydrobiinae (N = 50 occurrences) and Littoridininae (N = 27 occurrences).

The extreme endemism of the species surveyed, as measured by the number of occurrences and occupied range, suggests that they may be extremely vulnerable to human disturbance. Threats to viability were assessed or identified for 53 species (fig. 5). When more than one threat was identified for a species, the two most prominent threats were tabulated. Decrease in water quantity, due to aquifer depletion or surface water diversion, was identified as a threat for 33 species, with many of those species threatened by both aquifer depletion and surface water diversion. Declines in water quality, due to habitat destruction (from impoundment, dredging or cattle trampling), or pollution (nutrient or chemical), was identified as a threat for 21 species. Recreation, such as swimming or hot spring bathing, was identified as a threat for 10 species. A study by Reiter (1992) suggests that recreation may not be as severe a threat as a change in water quantity or quality. He found that swimmers at a spring in Florida displaced *Aphaostracon monas* from a small area favored by both swimmers and snails, but the snails repopulated the area following the swimming season. For 2 species, no threats were identified in threat assessment procedures.

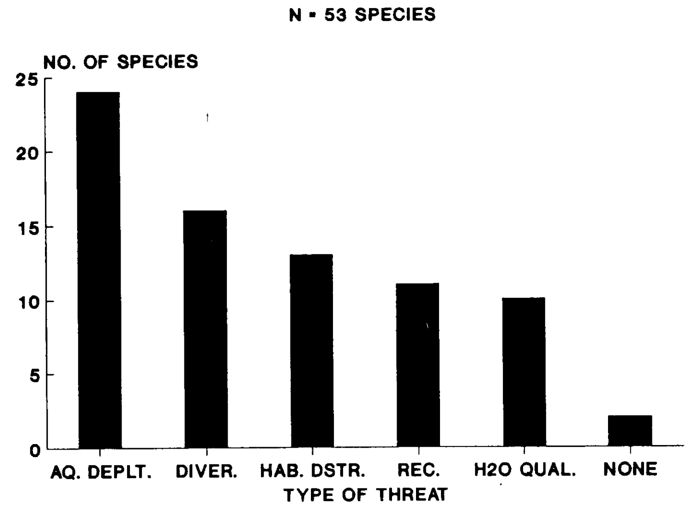

Figure 5.—Reported threats to snails in the family Hydrobiidae. AQ DEPLT = aquifer depletion, DIVER = water diversion, HAB DSTR = habitat destruction, REC = recreation, H2O QUAL = water quality, NONE = no threats found.

Ecosystem Sustainability

Species on public land and on private land designated for conservation offer some degree of long-term protection of ecosystems (Crumpacker et al. 1988). The number of occurrences for 59 hydrobiids was tallied by land ownership (fig. 6), multiple owners of any single occurrence were each counted as an owner. The greatest number of occurrences were on federal lands managed by the Bureau of Land Management (BLM) with private owners having the second greatest number. However, most of the occurrences on BLM lands were attributed to over 100 occurrences of *Pyrgulopsis bruneauensis* in springs along less than 10 miles of a water course (Mladenka 1992), a concentration of occurrences that has not been reported for other North American hydrobiids. If these are clustered as a single occurrence, 85 of the reported occurrences, or 65%, are on public lands or private conservation lands, 44 (33%) are on private lands other than those with a conservation interest and 3 (2%) are on tribal lands. Springs in western states are frequently in private ownership, often as inholdings or adjacent to large tracts of public land, while in Florida many are in the State Park system (Florida Natural Areas Inventory 1992).

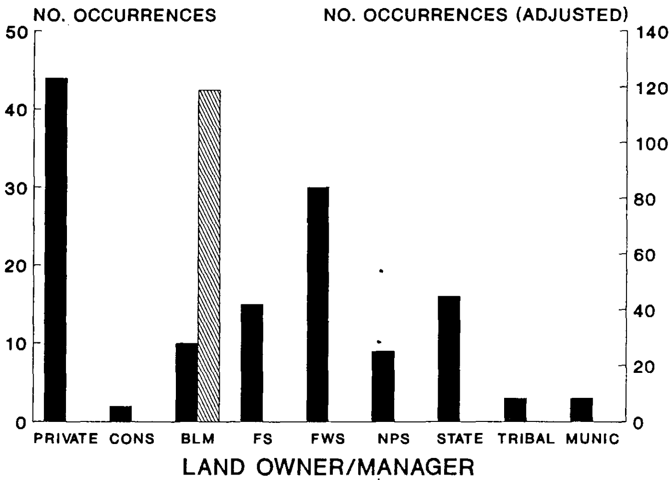

Figure 6.—Land owner or management agency of sites where hydrobiid snails in this study occur. When more than one owner was reported for a species occurrence, each owner was counted. The hatched bar shows the number of occurrences on BLM land without adjustment for the close clustering of over 100 occurrences of a single species. PRIVATE = private land with no formal protection status, CONS = private land with protection status, BLM = Bureau of Land Management; FS = USDA Forest Service, FWS = Fish & Wildlife Service, NPS = National Park Service, MUNIC = municipal ownership or control.

Recommendations for Ecosystem Sustainability

Most freshwater hydrobiids that have been reported as rare or threatened, or which occupy a narrow range, occur in one or a few artesian springs and their associated outflows (figs. 2 and 3). The aquifer source and hydrology of most of the spring systems is not well understood and because of this, hydrobiid ecosystems tend to be defined in reference to the surface waters of the host springs and outflows. When several springs are in close proximity to one another and have one or more hydrobiid species in common, they tend to be treated as a single system for management purposes (Deixis 1992; Federal Register 1991, 1992; Mladenka 1992). Hydrobiid-occupied springs are spatially small ecosystems, which is an advantage for management toward sustainability.

However, conservation and management planning needs to begin at a level higher than single spring ecosystems. For instance, a few spring systems, such as the Ash Meadows system in Nevada (Hershler and Sada 1987) and the Cuatro Cienegas system in Coahuila, Mexico (Hershler 1984, 1985) are quite large with several endemic species in various subsets of springs within the large system. In such cases, management needs to begin with the entire spring system. Artesian springs, especially those in arid environments, are analogous to islands in a sea of dry land that is inhospitable to aquatic species (Ponder et al. 1989). Striking regional species radiations have been demonstrated for both fishes (Soltz and Naiman 1978) and hydrobiids (Ponder et al. 1989, Thompson 1968). This argues for management perspectives that are at regional or large ecosystem levels rather than at the level of single isolated springs.

In many instances, springs are components of larger riverine ecosystems, though hydrologically distinct from them. Two examples of this are the Gila River ecosystem in southwestern New Mexico, which is a riverine ecosystem with eight known spring ecosystems occupied by hydrobiids (Mehlhop 1992 and unpublished data, Taylor 1987), and the middle Snake River with numerous associated springs (Deixis 1992, Federal Register 1992). In those situations, spring management must be a special target of management plans for larger ecosystems.

Most spring ecosystems examined in this survey are best sustained through threat analysis and control. Systems that are highly degraded with marginal hydrobiid populations probably cannot be restored without large financial expenditure and may not be worthy of investment if other, more naturally functioning spring ecosystems can be protected. Systems such as Torreon Spring in New Mexico, which has been impounded to an extent that the hydrobiid *Pyrgulopsis neomexicana* occupies less than 1 m^2 of its former range, is an example of an ecosystem that is no longer functional in its natural state (personal observation). The following recommendations for sustaining spring ecosystems for hydrobiids use a threat assessment and control approach.

1) Identify all springs in the landscape with hydrobiid snails and prioritize them for conservation.
2) Monitor and maintain water quantity in priority spring ecosystems.
3) Monitor and maintain water quality in priority spring ecosystems.
4) Identify and assess the need to abate other threats to ecosystem sustainability.
5) Quantitatively monitor occupied hydrobiid habitats within the targeted springs. In spring ecosystems with co-occurring hydrobiids, monitor relative numbers.

Monitoring will be the most time consuming action in sustaining many spring ecosystems. In most instances, it need not be elaborate, but it must be repeatable and occur at a frequency that will indicate decline in the parameters being monitored.

Hydrobiids are minute and easily overlooked by an untrained observer. To avoid investing in spring ecosystem management in lower priority spring systems, it is important to survey all springs and seeps in a large landscape (e.g., a National Forest and adjacent lands with similar landscape features). Primary threats to hydrobiid-occupied springs should then be identified and management actions prioritized based on assessments of species rarity, population size, degree of threat and amenability of threats to control measures.

Surface water diversion is readily detected and easily monitored. However, protection of surface waters alone is insufficient for many of the spring ecosystems. There are a large number of species for which ground water depletion has been identified as a major threat (fig. 5). Monitoring and protection of ground water flows for those systems is probably the single most important management need. This requires assessing the uses and regulation of the spring aquifer, for which depth and size are most often unknown. A long term monitoring program that roughly estimates water quantity at a spring may be an inexpensive, but adequate means of detecting ground water depletion.

For spring ecosystems that are a high priority for conservation, water quality should be measured initially to obtain baseline water quality data. The subsequent frequency of monitoring will vary with degree of threat. Results of this survey suggest that recreation is a threat to spring ecosystems only if spring outflows are altered substantially or if chemicals are added to the system. For instance, a hot spring in New Mexico is used for recreational bathing upstream from one of only two populations of a hydrobiid, and the population is maintained by flows of 0.3 cm and less over the snail substrate. While the probability of diversion or chemical pollution appears low, the consequences of such threats could be great.

Monitoring the snails themselves provides both a measure of the impacts of identified threats and a means of detecting unanticipated threats. Hydrobiid snail populations are difficult and costly to estimate, and methods used at one spring system may not be applicable to others (personal observation, T. Frest, personal communication). However, population stability can be estimated by monitoring the surface area occupied or the boundary of occupation. This needs to be done at approximately the same time of year due to seasonal population fluctuations generally associated with birth and death events. When hydrobiids co-occur in a spring, they usually cannot be distinguished with certainty without some disturbance to the population. However, some minimal monitoring is desirable to confirm that species proportions remain relatively stable.

UNIONIDAE

The unionid mussel fauna of North American freshwater is the most diverse in the world but is highly threatened. There have been major declines of mussel populations and species diversity in North American over the last century. Of the 283 species of native North American mussels, 131 species, or approximately 40%, are threatened with extinction: 17 species are presumed extinct, 44 species are actually listed as threatened or endangered, and 70 species are federal candidates for listing (Neves 1993, Master 1993) (fig. 7). Furthermore, all federally listed unionids are declining. There are no listed species with populations that are being maintained or increasing (Neves 1993).

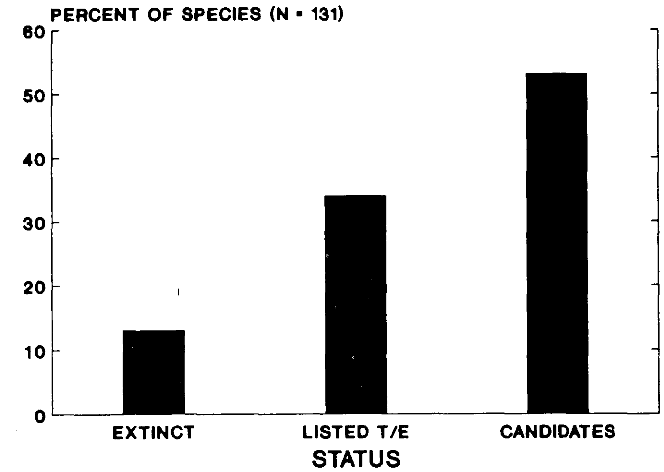

Figure 7.—Status of unionid mussels in the United States. N = 131. LISTED T/E = listed by the federal government as threatened or endangered, CANDIDATES = candidates for federal listing.

Unionid Characteristics

Freshwater mussels possess a suite of traits that make them highly vulnerable to habitat disturbance (table 1). Mussels have a complicated life history. The larval stage of freshwater mussels (glochidia) are temporary, obligate parasites on the gills or fins of fish. Many mussel glochidia can survive only on a narrow range of fish species hosts (Way 1988). Contact with an appropriate fish host and the location where young mussels are shed from the host is largely due to chance and only juveniles that reach a favorable habitat survive (Neves & Widlak 1987). Because only larvae can move between patches and juvenile survival is low, the potential rates of colonization are low. Reproductive maturity is not reached until age 6, most species live greater than 10 years, and some species live as long as 90 years (Haskin 1954, Imlay 1982, McMahon 1991). Once mature, adult mussels exhibit high survivorship (>80%) (McMahon 1991). However, adult mussels are sedentary; movements are

Table 1. — Life history characteristics of the Unionidae. Modified from McMahon (1991).

Life span	< 6 > - 100 yr
Age at maturity	6 - 12 yr
Strategy	Iteroparous
Fecundity	200,000-17,000,000
Reprod. efforts/year	1
Juvenile size	50 - 400 um
Rel. juvenile survivorship	Very low
Rel. adult survivorship	High
Larval habitat	Obligate parasite on fish

seasonal and on a scale of a few to an estimated maximum of 100 meters (Green et al. 1985). Therefore, unlike many stream organisms such as fish and aquatic insects (Townsend 1989), adult mussels have no refugia from disturbance events in streams. In addition, their filter-feeding habits make them especially vulnerable to sedimentation and chemical pollution events.

Threats and Causes of Decline

Species associations, species richness, metapopulation structure, and densities and population size structure of individual species are all potentially impacted by forest management practices. In addition, any effects on fish communities may ultimately affect mussels as well. Watters (1992) recently found high correlation between fish distribution and diversity and mussel distribution and diversity.

One major cause of mussel declines has been the fragmentation of river drainages through impoundments, channelization and other activities, such as timber-harvesting, which alter flow and sedimentation patterns. Declines in mussel species for various river drainages and the disturbance factor associated with these declines are shown in Table 2.

Timber harvesting operations can have significant effects on both stream water quantity and quality. The influence of catchment vegetation on stream discharge is dependent on a large number of variables, many of which are site-specific. However, in general, removal of forest vegetation increases stream runoff (Campbell and Doeg 1989). Increased flows have the potential to alter the distribution of sediment through scour, flushing, and deposition of newly eroded materials from the banks. Increased flows also have the potential to activate the bed. Bedload movement will wreak havoc on the survival of many mussels, particularly juveniles (Young and Williams 1983). Erosion caused by increased flows at one location results in deposition of this material further downstream. This "zone of aggradation" results in an increased width/depth ratio of that portion of the channel. As width/depth ratios increase the potential for bedload transport also increases. Thus, increased flows cause habitat loss through both sediment deposition and increased bed mobility. In the long term, higher base flow levels and shorter periods between peak flood periods will decrease habitat complexity by preventing the formation of islands, establishment of macrophyte beds, etc. (Frissell 1986). Stabilized sediments, sand bars, and low flow areas, are all preferred unionid habitats (Hartfield and Ebert 1986, Payne and Miller 1989, Stern 1983, Way et al. 1990). It is around these "complex" areas that most mussel beds, and indeed the highest diversity of stream fauna, are found.

Road-building activities and low water crossings associated with logging can lead to the development of "headcuts", or migrating knickpoints in the channel remote from areas of actual modification. Headcuts result in severe bank erosion, channel widening, and depth reduction and can have devastating effects on the mollusc fauna (Hart 1993).

Stream organisms, including mussels, have evolved in rivers that experience seasonal low-flow and high-flow periods (Meador and Matthews 1992). Fluctuating flows, especially if there will be lower flows for long periods of time, will result in the stranding of many mussels. Unlike fish species which can move rapidly in and out of microhabitats with changes in water levels, mussels move very slowly and are unable to respond to sudden drawdowns. Even if stranding doesn't actually kill a mussel, desiccation and thermal extremes will cause physiological stress and may reduce reproductive potential (McMahon 1991).

Fluctuating flows also mean that transport of particulates will vary. Depending on the flow schedule and the materials normally transported in the water column, there is the potential for loss of organics which are the food base for mussels.

Flow alteration not only has the potential to profoundly affect the stream fauna, but riparian fauna as well. Flood waters that normally recharge soils and aquifers may be rapidly exported

Table 2. — Reported loss of unionid mussel species from rivers and factors contributing to the losses.

Drainage	% Species Lost	Major Factor in Decline	Source
Upper Tennessee River	36%	Impoundments, sedimentation	Starnes and Bogan (1988)
Middle and Lower Tennessee R. sedimentation	13%	Impoundments, channelization,	Starnes and Bogan (1988)
Tombigbee River at Epes, AL	68%	Impoundment	Williams et al. (1992)
Stones River, TN	40%	Impoundment	Schmidt et al. (1989)
Upper Stones River, TN	25%	Gravel dredging, water quality	Schmidt et al. (1989)
Sugar Creek, IN	20%		Harmon (1992)
Illinois River, IL	51%	Impoundments, channelization, sedimentation	Starret (1971)
Kankakee River, IL	25%	Siltation	Suloway (1981)
Kaskaskia River, IL	38%	Siltation (80% reduction in numbers of individuals)	Suloway et al. (1981)
Vermillion River, IL	40%		Cummings (1991)
Embarras River, IL	39%		Cummings (1991)
Little Wabash River, IL	24%		Cummings (1991)

downriver. Lowered water tables may cause shrinkage of the riparian corridor and shifts in terrestrial species composition (Allan and Flecker 1993, Smith et al. 1991).

Mussels are most successful where water velocities are low enough to allow sediment stability but high enough to prevent excessive siltation (Salmon and Green 1983, Way et al. 1990). Thus, well-oxygenated, coarse-sand and sand-gravel beds comprise optimal habitat (McMahon 1991). Sediment deposition not only removes or moves habitat, but also clogs mussel siphons (i.e. smothers them) and interferes with feeding and reproduction (Dennis 1984, Aldridge et al. 1987). In addition, because mussels are sedentary filter-feeders, they are particularly sensitive to changes in water quality (Havlik and Marking 1987).

Demographic Consequences

Because of this dependence on the appropriate substrate and flow conditions, freshwater mussels are already naturally patchily distributed in rivers. Fragmentation acts to increase patchiness and to increase the distance between patches. These effects may have major consequences for the metapopulation (i.e. local or subpopulations connected by infrequent dispersal) structure of mussel species, particularly rare species and those with narrow fish-host requirements (Vaughn 1993). As some subpopulations are eliminated and dispersal distances are increased between other subpopulations, demographic and genetic constraints will diminish the ability of mussels to respond to even natural stochastic events much less human-induced environmental change (Wilcox 1986, Murphy et al. 1990).

Forest Management Strategies

Managing forests to maintain fully functional riverine ecosystems is the best way to protect unionid populations in National Forests. Best land-use practices should strive to maintain an uncut riparian corridor at least as wide as the predicted 100 year channel meander (Boon et al. 1992). Forest managers should seek to minimize the use of biocides and encourage selective logging rather than clear-cutting whenever possible. Disturbances such as low-water crossing which were thought to have temporary effects are now known to have long-term detrimental effects on mussel populations through the formation of migrating headcuts. Managing forests from an ecosystem perspective must include long-term monitoring of unionid populations.

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Appendix 1. Species of snails in the family Hydrobiinae included in this study and the sources of information used.

Apachecoccus arizonae	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973, Taylor 1987
Aphaostracon asthenes	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Aphaostracon monas	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Aphaostracon pycnus	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Aphaostracon theiocrenetus	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Aphaostracon xynoelictus	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia helicogyra	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia mica	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia monroensis	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia parva	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia ponderosa	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia vanhyningi	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Cincinnatia wekiwae	FL	Florida Natural Areas Inventory 1993, Thompson 1984
Pyrgulopsis aardahli	CA	California Natural Heritage Division 1993, Hershler 1989
Pyrgulopsis bacchus	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Pyrgulopsis bruneauensis	ID	Idaho Conservation Data Center 1993, Mladanka 1992
Pyrgulopsis chupaderae	NM	National Museum Natural History collections, Mehlhop (personal observation), Taylor 1987
Pyrgulopsis conicus	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Pyrgulopsis crystalis	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis davisi	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Pyrgulopsis deserta	AZ, UT	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Utah Natural Heritage Program 1993
Pyrgulopsis erythopoma	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis fairbanksensis	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis gilae	NM	Mehlhop (1992, personal observation), Taylor 1987
Pyrgulopsis glandulosus	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Pyrgulopsis isolatus	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis merriami	NV	Nevada Natural Heritage Program 1993
Pyrgulopsis metcalfi	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Pyrgulopsis montezumensis	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973
Pyrgulopsis morrisoni	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Pyrgulopsis nanus	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis neomexicanus	NM	Federal Register 1991a, Taylor 1987
Pyrgulopsis nevadensis	NV	Nevada Natural Heritage Program 1993
Pyrgulopsis n. sp.	NM	Mehlhop (1992, personal observation)
Pyrgulopsis owenensis	CA	California Natural Heritage Division 1993, Hershler 1989
Pyrgulopsis pecosensis	NM	Mehlhop (1992), Landye 1973, Taylor 1987
Pyrgulopsis pisteri	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Pyrgulopsis perturbata	CA	California Natural Heritage Division 1993, Hershler 1989
Pyrgulopsis roswellensis	NM	Mehlhop (1992), Landye 1973, Taylor 1987
Pyrgulopsis simplex	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973
Pyrgulopsis solus	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Pyrgulopsis thermalis	NM	Mehlhop (1992), Taylor 1987
Pyrgulopsis thompsoni	AZ, MX	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973
Pyrgulopsis trivialis	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973
Tryonia adamantina	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Tryonia alamosae	NM	Landye 1973; Mehlhop, P. personal observation, New Mexico Natural Heritage Program 1993, Taylor 1987
Tryonia angulata	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Tryonia brunei	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Tryonia cheatumi	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Tryonia elata	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Tryonia ericae	NV	Hershler and Sada 1987, Nevada Natural Heritage Program 1993
Tryonia gilae	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Landye 1973, Taylor 1987
Tryonia kosteri	NM	Landye 1973, Mehlhop, P. 1992, New Mexico Natural Heritage Program 1993, Taylor 1987
Tryonia margae	CA	Hershler 1989
Tryonia quitobaquitae	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988
Tryonia rowlandsi	CA	Hershler 1989
Tryonia salina	CA	Hershler 1989
Tryonia stocktonensis	TX	Taylor 1987, Texas Parks & Wildlife Department 1993
Yaquicoccus bernardinus	AZ	Arizona Heritage Data Management System 1993, Hershler and Landye 1988, Taylor 1987

Hypotheses Concerning Population Decline and Rarity in Insects

Kathryn J. Schaeffer[1] and Stacey L. Kiser[2]

Abstract — Although numerous insect species are considered "rare," many of them have not been listed as threatened or endangered species. There are numerous hypotheses as to the decline of specific insect populations. Among the most common of these hypotheses include mechanisms of habitat destruction, interactions with introduced species, and overkill or overcollecting. The Oregon Silverspot butterfly, *Speyeria zerene hippolyta,* is a federally threatened species under the Endangered Species Act, listed in 1980. Our work, in conjunction with the Nature Conservancy, the U.S. Forest Service, and independent researchers, has documented a decline in population numbers since the early 1960s. The Oregon Silverspot butterfly is found along the west coast, from San Francisco to southern Washington. In the 1960s, there were 15-20 strong populations recorded. Currently, there are seven to eight populations, with four of them containing fewer than 100 individuals. The current hypothesis for the decline in population numbers is from habitat destruction and fragmentation of original habitat due to development of coastal land, recreational use, and change in habitat management of current habitats. The goal is to understand the behaviors associated with habitat needs and, as a result, to implement effective management plans.

INTRODUCTION

The amount of attention that invertebrates receive from conservationists and governmental agencies compared to that of vertebrates is at least an order of magnitude less, if not more. This is despite the fact that there are far greater number of species of invertebrates than vertebrates. The majority of insect species (the largest class of invertebrates) are not favorably viewed by the public eye. The consensus on insects is that they are "pests" to humans. In direct competition with humans for certain food crops, billions of dollars are spent annually to eradicate local populations of insects. What is forgotten are the benefits humans gain such as pollination and decomposition from insects. However, as New stated in 1991, "The widespread conceptual barriers to conserving lower animals is gradually being overcome, and many people now admit their importance in natural ecosystems and in maintaining our natural world."

There is one group of insects that has historically escaped this negative image—the butterflies. Butterflies are spectacular insects, often depicting the epitome of nature, wonder, beauty, and tranquility. Because of the popularity of the Lepidoptera, it is no wonder that butterflies have received more attention from conservationists than any other insect taxa.

This paper concentrates on the Oregon Silverspot butterfly, *Speyeria zerene hippolyta* (Lepidoptera: Nymphalidae), which has been on the federal threatened and endangered species list since 1980. This butterfly has seen population declines since the 1960s and is now to the point where extinction of the species is possible within the next decade if appropriate management strategies are not implemented. We discuss the current hypotheses concerning population decline in insects in general and address the probable reasons for the decline of the Oregon Silverspot butterfly.

[1] *Department of Biological Sciences, Northern Arizona University, Flagstaff, AZ 86001 USA.*

[2] *Department of Biology, University of Oregon, Eugene, OR 97403 USA.*

LIFE HISTORY OF THE SILVERSPOT

The Oregon Silverspot butterfly is a medium-sized, dark, orange-brown fritillary with black veins and spots on the margins of the upper surface of its wings and bright, metallic silver spots on the side of the hind wings. The larvae are dark, with long spines and have two tan lines running laterally along the dorsal surface. Each line has a row of black patches running parallel to it on the outside (personal observation). The bases of the spines are a straw color which camouflage the larvae in the thatch. The larvae take shelter in dead vegetation when not feeding on *Viola adunca,* the common blue violet, their obligate larval host (Hammond and McCorkle 1984).

The adult female butterfly lays single eggs near the blue violet plant. Females oviposit 200 or more eggs between mid August and mid September. The eggs hatch within two to three weeks, although the time is variable depending on the microhabitat (personal observations). The larvae overwinter as first instars and emerge in the spring to feed. In July, larvae commence feeding and pupate. Adults emerge about two weeks after the beginning of pupation. Males emerge several days before the females, in order to attain proper thermal conditions for successful nectaring and quick maturation and to search and wait for emerging females (McCorkle 1980). Mating takes place within hours of female emergence, but can last through late August, with ovipositioning occurring through September. Eclosion of the adults occurs from early July until early September. The long emergence span appears to be an adaptation to an unpredictable environment (McCorkle 1980).

Currently, the Silverspot is found at seven to eight sites along the Pacific Coast (Fig. 1). The four strongest populations are 1) Bray Point, located eight miles south of Yachats, Oregon 2) Rock Creek, located ten miles south of Yachats, Oregon 3) Cascade Head, located six miles north of Lincoln City, Oregon and 4) Mount Hebo, located 12 miles inland in the Coast Range, south of Tillamook, Oregon. Smaller populations (approximately less than 100 individuals at each site) include 1) Clatsop Plains, located 20 miles south of Astoria, Oregon 2) Camp Rilea, located just north of Clatsop Plains (may be considered one metapopulation) 3) Long Beach, southern Washington and 4) Del Norte, northern California.

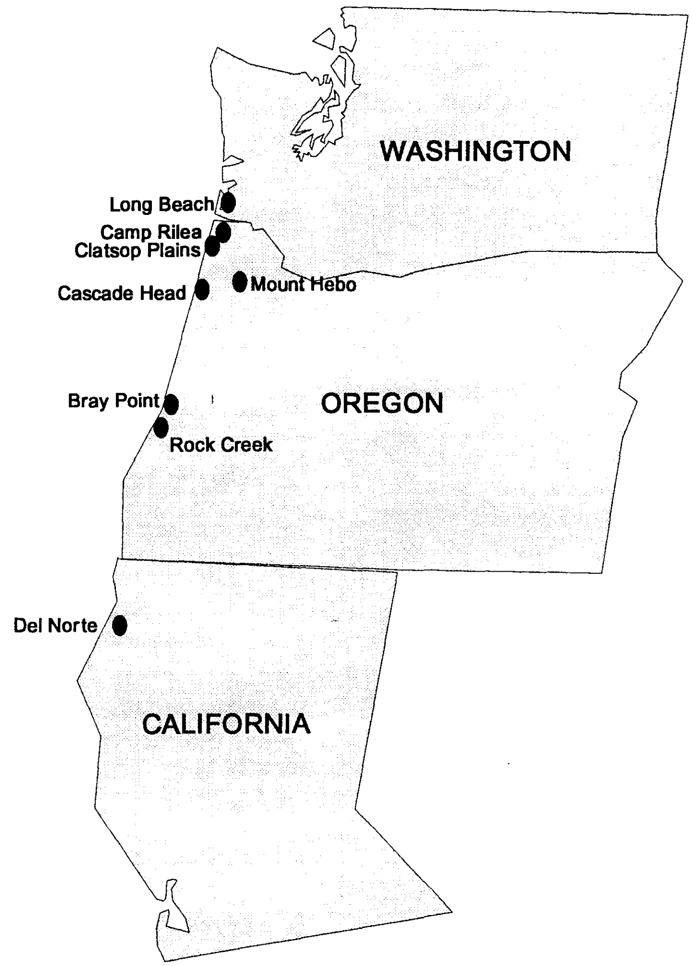

Figure 1. — Current locations of the Oregon Silverspot butterfly along the Pacific Coast.

REASONS FOR DECLINES

Overcollecting/Overkilling

Because of the threatened status of the Oregon Silverspot butterfly, overcollecting is not a problem since this activity is prohibited by federal law. But for other species of invertebrates, overcollection can pose a serious problem, especially if the species is already considered rare. While recreational collecting has been documented as being a minor factor in the decline of arthropods (Pyle et al. 1981), commercial exploitation may threaten populations of economically important species, especially species that are already declining due to other reasons, such as habitat destruction. Examples include the tropical birdwing butterfly, which are collected and sold for their beauty (Collins and Morris 1985, New 1991) and female tarantulas due to their unusual body shape and their increased popularity as pets.

For the Silverspot butterfly, other mortality factors which may affect total population numbers do not appear to have a significant impact (Stine 1982). Predators and parasites are known to attack the larvae (McCorkle 1980), but do not appear to pose a significant problem to the Silverspot population as a whole. Birds have been seen eating adult butterflies, and several adults can be spotted with "beak marks" out of their wings (personal observations).

Other factors, such as road kill and insecticides from nearby lands appear to have an insignificant impact on the population, although no statistics are available at this time. Currently, the Oregon Department of Transportation is conducting field surveys on the amount of Silverspot butterflies killed by automobiles at one site, Rock Creek. This site is divided by Interstate Highway 101 and ovipositing females must cross several times a day. This is discussed further under Habitat Destruction/Fragmentation. This study should provide statistics on the importance of roadkill as a factor affecting this population.

Introduced Species

Introduced Plant Species

The introduction of non-native species, both plant and animal, has been documented as a primary cause of invertebrate extinctions, especially on island communities such as Hawaii. There are over 2,000 species of non-native invertebrates that have successfully established on the islands of Hawaii (Howarth and Medeiros 1989).

For the Oregon Silverspot butterfly, the effect of introduced species on the decline of populations has not been seriously studied. Several species of introduced grasses are evident in the meadows which serve as Silverspot habitat, but their importance on the quality of the habitat is unknown. We can speculate that introduced grass species may have a large impact on the Silverspot (Schaeffer 1992). While it is known that the height of the vegetation in the meadows negatively affects the ability of females to oviposit on the larval host plant, *Viola adunca* (Schaeffer 1992), the types of vegetation present may also be important. Introduced species, such as *Anthoxanthum* sp., can dramatically change the overall vegetation height in the habitat (personal observations).

The impact of other introduced plant species becomes increasingly relevant if the habitat is left unmanaged. The non-native grass species could outcompete the blue violets, making for reduced violet density in the meadows. The elimination of the blue violet, being the only larval host plant, would lead to the local extinction of the butterfly. Sufficient data to support this hypothesis is not available at this time.

Management of the habitat for the introduced plant species (as well as overgrowth of native plant species) can be accomplished by a combination of mowing, slashing, grazing by animals, and burning to control the overgrowth of the meadows (Hammond 1980). A combination of these methods would provide the optimal solution for controlling the vegetation structure, since burning would not effectively control for bracken fern, *Pteridium aquilinum,* but would be the most rapid method for controlling large areas of habitat. Management needs to be site specific as well. For example, at Clatsop Plains, mowing of Scotch broom and reseeding of nectar species is of the utmost importance. However, at Cascade Head, bracken fern growth needs to be controlled, with recommendations including hand pulling and rotational burning (Kiser 1993).

Hammond (1993) reports that violet growth and butterfly numbers greatly increase in response to bracken fern removal. He performed an experiment where one quarter acre plots, adjacent to one another, were used. One plot had bracken fern removed in a two year treatment, the other was a control. The experimental plot produced 900 blooming violets compared to 21 violets in the control plot (Hammond 1987). Within a few years after treatment, there were ten times more butterflies utilizing the plots with bracken fern removed (Hammond 1993).

Kiser (1993) also reports that butterflies respond favorably to the removal of bracken fern. The females are better able to locate violets in areas of low vegetation height (Schaeffer 1992). Bracken fern inhibits the growth of violets by shading them, allowing more aggressive species to outcompete the violets, eliminating them from the meadows (Kiser 1993).

Hammond (1993) cites that the removal of Salal (*Gaultheria shallon*) and spruce trees (*Picea sitchensis*) made the habitat much more suitable for the Silverspot butterfly by allowing dormant plants and seeds of violets space to bloom. He reports an increase of butterflies utilizing the managed area within three years, giving the meadow sufficient time for succession to occur and give the habitat a natural appearance.

Introduced Animal Species

The effect of introduced animal species on the Silverspot butterfly has not been studied. In other systems, the introduction of non-native animals into a habitat can have adverse effects on the species in question. These effects may include competition for food, shelter, or territory space. Often, introduced animal species, especially vertebrates, can cause fragmentation of an invertebrate's natural habitat. The fragmentation can lead to decreased population size in any one given area, thereby restricting dispersal between the fragmented habitats and breaking up the gene pool. Loss of genetic variation can lead to local declines and possibly extinctions.

The effect of animal species on the Silverspot butterfly needs to be studied in order to determine proper management strategies concerning grazing of habitat. While grazing would positively affect the Silverspot by keeping vegetation heights low, it may negatively affect the survival of larvae by trampling and removal of nectar species. A study to determine the relationship between the positive and negative effects is recommended.

Habitat Destruction/Fragmentation

The degree of extinctions or, at best, population declines of invertebrates can be seen to closely follow patterns of human population growth (Opler 1976, New 1991). The main reason is that people are reducing natural habitats to accommodate human lifestyles, including more housing, more recreation areas, and more farmland (Arnold 1987, New 1991). The reduction and/or fragmentation of these natural habitats is the biggest threat to invertebrate diversity.

Loss of Land

The majority of work on the effects of loss of natural habitat has been done on butterflies in the United Kingdom. C. D. Thomas (1985a) documented that *Plebejus argus*, the silver studded blue, is one of the most rapidly declining species in Northern Britain, already showing a two-thirds reduction in population numbers since 1945. The primary reason for the rapid decline is from habitat loss to accommodate "agricultural 'improvement'" and forestry and urban development (C. D. Thomas 1985a). He further acknowledges the fact that, not only habitat loss but decline of traditional management of the existing habitat is responsible for the decline of the silver studded blue (C. D. Thomas 1985b).

J. A. Thomas (1984) estimated that of the 55 species of resident butterflies in the United Kingdom, 44 of them had declined in population numbers and in number of successful colonies within the past 25 years. Most of these 44 species have declined from habitat loss and lack of management.

The decline of butterfly species as a result of habitat loss is seen in the United States as well. The Palos Verdes blue butterfly, *Glaucopsyche lygdamus palosverdesensis*, has been endangered since 1980. However, habitat destruction by housing or recreational development continued even after listing. No Palos Verdes blues have been spotted since 1983 and very intensive management, including the creation of new sites and restoration of current sites, is needed or this species will become extinct, if not already extinct (Arnold 1987).

Habitat loss is unquestionably the main reason for the threatened status of the Silverspot butterfly. In the 1960s, there were 15-20 stable, viable populations along the Pacific coast, ranging from San Francisco to southern Washington. As prime coastal land began to be developed to make larger cities and resorts along the beach, the natural meadows were being eliminated. This happened at a phenomenal rate to keep up with the demand for ocean-front real estate. Presently there are only seven to eight populations left, with four of these populations containing fewer than 100 individuals. If left unmanaged, these small populations will likely go extinct by the turn of the century.

With only a few populations remaining, being tens of miles apart, there is little or no movement of individuals from one area to the next annually. Each meadow containing Silverspots is virtually an island, with no migration of animals in or out. Thus, it is critical that the remaining habitat be properly managed if we are to sustain viable populations of butterflies there.

The acquisition of new land, to be converted into suitable Silverspot habitat, needs to be seriously considered. More areas of habitat will strengthen the current population of Silverspot butterflies. Since the ultimate goal is to delist the butterfly, land acquisition is of utmost importance.

The problem then becomes, where does this land come from, who will manage it, and how will it be paid for? These questions need to be addressed before any management plan can be accepted. There are also other considerations to be addressed as well (after Eagles 1984):

1. Ecological considerations. This considers the need to emphasize long-term protection over short-term results. Also, studies need to be conducted that incorporate standard criteria and environmental impact assessments.
2. Legal considerations. There must be a balance between landowner's rights and the protection of the Oregon Silverspot butterfly under the Endangered Species Act of 1973. There also needs to be conformation with all local regulations concerning mowing, burning, etc.
3. Political considerations. This considers the acceptability by the city, county, and state governments and the general public to manage lands. It also considers where the monies come from, in what proportion, and how education of the public will commence.

Unsuitable Habitat Areas

Along with the loss of habitat due to development, there is also loss of habitat due to its unsuitability. Personal observations conducted in 1991 at Clatsop Plains indicate that cattle grazing on available nectar sources for the Silverspot may be responsible for the butterfly's limited presence. Part of this site is privately owned, bought for the sole purpose of supporting cattle. Not only do the cattle help keep the vegetation height low, but the cattle eat the flowering species there as well, such as *Senecio jacobae*, *Achillea millefolium*, *Solidago canadensis*, *Hypochaeris radicata*, and *Cirsium edule*. These species represent the majority of nectar sources available to the Silverspot, without which sustainable populations cannot exist. Therefore, meadows without sufficient nectar sources available (or nearby) are deemed as unsuitable.

The change in current management practices also may make parts of current habitat unsuitable. C.D. Thomas (1985b) has shown that the silver studded blue butterfly is declining from lack of traditional management of the remaining sites. If sites are left without proper management, no vegetation exists in "pioneer" condition and the butterfly may be eliminated (C. D. Thomas 1985a). Therefore, active management is required.

Allowing succession to progress in the meadows may be just as deleterious as loss of habitat for the Silverspot butterfly. The Silverspot requires early seral habitat in order to allow for the blue violet, the larval host, to grow and reproduce (Kiser 1993).

If succession is left unchecked, the blue violet may be outcompeted by other species of plants, making for unsuitable habitat for oviposition.

Another way that habitat becomes unsuitable is if barriers prevent animals from crossing over from one part of the habitat to the other. Barriers can be in the form of housing, recreational facilities, ravines, rivers, or roads. For invertebrates, most barriers are not a problem. However, for the Silverspot butterfly, a road can be a major obstacle to overcome, especially if crossing several times a day.

At the Rock Creek site, the habitat is divided into two main areas. The dividing line is Interstate Highway 101. Traffic is heavy on this highway, especially in the summer months when tourists are travelling up and down the coast of Oregon. While there are no statistics on the amount of loss of Silverspot butterflies that cross the road, females must cross at least several times a day when ovipositing. The area to the west of the highway is where most of the violet habitat is found and, consequently, where most of the oviposition events take place; the area to the east is where most nectaring occurs (personal observations). As the population declines, the effect of roadkill may become important enough to cause the local extinction of this population of butterflies, primarily since the females comprise the majority of crossers.

Genetic Problems Due to Fragmentation

Habitat destruction can lead to extinctions by restricting the genetic pool to a small number of individuals. As population sizes decrease as a result of habitat loss, restricted genetic variability and/or catastrophic events can ultimately cause the extinction of these small populations. The question then becomes, how long can small populations persist and is genetics relevant? Even if we can estimate the amount of time that a population can be self-sustainable given the population parameters, this is still not an accurate indicator of the viability time of the population. There are extenuating circumstances that are often overlooked, such as catastrophes or the interplay between population dynamics and the loss of fitness due to genetic drift (Soulé and Mills 1992).

One of the ultimate long-term goals of conservation genetics is to maintain a "minimum viable population" (Gilpin and Soulé 1986) that allows "enough genetic variation so that future adaptation, successful expansion, or reestablishment in natural populations is possible" (Hedrick and Miller 1992). For the Silverspot butterfly, this may mean the rearing of individuals in the laboratory and the reintroduction of larvae and/or adults from lab populations. The transfer of individuals from one population to another, separated by a considerable distance, would allow for additional genetic variation into any one population. At the very least, heterozygosity of Silverspot populations may be kept in the range suitable for maintaining genetic diversity and keeping inbreeding at a minimum.

Further research should focus on determining what the minimum viable population is for the Silverspot as well as delimiting populations from subpopulations. Most of the sites are far enough away to obstruct migration of individuals from one site to the next. But, at Clatsop Plains, there are two populations close enough that they may be working in a source-sink fashion (Kiser 1993). The understanding of the genetic makeup of each population may facilitate determining the placement of new sites and the transfer of individuals from site to site.

CURRENT POPULATION DECLINE OF THE SILVERSPOT

Population censuses have been done on a regular basis for the Oregon Silverspot butterfly for the past few years. Censuses consist of observational records of the number of butterflies counted when walking along a transect. The transect passes through the habitat, not overlapping areas, and butterflies are counted within 15 meters of the transect line in all directions. Data that we present in this paper will include the 1990 and 1991 censuses for Mount Hebo and Cascade Head, the two strongest populations, both conducted between July and October of each year. These data were collected by Kiser and Schaeffer, along with The Nature Conservancy (data are included in The Nature Conservancy 1991, 1992). These sites have vastly different management plans. Cascade Head is a Nature Conservancy preserve with little active management. Mount Hebo has active management, including mowing, slashing, and burning every year.

In Figure 2, the maximum number of butterflies are plotted for each area. As the figure indicates, there was an increase in the number of butterflies seen at Mount Hebo and a decrease in numbers at Cascade Head.

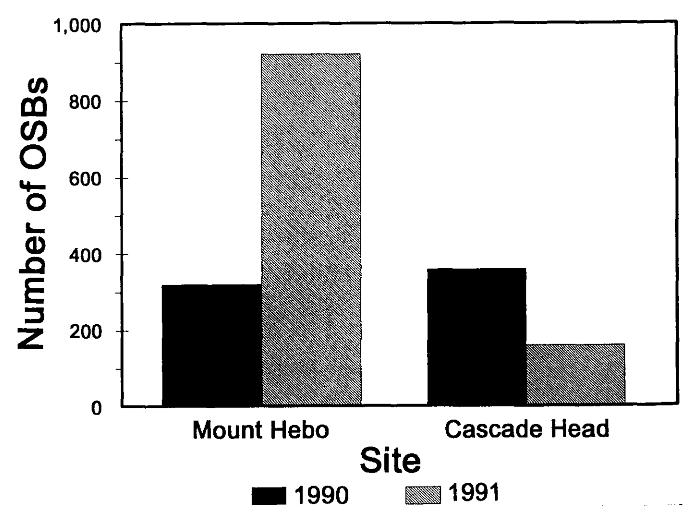

Figure 2. — Oregon Silverspot butterfly (OSB) maximum census counts for the 1990 and 1991 field seasons. Counts are done following a transect line and walking at a rate of 100 meters in 2.5 minutes. Adapted from The Nature Conservancy (1991, 1992).

Figures 3a and 3b represent the complete censuses for the two sites, plotting the number of Silverspot butterflies observed per given day. In Figure 3a, we observed an increase in the number of butterflies seen at Mount Hebo from 1990 to 1991. The peak number of butterflies occurs at around the same time of the year for two consecutive years. In Figure 3b, we observed a decrease in the number of butterflies at Cascade Head from 1990 to 1991.

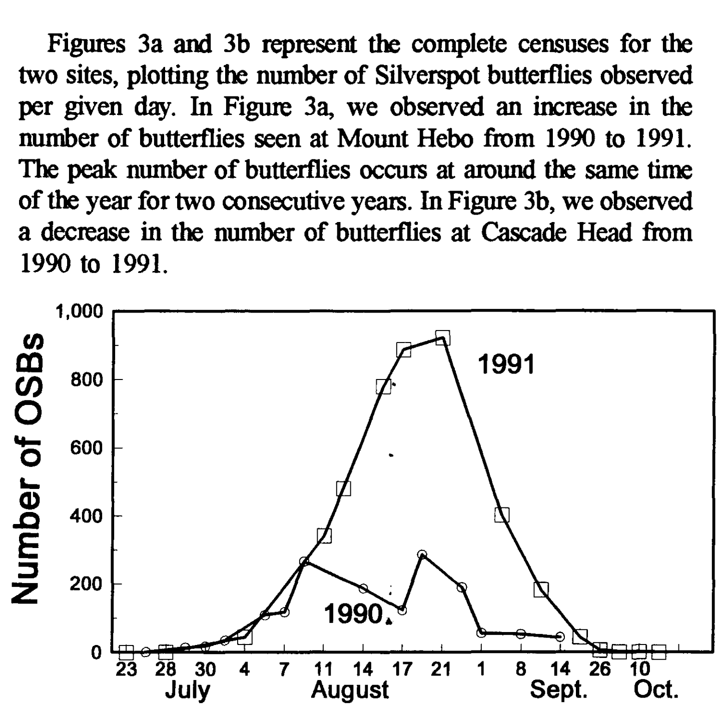

Figure 3a. — Graph of the complete censuses for 1990 and 1991 at Mount Hebo. Each point represents the number of individuals seen on any given day. Counts were done along a transect line. Adapted from The Nature Conservancy (1991, 1992).

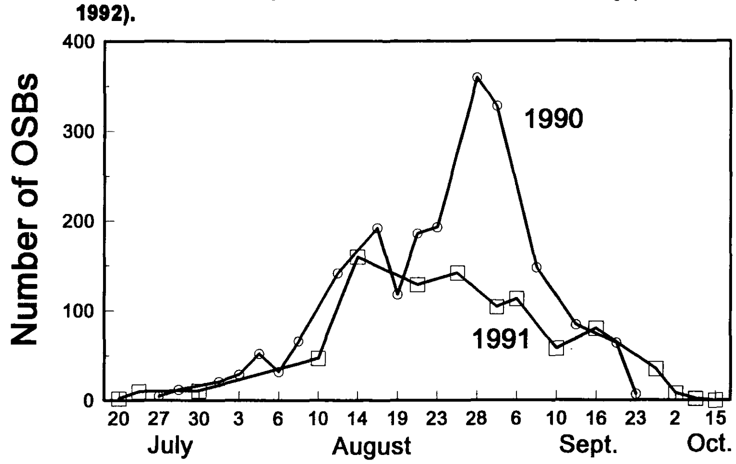

Figure 3b. — Graph of the complete censuses for 1990 and 1991 at Cascade Head. Each point represents the number of individuals seen on any given day. Counts were done along a transect line. Adapted from The Nature Conservancy (1991, 1992).

The discrepancies between sites may be due to the fact that Mount Hebo underwent intensive management in recent years to keep vegetation heights to a minimum and to remove bracken fern. On the other hand, Cascade Head has not had any management implemented in recent years, aside from removal of invading tree species. Cascade Head, being a nature preserve, is protected from grazing, recreational use except on a narrow path, and fires are suppressed as best as possible. Mount Hebo, U. S. Forest Service land, allows visitors to walk in the meadows, implemented mowing recently, and natural burns are not suppressed. If censusing continues in the current manner under similar conditions, we imagine that the current rate of decline of Silverspots at Cascade Head will continue due to habitat decline.

Weather differences at the two sites may be partially responsible for the discrepancies as well. Mount Hebo's increase in butterfly abundance could be attributed to the fact that the meadows are higher in elevation, often avoiding days of cool, foggy weather. Cascade Head, being along the coast, receives several days of rainy, cool, foggy weather, often occurring during the peak of the flight season (personal observations). Mount Hebo, on the other hand, is often above the fog and in the sunshine, allowing for longer butterfly activity for more days during the flight season (The Nature Conservancy 1992).

RECOMMENDATIONS

Effective management is crucial and urgently needed if we are to reach a goal of removing the Oregon Silverspot butterfly from the endangered species list. This management needs to be done at specific times of the year in order to minimize damage done to eggs, larvae, or pupae. Hammond (1993) in his vegetation management proposal suggests mowing twice a year, leaving vegetation three inches off the ground. The mowings should occur in late fall or spring and then again around the first of June. This would minimize damage to early larvae. More research needs to be done to identify the components needed for the adult and larval stages. Monitoring also needs to follow any management for the effects on the Silverspot populations.

We suggest that active management needs to be undertaken at all sites in a mosaic pattern. This limits the impact on the site as a whole, while creating new patches of early successional habitat required by the violet. Each site is unique and poses different problems in terms of management. For the optimal solution, separate management plans need to be designed at each of the sites.

Along with active management comes changes in management. This topic has been partially discussed so far, but changes need to consider the pooling of resources from all agencies concerned. The Nature Conservancy, the U. S. Forest Service, the U. S. Fish and Wildlife Service, and private organizations need to coordinate their efforts in the best interest of the Silverspot. Monies from these agencies can be combined and additional funding may come in the form of grants. Organization is the key to taking immediate action to implement appropriate management at precise times to prevent further population losses.

To ultimately delist the butterfly, habitat at current sites needs to be improved and new sites, once created, need to be repopulated. With the current low number of highly fragmented, unstable populations, the future looks bleak for the Oregon Silverspot butterfly.

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How Forest Fragmentation Hurts Species and What To Do About It

Daniel Simberloff[1]

HOW FRAGMENTATION HURTS SPECIES

Fragmentation of habitat is the major global environmental change occurring today and the one most likely to devastate biodiversity and ecological processes in the near future (Simberloff 1993a). Fragmentation always accompanies habitat destruction and the effects of fragmentation *per se*, as opposed simply to the loss of area, have been intensively studied only recently. An early suggestion that fragmentation could have important consequences concerned forests in Wisconsin (Curtis 1956), and most of the maps of habitat fragmentation that pepper conservation journals and texts are of forests. There is no satisfactory general theoretical framework for fragmentation analogous to the species-area relationship for habitat destruction (Simberloff 1993a). However, a number of intensive studies of particular systems suggest certain potential effects should always be considered. I will focus particularly on a system that surrounds my home, the longleaf pine (*Pinus palustris*) ecosystem of the Southeast.

Edge Effects

As fragments become smaller, they increasingly comprise edge habitat. This is because areas within the fragment are affected physically and biotically by the presence of the edge. The "edge effect" (Moore 1962, Williamson 1975)—the presence of species near an edge that characterize neither of the adjacent habitats—often results in increased diversity at an edge, so wildlife biologists have traditionally viewed edges as desirable (Harris 1988, Yahner 1988). However, species that colonize edges are often common elsewhere, while forest interior species that do not tolerate edges are often of special concern. Exactly how far inside a forest the existence of an edge is manifested depends on the forest, but it can be surprisingly far. Changes in wind currents, for example, can sometimes be detected at a distance 100 times the height of the vegetation; thus a forest with 20 m tall trees might need to be 2 km wide before any part of it would not suffer a meteorological edge effect (Saunders et al. 1991). Of course light and moisture regimes near an edge also differ from those of the interior. In Wisconsin forests, increased light can permit shade-intolerant vegetation to invade 30 m inwards (Ranney et al. 1981). In a well-studied fragment of old-growth longleaf pine forest, every introduced plant was within 2 m of a road or an artificially maintained clearing (S. Hermann, pers. comm. 1991).

Animals can also penetrate far from an edge. For example, tropical animals disperse seeds from secondary habitats into pristine forest tree falls 5 km away (Janzen 1983). Similar effects are known in other temperate and tropical forests (e.g., Janzen 1986; Wilcove et al. 1986). Most research on effects of animals penetrating an edge is on how they eat forest interior organisms.

Increased Predation and Herbivory

Nest predation may increase greatly in a fragmented landscape. Wilcove (1985, 1990) placed artificial nests with quail eggs in eastern U.S. forests of different size, ranging from small woodlots to the continuous forest of the Great Smoky Mountains National Park. In the latter, only 2% of nests were preyed upon within a week, while nests in rural woodlots of 4-10 ha averaged 48% predation, and similar sized suburban woodlots reached 70%. This study and similar results (e.g., Andren and Angelstam 1988, Small and Hunter 1988, Yahner and Scott 1988) inspired the "intermediate predator hypothesis" (cf. Terborgh 1988), which states that medium sized predators—raccoons, squirrels, blue jays, crows, dogs, cats, etc., in eastern forest—are greatly increased in a patchwork quilt of housing, farmland, second growth, and forest fragments. These predators, in turn, invade the forest fragments and prey on its denizens. In the continuous forests of the past, according to this hypothesis, large predators like wolves, mountain lions, and raptors were much more numerous and greatly suppressed populations of the intermediate predators.

In longleaf pine forests, nest predation of the state-listed gopher tortoise (*Gopherus polyphemus*) is enhanced by habitat destruction and fragmentation, as elevated populations of skunks, raccoons, crows, and introduced fire ants (*Solenopsis invicta*) thrive in the agricultural and second-growth matrix that surrounds longleaf fragments but attack nests in the longleaf (references in Simberloff 1993a). Nest predation of two common

[1] *Department of Biological Science, Florida State University, Tallahassee, Florida 32306.*

game species, bobwhite quail and turkey, is apparently similarly elevated, with many of the same culprits (references in Simberloff 1993a). Not only nests are preyed upon. Adult fox squirrels (*Sciurus niger*), gopher tortoises, and turkeys are all heavily preyed upon by species typical not of their favored old growth pine habitats, but of the varied, disturbed landscape that now prevails.

Herbivory of forest plants can be increased in a fragmented landscape just as predation can, if numbers of herbivores and/or their access to forest habitat are increased. The national forests of northern Wisconsin were once ca. 80% old growth and contained small fragments of earlier successional stages generated by fires and storms. Nowadays, in the wake of intensive logging, the landscape is the reverse: a patchwork quilt with 95% earlier successional stages dominated by aspen and only 5% old growth fragments of 5 - 200 ha. Alverson et al. (1988) have found that white-tailed deer populations have more than doubled in this landscape of excellent browse, and their browsing modifies even the old growth fragments. The deer select many old growth ground cover plants as well as seedlings of old growth trees like eastern hemlock, white cedar, and Canada yew. Alverson et al. (1988) believe it will be impossible to maintain old growth in small fragments unless deer populations are controlled.

Seedlings of longleaf pine suffer a similar fate in some regions (references in Simberloff 1993a). Pocket gophers (*Geomys pinetis*) and especially wild hogs eat longleaf seedlings and both animals thrive in the mixed agricultural/early successional stage communities surrounding longleaf fragments.

Failure of Metapopulation Dynamics

Metapopulation dynamics as a hedge against extinction are all the rage nowadays. The first model (Levins 1969) has been supplemented by numerous others (Hanski and Gilpin 1991), and the overall theoretical result is clear: populations that would not persist in one large population might do so in a metapopulation of populations, given sufficient rates of intersite movement. Metapopulation theory has superseded island biogeographic theory as a way of thinking about nature among conservation biologists (Merriam 1991). Many authors (e.g., Carter and Prince 1988, Wilson 1992, Noss 1993) contend that most species *are* distributed as metapopulations, but there are few data. If species are, in fact, maintained by continual recolonization of temporarily empty sites, it is easy to see how fragmentation could cause a metapopulation to collapse. As fragments get smaller and more isolated, the number of individuals moving from site to site decreases and may surpass a threshold below which the entire metapopulation collapses. But this is the rub: actual rates of movement between sites is rarely known, so it has proven almost impossible to assess whether most species *are* maintained as metapopulations. Cursory reviews (Harrison 1991, Simberloff 1993b) cast doubt on the proposition. Rather, it seems that many species are not metapopulations at all and many others are metapopulations of the sort envisioned by Boorman and Levitt (1973) and Pulliam (1988). In this model, large central populations are continuous sources of colonists for smaller ephemeral populations, which are "sinks" in the sense that they are maintained only by this recruitment and do not contribute to the persistence of the central populations. Until many more data are available on movement, any hypothesis of metapopulation-collapse induced by fragmentation is just that: an hypothesis.

Other Effects

Other effects of fragmentation are not as general as edge effects, increased predation and herbivory, and (potentially) failure of metapopulation dynamics. Some, however, probably occur in many systems.

Some species simply cannot maintain a population in a small fragment and, if fragments are sufficiently isolated, cannot maintain a population or metapopulation in the entire constellation of fragments. For example, whatever forces determine minimum viable population sizes (reviews by Shaffer 1981, Simberloff 1988), large carnivores are likely to disappear from small fragments for thermodynamic reasons alone unless they are very good at getting from site to site. There simply is insufficient food and space to support a population of bears or bobcats in a 10 ha site.

Introduced species are likely to be a far greater problem within forest fragments in a variegated landscape than they would be in intact large expanses. Not only are some introduced species highly adapted to the anthropogenous habitats that surround forest fragments but these habitats provide access to the forest proper (Simberloff 1994). Longleaf pine forests are almost devoid of introduced fire ants (*Solenopsis invicta*) except along roads or edges (Tschinkel 1988). The same is true of introduced plants.

Fragmentation can disrupt a fire regime and thereby change an entire community. Longleaf pine forests are fire disclimaxes maintained by frequent fires. Previously, lightning-induced fires spread widely and every site was thus burned every few years whether it was struck by lightning or not. Now the situation is completely changed, because the forest fragments are widely separated by farms, commercial plantations, roads, housing, etc. (Simberloff 1993a). Managers must perform regular controlled burns. Disruption of fire regimes can also be induced by introduced plants, as has occurred with the introduction of *Melaleuca quinquenervia* (Ewel 1986).

HOW TO COUNTER THE EFFECTS OF FRAGMENTATION

Corridors

The most highly publicized approach to mitigating problems engendered by fragmentation is to connect the fragments by corridors. These proposals range from rows of trees 10 m wide (Hussey et al. 1990) through mega-corridor proposals such as the Wildlands Project (Mann and Plummer 1993) or the proposed corridors 300 meters wide and thousands of kilometers long to provide for movement in the face of global warming (Hunter et al. 1988). What these proposals almost all have in common is a dearth of evidence that the target organisms will actually use the corridors and scant consideration of the cost of the corridors relative to the cost of other possible conservation measures (Simberloff and Cox 1987, Simberloff et al. 1992). Because these problems have been thoroughly aired (e.g., Hobbs 1992), I will not belabor them here.

Suffice it to say that even defenders of the proposition that corridors will often be very useful still qualify their defenses by admitting there are few data showing this, and that corridor proposals almost never include a discussion of possible alternative uses of funds.

Landscape Management

Another way to mitigate fragmentation, or one that might operate simultaneously with corridors, is to manage the entire landscape so that, as a whole, it supports a large fraction of the community. In other words, granted that small refuges are important but insufficient and that large enough refuges may not be attainable for economic reasons, is there some way that the land outside refuges can be managed so that the refuges do not appear, to the species of concern, as islands in an inhospitable sea? This is the premise behind the "new forestry" (e.g., Franklin 1989, Swanson and Franklin 1992): can timber be extracted from a major portion of the forest without major harm to resident species? The idea of "habitat variegation" (McIntyre and Barrett 1992) proposed for the northern tablelands of New South Wales is very similar. In both instances, the goal is to manage a landscape so that, even if it is far from pristine, and even if many resources are extracted, the threat to all species is vitiated. The Forest Service calls its version of this philosophy "New Perspectives" (Kessler et al. 1992).

Is the new forestry truly new? It and related ideas seem to be versions of a multiple-use strategy at the landscape level. The Forest Service's planning regulations under the National Forest Management Act of 1976 (36 C.F.R. pt. 219) require that the Service manage the land for multiple use (sec. 219.2(b)(1)). Well before then, the Forest Service applied a multiple-use philosophy to forest management (Kessler et al. 1992), and the Service has for years proclaimed the national forests the "Land of Many Uses" with countless signs. Thus there is no radical shift in direction indicated in the letter, "Ecosystem Management of the National Forests and Grasslands" sent by Chief F.D. Robertson on June 4, 1992. Rather, he says that now "an ecological approach will be used to achieve the multiple-use management of the National Forests and Grasslands." There seem to be two main components to the new approach: more science and an ecosystem focus. Will a more ecosystem-focussed management and closer interaction with scientists lead to successful maintenance of biodiversity while allowing continued other uses of the habitat, such as recreation and harvest of wood, at levels acceptable to all users? Only time will tell.

Just as with the rush to create corridors, however, there seems to be an element of faith in the New Perspectives. That is, one would expect a scientific approach to forest management to be founded on a falsifiable hypothesis and a commitment to discard the hypothesis if it is falsified. Neither the Chief's letter nor the more formal statement of the New Perspectives (Kessler et al. 1992) really presents the approach in this way. Neither considers the possibility that adequate maintenance of biodiversity might be incompatible with other uses at desired levels. Worse, the terms of these manifestoes are sufficiently vague and general that it is difficult to imagine a possible future result in some specific ecosystem or landscape that would definitively falsify the hypothesis. That is, is there a particular set of observations that could cause the Service, or its Chief, or its scientists, to proclaim that the New Perspectives cannot achieve their desired goal? The explicit method proposed by the scientists is adaptive management, in which "information from monitoring is used to continually evaluate and adjust management relative to predicted responses, management objectives, and predetermined thresholds of acceptable change" (Kessler et al. 1992, p. 225). It is unclear in this approach exactly when the entire framework for conceiving the problem might be rejected, if ever.

The Service itself clearly views the New Perspectives as something very different from what had gone before, and they are "new" in the sense of "recent." So it is important to remain optimistic and open-minded until some results are in. However, the history of conservation is littered with bright ideas of great intuitive appeal that turned out not to solve many or any conservation problems (Simberloff 1988), and one should take a lesson from this fact: remain skeptical and conceive of every idea as an hypothesis. Partnerships play a key role in the New Perspectives—Robertson's letter speaks of "partnerships with State and local governments, the private sector, conservation organizations...". For maintenance of biodiversity, it is clear in some regions that partnerships are necessary if only because the Service (in fact the entire federal government) does not control a large enough fraction of the land to ensure continued persistence of all species. Half of all federally listed species and subspecies are not found on any federal lands; 64% of all occurrence records for these taxa in Natural Heritage Data Centers are not from federal lands (Natural Heritage Data Center Network 1993). Consider the South: 90% of southern timberland is privately owned (Norvell 1993). Corporate timber companies own 17% of forested areas (Doster 1993). Evidently the private sector will have to be a partner if biodiversity is to be maintained.

A Case-Study: The Red-Cockaded Woodpecker

The red-cockaded woodpecker (*Picoides borealis*) is the most publicized problematic species in southern forests. Probably it would have achieved the global notoriety of the northern spotted owl but for two facts: 1) The longleaf pine forests that are its prime habitat, though beautiful, do not match in visual impact the majestic rainforests of the Northwest. 2) Almost all of the southern old-growth was cut down long ago (Tebo 1985), before conservation of biodiversity was even an issue and before heavy logging operations moved to the Northwest. Of about 28 million ha of original longleaf pine forest in the Southeast, fewer than 600 ha remain (references in Simberloff 1993a).

The woodpecker has been viewed as endangered since at least 1968 (U.S.D.I. 1968) and was listed as endangered in 1970 under the Endangered Species Conservation Act of 1969. It was one of the first species listed under the Endangered Species Act of 1973. Two recovery plans have been written; the first was never implemented, and the second (Lennartz and Henry 1985) was severely criticized by a committee appointed by the American Ornithologists' Union but has not been revised (Jackson 1994). The number of birds has declined more than 20% during the last decade, and much of the decline has been in populations designated as recovery populations in the 1985 recovery plan, including populations in national forests (James 1994). The bird is important not only in its own right, but because the cavities it laboriously constructs in large, diseased trees are used by many other species (Engstrom 1993).

Human activity has affected the woodpecker primarily in two ways. First, through loss of active and potential cavity trees, and second, through fragmentation and loss of foraging habitat through conversion of forest to other habitats or change in forest type because of short-rotation, even-aged management or limitation of fire (Jackson 1994). The bird does not regularly disperse more than about 8 km (Walters et al. 1988), and small, isolated sites that lose their woodpecker colonies yet appear to constitute suitable habitat often remain without birds for a long time. The decline of numbers is undoubtedly partly due to this loss of isolated populations, but it is an open question whether the failure of this aspect of metapopulation dynamics threatens the larger aggregations; over half of all sites are in six areas (James 1994).

Given the large fraction of southern forests in private ownership, it seems that recovery of this species could be greatly aided by partnerships with private landholders. Of approximately 4,000 known active sites, half are on national forests, a fourth on Department of Defense lands, and only an eighth on private lands (Costa 1993). This disparity between fraction of privately held lands and fraction of woodpeckers reflects the fact that management of private lands has generally been even more inimical to the bird than management of the national forests. Not all private lands have been poorly managed from the standpoint of the woodpecker. The Red Hills Hunting Plantations of southern Georgia contain the sixth largest aggregation of birds (James 1994). This region is dominated by uneven-age management and selective cutting (James 1994), the antithesis of the methods primarily used on national forests and large timber plantations (Jackson 1994).

The generally poor situation on private lands has led the U.S. Fish and Wildlife Service to formulate a strategy for private landholders that they claim will aid the recovery effort (Costa 1993). They view small landowners as unlikely targets for this effort because they feel the costs would be too high for them to bear and the birds on their lands are doomed anyway, so they focus on large landowners. The strategy has three parts: 1) a procedures manual for private lands, 2) a rangewide habitat conservation plan, and 3) individual habitat conservation plans or memoranda of understanding. The last element, the memoranda of understanding, is viewed as "probably the best hope of maintaining the remaining, relatively large RCW populations on private lands" (Costa 1993, p. 13).

Given the importance of these memoranda, it is not surprising that the first one, with the Georgia Pacific Corporation, by far the largest southern timberland holder (Norvell 1993; pulp and paper companies not included), was front page news in the national press (e.g., Schneider 1993). Secretary of Interior Babbitt said that the most important effect of this agreement "could well be the precedent it sets in helping to establish a less politically incendiary approach to safeguarding endangered wildlife and their habitat" (Schneider 1993). It is thus crucial that this agreement be a sound one.

The company agrees to restrict operations on some 20,000 ha in return for a government promise not to invoke the Endangered Species Act to restrict logging on the remaining 1.68 million ha of Georgia Pacific timberland in the South. The restriction consists of not clearing at least 4 ha of land around each colony site on those 20,000 ha, and reducing the stocking rate to 4.59 m2 basal area of pine/ha over 61 ha. On the face of it, this agreement seems quite remarkable, given that home ranges often exceed 80 ha and may exceed 400 ha in poor habitat (references in Jackson 1994). In prime homogeneous habitat in Florida, ca. 25 ha per social group apparently suffices for population persistence (James, pers. comm. 1993). This disparity is less mysterious, perhaps, in light of the heavily criticized 1985 recovery plan (Lennartz and Henry 1985), which calls for a 4 ha core area around each cavity tree. The stocking rate is more impressive than the area, as it is quite low and constitutes a 3-fold reduction of the original plan of the company (Wood and Kleinhofs 1992). From the standpoint of the company, this agreement may be acceptable in that the sacrifice of income does not unduly affect profit margin (Wood and Kleinhofs 1992).

From the standpoint of the U.S.F.W.S., it is hard to imagine why this agreement is acceptable; certainly it does not reflect an abundance of scientific evidence.

The rangewide habitat conservation plan of the U.S.F.W.S. seems peculiar as well. It will apparently consist of memoranda of understanding with large landholders, such as the one with Georgia Pacific, plus a global agreement with small landholders by which woodpeckers on their lands would be moved to federal lands or to larger private lands (Costa 1993). It is again clear that such cooperation benefits the landholders: it relieves them of the onus of managing for an endangered species. It is again not clear that the U.S.F.W.S. will benefit. The U.S.F.W.S. sees these birds as a potential aid in designated, larger recovery populations. Two recent developments (reviewed by Jackson 1994) spur such reasoning: movement of young females from natal sites to clans lacking a female (DeFazio et al. 1987), and the construction of artificial cavities (Copeyon 1990, Allen 1991). Although both techniques may be of great use in recovery, they are sufficiently new that one cannot be certain how well they will work in the long term, and they are expensive.

Although the Endangered Species Act does not address the fate of species that interact with a listed species, an ecosystem approach to forest management surely would, and there has been no substantial study of how artificial cavities are used by the many species who depend on red-cockaded woodpecker holes. In fact, if the new thrust of forest management is to be ecosystemic, a goal proclaimed not only by the Forest Service but by Secretary Babbitt (U.S.D.I. 1993), species-specific remedies such as movement of individuals and provision of artificial habitat will likely play a reduced role. Rather, a real ecosystem approach necessitates addressing entire native ecosystems. The longleaf pine forests are ripe for such an approach, with numerous species of special concern, some understanding of the problems of many of them, and a good scientific basis for management (Hermann 1993). One hears much talk about "getting ahead of the extinction curve" nowadays; we will never get ahead of this curve if we attempt to save one species at a time.

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CONSERVATION BIOLOGY AND RESTORATION ECOLOGY

Session Summary

W. Wallace Covington, Chair

This session began with a presentation by Wallace Covington on changes since Euro-American settlement in southwestern ponderosa pine ecosystems in the context of conservation biology. Drawing on the conservation biology postulate that outbursts reduce diversity, he related the postsettlement irruptions of pine populations to decreases in the diversity and stability of native flora and fauna. He closed with a description of the field of ecological restoration and proposed treatments for restoring and maintaining more nearly natural conditions in the southwestern ponderosa pine/bunchgrass type. Next was a presentation by Tim Allen entitled, "Toward a definition of sustainability." Allen developed a definition of sustainability which included human cultures functioning to substitute for natural processes in stabilizing the landscape. He then took an historical look at the failures of various cultures in sustaining their natural resource bases including examples ranging from the early Greek and Roman times to the present. He closed by developing a scale-explicit framework for addressing the issues of sustainability and ecosystem management.

Allen's talk was followed by Tom Bonnicksen who discussed social and political issues in ecological restoration. In addressing the question, "Should we attempt to restore ecosystems?", Bonnicksen stressed the importance of including humans in the restored landscape. He went on to elaborate on social definition of a desired condition for a restored landscape and then on to the politics of ecological restoration.

Dan Neary presented the next talk, which concerned the restoration of degraded soils. Neary discussed the soil as the foundation of sustainability for ecosystems and then described the factors which control soil fertility in various types. He then followed with factors which lead to soil degradation and closed with an overview of management techniques designed to restore degraded soils.

Next, Steve Sackett presented a discussion of ways in which prescribed burning can be used to reduce heavy fuel accumulations, thin dense stands of trees, and prepare a seedbed for tree establishment. He then addressed specific examples of the use of prescribed fire in southwestern ponderosa pine ecosystems to improve forest health and productivity.

This session closed with a presentation by Victoria Yazzie Pina in which she described a Navajo perspective on the postulates of conservation biology and the principles of ecological restoration. Pina concluded that the holistic philosophies inculcated in Navajo culture are consistent with key concepts of conservation biology and ecological restoration. She closed by describing the Navajo religious concept of *Sa'a Naghai Bik'e Hozho* (walking toward the sacred way) which expresses the health, beauty, and harmony of humans with the land and relating this to the ecosystem approach to land management.

Implications for Ponderosa Pine/Bunchgrass Ecological Systems

W. Wallace Covington[1]

Astract — When viewed from a conservation biology perspective, postsettlement outbursts of ponderosa pine trees in ponderosa pine/bunchgrass ecosystems not only reduce biological diversity but also lead to nonadaptive catastrophic processes. These changes, in conjunction with parallel decreases in natural resource conditions, are compelling reasons for beginning ecological restoration treatments designed to establish landscape conditions which more closely approximate the conditions which these ecosystems have experienced over evolutionary time.

"Between the two extremes of passively following Nature on the one hand, and open revolt against her on the other, is a wide area for applying the basic philosophy of working in harmony with natural tendencies" (H. J. Lutz 1959).

INTRODUCTION

This document presents an overview of some unintended consequences of failure to manage in harmony with natural tendencies in the Southwest. Although the paper focuses on ponderosa pine/bunchgrass ecological systems, parallel changes of ecosystem structure and function have occurred throughout other forest and woodland types in the Southwest.

This discussion will be placed in the context of key concepts of conservation biology and restoration ecology. The paper begins with a quick overview of some consequences of overly simplistic approaches to resource management. This is followed by a brief outline of some central postulates of conservation biology. Next will be a synopsis of changes since settlement in ponderosa pine/bunchgrass ecosystems. This synopsis is followed by a brief outline of ecological restoration concepts as stated in draft policy statements by the Society for Ecological Restoration. Then comes a presentation of some ideas of how we might apply these principles to the restoration of more nearly natural condition in southwestern ponderosa pine/bunchgrass ecosystems. Finally, the paper closes with a challenge for action and an alarm regarding the impending loss of key components of our natural resource management infrastructure.

OVERSIMPLIFICATION IN NATURAL RESOURCE MANAGEMENT

It is interesting to think about ecological management problems in the context of the "exploitation heritage" of contemporary natural resource management. In large part, this notion stems from a reductionist, anthropocentric view of ecological systems, in contrast to a more holistic, ecocentric view (Leopold 1933, 1939, and 1949; Flader 1974; Devall and Sessions 1985). The exploitive view traces its roots to the industrial revolution and specifically to a commodity view of the land, in which the land is viewed merely as a source of resources for the "engine" of economic growth (Flader 1974 and this volume; Callicott, this volume). The cornerstone in such a view is that the role of humans is to exploit natural resources by channelling the "machinery" of natural resource production to support the accumulation of wealth. A consequence of this thinking is the conclusion that the best and highest value of the land will be achieved by killing all of the predators, spraying all of the insects, putting out all of the fires, and replacing all of the slowly growing and "decadent", old-growth trees with rapidly growing and "vigorous", young trees. These management actions are viewed as eliminating the "waste" from, and increasing the "efficiency" of, natural resource production.

[1] *Wallace Covington is Professor of Forest Ecology, School of Forestry, Northern Arizona University, Flagstaff, AZ 86011.*

Ironically predator control (extermination) programs, insect spraying programs, fire suppression programs, and old growth liquidation programs all appear to be successful at first and thus become strongly entrenched policies (e.g., see Holling 1981). Nature's backlash occurs fairly rapidly in predator:prey systems. For example, land degradation caused by overpopulation by deer (in turn caused by wolf and lion extirpation programs) was obvious within the first decade of this century (Leopold 1949, Flader 1974). It can take longer in insect spraying programs. C.S. Holling (1981 and elsewhere) has documented the backlash in spruce budworm spraying programs in northern coniferous forests. While successful initially, eventually (within 2 to 4 decades) so much of the forest becomes susceptible to budworm outbreak that spraying could not prevent mortality on an unprecedented scale.

In the case of fire suppression, the lag in Nature's backlash is even longer, perhaps 5-10 decades depending upon the interplay between fuel production and decomposition and between new tree establishment and mortality (Sando 1978, Holling 1981, Kilgore 1981, Covington and Moore 1992). Eventually though, enough fuel accumulates so that no amount of fire suppression effort can contain ensuing wildfires.

Replacement of old-growth trees and stands with younger stands was a very successful strategy while wood fiber production was the dominant goal of public forest management in the U.S. (arguably almost the entire history of public forestry, except the last few decades). Today, naturally functioning old-growth trees and stands are widely viewed as an integral component of forest landscape ecosystems and one which has become exceedingly rare, if not totally absent, in most forest and woodland types (Hoover and Wills 1984, Thomas 1979, Booth 1991, Kaufmann et al. 1992).

SOME POSTULATES OF CONSERVATION BIOLOGY

Michael Soule, in his 1985 paper, presented some postulates of the discipline of conservation biology which are relevant to interpreting the ecological consequences of changes since settlement in ponderosa pine ecosystems. He proposed two sets: a functional, or mechanistic set and an ethical, or normative, set. For this discussion I will focus on the functional postulates. Soule defined the functional postulates as working propositions based partly on evidence, partly on theory, and partly on intuition:

The first, the evolutionary postulate states: Many of the species that constitute natural communities are the products of coevolutionary processes.

The second functional postulate concerns the scale of ecological processes: Many, if not all, ecological processes have thresholds below and above which they become discontinuous, chaotic, or suspended. Two major assumptions, or generalizations, underlie this postulate. First, the temporal continuity of habitats and successional stages depends on size. Second, outbursts reduce diversity.

Finally, genetic and demographic processes have thresholds below which nonadaptive, random forces begin to prevail over adaptive, deterministic forces within populations.

CHANGES IN SOUTHWESTERN PONDEROSA PINE ECOSYSTEMS

The most extensive study of postsettlement changes in southwestern ponderosa pine to date is the monograph by Cooper (1960). In this dissertation work, Cooper used a combination of historical methods and direct observations of stand structure to document changes since settlement in west-central Arizona. He used reports from early travelers to illustrate the changes in appearance of the ponderosa pine forest since settlement. For example, E. F. Beale's 1858 report is quoted by Cooper (1960) as follows:

"We came to a glorious forest of lofty pines, through which we have travelled ten miles. The country was beautifully undulating, and although we usually associate the idea of barrenness with the pine regions, it was not so in this instance; every foot being covered with the finest grass, and beautiful broad grassy vales extending in every direction. The forest was perfectly open and unencumbered with brush wood, so that the travelling was excellent" (Beale 1858).

Cooper (1960) concluded that, "The overwhelming impression one gets from the older Indians and white pioneers of the Arizona pine forest is that the entire forest was once much more open and park-like than it is today."

Before European settlement of northern Arizona in the 1860's and 70's, periodic natural surface fires occurred in ponderosa pine forests at frequent intervals, every 2-12 years (Weaver 1951, Cooper 1960, Dieterich 1980, Stein 1988, Swetnam 1990). Several factors associated with European settlement caused a reduction in fire frequency and size. Roads and trails broke up fuel continuity. Domestic livestock grazing, especially overgrazing and trampling by cattle and sheep in the 1880's and 1890's, greatly reduced herbaceous fuels. Active fire suppression, as early as 1908 in the Flagstaff area, was a principal duty of early foresters in the Southwest. A direct result of interrupting and suppressing these naturally occurring, periodic fires has been the development of overstocked forests.

Changes in the forest structure (e.g., tree density, cover, age distributions) in southwestern ponderosa pine forests since European settlement have been blamed for many ecosystem management problems (Cooper 1960, Biswell 1973, Weaver 1974, Covington and Sackett 1990, Covington and Moore 1992). Problems attributed to fire exclusion and resulting increased tree density in ponderosa pine include:

1. an increase in tree density, especially of small diameter trees — Arnold (1950), Cooper (1960), Biswell (1973), Weaver (1974), Steele et al. (1986), Barrett (1988), Laudenslayer et al. (1989), Savage (1989), Keane et al. (1990), Covington and Moore (1992)
2. a decrease in herbaceous and shrub production — Arnold (1950), Cooper (1960), Biswell (1973), Weaver (1974), Steele et al. (1986)
3. a consequent decrease in the diversity of net primary production and hence food web diversity (i.e., a tendency toward a monotypic photosynthesis concentrated in ponderosa pine trees) — This conclusion comes from the fact that NPP in open park-like stands was spread across 50-200 vascular plants in addition to ponderosa pine, whereas today it is concentrated primarily in ponderosa pine.
4. a shift in wildlife habitat from one favoring species requiring open, park-like stands dominated by large trees to one favoring species which are more successful in dense forests composed of smaller diameter trees (Covington and Moore 1992)
5. accumulation of pine litter on the soil surface as forest floor fuels (pine litter is very high in lignin compared to herbaceous litter; lignin is a broad-based metabolic inhibitor.) — Ponderosa pine litter has one of the lowest decomposition rates ever observed (Olson 1963 and van Wagtendonk 1985)
6. disruption of organic matter processing and nutrient cycling — Covington and Sackett (1984, 1986, 1990)
7. increased crown fuel loading and increased crown closure — see references under item 1 (above) plus Barrows (1978), Sando (1978), and Covington and Moore (1992)
8. increased fuel ladder (vertical fuel continuity) — Barrows (1978), Swetnam and Dieterich (1985), Swetnam (1990), and Covington and Moore (1992)
9. increased patch and landscape crownfire hazard and occurrence — see references in item 7 (above)
10. decreased tree vigor, especially the oldest age classes (300 yrs old) — Avery et al. (1976), Sutherland (1983), Waring (1983), Covington and Moore (1992)
11. increased tree mortality due to insects and diseases which attack trees of low vigor — Sartwell (1971), Sartwell and Stevens (1975)
12. ecosystem simplification at all levels in the biotic and landscape hierarchy (decreased nutrient recycling, forest floor fuels steadily accumulating, simplification of NPP and food webs, decreased species diversity, larger and more homogenous disturbances, decreased landscape diversity) — Mooney (1981), Covington and Moore (1992)

Covington and Moore (1992) present quantitative estimates of changes since settlement for two study areas in the Arizona ponderosa pine type. Their estimates of 23-56 trees per acre (with most trees being large, old, "yellow" pine) at the time of settlement in the Flagstaff area and on the Kaibab Plateau are consistent with the results of other studies including those of Woolsey (1911), Rasmussen (1941), Cooper (1960), and White (1985). This open, presettlement forest structure stands in stark contrast to today's dense, postsettlement stands containing 200-1,200 trees per acre with very few remaining old-growth trees. The magnitude of such a population irruption is staggering. For example, a "back of the envelope calculation" would yield an estimate of an excess of over one billion trees in Arizona alone (this estimate is based on a presettlement density of 40 trees per acre, a current density of 350 trees per acre (Fox et al. unpublished), and a total of 3.35 million acres of the ponderosa pine type in Arizona). Such a population irruption dwarfs the irruptions in deer populations estimated by Leopold (cited in Flader 1974).

Covington and Moore went on to estimate changes in resource conditions since settlement. These results indicated, among other things, decreases in water availability and runoff, in aesthetic values, and in forage production. Although their inferences regarding changes in wildlife habitat have been controversial, there can be little doubt that the change from a landscape dominated by prairie vegetation with patches of pine trees to one where pine trees dominated the net primary productivity has wrought substantial changes in both the composition and the population sizes of animal communities.

ECOLOGICAL RESTORATION

Given that many of these changes are deleterious, the question then becomes, "What can we do to remedy these problems?" In a predator:prey system, if predators are suddenly reintroduced into a prey population that is too low in vigor or too few in number, the whole system might well crash. Wholesale cessation of insect spraying programs could result in very large and massive tree mortality requiring many decades for recovery (Holling 1981). Similarly, allowing fires to burn freely in ecosystems which have unnaturally heavy fuel accumulations might cause extensive long term damage to food webs, nutrient cycles, and soil development (via accelerated erosion). In fact, research in giant sequoia/mixed conifer, oak savannah, and ponderosa pine/bunchgrass indicates that manual removal (thinning) of trees and spot fuel treatment may be necessary prerequisites for restoration of fire as a natural component of ecosystems adapted to a frequent, low intensity fire regime (see Parsons (1981), Bonnicksen and Stone (1985), Parsons et al. (1986) for a lively discussion of policy concerns). The need for manual thinning and spot fuel treatment as components of an ecological restoration program in southwestern ponderosa pine are indicated by the difficulty of thinning postsettlement trees by prescribed burning and the high mortality rate of old-growth trees following prescribed burning of current heavy forest floor loads (Harrington and Sackett 1992).

Answering the question, "What can we do to remedy these problems?" is what the field of restoration ecology and management is all about (Jordan et al. 1987, Jackson 1992). It deals specifically with research and management experimentation to determine ways to safely restore degraded ecological systems to more nearly natural conditions. Restoration ecology was founded by Aldo Leopold after he abandoned the sustained yield view of game management, shortly after his arrival in the Southwest. As Leopold said, "The first step is to reconstruct a sample of what we had to begin with." Ironically, one of Leopold's first (1924) professional publications (written while he was a forester with the Southwestern Region of the Forest Service) dealt with the postsettlement decrease in grasses and the increase in shrubs, trees, and fuels in Arizona.

Although some of the principles for ecological restoration are still in the development stage, several have received broad-based support. Various authors in Jordan et al. (1987) provide stimulating discussions of some of these principles. Perhaps the most useful restoration definitions and principles in the context of this paper are those presented in the 1992 draft policy statements of the Society for Ecological Restoration. In that document "ecological restoration" is defined as the process of intentionally altering a site to establish a defined, indigenous, historic ecosystem. The goal of this process is to emulate the structure, function, diversity and dynamics of the specified ecosystem. The policy statement goes on to state that while human use of restored landscapes is not only inevitable but also desirable, these uses should be designed to be compatible with the principle of sustainability.

Regarding the preservation of biodiversity and endangered species, the policy statement recognizes that endangered species cannot be sustained satisfactorily apart from viable ecosystems. This has lead the society to advocate that resource agencies charged with preserving biodiversity and protecting endangered species focus attention on restoring and maintaining the ecosystems upon which endangered species depend.

CONSERVATION AND RESTORATION OF SOUTHWESTERN PONDEROSA PINE ECOSYSTEMS

Given the well documented outburst of ponderosa pine since Euro-American settlement and the consequent declines in both ecological conditions and resource values, it is incumbent upon today's generation of natural resource managers to begin to set things right. Although little practical thought has been put into how exactly to accomplish this on a large scale, such a program would clearly involve site-specific adaptations of the following elements:

1. Preserve all trees which predate grazing and fire exclusion.
2. Thin all postsettlement trees, except for those needed to emulate presettlement densities and diameter distributions.
3. Manually or mechanically remove heavy forest floor material from under presettlement tree canopies.
4. Prescribe burn — Initial cool season prescription (ideally wet soil, cool air temperatures; eventually warm season maintenance burning or burning alternating with livestock grazing to approximate effects of natural fires while minimizing air quality degradation by smoke).
5. Reintroduce indigenous biota (plants and wildlife, in particular) when necessitated by local extinction.

CONCLUSIONS

In summary, owing largely to the lack of an ecological view of the land, the history of Euro-American settlement of the southwestern ponderosa pine/bunchgrass type has been characterized by open revolt against Nature. While there can be little doubt that much remains to be discovered about ponderosa pine ecosystem structure and function, what we do know is that inaction is indefensible, with long-term negative ramifications for ecosystem structure and function. Reliance on piecemeal approaches (one species at a time, one process at a time) is overly simplistic and likely to have undesirable consequences for the land system as a whole. Instead, it is becoming increasingly apparent that large-scale, whole-system, management experiments are necessary for discovering how best to restore the health (inherent ability for self-renewal) and integrity (coevolved biological diversity) of ponderosa pine ecosystems.

Finally, removal of excess trees and prescribed burning possibly in conjunction with carefully controlled livestock grazing are necessary steps not only in restoring but also in maintaining the health and integrity of our southwestern forest ecosystems. A failure to understand conservation biology and restoration ecology by many in the debate over forest management in the Southwest has lead to constraints which may well result in the destruction of much of the "tree removal" and forage (herbaceous fuel) management infrastructure essential for restoring and maintaining ecosystem health and integrity while maintaining a culturally acceptable fire regime. If we allow this to happen, it seems probable that, within the next generation or two, our children and grandchildren will have to invest tax dollars to rebuild that infrastructure — unless insects, disease, and wildfire preempt their options.

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Toward a Definition of Sustainability

T. F. H. Allen[1], and Thomas W. Hoekstra[2]

Abstract — Sustainability is not an absolute, independent of human conceptual frameworks. Rather it is always set in the context of decisions about what type of system is to be sustained and over what spatiotemporal scale. There is a duality of the material system itself, as opposed to human frameworks for communication or management action. Exclusive focus on the material system gives the decision-maker an impossible number of choices, and no definitions; exclusive focus on scale and type gives narrowly directed capricious action that ignores lessons from the material system. An ideal is guided by the principal physical and biological material flows, as the scientist erects a rich system definition that explicitly links different types of system, like landscape and ecosystem, across a range of scales, in a coherent complex management scheme. Sustainability is not a matter of degree, because the material imbalances of incomplete sustainability will bring all down like the ancient failure of Sumerian agriculture through salination. True, sustaining at one scale may deny sustainability at another, but if it is in a scale- explicit framework, trade-offs can be calculated and weighed. Sustainability must work with natural processes, but they are not those of the pristine system. Rather management must accommodate to new structures and their patterns of process which naturally emerge far from equilibrium as a result of a substantial human presence. In a world with 5 billion people, managing towards a pristine system is irresponsible.

INTRODUCTION

Sustainability, An Emerging Concept

Over the last decade a set of terms has emerged in the arena of resource management that indicate an alternative style of applied ecology. This new vocabulary is a response to past and present piecemeal approaches to natural resources, research and management. Terms include biodiversity, ecosystem health, ecosystem management, viable populations, conservation biology, restoration ecology, and global change. One of the most important of these terms is sustainability. Like the other concepts listed above, sustainability is an immature notion. It conjures up different images for each environmental scientist and manager, although there is a common, general understanding. For example, everyone agrees that sustainability is a good thing, and that desirable situations last longer under it. Sustainability is appealing because, despite differences as to how to achieve sustainability, both "green" environmentalists as well as those investing in commodity production favor it. Not only is sustainability a desirable ecological condition, but its reliable context is a requirement for a return on long-term capital investment. The wide spectrum of agreement on the virtues of sustainability make sustainability a touchstone for mutual consent.

Problems Of Defining Sustainability

These new terms, including sustainability are somewhat vague. Many of them have arisen because modern problems require environmental scientists and managers to grope up-scale to larger issues, such as global warming and global amphibian decline, where we have little experience to date. We have found it very helpful to fall back on the ideas and protocols of our new book, *Toward a Unified Ecology* (Allen and Hoekstra, 1992). In this paper, we will apply the general approach used

[1] *Botany Department, The University of Wisconsin, Madison, WI., USA.*

[2] *Assistant Director for Research, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO., USA.*

in our book to defining and unraveling the notion of sustainability. By the end of this presentation, we hope to have given a rationale for a definition of sustainability that will offer common ground for future communication and management action.

Defining sustainability is not simple because it must apply to many ecological and social situations. To make this point, we need to draw attention to the difference between the observer and the material system. We must therefore define what we mean by the term "material system." The material system is the physical substance toward which a discussion is directed, as opposed to the abstraction of that system which emerges in words and concepts (figure 1). The material system includes humans, if they are physically present.

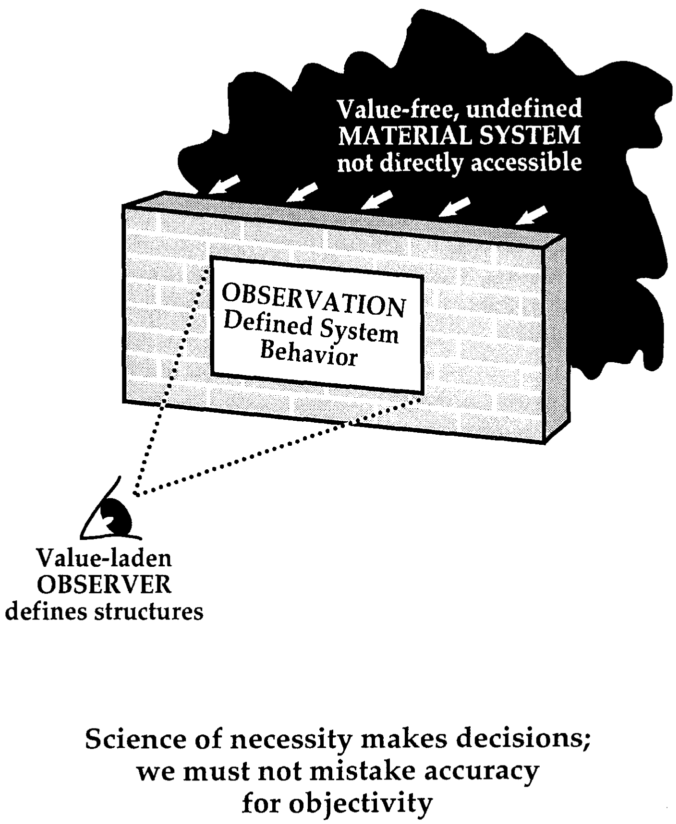


Figure 1. — Scientists do not have access to the complete material system as such, they can only collect and analyze data. The full material system is undefined and involves no values in and of itself. By contrast, the human observer experiences the material system through a set of value judgements and decisions as to observation protocol. The observation is of the behavior of a defined system.

Sustainable ecological systems can be different in two separate ways. First sustainable systems can be different because the observer recognizes different aspects of the material system as important. Those characteristics define what is in the foreground. Different material systems will suggest different criteria for what is important, but even one material situation can be viewed according to many criteria such that the ecologist recognizes an ecosystem as opposed to a community, population or landscape. For example, a given tract of land that makes up the material system can be viewed as a spatially defined and ordered place, a landscape; however, that same piece of land may be seen as a physical setting in which a population is growing or declining. Both views can be reconciled with the material system, but in the first case the system is identified as a landscape, while in the second case it is a population and its environment.

The second way that sustainable systems can be different is a matter of scale. Scale is entirely separate from differences of system type. A given material system will appear very different when it is viewed at a different scale, even if the observer recognizes the same system type. A physically small landscape can appear as different from a large landscape as it can from viewing the same material system in population and population environment terms. Appropriate action for achieving sustainability will be altered by the spatial or temporal extent of the universe to be sustained.

SYSTEM TYPE AND SCALE

Richness Of Perspective

In our book (Allen and Hoekstra, 1992) we point out that the type of ecological system must be explicitly identified by the scientific manager. System type is not self-evident and needs to be stated before any discussion. Even in the simplest setting, no two observers will recognize exactly the same features of the material system as being critical. Observers will disagree on what is in the foreground, and conversely what is in the background (figure 2). An example might be when focus on genetic variability in a population may involve ignoring the processes of nutrient flow in which the population participates; genetics comes to the foreground while nutrient cycling becomes part of the background. Choosing a point of view is an inescapable responsibility of the manager and scientist alike. Neither serious science nor effective management can proceed until the type and scale of the system to be sustained is stated explicitly.

The whole material system cannot be sustained in its every facet, and we would not want to do that if we could. Life precisely works as a process of building up and breaking down materials and relationships. In all healthy biological functioning, things persist and grow because other things are not sustained, as when prey succumbs to predator. Absolute sustainability where nothing is broken down might be possible on the moon, for that is a suitably static place, but here on Earth, a completely sustainable system in every detail cannot, and has never existed on it. So by sustainability we must mean something different from the potential for absolute and complete persistence.

Various criteria or perspectives on the ecological system are more popular than others, sometimes because they have become mistaken for the perfect or somehow true sustainability. One such criterion, which is now being redefined by a more diverse

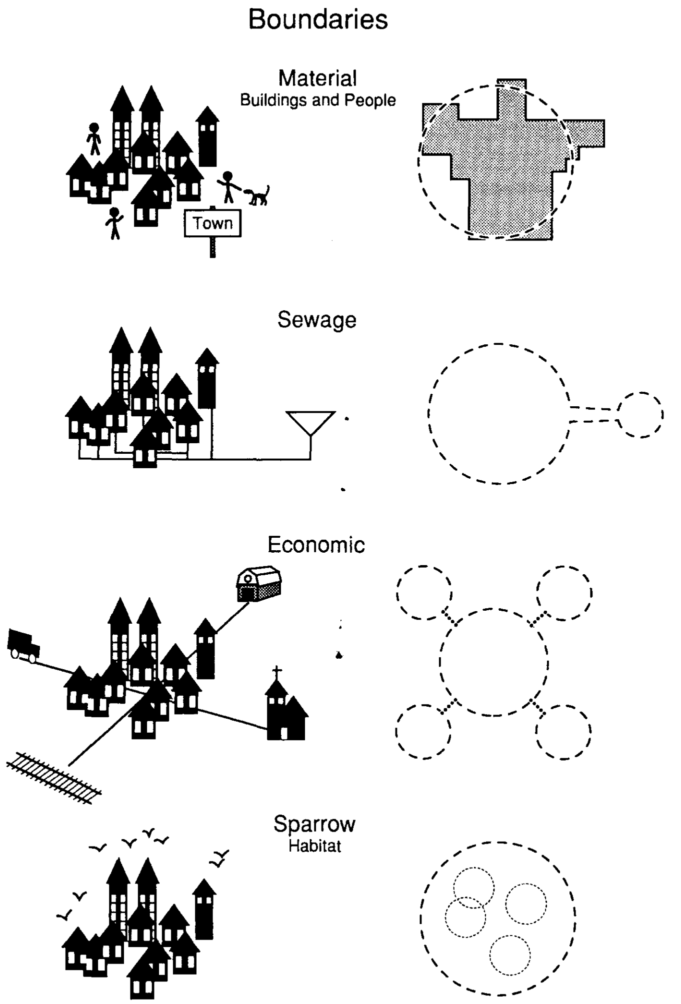


Figure 2. — A material town consists of all the buildings, ground, air and biota, including humans (top). Sustainability may involve mineral nutrient flow for recycling and waste heavy metals that must be kept out of the local food chain. The town in those terms involves connecting buildings to the sewage works. However, ecological remedial action will cost money, and so an equally valid perspective on sustainability and the town will emphasize economic considerations. Yet a third perspective might view the town in terms of the habitat for birds or other biota. Under this view, the town takes on yet another form with yet other parts (eg. nesting sites) linked together by connections important to the animals in question, but unimportant for sewage collection and economics.

ecosystem management approach is sustainable production of commodities such as timber, livestock and minerals. Another criterion that has become iconic is sustaining populations of individual species identified as critical, like the Spotted Owl. These criteria are valid for at least local, particular situations and intentions, but the mistake is using them zealously and extensively for profit or preservation to the exclusion of other criteria for sustainability. The requirement for being explicit as to criterion is not an excuse for fixating on a narrow criterion when the situation demands subtlety and complex criteria to deal with competing interests. Other different criteria for organizing sustainability might include aesthetics or human cultural preservation.

The critical point here is that sustainability must always involve a chosen perspective if it is to be meaningful. Without a suitable definition of ecological system type, it is not possible to set unequivocal standards of achieving sustainability. Without a criterion to assess results, sustainability is vacuous. Nevertheless, a criterion is a matter of human decisions, not something that follows in any necessary way from the material system, and so explicit statement of system type must be tempered by a willingness to suspend one definition and turn to another as the situation warrants. Intellectual flexibility is crucial, because rational action to make a system sustainable under one criterion might well create surprises under another that has not been considered. The problem may well be a great shock as we let the consequences of the planned management action take their course in real time. The challenge is to link different explicit types of sustainability so that a suitably rich management process is set in place. That may be something of an iterative process tested by upsets.

Scale Of Sustainability

Just as no one criterion is particularly correct, there is no nature-given scale at which a system is sustainable or otherwise. Sustainability without a stated scale has no meaning (figure 3). Since the biosphere is only as sustainable as the sun that supports it, then all ecological sustainability has an upper temporal limit. "Sustainable for how long?" then becomes a fair question. Therefore, a system that is only sustainable for a relatively short time may be well worth sustaining over that brief period. Sustainability applied to a microcosm is likely to be a critical aspect of it, even though a matter of months may be enough. Most uses of the concept of sustainability will be in between months and eons. Although the options for scale of sustainability are many, failure to be explicit makes plans ambiguous. Allen & Starr (1982) identify a corn field as sustainable over a period of two years to about half a century, but not sustainable at temporal scales of only a single growing season including the first frost or periods longer than a few centuries.

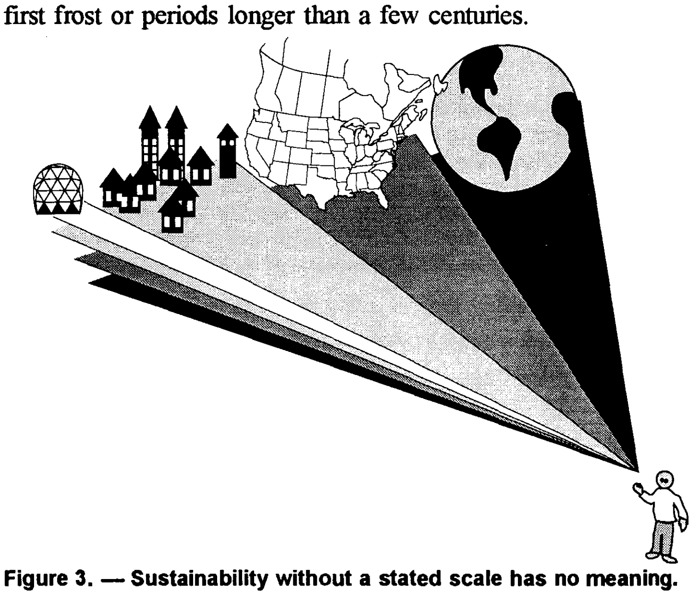


Figure 3. — Sustainability without a stated scale has no meaning.

Both the spatial and the temporal extent of sustainability must be stated for each case. Actions to sustain a local rare population are likely to be different from sustaining a large landscape mosaic across which the species moves over millennia. Often we will want sustainability for a larger spatial area to pertain to longer time frames, but the link between temporal and spatial scales is not a requirement. It may be appropriate to sustain for a very long time a small system of special cultural significance, such as a grove of sacred trees. It may also be appropriate to sustain very large systems for only a few years, as in the genetic characteristics of the crop across the entire corn belt.

CONCEPTUAL FRAMEWORK VERSUS THE MATERIAL SYSTEM

Although ecology is a matter of modeled types and scaled conceptions, do not forget that the discourse relates to a material system. It is easy to put too much emphasis either on the complete material system, on the one hand, or the conceptual framework, on the other hand. An overcommitment to either the material system or the conceptual aspects of sustainability will have unfortunate results. A complete focus on the material system leads to undefined and therefore unscientific understanding. Conversely, a complete focus on the conceptual framework leads to decisions that are not only arbitrary but also capricious. We recognize two classes of misconception about sustainability. One comes from placing an overemphasis on the observer side of the duality as opposed to the observed system. The other comes from an overemphasis on the material, observed side of the duality.

An overemphasis on the material system relates to some of the problems mentioned above in failing to type and scale the system under discussion. One manifestation of this error would be an insistent focus on the material system that existed before there was any significant human influence. We see this archaic system before the coming of our species as being of historic interest, but irrelevant to current management. It is inappropriate to strive for a completely pristine system without humans and use that as the benchmark for sustainability. The first problem with that agenda is that it cannot be achieved, even to a significant degree. Second, we would not want to do it if we could. Sustainability is appropriately set in the context of material human presence and must be prescribed by human value systems (figure 4).

All material systems can be observed in an enormous number of ways without much effort on the part of the scientist. This fact presses itself upon us when the material system offers as rich a primary experience as does ecological material. Therefore it seems particularly inappropriate in an ecological setting to hold up the full, somehow "natural," material system as the standard against which management action should be judged. If one happened to achieve sustainability of an ecological material system independent of any values, nobody would be able to tell that to be the case. There would be no way to know whether

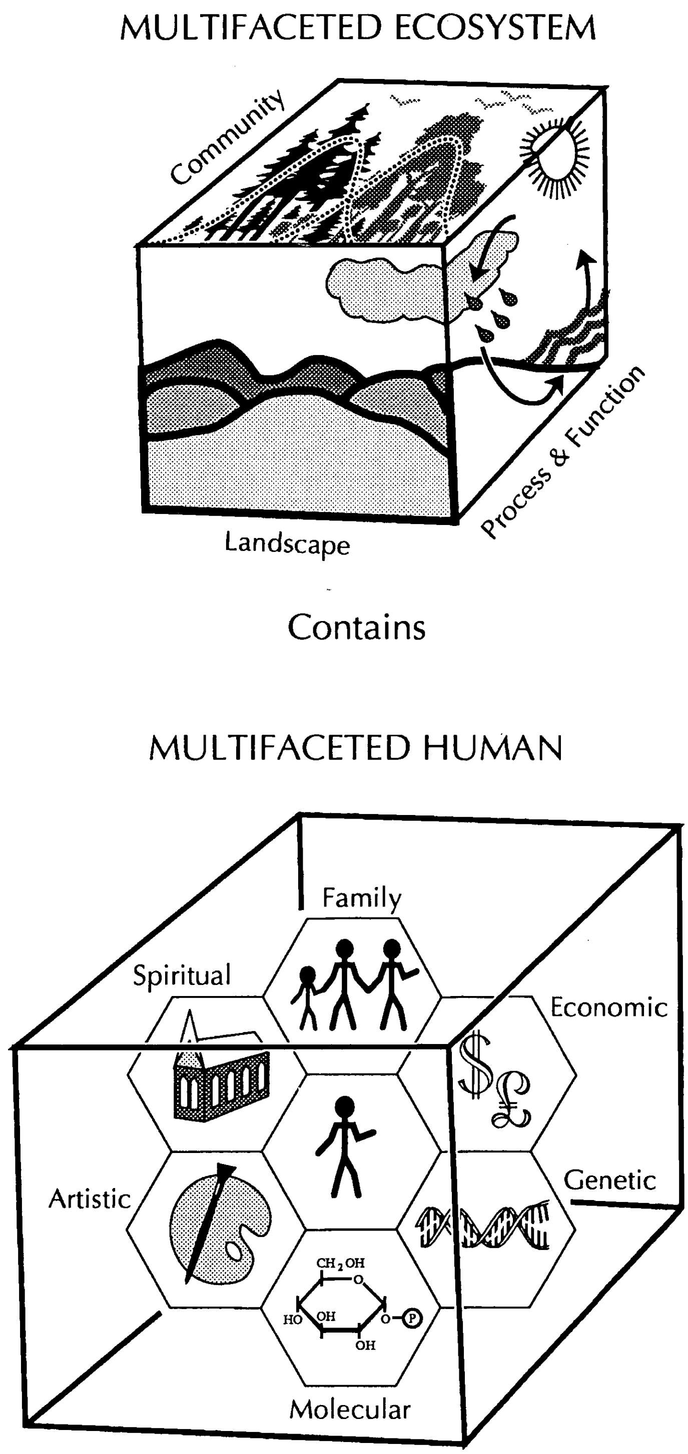


Figure 4. — Not only does an ecological assessment of a system involve particular views of the ecological context in which humans are set (eg. landscapes, process/functional ecosystems, or communities) but the humans that are an integral part of all contemporary ecological systems are themselves multifaceted. Criteria for ecological sustainability must be explicitly stated, but in a system requiring so many different perspectives, they must be employed with flexibility.

some as yet neglected perspective would indeed indicate a lack of sustainability. A manager using the undefined material system as the reference, attempting to achieve complete sustainability in every way, could only expend large amounts of energy and resources to no avail.

Having warned against oversubscription to a "natural" system as the one which is most ultimately sustainable, there are caveats for the obverse position. It is inappropriate to insist on the pristine material system as the reference, but even so this is not a license to ignore the material system and manage for capriciously chosen intensive commodity production. An attempt to maintain an untenable intensive production system is not only doomed to failure, but it is likely to have deeply undesirable side-effects. Just because there is utility in holding a system in a certain state, it does not mean that it is possible or is, in the long run, desirable. As human observers of the material world, we cannot prescribe situations to be sustained that are at odds with the way the material world works.

In the crudest version of this caveat, we humans cannot do the impossible, no matter how much we may desire a particular outcome. Beyond that, long before the impossible appears on the agenda, insurmountable problems will emerge if the intended human manipulation flies in the face of significant material flows. We refer here not to the particulars of the pattern of flow, but to inexorable forces that underlie those patterns, like the truism that water always flows down hill. For example, it is possible to change a pattern of flow as in a large river diversion, but the new pattern cannot defy gravity without unimaginable expenditures of energy spent in pumping. Note that large dams use rather than defy the force of gravity. There are subtle inexorable processes that, if ignored, will bring the best laid plans crashing down. The same applies to plans that may not be the best laid, but are plans to which society is prepared to devote enormous resources anyway. For example, fighting against processes of evaporation by flagrantly introducing yet more water will end, as it did for the Sumerians, with irretrievably salinated soils. California beware; even the greatest economic profits will be unable to bear the cost of restoring a heavily salinated Central Valley to a sustainable condition. Much better to recognize the process of evaporation, and drip water to the plants underground.

Ecological theory suggests that sustainability must involve general systems principles that relate to the tightness of control of the system. In formal analyses of ecological systems (Holling and Ewing, 1971; Holling, 1986) and more intuitive analyses of the course of civilization (Jenkins, 1973) it emerges that systems become fragile unless they have a significant amount of slack. The constant pressure used against inexorable forces of nature in an over-managed system leaves very little slack in the system. The tightness of the control required for system maintenance leaves the system with very little resilience. If a system is to persist a relatively long time, then it must have resilience so that it can come back from inevitable large perturbations, like a hundred year flood, that must come eventually. Thus part of the problem with fighting against principal material flows in management is loss of system slack. This leads to a system that is less sustainable. In a changing society with new demands, loss of slack might also lead to an inability to meet changing demands. A very tightly run timber production system with no slack invites an inability to respond to different timber quotas.

Thus insistence on a capriciously chosen system configuration undermines sustainability, while the converse striving for an undefined "natural" sustainable situation is impractical. Fortunately, there is a middle position that neither aims for an undefined utopia nor a narrowly specified, capriciously set action plan. Management operates on a material system that has a prescribed spatiotemporal extent. Given the infinite possibilities, management also comes from a position that recognizes a given type of system. Necessarily this means that other facets of the system, real as they may be, are put in the background. The most effective efforts to achieve sustainability will be guided by explicit definitions of the system scale and type, and specifications of goals. Action plans will also have to be cast so that the influences they exert line up with the principal material flows in the system, given the definitions and objectives. Rich definitions of the system will be required of course, and they might not fall neatly into conventional types of ecological systems, such as a highly focused population view, or a conventionally specified community perspective. Imaginative solutions are to be found working unlikely interfaces among all sorts of conventional ecologies.

Process And Structure

The caveat about material flows denies a strategy that might otherwise have appeal. Given that perfect sustainability is impossible, it is tempting to consider sustainability to a degree. However, sustainability to a degree is an internally inconsistent notion, it is an oxymoron. Theorists have identified (Allen & Starr, 1982) the need for a clear distinction between system structure and system behavior. This analysis turns on the concepts of rate-dependent dynamics and rate-independent structure. While an ecosystem may recycle nutrients at a rate, it is not an ecosystem at a rate. The ecological system either meets one's definition of an ecosystem, or it does not; "ecosystem" is a state of being not a process of becoming.

In a similar vein, a system is either sustainable or it is not. Sustainability is a state, not a process. Accordingly, degrees of sustainability make no sense. Leave even a subset of processes at work that undermine sustainability, and even if they are slow and are a small part of the material flow, it is only a matter of time before they take the system their own way. The accumulation of salt in Sumerian irrigation was a gradual process. The agroecosystem was almost sustainable. It took a thousand years, with the center of culture being pressed to the northwest from the Persian Gulf, for Sumerian civilization to disappear two millennia before Christ. "Almost sustainable" means "not sustainable." Therefore, seeking sustainability to a degree denies sustainability altogether. Sustainability to a degree is a cruel trick, for it appears an innocuous compromise, but in fact it compromises the entire enterprise.

SYSTEM FRAGILITY AND FREQUENCY CHARACTERISTICS

Relative to robust systems, fragile systems can go wrong in a larger number of ways. Also they will break down more suddenly and with less warning signs. In a fragile system, there is a larger number of local components with narrow tolerances, the failure of any of which would bring the entire system down. Thus a fragile system could be less stable than a robust system, but the message we wish to give is that, if fragile systems are to be as stable as robust systems, they will require more maintenance and planning.

When an ecological system is altered by human activity, it often becomes more fragile. While this fragility may play a role in ecosystem collapse, fragility does not necessarily lead to lack of sustainability. Indeed, the whole discourse of sustainability through management action turns exactly upon how systems greatly changed by man may be maintained. In pristine systems that can quietly evolve and function indefinitely without intervention, the ecologist seeking sustainability is an irrelevance. Sustainability only becomes an issue when one accepts human presence and influence as something that will not go away and with which we must deal.

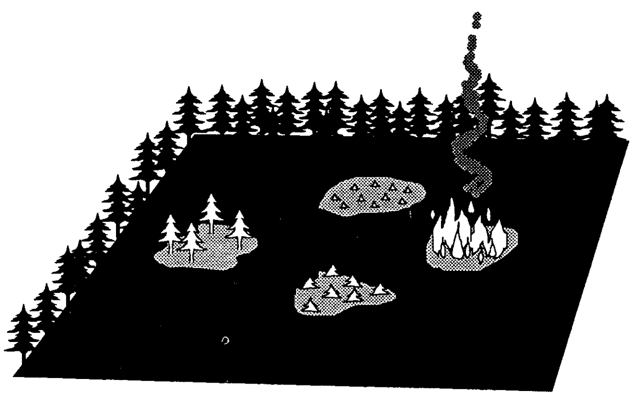
To get a clear picture of the role of fragility, we may learn more from systems that have been greatly modified. Appropriate action that sustains such systems should be able to sustain systems where more of the original fauna and flora are in place. Consider the modern landscape of Greece. It may be beautiful, but it is far from unspoiled, with its topsoil washed into the Mediterranean, it is a clear victim of lack of sustainability. However, the story of how it got to the modern condition is complicated, and is not a matter of the Ancient Greeks failing to sustain their ecosystem. It was more that Greek civilization itself was destroyed from the outside. The role of the Greeks was to make their system fragile and dependent on their civilization. It fell apart when they were not there to maintain it.

With the coming of Iron Age technology, Ancient Greece flourished under wise agricultural management. However, sound as the land ethics of the Greeks may have been, their landscape was importantly altered by their civilization. On many criteria, such as faunal diversity, the system was drastically altered, although Aristotle, who lived early in the process of change, reported unusual amphibia that nurtured their young, and they are found today in the place where he saw them. Development of agriculture caused the significant removal of forests. The second century A.D. traveler, Pausanius, commented on trees when he found them, implying that the primitive forest was essentially gone (Hughes, 1975). However, deforestation appears not to have been the direct cause of the lack of sustainability. The system was surely highly modified by deforestation, but it was not at that time critically damaged. With a full human population to tend the terraces, the agroecosystem was stable; it was not only sustainable, but it was being sustained, and might have been sustained until today but for outside pressures.

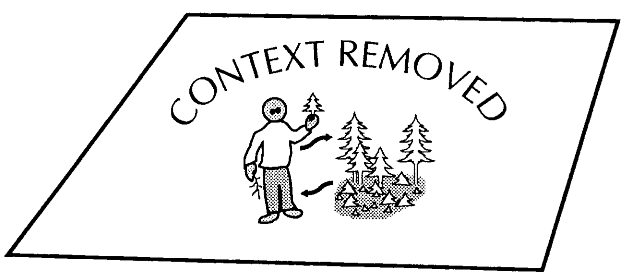
While it did not make the system unsustainable, the human modifications of the Classical Greek landscape had made it fragile. The ultimate destruction of the ancient ecosystem was the consequence of Romans taking slaves and reducing the population. With too few people to tend the fragile landscape, it was washed off into the sea (Heichelheim, 1956). Thus human modification will often lead to fragility, although fragility does not mean that the system is unsustainable. For example an equivalent agroecosystem that was equally fragile did survive, even in the face of the collapse of the central power. Roman agriculture left Italy in a sustainable but fragile condition. Aerial photography by the RAF during World War II revealed landscape patterns of a fully functional farming system well after the decline of Rome (Heichelheim, 1956). The destruction of the landscape of the Italic Peninsula did not occur for a thousand years after the Romans, being caused by "Spanish destructive methods of sheep-breeding after A.D. 1300," (Heichelheim 171, 1956).

The source of the fragility in heavily human influenced systems is twofold, one relating to slow and the other to fast behavior. First, the altered system has lost at least some of its controlling negative feedbacks. This is a matter of the removal of the slowest system components, the reliable context in which the system normally functions. The second source of system fragility is the high frequency behavior that commonly accompanies human system modification. Humans work to maintain the system in a state that they desire. Since that state is not where the system would rest left to its own devices, maintenance requires many fine grain adjustments. Management involves constantly directing the system to where we want it to be.

The two sources of system fragility deserve to be put in more concrete terms. In the example of Ancient Greece, the alteration of the context was the removal of the forest. In less human impacted systems, the context will remain without any particular effort to maintain it. The context of a modified ecosystem needs to be substituted by humans performing the services of the primitive context (figure 5). On the landscape that existed before Ancient Greek agriculture, the forest had been there for thousands of years, maintained by processes normal to forest regeneration, making sustainability a moot point. The problem was not the removal of that forest by agriculture; rather it was an inability of the society debilitated by slaving to continue to perform the functions of the forest, like soil conservation (figure 5). Thus, promoting sustainability is almost never the preservation of a primeval condition, but rather it means maintaining the critical functions of the primeval system, or something like it. Allen and Hoekstra (1992) have argued that management exists to perform the services normally provided by the now removed context. When that is done effectively, the fully serviced, orphaned system functions as it would in the pristine setting. In Ancient Greece, crop cover and holding walls held the soil in place as the forest would have done.



Mosaic of Patches in a Contextual Matrix



Humans Subsidize Local Unit

Figure 5. — In a pristine system, or even one with minimal human intrusion, local ecological systems rely upon a context for services. Perhaps primeval context is a forest matrix that offers a humid nursery for the local patch after a fire, or supplies seeds that will start the process of recovery. The reason management is necessary is to make up for the absent context removed by human resource consumption. The manager offers the services of the missing context. Management is best conceived as contextual.

Now let us expand on the second cause of fragility, the high frequency behavior that comes from humans constantly grooming the system. System modification amounts to moving and keeping the system away from the equilibrium that would prevail if the system were unmodified. The high frequency human activity keeps moving the system up a gradient away from the more primitive condition and counteracts any tendency for the system to regain that condition. In Ancient Greece this was the constant tilling and weeding of woody plants that, left to grow, would lead back to the forest through succession. In structured systems that exist far from equilibrium, like convection cells that make thunder storms, whirlpools, or agroecosystems, energy is dissipated particularly rapidly in the maintenance of the distinctive structure. The distinctive structures in the three examples are the thunderhead, the vortex, and the plowed field respectively. If the high frequency control of the system is suspended, there will be rapid change as the system moves down a steep gradient, sometimes back to the primitive condition, but sometimes to something else (Kay, 1991). In the case of the wholesale abandonment of intensive agriculture in Classical times, the system moved quickly to a condition where the unprotected soil washed away.

Another example of a highly contrived human system that was sustainable, but also collapsed when invaders altered the pattern of exploitation, was the chinampa agriculture of the Aztecs. In that example, the importance of dependence on a viable context is even more apparent than in the Greek case. The Aztec system too had all the properties of fragility and great effort put into persistent local management action to maintain the system. In the tropics, decomposition and high rainfall puts mineral nutrients at risk. Those that are not captured and stored in vegetation flow away in watercourses and end up in the lakes. The Aztecs cleverly recycled those nutrients by scooping them up onto raised beds in marshes. The raised beds were called chinampas and the Aztecs grew crops on them.

By recycling inside the nutrient sink, Aztec farming diverted the flow of energy through humans without long-term depletion. In no way do the Aztecs represent a return to nature, for their system was intensely worked. However, they did form a subtle accommodation with the natural flows of nutrients into the marshes. Unlike the Greek system, deforestation in Mexico not only modified the landscape, but it also made it non-sustainable. Deforestation on the surrounding hills following the Conquistadors, not collapse of the farming system itself, brought the sustainable Aztec system down. The Mexican botanist, Gomez-Pompa has suggested that chinampa farming is the only way to deal with tropical farming and burgeoning populations in an ecologically sound but humane fashion. This suggests the general model of using historically sustainable management, but in the knowledge of how such systems were turned from fragile to non-sustainable.

MANAGING FAR FROM EQUILIBRIUM

Often there will be important turnover rates that indicate different levels of functioning, all of which must be preserved in a sustained system. Some models of grasslands have been able to show the link between cropping and system sustainability by putting carbon into three pools, one with fast turnover, another with moderate turnover rates, and a third which constitutes the long term storage of carbon in the system. Production of human resources often involves cropping the small pool in the highest frequency compartment. The slower compartments replenish the carbon removed. Sustainability involves keeping viable quantities of carbon in the slowest storage compartment. Thus human activity may be local, but it is importantly linked to long term aspects of the system.

The contextual temporal frame of reference for sustainability could be short for microcosms to very long for forests. However, in both cases, relative to the specific time frame in question, sustainability is by definition concerned with the long run. The long run for microcosms may be months, while in forests it is at least millennia. Once again relative to the time frame in question, human management generally involves short term manipulation of the ecological system: perhaps second by second in microcosms to decade by decade in forests. Thus extending long term aspects of the system through sustainability does not fit intuitively with the immediate effects of human manipulation of ecosystems. Local adjustment is used to enhance long term outcomes.

Expressing this in more explicit systems terminology, in efforts to achieve sustainability, dominant aspects of system behavior are made to operate more slowly with longer cycle times through enhancing high frequency, energy demanding activity. That activity fights the tendency to degeneration of the emergent structure. Such energy demanding systems with rapid internal functioning are now recognized as stable energy dissipating structures that exist far from equilibrium. They are the appropriate model for the nature of sustainable systems.

Kay and Schneider (1992) suggest that life itself is exactly such a dissipative structure that requires energy dissipation for its continued existence. Sustainable systems owe their long term persistence to energy dissipation. In the creation of a sustainable system, one does not seek a low level of organization that persists only by being torpid. Rather one seeks stable configurations that may well be demanding of considerable energy inputs and work to keep the system going. Life in general does it by capturing more energy through photosynthesis. It does this using precisely the structure created by the energy dissipation that demands that increased energy capture in the first place. Leaves do not come cheaply, but plants are ruthless in their abandonment of leaves that fall below the compensation point; expensive structure that cannot pay for its structural maintenance has no place in a far-from-equilibrium system. Parts of far-from-equilibrium systems that are not critical to the maintenance of the special configuration are usually pruned away.

In the systems that ecologists wish to make sustainable, it is not a primitive unorganized condition that is sought. Rather the existence of human activity as part of the system is taken as a given. The goal is a system where the human presence bears the cost of its own inclusion by actively maintaining the context. Humans will have to pay energetically for that activity by channeling the energy of the biosphere increasingly through human institutions. All major primitive ecological systems have already succumbed to that diversion of resources, so a program of sustainability of humanly altered systems is the only course left. It is crucial that the energy diverted through society be used to maintain viable ecological regimes that are stable in the long term. It will not be possible to force our way past ecological impasses with the expenditure of material resources. Pumping the ozone smog of our industrial centers up to the stratosphere to replace lost ozone there is so far from being an option that anything of that ilk must be laughed out of consideration.

Our energies, in literal terms, must be pointed toward achieving ecological balances in line with principal flows of the system. Once again lessons are to be learned from generalized far-from-equilibrium systems. A whirlpool dissipates the kinetic energy of the head of water particularly fast. It is through that vigorous expenditure of energy that the whirlpool maintains other very unusual gradients. In a whirlpool, the spinning water allows the water in the middle of the vortex to stand vertically. Of course, water does not usually form vertical surfaces with air, and it is that striking gradient that is maintained by the increased energy dissipation of the flow that characterizes whirlpools (figure 6).

So it is with human activity in agroecosystems and other highly manipulated systems. The energy generated by agriculture is used to pay for plowing the field, thus keeping the site permanently in the first helter-skelter phase of succession. In sustainable systems, energies entrained by system structure must be employed in the careful maintenance of those aspects of the system that perform the entraining. Since water is being held in a vertical wall in the vortex of the whirlpool, the energies entrained by the system are employed in the most efficient manner possible to hold the water in that configuration.

In similar manner, far-from-equilibrium, human-controlled systems may hold the material system in some extremely unlikely and highly contrived configurations, but they must do it in the manner that employs system energies most effectively. Human activity involves highly contrived ecological circumstances, so the pristine natural configuration is irrelevant. However, the energy entrained by human activity must be in line with the principal flows and gradients that emerge in the far-from-equilibrium configuration. Thus human activity directed toward sustainability does not promote the pristine, but it must line up with the natural ecological flows that emerge in anthropogenic settings.

As a way out of finding and holding the system in some unworkable pristine straitjacket, there are moves to declare human-manipulated systems as sustainable so long as they vary within the range of variability manifested by unspoiled primitive systems. In that range-of-variation management strategies demand less precision and look close to achievable, such approaches appear at first sensible and attractive. Of course, the variation of the primitive system is often calculated rather than observed, but that is not the problem with the approach.

The error of managing within ranges of natural variation is in the assumption that natural ranges of variation have anything to do with normal behavior of a system that contains large human populations and the large expenditures of energy that come with modern human occupancy of a site. If the human system is characterized as being a far-from-equilibrium dissipative structure, then the close to equilibrium variation of

Far From Equilibrium – Dissapative Structure

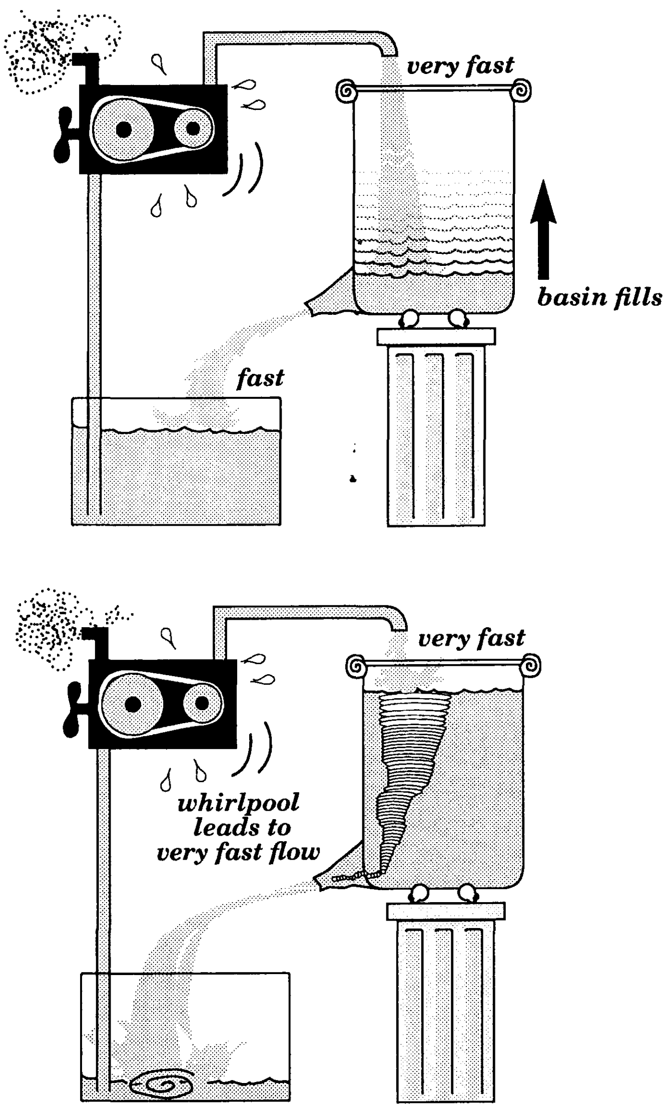


Figure 6. — If water input to a system is great, a large head will be created. Consequently, water pressure will become so great that laminar flow out of the bottom will organize so as to maximize water flow. A whirlpool will emerge that 1) increases flow, and 2) uses the increased energy dissipation to maintain an interface in the vortex.

Homeostasis – Dynamic Equilibrium

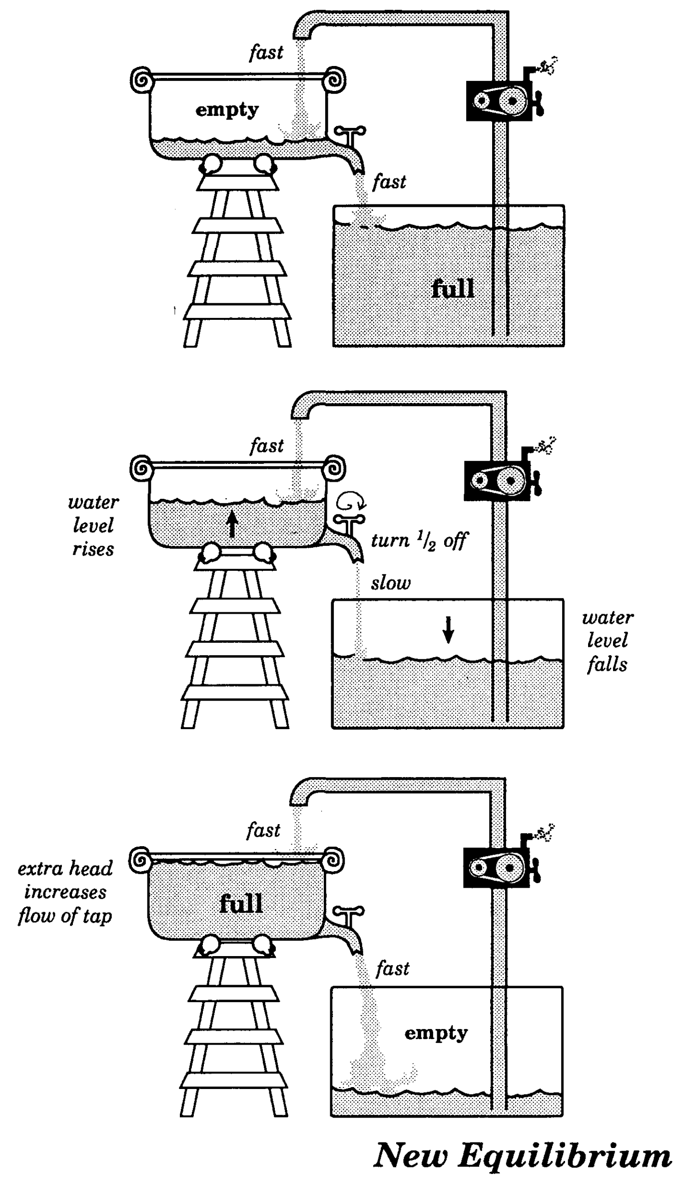


Figure 7. If a whirlpool is analogous to the highly contrived, energy-dissipating system that emerges naturally with dense human populations and anthropogenic manipulation, then a simple tap with a moderate flow, as figured here, is analogous to the pristine ecosystem that pertained before the coming of our species. A simple and unstructured homeostatically balanced flow of water clearly operates over a different range than the spate and the whirlpool. Similarly, highly contrived, energy-dissipating, human-dominated systems would be expected to function normally and healthily outside the range of variation of the pristine ecosystem. Range-of-variation management that uses the pristine system as its benchmark is ill-advised.

a system without humans in it is irrelevant (figure 7). A perfectly healthy dissipative system may exist well outside the range of the primitive system, in fact we would expect that to be the case.

Consider for the last time the whirlpool. The variation of states in which a whirlpool exists occur well outside the range of variability found in a pond with a trickle of water coming in and another leaving. If the pond is the pristine system without humans, the whirlpool is the system with present human populations in it. We cannot abandon agriculture, and the structures that occur therein are held well outside the range of natural variation. It is no response to say that if water flowed in and out of the pond in a torrent, then whirlpools would become a natural part of the system, so whirlpools are natural and therefore cannot correspond to unnatural human influenced systems. It is our point exactly that when more energy goes through a system, far-from-equilibrium structures arise spontaneously and naturally. It is to that far-from-equilibrium nature that we must accommodate. Fields are as natural in a world with five billion people in it, as whirlpools are natural in a spate. Sustainable ecological systems with the present human population in the world will occur naturally well outside the range of ecological systems before agriculture 12,000 years ago.

Thus sustainability is precisely not a matter of a return to some mythical pristine past, nor even an attempt to approach such a condition. Rather it is a process of evolution that is incorporating humans and their institutions into a larger ecological system. In this new ecological arena, the human creature must pay its way in maintaining system structure. This is precisely a cooperative enterprise, for our species does not have the resources or cunning to dominate nature for very long. That is why it is so important for sustainability to work with the major processes in our material setting. Thus efforts to achieve sustainability are neither a journey back to nature nor a dominance over it. In positive terms, it is a new collaboration with nature that will produce something not often seen in the world before.

CONCLUSION

Our arguments with respect to sustainability also apply in large part to the other concepts mentioned at the outset of this paper: biodiversity, ecosystem health, ecosystem management, viable populations, conservation biology, restoration ecology, and global change. All those issues share with sustainability the need to define what we mean with respect to scale and system type. They all require a more sophisticated view than a return to an undefined nature. Elsewhere we have laid out these arguments with regard to restoration ecology (Allen and Hoekstra, 1987). The position we take does not support either commodity exploitation at the expense of environmental preservation nor its opposite. It can help to bring otherwise extreme positions into an arena of rational discussion. Application of the principles we suggest should help bring the virtues of sustainability as seen by environmentalists closer to the value of sustainability that applies to those concerned with commodity production.

Other major civilizations have exploited resources and paid the price. Less grand cultural adventures, that have lasted longer, have been held in the vice grip of what nature can spare: the hunters and gatherers. We as a civilization find ourselves at a cultural watershed where we cannot return to the existence of a noble savage, nor can we persist in the reckless activities of rapacious exploitation. A rapprochement is required; we must take a third path, that of seeking sustainability and positive solutions associated with conservation of viable populations to maintain adequate levels of biodiversity, in the face of global change. It will involve working with processes in the world around us but without the sentimentality of a search for a mythic natural world. We seek something as unromantic as a stable configuration with post-industrial production systems as a working component. Only through hard-nosed decisions mediated by recognition of our special role is sustainability going to be achieved. Without it ours will come crashing down, like 21 major civilizations before us (Moore, 1973).

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Social and Political Issues in Ecological Restoration

Thomas M. Bonnicksen[1]

Abstract — There are four major questions affecting the future of ecological restoration. The first and most serious question is philosophical. Should we attempt to restore ecosystems? Some people want to separate humans from nature because they believe that human intervention is bad or imperfect. They define "natural" as the absence of human influence. They also think restoration should consist of drawing lines around ecosystems and keeping people out. If this philosophy prevails, ecological restoration has no future. The second question is social. What do we want to restore? The third question is scientific. What can we restore? The fourth question is political. Who decides what we will restore? Large-scale restoration projects cannot begin without answering these questions. This paper explores the implications of these questions.

INTRODUCTION

Ecological restoration can trace its roots back to three scientists who had the foresight to see that people can play a constructive role in preserving ecosystems. It began with Aldo Leopold who advocated constructing samples of native plant communities in the University of Wisconsin Arboretum (Jordan 1983a). In his dedication speech for the Arboretum on June 17, 1934, Aldo Leopold said "The time has come for science to busy itself with the earth itself. The first step is to reconstruct a sample of what we had to start with" (Jordan 1983b). Aldo Leopold's son, Dr. A. Starker Leopold, emphasized the importance of using historical ecosystems as a model for future management. He also recognized that Indians played an important role in creating and maintaining those historical ecosystems. For example, as chair of the Committee on Wildlife Management in the National Parks (the Leopold Committee) he helped clarify the goal of national parks. The committee recommended that "the goal of managing the national parks and monuments should be to preserve, or where necessary to recreate, the ecologic [sic] scene as viewed by the first European visitors" (Leopold et al. 1963). A National Academy of Sciences Advisory Committee supported this goal and it was incorporated into the administrative policies of the US Park Service (National Academy of Sciences 1963; US National Park Service 1968). In 1965, Dr. Edward C. Stone published a paper in Science that advocated training "vegetation preservation managers" to carry out the recommendations of the Leopold Committee (Stone 1965). He also developed criteria for educational programs to train these specialists. Such educational programs do not yet exist but the rapid growth of restoration ecology ensures that they will exist in the future.

Today, restoration ecologists pursue a well defined set of professional goals (Bonnicksen 1988a). First, ecological restoration involves repairing ecological communities, or reestablishing them on the same sites if they are destroyed, or replacing those communities with synthetic communities on other sites if the original sites can no longer be used. Second, ecological restoration involves maintaining ecological communities, or protecting communities from unwanted influences so that they can change in desired ways. Third, ecological restoration involves using restoration projects to advance knowledge about ecological communities. In each case, restoration ecologists use the historical or indigenous structure and function of an ecosystem as the model for restoration.

There are four major questions that must be answered to further develop restoration ecology as a field of science and management. The first and most serious question is philosophical. Should we attempt to restore ecosystems? The second question is social. What do we want to restore? The third question is scientific. What can we restore? The fourth question is political. Who decides what we will restore? This paper explores the implications of these questions.

[1] *Professor, Department of Forest Science, College of Agriculture and Life Sciences, Horticulture/Forest Science Bldg., Texas A&M University, College Station, Texas.*

SHOULD WE RESTORE ECOSYSTEMS?

Some people believe that nature is sacred. This belief reifies nature, or converts nature in the abstract to nature as a real thing. Since nature is a sacred thing, adherents to this philosophy define humans as unnatural. They exclude humans from nature. As Frankena (1979) points out, they believe that what is natural "is right and the virtuous." They also believe that humans are inherently destructive and that beauty only exists in dehumanized landscapes, so nature must be left alone. Such misanthropic beliefs form the foundation of many environmental organizations. Consequently, they believe that restoration should only protect ecological communities from human influence. Supporters of this philosophy assume that "nature" will restore itself without human help. What "nature" creates is not important, only the absence of human influence is important. Their watchword is to "let nature take its course," despite the potential for sacrificing other values. In short, if the "nature as sacred thing" philosophy dominates resource management then ecological restoration has no future. Supporters of this philosophy would answer the question posed above by saying that people should not restore ecosystems.

The opposing philosophy accepts humans as part of nature. Supporters believe that ecological communities should serve human needs, but that the needs of other beings must be considered. They believe that excluding humans from nature is an unnatural change that would ultimately destroy ecological communities. Examples include the rapidly deteriorating ancient forests within national park and wilderness areas throughout the United States. They argue that the removal of humans as a natural force will begin unnatural chains of events and create new and artificial ecological communities. If this philosophy dominates resource management then the future of ecological restoration is assured. Supporters of this philosophy would answer the question posed above by saying that people should restore ecosystems.

Since restoration ecology uses historical or indigenous conditions as a model for restoring ecological communities, it includes an implicit recognition of the effects of past human use. Restoration ecologists point out that humans played a natural and decisive role in guiding evolutionary change for at least 2.6 million years. Humans used tools and fire to help shape and maintain plant and animal communities throughout the world. Thus restoration ecologists use the past, including historical human influences, as a model for the future. On the other hand, people who believe that humans are not part of nature place no value on historical conditions. Instead they value the abstract idea of "letting nature take its course." To them, future ecological conditions are "good" no matter what changes occur. The remainder of this paper assumes that people will accept their role in nature and that restoration ecology will grow in importance.

WHAT DO WE WANT TO RESTORE?

Ideology

Most of the legislation creating US national parks, wildernesses and reserves refer to the goal of maintaining natural conditions (Bonnicksen and Stone 1985). Regulations governing Canadian national park and wilderness areas also refer to maintaining the "natural state" (Parks Canada 1983). However, naturalness remains undefined. Some people advocate "letting nature take its course," others advocate restoring historical conditions, still others argue that everything is natural. In each case the definition of naturalness seems clear to advocates, but ambiguous to managers.

Ambiguous definitions of naturalness provide a false sense of understanding that often leads to useless debates over ideology. For instance, the US National Park Service argues strongly for the "let nature take its course" ideology. They even allowed huge and unprecedented wildfires to burn 50 percent of Yellowstone National Park in 1988 because of this ideology (Bonnicksen 1989). In contrast, however, the Park Service also uses logging, burning and mowing to remove native herbaceous plants, shrubs and trees for aesthetic purposes in other national parks. Why is it unnatural for native shrubs to invade a meadow in one park, and natural for human-caused wildfires to burn large areas in another park? Unfortunately, there are no criteria for making this choice, so they are ad hoc decisions made by local Park Service officials. This inconsistency shows that ambiguous ideological statements cannot serve as useful goals for resource management.

The Canadian Park Service avoids such inconsistencies by requiring an approved vegetation management plan for all units of the system. These plans emphasize the goal of restoring or maintaining "ecological and historical integrity" that includes the effects of past use by native people (Parks Canada 1983). In short, instead of debating the meaning of naturalness or "letting nature take its course," Canadians manage their parks. They decide what they want in each park and then they find the best way of getting it. This is what the Leopold Committee recommended for US national parks back in 1963. Dr. Leopold reiterated this recommendation in a letter dated June 9, 1983, (his last written statement on restoration). He told the Park Service that restoration issues "involve judgment, followed by action" and that such issues "are not resolved simply by allowing natural ecosystem processes to operate.'" He concluded by saying that "I still espouse the idea of active manipulation." The US Park Service still has not carried out the Leopold Committee recommendations. In contrast, the Canadian Park Service took the recommendations seriously and applied them successfully.

Vegetation management plans for Canadian national parks must conform to a set of overarching principles. First among these principles is the prudent goal of "minimal interference" (Parks Canada 1983). Managers can manipulate park resources when neighboring lands, public health and safety, and park facilities are threatened. They can manipulate resources to "restore the natural balance" or to substitute human action for "a major natural control" that is absent. They also can interfere in natural processes to protect rare and endangered plants and animals. Most important, they can manipulate resources when "the population of an animal species or stage of plant succession which has been prescribed in the objectives for a park, cannot be maintained by natural forces." Unlike the United States, the Canadian people decide what they want to restore in their parks in unambiguous terms. Then they provide their Park Service with the flexibility and resources to achieve the goal.

Restoration Goals

Goals define what should be done. They provide an idealized sense of direction for restoration projects. There are three broad categories of restoration goals: structural, functional, and wholistic (Bonnicksen 1988a). Structural goals concentrate on the parts of an ecological community, functional goals concentrate on processes and wholistic goals include both.

Structural Goals

Structural goals use physical features to describe the desired future condition of an ecological community. The type of function that is used to produce the desired condition is less important because function is a means to an end, not the end itself. Unless prohibited, chain saws, prescribed fire and chemicals are legitimate means to restore the structure of the ecological community. The Canadian Park Service, for example, must use restoration techniques that "will duplicate natural processes as closely as possible" (Parks Canada 1983). Nevertheless, a historically authentic function, such as the use of old agricultural practices, may be essential for perpetuating an ecological community in some historical structural condition. Structural goals include: 1) the biotic diversity goal, 2) the special species goal, 3) the special community goal, and 4) the cultural landscape goal (Bonnicksen 1988a, 1990).

1. The biotic diversity goal focuses on the number and kinds of "things," such as native species, in a particular area. The arrangement of "things" in space and time may also be an essential attribute of biotic diversity. Biotic diversity is only used for ecological restoration when it is based on a historical or indigenous model.
2. The special species goal focuses on favoring one native species over another. Animal or plant species that are identified as more important to humans than other features of the ecological community, such as the northern spotted owl, are known as special species. Special species include those that are threatened with extinction, outstanding specimens of the total population, or species that are highly valued by some social groups for other reasons.
3. The special community goal focuses on restoring historical associations of native plants and/or animals. Past human activities may or may not have been important as the dominant force responsible for creating a special community. Society may value special communities, like special species, because they are rare, spectacular, or important to a particular social group. Special communities can also serve as historically accurate ecological settings for cultural artifacts.
4. The cultural landscape goal focuses on restoring culturally derived associations of plants and/or animals. Cultural landscapes are ecological communities that resulted from, or coexisted with, human habitation. Artifacts, such as buildings and quarries, may or may not be important elements of the landscape. Cultural landscapes range from those that appear unoccupied, but were maintained by aboriginal peoples, to intensively managed agricultural landscapes.

Functional Goals

Functional goals do not include the structure of ecological communities because function, such as wildfire and plant succession, are more important. Thus any structure is acceptable if it is created by, or sustains, the desired function. What is important here is not the authenticity of the structure but the authenticity of the function. Functional goals include: 1) the unimpeded processes goal, and 2) the analogical processes goal (Bonnicksen 1988a, 1990).

1. The unimpeded processes goal is designed to perpetuate a desired historical function rather than the structural attributes of an ecological community. It is laissez-faire or passive management. Humans simply observe historical non-human forces at work. These forces are allowed to operate freely despite alterations to the structure of an ecological community. This is an abstract goal because the presence or absence of a function, such as wildfires, determines success. However, structure and function are inseparable. Therefore, in order to sustain the historical function, the starting structure of the community must approximate past conditions, or the condition that would have existed without degrading influences.
2. The analogical processes goal focuses on reestablishing a desirable historical function, such as plant succession or the cycling of soil water reserves, or eliminating an undesirable function,

such as soil erosion. The structure of an ecological community can be modified as needed to support the desired function.

Wholistic Goals

Wholistic goals consider both the structure and function of an ecological community. Wholistic goals include: 1) the controlled evolution goal, and 2) the synthetic community goal (Bonnicksen 1988a, 1990).

1. The controlled evolution goal is based on an evolutionary perspective that accepts changes in the structure and function of ecological communities. However, selected attributes of these communities are controlled by keeping them within the limits that society finds acceptable and desirable. The starting point for controlling evolutionary change can be the historical condition or an estimate of what the current ecological condition may have been without degrading influences.
2. The synthetic community goal uses structure and function as equally important measures of authenticity. Synthetic communities resemble other ecological communities that may have been lost. It means starting from nothing and knowing enough to include the relevant parts of the system, along with essential interconnections and ecological processes.

WHAT CAN WE RESTORE?

Restoration ecologists follow a systematic procedure for carrying out restoration projects to achieve a goal. Underlying this procedure is the principle that a historical or indigenous model, or reference ecosystem, is always used as the target for restoration. Standards for assessing the success of restoration come from measurable attributes of the reference ecosystem. Whenever possible, restoration practices mimic the historical or indigenous processes that operated to maintain the reference ecosystem. Thus restoration usually involves 1) selecting a reference ecosystem and documenting the difference between current conditions and the reference ecosystem; 2) developing measurable standards from the reference ecosystem that serve as a target for management; 3) documenting historical processes and developing restoration practices that mimic the effects of those processes; 4) projecting the consequences of management to improve restoration practices before intervention; 5) monitoring the results of intervention and revising management practices to ensure success. The first three steps in this restoration procedure determine what can be restored.

Reference Ecosystems

The most important decision in ecological restoration is selecting the reference ecosystem. Since future changes in an ecological community will always be dictated by its starting structure, the starting structure must accurately represent the reference ecosystem during the historical period. Restoration can only proceed after the reference ecosystem has been documented using measurable standards of authenticity (Bonnicksen 1990, 1988a, 1988b; Bonnicksen and Stone 1985, 1982a, 1982b).

The historical structure of a reference ecosystem can be documented by several means. Sources of evidence include archeological materials, historical accounts, old photographs, early land surveys, sediment analysis, pollen analysis, soil maps, climate maps and existing vegetation. For example, pollen analysis was used to describe the vegetation surrounding Fort Necessity, Pennsylvania, as it appeared in 1754 (Kelso, Karish and Smith 1993). The fort was built by Lt. Col. George Washington to defend against a French-led Indian force. However, using existing vegetation is the most direct and accurate approach for reconstructing historical conditions (Bonnicksen and Stone 1982b; Henry and Swan 1974).

Using existing vegetation to reconstruct historical conditions involves rolling woody plants back in time and developing a description of the historical structure (Bonnicksen and Stone 1982b, 1981). Spatial patterns of seral stages, and non-woody vegetation, which comprise the vegetation mosaic are also important structural features. Differences between the current and historical conditions are then used to describe a target condition for restoration.

This approach to documenting a reference ecosystem provides a sound scientific basis for management. For example, the ancient mixed-conifer forests of the Sierra Nevada mountains of California are seriously degraded due to a century of fire suppression and the elimination of Indians. Today's forest is thicker and older than the ancient forest. Shrubs, oak trees and wildflowers are less abundant, and white fir is gradually becoming the dominant species. These changes present a serious threat to wildlife and the biological diversity of the forest. Unfortunately, many people that advocate restoring these forests use unscientific images as a guide for restoration.

A persistent myth about ancient mixed-conifer forests is that they were composed mostly of large old trees. Old trees were present, but young and middle-aged trees, shrubs and wildflowers also were a prominent part of the ancient forest. Studies by Bonnicksen and Stone (1982b, 1981) within a 2042-hectare watershed in Kings Canyon National Park showed that aggregations of sapling size trees covered 17 percent of the watershed when it was an ancient forest. Aggregations of pole-size trees covered 15.4 percent of the watershed, and 19 percent was covered by shrubs. Only 17.6 percent was covered by aggregations of large old trees when it was an ancient forest. The remainder of the watershed consisted of meadows, gaps, tree seedlings and rocks. Therefore, the ancient forest was a mosaic of vegetation, not a dense forest of large old trees. Such scientific studies are essential to prevent using myths in the description of reference ecosystems.

Sometimes existing vegetation cannot be used to reconstruct historical conditions. The 39,000 acres of cutover redwood forest added in 1978 to Redwood National Park, California, is a dramatic example. Not only were these lands clear-cut by logging companies, but they were seeded to Douglas-fir and hand planted to redwood before being added to the park. Fortunately, uncut old-growth redwood forests, which have changed little over the past century, surround these cutover lands. Thus relict native ecological communities, such as these uncut redwood forests, are especially valuable as reference ecosystems for restoring severely damaged communities.

Restoration Standards

If restoration goals define what should be done then standards provide a way of determining how well it was done. Standards are equivalent to objectives because they provide measurable targets that are supposed to be achieved in a specific period. They lead toward goals, but they are not the equivalent of goals. Standards are also imperfect representations of the reference ecosystems they document. Everything in an ecosystem cannot be measured nor can the measurements themselves be flawless. Thus standards represent the goal and measure how successfully it has been achieved. In short, standards define what can or will be restored.

Restoration standards can be illustrated with the controlled evolution goal. This goal requires taking repeated measures of both the structural and the functional attributes of an ecological community and comparing them with predetermined quantitative standards. Monitoring pinpoints undesirable changes at an early stage so that manipulations can be used to guide the ecological community back to the desired trajectory.

Structural standards for the controlled evolution goal could include the presence, number, size, vigor, genetic composition, and horizontal and vertical arrangement of species. The pattern characteristics of mosaics of plant aggregations that comprise ecological communities may also be important standards, such as random, uniform, or clumped patterns, and their intensity and grain. Several diversity indices also measure evenness in the distribution among species, including soil biota and plant aggregations. Measures of microbial biomass and the insularity of communities also may be critical to sustainable management. Functional standards for the controlled evolution goal could include fire cycles and burning patterns, microsymbiont effectiveness, biomass productivity, and biogeochemical and soil nutrient cycling indices. The standards used to guide restoration will depend on what is feasible and desirable in particular situations. The problem is finding the mix of standards that come closest to representing the goal.

Restoration Practices

Restoration ecologists recognize that ecological communities are too complex to either completely understand or fully control. Therefore, restoration will always be imperfect. Nevertheless, restoration can help to counteract the continued and widespread degradation of ecosystems. Like a doctor of medicine, a restoration ecologist does not have to fully understand how an organism or an ecosystem works to restore it back to health (Jordan 1983c). Thus restoration ecologists are more like gardeners than engineers because they can only guide ecological communities toward a goal (Bonnicksen 1988a).

The first step toward developing effective restoration practices is to better understand the historical processes that led to, and sustained, the reference ecosystem. Changes in most ecological communities are driven by periodic disturbances. For example, forest aggregations can be traced back to some destructive event, such as fire or wind throw. Others can be traced back to insect outbreaks and the effects of root pathogens. Thus the types of disturbances that affect particular ecological communities must be determined, also their scale, frequency, intensity and impact.

It is important to know the historical scale and frequency of disturbances. The size of the area undergoing restoration, and the period for assessing past conditions, must fit the scale and frequency of disturbances in the reference ecosystem. Some types of destructive events can cover a wide area, such as crown fires, hurricanes and avalanches, producing correspondingly large aggregations. Such large-scale disturbances usually occur infrequently. Small-scale disturbances may involve single tree falls or frequent light surface fires that open gaps in a forest canopy and create small aggregations. Thus the size of the area and period for assessing historical conditions must be larger for communities affected by infrequent large-scale disturbances than for communities affected by frequent small-scale disturbances.

It is also important to know the agent responsible for disturbances in a reference ecosystem. In the northern Rocky Mountains, for example, many open ponderosa pine forests appeared untouched when first seen by European settlers, but they were kept open by an interaction between frequent Indian burning and lightning fires (Barrett and Arno 1982). The elimination of Indians and the suppression of lightning fires resulted in succession toward more shade tolerant tree species, thickening understory vegetation, heavier fuel accumulations, and a concomitant increase in the potential for massive wildfires. Without Indian burning, lightning fires cannot be relied upon to restore these forests because they occur too infrequently to prevent fuels from building up and causing catastrophic fires (Bonnicksen 1990). Since the agents of disturbance are gone the effects of burning must be simulated using either prescribed fire or mechanical methods. As Dr. Leopold said in his 1983 letter to the US Park Service, "A chain-saw would do wonders."

Regulations to control air pollution, and the reluctance of Congress to appropriate funds for prescribed burning, are serious barriers to restoration. As a result, future restoration efforts may require a greater emphasis on mechanical methods. Mechanical methods may also be needed to harvest resources that can be sold to pay for restoration. For example, old growth forests cannot be sustained unless a continuous supply of young trees is produced to replace the old trees that die. In the past, Indian and lightning fires created the openings in the forest needed to regenerate young trees. Today restoration ecologists can mimic the effects of these fires by creating similar openings with carefully managed logging.

The best way to mimic the effects of ancient fires is to cut groups of trees in a way that ensures that all essential ages of trees and associated vegetation exist in the forest mosaic. The sizes of openings, and the optimum mixture of old growth and other stages of tree growth, will vary depending upon local ecological conditions. Restoration cuts could maintain the same proportion of old growth in the future forest that existed in the ancient forest. Thus decadent old growth cut in one part of the mosaic would be replaced with renewed old growth as the trees grow larger in another part. Thus dramatic stands of old growth would float around the future landscape in the same way that they floated around the ancient forest landscape. Using logging as a substitute for Indian and lightning fires would sustain old growth, increase biodiversity, provide a secure economic future for local communities and pay the cost of restoration.

WHO DECIDES?

Since restoration goals are value judgments that describe the preferred condition of an ecological community, goal-setting is a social or political decision, not a technical or professional decision. The courts provide an inappropriate forum for setting restoration goals because they address specific cases that usually involve an alleged violation of law. Similarly, resource managers are no better qualified than the public to choose goals for restoration projects. Scientists possess essential technical knowledge, but they are even less qualified than managers to make value judgments for the public. Therefore, restoration goals are best set through legislation or cooperative decision-making.

Since most legislation is vague, cooperative decision-making should be used whenever possible to formulate restoration goals and establish standards for management. Cooperative decision-making involves managers and the affected public, or stakeholders, working together as partners to formulate and carry out decisions (Bonnicksen 1993). It is based on the idea that it is wiser to include affected groups in making decisions than to try to guess how they may react. It is also wasteful to ignore the knowledge possessed by people who spend their lives dealing with an issue. Cooperative decision-making also discourages conflict and fosters teamwork. Thus it is the best method for setting restoration goals because it provides opportunities for stakeholders to exchange information, weigh arguments and make the tradeoffs that are needed to reach acceptable compromises.

CONCLUSIONS

Ecological restoration requires a new perspective in resource management. It requires thinking about how to put back together the ecological communities that analytical studies have taken apart. It requires working with a variety of disciplines so that the essential parts of a community can be reassembled and sustained. It also requires working with the public to select restoration goals and the standards needed to measure success in achieving those goals. Finally, restoration requires accepting the constructive role of humans in nature and working cooperatively to restore and maintain ecological communities.

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Restoration of Southwestern Ponderosa Pine Ecosystems With Fire

Stephen Sackett, Sally Haase, and M.G. Harrington[1]

Abstract.— Heavy grazing and timbering during settlement by Europeans, and a policy of fire exclusion shortly after caused extensive structural and compositional changes to the southwestern ponderosa pine ecosystem. These changes have resulted in forest health problems, such as increased insect and disease epidemics, reduced wildlife habitat, and a serious wildfire hazard. Prescribed burning can reduce heavy fuel accumulations, provide adequate sites for natural regeneration, thin dense stagnated thickets, and create an edaphic and stand environment conducive to better forest health and productivity. Although presettlement conditions may never be restored, forest condition and health can be improved by means of prescribed fire.

Prior to European settlement, the composition and structure of southwestern ponderosa pine (*Pinus ponderosa*) forests were quite different from today. The open, park-like presettlement stands, characterized by well-spaced older trees and sparse pockets of younger trees, had vigorous and abundant herbaceous vegetation (Biswell and others 1973, Brown and Davis 1973, Cooper 1960). These forest conditions were maintained by naturally-ignited fires burning on a frequent, regular basis in light surface fuels of grass and pine needles. Light surface fires burned at intervals averaging less than 10 years and as often as every 2 years (Dieterich 1980, Weaver 1951). Warm, dry weather common to the Southwest in early summer, the continuity of grass and pine needles, and the high incidence of lightning caused this short fire interval. Light surface fuels built up sufficiently with the rapid resprouting of grasses and the abundant annual pine needle cast. Large, woody fuels in the form of branches or tree boles, which fall infrequently, rarely accumulated over a large area. When they were present, subsequent fires generally consumed them, reducing grass competition and creating mineral soil seedbeds which favored ponderosa pine seedling establishment (Cooper 1960). These effects created an uneven-age stand structure composed of small, relatively even-aged groups.

The decline of the natural fire regime in southwestern ponderosa pine ecosystems started with extensive livestock grazing in the late 19th century when fine, surface grass fuels were reduced (Faulk 1970). Subsequently, ponderosa pine regeneration increased because of reduced understory competition, less fire mortality, and more mineral seedbeds (Cooper 1960). In the early 1900's, forest practices, primarily fire suppression, further reduced the ecological role of fire. These practices lead indirectly to stagnation of naturally regenerated stands and unprecedented fuel accumulation (Biswell and others 1973).

Stand stagnation has been reported on tens of thousands of acres throughout the Southwest (Cooper 1960, Schubert 1974), and still persists where natural or artificial thinning has not taken place. Sites with dense thickets are not only unproductive but also represent a severe wildfire hazard.

For several decades, trees of all sizes have been showing signs of stress with generally poor vigor and reduced growth rates (Cooper 1960, Weaver 1951). This condition is likely due to reduced availability of soil moisture caused by intense competition and by moisture retention in the thick forest floor (Clary and Ffolliott 1969). Thick forest floors also indicate that soil nutrients, especially nitrogen, may be limiting because they are bound in unavailable forms (Covington and Sackett 1984, Covington and Sackett 1992).

During the last 75 to 100 years with a greatly altered natural fire cycle, unprecedented and unnaturally large amounts of surface and ground fuels have accumulated (Kallander 1969). Sackett (1979) reported average loadings of naturally fallen fuels

[1] *Stephen S. Sackett and Sally M. Haase are Research Foresters with USDA Forest Service, Pacific Southwest Research Station, Riverside, California, and Michael G. Harrington is a Research Forester with the Intermountain Research Station, Missoula, Montana.*

at 22 tons per acre for 62 southwestern ponderosa pine stands. Harrington (1982) verified the heavy fuel loadings with an average of 34 tons per acre in southeastern Arizona.

Forest floor fuels can accumulate to 9 tons per acre in sapling thickets and to more than 50 tons per acre on old-growth sites. Annual fuel accumulation on those sites can range from 0.6 to more than 3.5 tons per acre (Sackett and Haase in preparation). The decomposition rate (k) (Jenny and others 1949) in these forests is extremely slow, resulting in the large buildup of forest floor fuel. K values range from 0.076 to 0.059 and 0.050 for sapling, pole, and old-growth substands respectively (Sackett and Haase, in preparation).

Large, woody fuels, formerly uncommon in the Southwest, now average about 8 tons per acre but are frequently found at twice that loading (Sackett 1979). Much of the heavy fuels have accumulated in sapling thickets, creating an even more severe hazard.

A combination of heavy forest floor fuels and dense sapling thickets, coupled with the normally dry climate and frequent lightning- and human-caused ignitions, has resulted in a drastic increase of severe wildfires in recent decades (Biswell and others 1973, Harrington 1982). Data summaries from USDA Forest Service Smokey Bear Reports show (fig. 1) a great increase in the number of acres burned by wildfire since 1970. Of all the years since 1915 with over 100,000 acres burned, almost 70 percent occurred between 1970 and 1990, indicating a worsening problem.

A final characteristic of the present southwestern ponderosa pine stands is the sparseness of understory vegetation, including pine regeneration. The thick organic layers and dense pine canopies have suppressed shrubby and herbaceous vegetation (Arnold 1950, Biswell 1972, Clary and others 1968). In openings left by overstory mortality where pine regeneration is desired, conditions for establishment are poor, again because of the deep forest floor (Sackett 1984, Haase 1981). This condition has reduced the wildlife, range, and timber production value of these forests and has generally resulted in minimal biodiversity.

REESTABLISHING FIRE TO ITS NATURAL FUNCTION

Because natural fire was the major presettlement factor in shaping and maintaining southwestern ponderosa pine ecosystems, it is logical to consider applied fire in a management scheme to relieve the serious problems that plague these forests due to years of fire exclusion. Fire has been used in the southeastern United States for many years to maintain pine in an environment that would naturally shift to hardwoods. It is also recognized as the key factor in keeping healthy, seral ponderosa pine stands from becoming stressed, wildfire-prone, mixed-conifer stands in the interior West (Arno 1988).

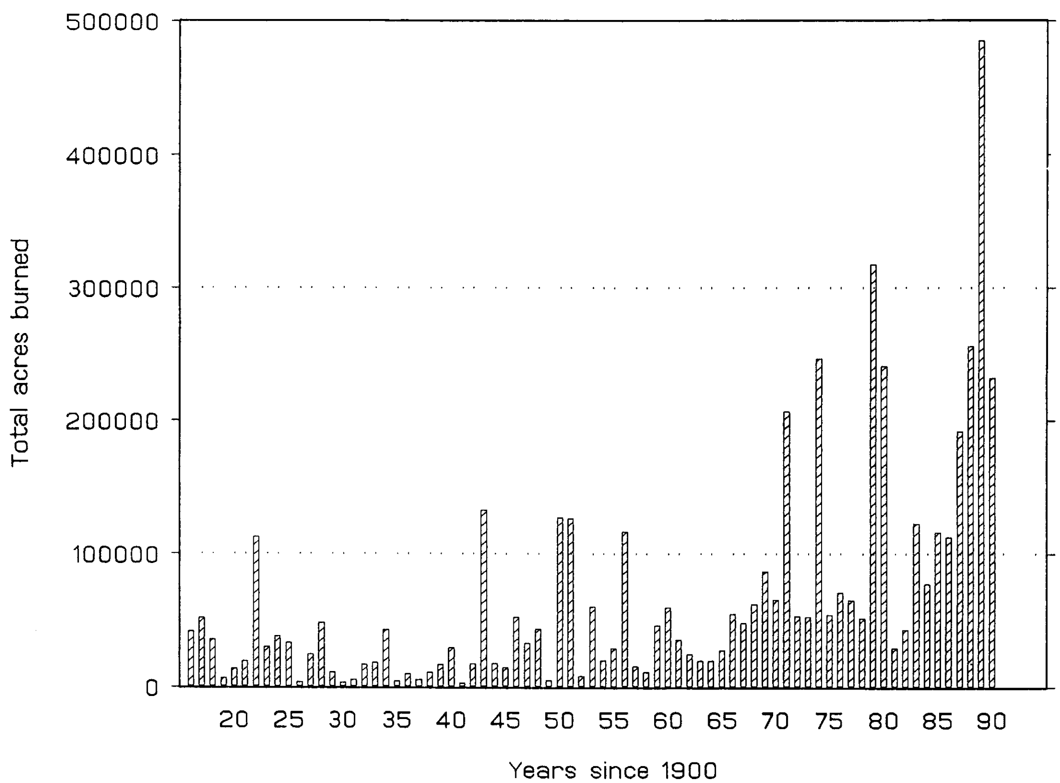

Figure 1. — The total number of acres burned by wildfires in Arizona and New Mexico from 1916 to 1990. Data obtained from USDA FS Smokey Bear Reports.

The hazardous conditions described make the wide-spread application of prescribed fire difficult and costly. Heavy fuels and dense stands can create control problems, and overstory mortality from excessive above- and below-ground heating is a certainty. An acceptance of these risks and economic losses, however, seems necessary in the short-term if ecological sound management is sought.

For prescribed fire to be an effective natural force in ponderosa pine, it must be applied at regular intervals — as were presettlement fires — and, most importantly, its use continued once started (Sackett 1975, Sackett 1980). Too often, fire use means one treatment with no consideration for future applications. Frequently, the fire hazard remains high after one application because of the addition of fire-killed fuels (Harrington 1982). To be effective, maintenance burning is necessary to keep recurring fuels to a minimum (Davis and others 1968, Gaines and others 1958, Harrington 1981, Sackett 1975, 1980). Generally, repeat burns in light, needle fuels are easily manageable.

Historically, natural fire in presettlement times probably burned during the period just after the spring dry season, just as the first storms developed announcing the start of the monsoon season in the Southwest. These first storms are typically dry, and the accompanying lightning could start numerous fires. With the increased fuels and dense stands of today, spring prescribed burning would be unwise because the most severe part of the wildfire season is imminent. Fuel reduction and overstory thinning have to be done in stages over time. Fall, then, becomes the season of choice when weather and fuel moisture conditions are more moderate, and high winds not as likely. Once stands have been conditioned over a period of years of regular, close-interval burning, spring burning becomes a more realistic option to lengthen the burning season. Summer prescribed burning can also be successful as an alternative to fall when conditions are often poor for burning (Harrington 1981, 1987).

The real premise of prescribed fire in ecosystems that naturally had frequent fire, is to provide for interval burning on a rotation that promotes healthy, wildfire-resistant, productive forests.

TWO CASE STUDIES

In 1976 and 1977, companion studies were established near Flagstaff, Arizona, to investigate the effects of reestablishing fire in ponderosa pine. Study areas were established on the Fort Valley Experimental Forest in 1976 on a basalt soil site now referred to as Chimney Spring. One year later, a research site was established on the Long Valley Experimental Forest on a limestone/sandstone soil now known as Limestone Flats (Sackett 1980).

The initial objective of these sister studies was to determine a burning interval that would adequately manipulate fuels and stocking of a post-settlement ponderosa pine stand so that it would survive a stand-replacing wildfire. The study objective assumed the need for reestablishing fire as a natural, necessary function in southwestern ponderosa pine ecosystems. The primary focus of the study was to deal with the most apparent problem in the pine ecosystem, that of heavy, unnatural forest floor fuels.

Initially, both Chimney Spring and Limestone Flats had essentially the same forest floor fuel loadings, 15.2 and 15.7 tons per acre, respectively. Limestone Flats had more than 16 tons per acre of woody fuels greater than 1-inch diameter, whereas Chimney Spring had about 7 tons per acre (Sackett 1980). The importance of fuel moisture on fuel consumption and fire effects was demonstrated when all the interval burning treatment plots (1-,2-,4-,6-,8-, and 10-year) were initially burned in 1976 at Chimney Spring and in 1977 at Limestone Flats. A dry summer and fall in 1976 caused fuel moistures to remain low, the initial burn at Chimney Spring was therefore done at night when the humidity was higher and temperatures were lower. As a result, 63 percent of the forest floor fuel was consumed, as was 69 percent of the woody fuels greater than 1-inch diameter. In contrast, the Limestone Flats area was burned in fall 1977 after an extremely wet summer that continued into fall. As a result, only 42 percent of the forest floor material and 44 percent of the woody fuels greater than 1-inch diameter were consumed.

FIRE BEHAVIOR

Annual burning (1-year interval) is a rotation established to determine the feasibility and effects of such frequent burning. We have found that annual burning is not possible, not because of insufficient fuels to carry a fire, but because weather and fuel conditions in certain years are too damp. Windspeeds compensate sometimes for damp fuel conditions, allowing fire to carry in these light fuels.

Repeat burns every 2 years are generally more successful because of the slightly heavier fuel loads. Again, marginal weather in fall makes biennial burning dubious. Biennial burns may be effective in wildland/urban interface situations.

The most effective prescribed burning rotation observed at Chimney Spring is the 4-year interval. Although this rotation has burned well each interval and has not damaged the healthy overstory, it is not certain whether optimal weather has occurred synchronously with those years or if 4 years is the optimum burning cycle. This rotation appears to be effective because of the consistent ease of carrying out the treatment in keeping fuels to a minimum. To test whether each rotation meets the objective of reduced wildfire hazard, heading fires are ignited for each burn to determine if the stand is protected from a wildfire situation. To date, 4-year-rotation burns have done well to meet the objectives.

Six-year burning rotations begin to accumulate fuel loads that stretch the fire intensities to an upper limit that may cause undesirable damage to the residual overstory. The two

6-year-intervals burned have yielded contrasting results. But, fuel loads are such that, under severe fall conditions, fires could be a control problem and lead to undesirable fire effects.

This high fire intensity problem occurred in the fall of 1992, which was warm and dry, and frequently windy. With 42 rainless days, the heavy, woody fuels had thoroughly dried out from the summer monsoons. Rotations of 1-,2-,4-, and 8-years were burned at the same time. All except the 8-year rotations burned well and did not result in excessive crown scorch. However, 8 years of fuel accumulation (5 tons per acre), low fuel moisture (4 percent to 6 percent), low humidity (21 percent), and only moderate winds, resulted in a .1-chain-deep strip heading fire that heavily scorched most of the pole and smaller size trees in a one-half-acre area. Continuing with heading fires would have completely devastated the entire plot. By allowing the fire to continue as a backing fire well into the night, the 8 years of fuel accumulation was safely consumed. The severity of this 8-year-interval burn points out clearly the need for continuous, short-interval burning in an ecosystem so demanding of fire for its existence.

The only test of 10-year burning intervals occurred in 1986. Fall conditions were too damp for effective fire spread. Forest floor fuel had accumulated in 10 years to more than 7 tons per acre, so experience from the 8-year burns would suggest severe overstory damage would have occurred if conditions had been warm and dry.

REGENERATION

Regeneration of ponderosa pine has obviously been sufficient to perpetuate the ecosystem over many thousands of years. Except in isolated situations, attempts to regenerate southwestern ponderosa pine stands naturally or by direct seeding have failed (Heidmann and others 1977). Schubert (1974) identified several conditions necessary for successful regeneration of ponderosa pine. In the past, fire functioned to prepare competition-free, mineral microsites that gave the highest probability for pine seedling establishment. Prescribed fire can provide mineral soil seedbeds for superior germination and early growth.

Especially at Chimney Spring and to a lesser extent at Limestone Flats, natural regeneration and seedling survival have been satisfactory. As a result of the initial burns at Chimney Spring, mineral soil was exposed on 19 percent of the area, mostly around large, mature, old-growth trees and where rotten logs were consumed (Sackett 1980). Seedlings began to appear soon after summer rains started in the year succeeding the initial burns, and were concentrated in areas where forest floor consumption was sufficient to expose mineral soil (Sackett 1984). First inventories made in August 1977 indicated that an equivalent of 2,600 seedlings per acre were present on the 18 burned plots. To become established and survive, seedlings must develop a long tap root to avoid desiccation from fall drought and to resist frost heaving. Seedlings excavated on burned sites had long tap roots, giving them a survival advantage. Roots of seedlings in unburned plots generally remained in the heavy forest floor and never penetrated mineral soil, resulting in high fall and winter mortality.

In 1993 at Chimney Spring, many of the 1977 seedlings are now 4- to 8-foot-tall saplings. The trees that have survived are found on sites where large, old-growth trees were killed by the initial burns. On these sites, fine needle fuels have not been available for fire spread. Obviously, these are the very sites where pine regeneration is desired.

Since 1976, there have been two other good seed years where seedlings have flourished at Chimney Spring. On one burned plot, the equivalent of 650,000 seedlings per acre were counted (Sackett and Haase, data on file). Seedbeds remain viable for up to 7 years after a fire (Sackett and Haase, data on file). Needles cast during this interval do not have time to combine as heavy, tightly held mats like old, undisturbed forest floor material does. Seeds are able to fall through the loose mat of new needles to settle on mineral soil (Haase 1981). Without fire as a natural disturbance to the forest floor, pine regeneration will be unsuccessful.

THINNING OF STANDS

A major role of natural fire in the presettlement era was the thinning of young trees, giving the landscape the open, park-like look. The dense structure and composition of southwestern ponderosa pine forests today forces managers to consider alternative methods of thinning. Much of our forest lands are thinned mechanically. Prescribed burning can be used effectively, however, to thin stands back to some reasonable density. Naturally ignited fires in past centuries merely eliminated excess seedlings where fuels were sufficient to carry fire over the seedlings. Where heavy fuels and fallen trees burn out, seedlings are able to germinate and become established because of the elimination of fuels and competition. Using prescribed fire within stands as they exist today is different because of dense "dog hair thickets" of pine saplings that resulted from good seed years (1914 and 1919) after curtailing heavy grazing. Although the saplings in these thickets are of small diameter due to close spacing and competition, the bark is relatively thick. Prescribed fires in dog hair thickets are usually not as intense as in open stands. Shade, higher fuel moistures, and minimal amounts of humus in the forest floor prevent temperatures around the bases of most trees from being high enough to girdle them. We have found that heavy crown scorch and/or consumption is necessary to thin dog hair thickets.

Initial burns at Chimney Springs reduced the number of stagnated reproduction and sapling stems from an average of 1553 to 912 stems per acre (Harrington and Sackett 1990). Small poles, many of which are also stagnated in thickets, were reduced from 192 to 156 stems per acre. Limestone Flats, as mentioned previously, did not burn well due to wet conditions; an average of only 180 stems per acre were killed by the fire in reproduction/sapling size classes.

It has become apparent that only the newly cast needles (L layer) and upper portion of the fermentation layer (F) actually burn as flaming combustion in heavy, old forest floor accumulations. The lower F layer is matted and bound tightly together by mycelium hyphae. As a result, the lower portion of the F layer acts more like a solid piece of fuel rather than as individual particles, and does not burn well (Harrington and Sackett 1990).

In an undisturbed, well-developed forest floor, newly cast needles become rapidly colonized and bound by mycelium and therefore less burnable. Fire spreading over the forest floor destroys most of the fungi. Needles that fall after a fire do not become readily infected and a much deeper layer of pure litter accumulates. Under good burning conditions, repeat fires consume most of the needles and small twigs. Fire behavior, rate of spread, fire intensity, and flame lengths are much higher in response to the greatly increased amount of available fuel. This increased fire behavior potential can be used advantageously to eliminate stagnated, dense sapling crowns.

At both prescribed fire research areas, thinning of dense stands has been an objective to relieve the dense, stagnated condition. Ability to manipulate the fire through ignition techniques and the fire environment to achieve slow-dissipating, high temperature air in the crowns is necessary to use fire as a thinning tool (Harrington and Sackett 1990, Sackett 1968). Adjusting the direction of fire spread relative to windspeed is the most common technique. Heading or uphill fires move at a speed commensurate with windspeed, creating more intense fire behavior. On the other hand, backing fires, moving against the wind (or downhill), progress with short flames and low intensities, and seldom thin stands.

Season of burning can also affect thinning. Burning at different times of the year to take advantage of various phenological and physiological conditions of the trees to modify their susceptibility to fire damage is an added condition to consider when thinning. Although it was mentioned that fall burning was recommended for initial burns, repeat burns might well take advantage of spring and summer conditions for thinning (Harrington 1987).

Skillful manipulation of prescribed fire techniques and conditions is required to thin dense ponderosa pine thickets. It is, however, another way prescribed burning can be used to relieve unnatural conditions in a fire-dependent ecosystem.

UNDERSTORY VEGETATION RESPONSES

In southwestern ponderosa pine forests, understory vegetation has declined steadily from the presettlement era. The decline has long been attributed to the exclusion of fire and the subsequent increase in heavy forest floor accumulations, and increased overstory densities (Cooper 1960, Biswell 1972). Burning at Chimney Spring and Limestone Flats has resulted in substantial changes in the understory. Most evident is the abundance of disturbance invader species like mullein (*Verbascum thapsus* L.), toadflax (*Linaria dalmatica* L. Mill), and thistle (*Circium pulchellum* [Greene] Woot and Standl.). Mullein and toadflax are dominant on heavily burned sites around large, old-growth trees that have died since the initial burns. Although some animals use these plants (Patten and Ertl 1982), none are considered favored by wildlife or cattle.

Grass species respond to prescribed fires and wildfires differently, as noted throughout the literature. Generally, production is increased, but this depends on fire severity, season of burn, and overstory characteristics. Individual species will also respond differently. Arizona fescue (*Festuca arizonica* Vasey.) and squirrel tail (*Sitanion hystrix* [Nutt.] J.G. Smith) usually show an increase in production 1 year after a fire (Harris and Covington 1983, Sackett and Haase, unpublished data, Vose 1984) whereas mountain muhly (*Muhlenbergia montana* [Nutt.] Hitchc.) requires a longer recovery period.

In 1992, vegetation was surveyed at Chimney Spring study area on the control, 1-,2-,4-, and 8-year rotation plots before burning. Individual plant occurrences were measured on subsample plots. Preliminary review of the data substantiates previous research. Production of mountain muhly and buckbrush (*Ceanothus fendleri* Gray) was reduced immediately following the prescribed burn. On the 4-year-interval plots, mountain muhly had almost recovered to the level of the control plots (46 observations on burned plots, 53 observations on control plots), and the 8-year-rotation plots had a much greater number of observations (92-burned, 53-control). The 2-year-interval plots showed a small increase in number of observations from the 1-year-interval plots (38 and 32 respectively). Buckbrush appears to require a longer recovery time also. The 1-,2-, and 4-year rotations had substantially fewer observations (6, 2, and 6 respectively) than the 8-year rotation and the control plots (17 and 19 respectively).

These data reflect density differences between burning treatments. Evaluation by cover class should show that overall biomass production is greater in the burned plots because plants were visibly larger than those in the control plots. Much of the current vegetation response research takes into consideration the effect of the small, even-aged groups of ponderosa pine (Oswald and Covington 1984, Harris and Covington 1983, Vose 1984). The greatest vegetation response occurs in open mature timber stands or directly beneath the mature timber canopies. Generally, little change in vegetation is seen in pole stands or in the dense sapling stands.

Most current studies have measured responses on fall prescribed fires. It would seem that if we are able to increase understory vegetation production by burning in this unnatural time of year, we may see a larger increase in production when burned earlier in the year when green grass is not readily consumed.

CONCLUSIONS

Very few forest ecosystems compare with southwestern ponderosa pine in the frequency of presettlement fire, which substantiates its importance for maintenance of forest health and stability (Harrington and Sackett 1990). Fire history studies from this region confirms this. Prescribed fire, then, can be an ideal tool for changing the ecosystem back to a more natural condition. Although exact presettlement conditions may never be achieved, forest condition and health can be improved using prescribed fire.

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Conservation Biology, Restoration Ecology, and a Navajo View of Nature

Victoria Yazzie Piña and W. Wallace Covington[1]

Abstract — The renaissance of ecologically based forestry over the past decade has led some individuals within the natural resource management professions to incorporate concepts articulated by conservation biologists and restoration ecologists in resource management decisions. However, many within these professions who embrace the traditional western science tradition of natural resource management resist some of the premises advanced by conservation biologists and restoration ecologists as unscientific and too metaphysical. Navajo traditionalists, on the other hand, hold values which strongly support many of these premises. This paper explores key concepts of conservation biology and restoration ecology from the perspective of traditional Navajo culture. Central to Navajo "religion" and culture is the concept of *Sa'a Naghái Bik'e Hózhó* ("walking toward the sacred way"), which expresses happiness, health, and beauty of land as well as the harmony of the interrelationship of individuals with their environment. Holistic thinking in maintaining a harmonious relationship with the land is a central foundation of a Navajo cultural perspective.

An awakening of attitudes toward the quality of the human world and of the preservation of nature has resulted in a broadened ecological approach to man's relationship with his environment. Ecosystem management, new perspectives, biological diversity, sustainable ecosystems, and new forestry are only a few of the phrases used to characterize this changing view. The relationship of man to the environment has been the center of thought for many native people across the United States. These current concepts are not new. In fact, western philosophers such as Henry David Thoreau, John Muir, and Aldo Leopold have long been influenced, in subtle ways, by an understanding and interpretation through lessons of experience founded in Native American cultures.

The attitude of ecological interrelationships is a progression of thought incorporating new observations and adjusting to influence a completely new interpretation. It is through observation, experience,and intuition that these philosophers evolved to this mature attitude toward a dynamic system. This study is intended to examine the key elements of conservation biology and restoration ecology, and those of Navajo philosophy of land ethics.

[1] *Victoria Yazzie Piña is a graduate student and W. Wallace Covington is Professor of Forest Ecology at the School of Forestry, Northern Arizona University, Flagstaff, AZ 86011.*

NAVAJO PHILOSOPHY OF NATURE

The Navajo, called *Diné* (the People), live in a vast and beautiful land in northeast Arizona, southwest Utah, and northwest New Mexico.

The Navajo culture has survived through an innate sense of "oneness" that compels them to help each other both in times of wealth and in times of poverty. Their concept of family relationships, of man's relation to the world around him and his place in the order of things, is far from that of Anglo-American society. The Navajo concept of religion is so total that, in a sense, it can be said that there is no such thing as religion in the Navajo culture. Everything is religious. Everything the Navajo knows, his home, his fields, the land, and the sky above is "holy." Religion is not a separate entity to be believed in or subscribed to, it is ever present. Inseparable from a traditional Navajo's daily life more than eating and breathing.

An attempt to portray a complete account of the origins, and developments regarding Navajo philosophy is beyond the scope of this paper. Navajo philosophy is built upon traditions that began in oral epics of inanimate earth-surface creatures, and have developed through interpretations and understanding of past events depicted through legends and stories.

In the Navajo pantheon there is no single deity who can be described as a supreme being. The most important deities include Changing Woman, who created human beings and is associated with the Earth, Sun, First Man, First Woman, the Hero Twins (Monster Slayer and Born of the Water), sons of Changing Woman, and her sister White Shell Woman (Changing Woman and White Shell Woman are one entity in some stories). Other entities, or Holy people, occupy less dominant or minor positions without, however, the clear cut divine hierarchy which characterized the Greek and Roman pantheons. The central concept of Navajo philosophy and vital requisite for understanding the whole, is *Sa'a Naghái Bik'e Hózhó*.

According to my Grandmother Tsinnie, "*Sa'a* is harmonious or desirable destiny or even restoration to youth," the attitude encompasses respect and reverence to nature.

A story that describes the importance of Sa'a Naghái Bik'e Hózhó, is when a deity named First Man left his medicine bundle behind (Sa'a Naghái) in the underworld. Today the process is repeated, both in the sense of the curing achieved through the ceremonial, of which this journey is a necessary part and more generally through knowledge acquisition, where all of us necessarily return to the source or the beginning (Farella 1984).

This doctrine within the Navajo culture stems from the idea that tries to account for everything in the universe, by relating it to man and his activities (Reichard 1974). The activities of man are viewed in the light of the supernatural ventures founded in the stories, and ritualistic explanations in songs used by the medicine men. Navajo "religion" means ritual and the beliefs tied to these rituals, according to Reichard (1974). Each ceremony has its own story from which it derives its authority.

CONSERVATION BIOLOGY

Conservation biology is a science that does not fit into the familiar mold of classic western science. Conservation biology is a crisis- or mission-oriented discipline that deals with phenomena which frequently addresses human sensibilities including, ethics, morality, and the relationship with animal communities and ecosystems as a whole dynamic system.

To paraphrase excerpts from M.E. Soule's (1985) article, "What is Conservation Biology?":

Conservation biology tends to be holistic. Ecological and evolutionary processes must be studied at their own macroscopic levels and reductionism alone cannot lead to explanations of community and ecosystem processes. Second, the assumption is that multidisciplinary approaches will ultimately be the most fruitful.

The Universe, as viewed by the Navajo, is an orderly system of interrelated elements, an all-inclusive whole that contains both good and evil. Hence, the universe is simultaneously good, benevolent, and dangerous. Humans are not seen as having dominion over nature. Instead, nature is seen as powerful and capable of causing great harm if not treated with respect. Thus, a person's attitude and actions toward nature must be respectful. If not, then harm will come which may only be rectified by performance of an appropriate healing ceremony.

Another distinguishing characteristic of conservation biology is its time scale and view of system components (Soule 1985).

In Navajo culture, past, present, and future are essential for acquiring knowledge, symbolic of the emergence of the Navajo, and shedding the chrysalis of ignorance in the lower worlds. The legends are not only the basis of the complex ceremonies, they are the history of the Navajo.

For decades man has been advancing toward oneness with the universe, the Navajo identifies with all its parts. The Navajo does not separate himself from the natural, he regards himself as a part of something larger rather than having a separate existence. The Navajo accent is on repeated creation which is often, if not always, cyclical. It is to be contrasted with the lineal, progressive view of time that dominated much of western science (Reichard 1974 and Farella 1984).

Postulates of conservation biology as suggested by Soule form two sets: a functional/mechanistic set and an ethical/normative set. The first functional postulate is the evolutionary postulate which states that many of the species that constitute natural communities are the products of coevolutionary processes. The second functional postulate concerns the scale of ecological processes: Many if not all ecological processes have thresholds below and above which they become discontinuous, chaotic, and suspended.

The second postulate is consistent with the dualism which is associated with *Sa'a Naghái Bik'e Hózhó* of good and evil ... *Hózhó* and *Hochxó*. Everything in the universe, including but not limited to knowledge, people, gods, behavior, ritual, thought, and language are divided into the good and evil, and are points in process that is continual and ongoing (McNeley 1981, and Farella 1984). One portion is not preferred more over the other, rather they are interdependent, that is, if evil were eliminated, there could be no good. In a sense, evil and good are seen as two sides of the same coin.

Soulé (1985) describes the normative/ethical postulates as: value statements that make up the basis of an ethic of appropriate attitudes toward other forms of life. They provide standards by which our action can be measured. Following is a synopsis of these normative postulates:

Diversity is good. A corollary of this postulate is that untimely extinction of populations and species is bad. Natural extinctions are rare events on a human time scale. Ecological complexity is good. This postulate parallels the first one, but assumes the value of habitat diversity and complex ecological processes. This postulate expresses a preference for nature over artifice, for wilderness over gardens (cf. Dubos 1980). Biotic diversity has intrinsic value, irrespective of its instrumental or utilitarian value. In emphasizing the inherent value of non-human life, it distinguishes the dualistic exploitive world view from a more unitary perspective.

This preference for nature over artifice is obvious in Navajo attitudes toward animals. Although for years the tribe has depended upon domesticated animals for subsistence, the religion still emphasizes wild animals. The belief that wild animals are helpers of human beings has not been laid aside now that game has been supplanted by the more easily obtainable sheep, goat, or steer. Domesticated animals have little religious respect (Locke 1992). They are property (economic value) rather than sentient (ceremonial value) beings, such as the feared bear and snake. This view of domesticated animals parallels the conservation biology postulate of a preference for natural systems over artificial systems, and for ecological integrity, or the coevolved diversity of life (Leopold 1949).

The Navajo attitude toward plants is one of appreciation of abundance. Every plant is viewed as an important component of all of the vegetation upon which man and animals depend. Thus, flowers and other plant parts from many species are treated ceremonially and used in healing/curing ceremonies.

Many Navajo medicine men, and traditionalists believe that the People live in disharmony today. Medicine men ascribe many of today's problems to being a result of disharmonious and chaotic lifestyles. One explanation of this disharmony is the over abundance of domestic animals that are steadily overgrazing the once plentiful grasslands, and many have migrated into the sensitive riparian zones, decreasing the plants and grasses which are needed both to sustain wildlife and for ceremonial uses.

What are the answers for restoring harmony of man and nature? Aldo Leopold stated, "The first step is to reconstruct a sample of what we had to begin with", similar to the journey of First Man, when he returned back to the beginning and/or source to retrieve his medicine bundle.

RESTORATION ECOLOGY

Restoration ecology is the discipline that provides the theoretical foundation for the practice of ecological restoration. In turn, ecological restoration provides the ultimate testing ground for theories of restoration ecology (Jordan et al. 1987). In a nut shell restoration ecology is the interrelationship of ecological theory and practice.

Restoration ecology, as a central challenge, acquires not only an identifiable goal (understanding the system and being able to demonstrate this understanding in an objective, unambiguous way), but also a mission (being able to heal the system). To heal the system like a form of medicine, a science, and art of healing at the community and ecosystem level (Leopold 1949, Jordan et al. 1987).

Ecological restoration deals with restoring degraded habitats to more nearly natural conditions using research and management experimentation. Restoration ecology traces its forestry origin to Aldo Leopold after he adapted a stance of conserving ecosystem integrity and the concept of coevolution (diversity of life). The history of ecological and evolutionary thought integrate to form a scientific basis for conservation management (Jordan et el. 1987).

"Restoration is more than a step toward a better relationship with the environment and a deeper understanding of it, but one which went hand in hand with it", surmised Aldo Leopold (1949). Such thinking is consistent with the Navajo concept of Sa'a Nagháí Bik'e Hózhó, walking the sacred path.

CONCLUSION

The Navajo perception of the land ethic reflects and reinforces the design of the community to which it is correlative. The basic concept of *Sa'a Nagháí Bik'e Hózhó* accents and unsurprisingly parallels many premises of conservation biology. A holistic view of nature does not devalue the dynamic dimension of nature, but broadens the scope for incorporating management strategies to enhance a self-perpetuating system.

In the Navajo culture, the earth is a sacred component of a unit family, a revered and respected member called Mother Earth. The mountains are sacred, for the Navajo came from them and depend upon them. The water courses are veins and arteries. They are the mountain's life, as our blood is to out bodies (Reichard 1974).

Restoration to youth is the pattern of the earth. A deity named Changing Woman renewed her youth as the seasons progress. This restoration to youth is something for which the Navajo lives, for he deduces that what happens to the earth may also happen to him. It is taught that the earth should not be injured. If the earth is damaged, the People will suffer.

Sa'a Nagháí Bik'e Hózhó sets value to life, living, and the unity of thought and action. It is an understanding of the whole, it is the whole. It is the substance which adds value to life and living.

A change from the inside, from a point of view of the community with evolved moral and ethical sensibilities to inherent value and biotic rights, can dramatically enhance the land use strategies of today not only in Navajo culture but also in Anglo-American culture.

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DEVELOPING AND APPLYING ECOLOGICAL THEORY TO MANAGEMENT OF ECOSYSTEMS

Session Summary

W.H. Moir, Chair[1]

The human species, approaching a population of 6 billion on Earth, is one of Earth's most efficient predators. Effects of human activities ripple in complex ways throughout the planetary ecosystem. Cities swollen with humans are great heterotrophic sinks, concentrating nutrients and pouring forth respiratory and industrial gases. Their autotrophic counterpart, the vast agricultural land systems, are an essential production base to support great cities. Globally, both the urban and agricultural regions are expanding ever more into remnant wildlands and forcing many other of Earth's inhabitants into marginal environments, if not outright extinction (some species are highly adaptive to human ways). Global effects of human activities influence Earth's continents, great rivers, oceans, and atmosphere. Some effects, such as radioactively contaminated sites, will last long into the future. Great issues arise about human dominance. What is the nature of the global ecosystem that will support Earth's human population at some sustainable level and at some quality of life? How is this global ecosystem composed of hierarchically organized parts? How do we keep these ecosystems sustainable, resilient to change, and productive?

Papers in this session all play upon the above themes. At the global scale we must monitor the movement of nutrients and keep track of primary productivity (from photosynthesis) along major climatic and nutrient gradients (Wessman and Nel). Each presentation in this session addresses the need to understand ecological processes and effects at scales of space and time ranging from macro to micro. This is a recurrent theme of ecosystem management: that the effects of populations (not just humans) upon ecosystems in which they function can vary, depending upon the scales of space and time. At what scales are analyses of ecosystem function appropriate (Salwasser, Urban, Wessman and Nel)? How do functions at one scale influence functions at another scale?

Ecosystem patterns are affected by periods and intensities of disturbance regimes at whatever scales (Urban). For example, an ant can harvest green leaves from certain trees in a tropical forest, or a hurricane can level thousands of hectares of forests. A small activity, local in space and of short time interval, can have cumulative effects far more than the arithmetic sum of the individual activities. A complex interaction of disturbances with space-time scales can affect long-term ecosystem equilibria (e.g. the condition around which they tend to fluctuate in biotic and abiotic conditions). Some ecosystems may behave chaotically under certain conditions (Moir and Mowrer), and some may flip from one equilibrial state to another, such as pinyon-juniper woodlands of the American Southwest (Jameson).

Ecosystem analysis is very much complicated by the necessity to include interactions or disciplines that are difficult to quantify or measure. Three papers in this section illustrate the importance of cultural, political, economic, and sociological nature of human activities for ecosystem analysis. Salwasser discusses how "founding principles" of ecosystem management must come from the social sciences as well as from the biological and physical sciences. The paper by Ayn Shlisky, based on work by Nancy Diaz and Dean Apostol, shows how a blend of ecosystem analysis and landscape design can result in a culturally acceptable, functional, and sustainable landscape. Their analysis transforms narrative landscapes of desired future conditions into concrete form at a local community or watershed scale, although the analysis must necessarily also consider effects of management at other space-time scales. At a more regional scale in the Sierra Madre Occidental of northern Mexico, Aguirre-Bravo shows how cultural factors of ecosystem analysis are more limiting and problematical than biological factors. The region is currently in tumultuous disequilibrium as forest, woodland, agricultural, and pastoral ecosystems suffer intense human commodity demands. There are also cultural

[1] *Ecologist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.*

insensitivities that threaten ecological balances and well-being of people, plants, and animals. Dysfunctional rural ecosystems display ethnic hostilities, drug trafficking, political unrest, and execution of local leaders. Other biological consequences include loss of biological diversity and loss of the productive base of ecosystems, including soil erosion and loss of human know-how about maintenance and values of indigenous crops and medicine.

The papers which follow, therefore, illustrate the breadth, difficulty, and urgency of ecosystem management viewed holistically. The reader is challenged to the near impossibility of truly understanding ecosystems in all their great complexity. In learning this lesson, we may arrive at the conclusion that the human species must first come to a deeper understanding about itself. We must be able to distinguish greed, which can lead to ecological dysfunction, from true need, which links humans into supportive and sustainable ecosystems.

Landscape Ecology and Ecosystem Management

Dean L. Urban[1]

Abstract — Landscape ecology is an interdisciplinary field that embraces spatial heterogeneity and pattern in ecosystems. Of several key concepts in ecosystem management, landscape ecology has much to say about scaling issues and "the natural range of variability" as this applies to the dynamics of landscape pattern. Over a sufficiently large area, dynamic habitat pattern—a consequence of biotic processes, environmental constraints, and disturbances—exhibits a scaled equilibrium over an area that is sufficiently large to maintain a constant distribution of habitats of all types and ages. This area that incorporates the full range of landscape variability for habitats and their resident metapopulations is the "unit pattern," and to maintain this pattern is the ideal goal of ecosystem management. Simulation studies suggest that this fanciful ideal will rarely be met in real systems, but these studies can provide useful predictions of the natural range of variability one might expect for a system, given the scaling parameters of its disturbance regime and successional dynamics. This approach can be extended to incorporate explicit spatial considerations, environmental gradients, and more realistic ecological details. Meeting this challenge will require the integration of landscape models into research and management. Uncertainty in dealing with landscapes from an ecosystems perspective calls for creative research using "experiments" provided by management activities, coupled with aggressive efforts to educate ourselves and the public about this changing perspective.

INTRODUCTION

Landscape ecology is a rapidly evolving field that crosses a bewildering spectrum of disciplinary boundaries. Although the field is still defining itself (Wiens 1992), its hallmark as a discipline is its focus on spatial heterogeneity and pattern (Risser et al. 1984, Urban et al. 1987, Turner 1989, Turner and Gardner 1991). Specifically, landscape ecology is concerned with (1) detecting and characterizing pattern; (2) explaining how pattern develops; (3) discovering its implications to populations, communities, and ecosystems; and (4) describing how pattern changes through time. As in other fields, there is a healthy interaction between those interested in more academic or theoretical issues in landscape ecology, and those driven by more practical issues related to land management.

[1] *Forest Sciences Department, Colorado State University, Fort Collins, CO 80523*

Here I address the question, *What does landscape ecology offer to sustainable ecosystem management?* This is a natural question, as the goals of ecosystem management (Behan 1990, Kessler et al. 1992) overlap substantially with the principle concerns of landscape ecology. There is a special resonance on scaling issues and in characterizing the natural range of variability in large-scale systems. I focus here on vegetation pattern on landscapes, but most of my arguments could be extended readily to animal metapopulations in habitat mosaics.

Landscape ecology offers no simple recipe for managing ecosystems; yet, it does offer some useful insights as to how we might approach this task. Three general insights provide an outline to the remainder of this paper:

(1) An ideal approach to sustainable landscape management aims to preserve landscape pattern as a stationary distribution of patch types. This ideal is not likely to be met except in simple systems.

(2) Pattern-based approaches can be extended by explicitly considering the agents of pattern formation on landscapes.
(3) Landscape (ecosystem) managers must invest heavily in models, especially spatial simulators, as tools for exploring alternative scenarios for systems that cannot be manipulated easily.

Pattern and Process in Ecology

Much of ecology today labors under the "pattern-process paradigm," which might be loosely stated as: *Ecological processes generate patterns, and by studying these patterns we can make useful inferences about the underlying processes.* There is an implicit concession here that it is actually the processes we are most concerned about, but these are often too difficult (perhaps for logistical reasons) to study directly. Thus, we measure the result of these processes, and infer the rest.

Landscape ecology labors under an additional onus, in that we recognize that pattern constrains ecological processes, providing a feedback between generating process, resultant pattern, and constrained process (Turner 1989). To my knowledge, landscape ecologists have not explicitly considered the extent to which ecological processes can be inferred from measured pattern in this feedback relationship. To be fair, the discipline has probably invested more in describing pattern and its implications than in explaining how pattern actually develops.

I digress about pattern and process for this simple reason: I believe we may limit ourselves by emphasizing pattern itself, and we should be investing more effort to understand how ecological processes work. Much (most?) of our theory is about pattern; much less so, about processes. For example, we have a "law" about the relationship between stand biomass and density (the $^{-3}/2$ thinning law), but the precise reasons for this law—the processes generating it—are somewhat debatable (Weller 1987). Likewise, species-area relationships are readily observable patterns in nature, but the underlying processes—and there are several—are not always obvious (Conner and McCoy 1979). The list of examples could go on: we observe log-normal distributions of species abundances (why?), and so on.

A few studies have looked into the inference of process from pattern, and results suggest we should not push such inferences too far. Cale et al. (1989) studied a simple model of two populations to determine whether the generating processes (competition and reproduction) could be inferred from observed pattern (species abundance). Even in their model, they found that it was difficult to infer the relative importance of the underlying processes: patterns were not isomorphic (different processes could generate similar patterns), the modeled processes sometimes yielded patterns that appeared random, and in a few cases the pattern suggested an inference which was simply incorrect. In another study, Moloney et al. (1992) used a simple disturbance model to assess whether disturbance parameters (patch size) could be inferred from the resultant pattern as summarized by spatial statistics (autocorrelation and power spectra). In very simple cases, this inference worked quite well. But when they introduced a range of disturbance patch sizes, or allowed these patches to overlap, inferences ultimately were degraded and processes were not derivable from pattern.

The message here is important: pattern does not map 1:1 with generating processes, and so for complex (i.e., real) systems the logical coupling whereby we emphasize pattern as the key to underlying processes should not be over-interpreted. This is a crucial point, as an implicit working hypothesis in landscape (or ecosystem) management seems to be, *Save the pattern and you'll save the process as well.* Nonetheless, it is pattern that we know best, and for which we have the most readily available data (e.g., maps and surveys). And so, it is still reasonable to attempt to base a management strategy on maintaining landscape pattern.

PATTERN PRESERVATION AS A MANAGEMENT GOAL

The ideal goal in managing a landscape based on its pattern may be to maintain a statistically stationary pattern over time. This, of course, requires that the reference pattern be defined beforehand, in terms relevant to the management objectives (timber classes, habitat types, or whatever). The notion of a "stable" landscape (or ecosystem) as a statistically stationary pattern (however defined) is as fundamental to ecology as the pattern-process paradigm itself (Watt 1947). This concept has been rediscovered repeatedly by ecologists recently (Bormann and Likens 1979, Shugart and West 1981, Urban et al. 1987, Turner et al. 1993).

The "Unit Pattern" as a Model System

In his seminal paper, Watt (1947) emphasized the relationship between demographic processes (establishment, growth, and mortality) and forest pattern (distribution of forest age classes or seral stages). Watt defined the "unit pattern" as the basic entity of the forest community—a full representation of the pattern in all its phases (fig. 1). The unit pattern is a two-levelled depiction of a forest: at a fine scale, each patch-scale element of the community is undergoing continual change, yet at a larger scale the distribution of patch types—the pattern—is stationary. This depiction was later developed as the "shifting-mosaic steady state" for northern hardwood forests (Bormann and Likens 1979); it was further extended by Shugart (1984), and has been illustrated in a statistical framework by Smith and Urban (1988).

While Watt's focus was the plant community, this same logic extends to landscapes or ecosystems. Indeed, Whittaker's (1953) redefinition of the "climax" as a stationary distribution of various successional stages and edaphic types is as appropriate a model for landscape pattern as any definition more recent landscape ecologists have proposed (Urban et al. 1987). To apply

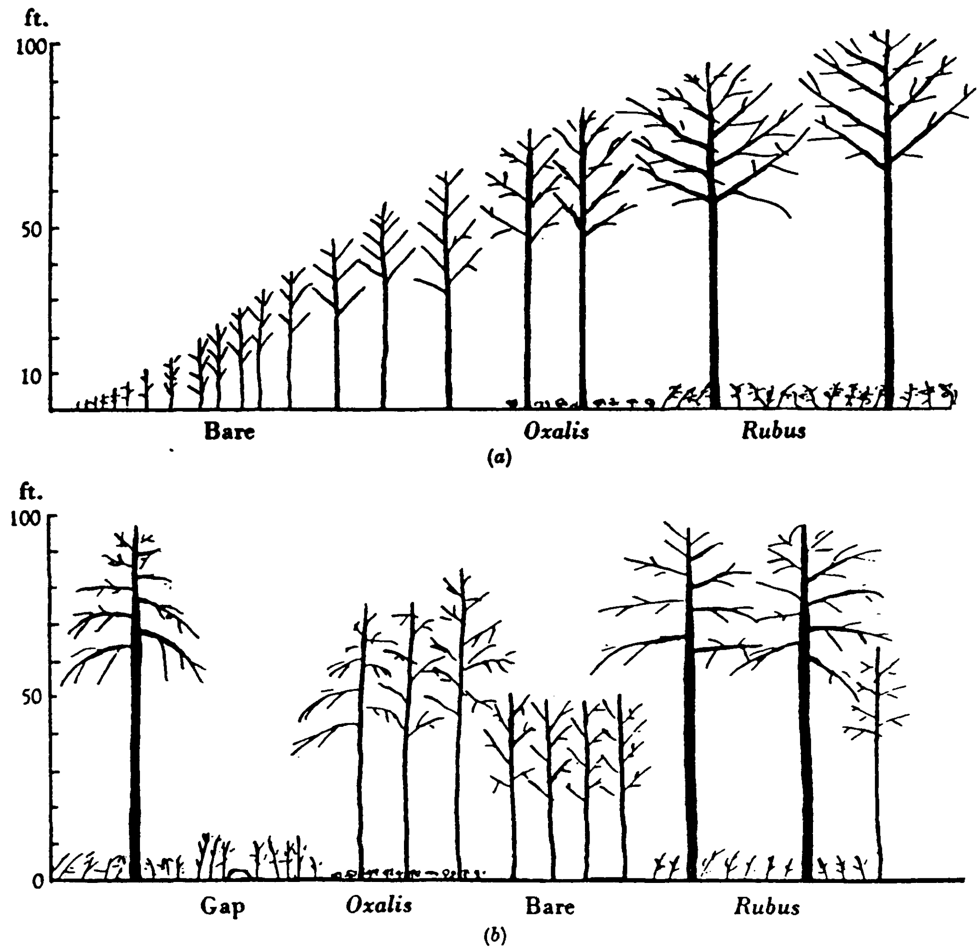

Figure 1. — Watt's concept of the unit pattern: (a) idealized successional pattern and (b) this pattern as a distribution in space "when the old wood is left to itself" (Watt 1947:14).

the unit pattern concept more fully to landscapes, it merely must be extended to include the primary agents of patch formation on landscapes. These agents are *biotic processes* (e.g., demographics and competition), *abiotic constraints* (edaphic pattern, topographic constraints), and *disturbances* (see below).

Two implications of the unit pattern are pertinent here. First, a sample of a system (landscape or ecosystem) smaller than the unit pattern is an inadequate representation of the system in the sense that it cannot represent all of its phases. Secondly, in a constant environment and over a sufficiently large area, a system will show a steady state of definite proportions among constituent phases, with the area in each phase in proportion with the duration of the phase. This latter notion (Watt's "phasic equilibrium") is the exact goal of sustainable management.

Thus, an obvious goal in managing an ecosystem (or landscape) is simply to preserve the unit pattern. This strategy is neither profound nor novel. Indeed, one of the basic tenets of timber management in forestry is to maintain a stationary age distribution across cutting units, as this ensures sustained yield. This is the so-called "fully-regulated forest" in modern forestry (Davis 1966), or the "normal forest" for German foresters of centuries ago.

So if this strategy is so simple, why don't we do it already? The fact is, the strategy *is* simple but actually fulfilling this strategy is much less simple. The area required to stabilize a distribution of habitat types can be estimated by simulation, in a way exactly analogous to constructing a cumulative variance curve to estimate a minimum sample size in study design. Shugart and West (1981) and Urban et al. (1987) provided heuristic examples whereby they estimated the land area needed to ensure stationarity for systems driven by episodic disturbances (fig. 2). In many cases, the temporal variability was such that the implied unit pattern was much larger than the area available as bounded reserves (e.g., National Forests or Parks). That is, the ideal goal can probably be realized for very few systems. Indeed, the example of the normal forest is perhaps one of very few cases where the goal of stationarity can be met in a real system, and only then if there are no larger-scale disturbances acting on the system.

Turner et al. (1993) used a simulation model to further explore the idea of landscape-scale variability for systems driven by disturbances. In their model, the landscape vegetation (which grows deterministically on a featureless landscape) has a recovery time t indexing the rate of succession, and the

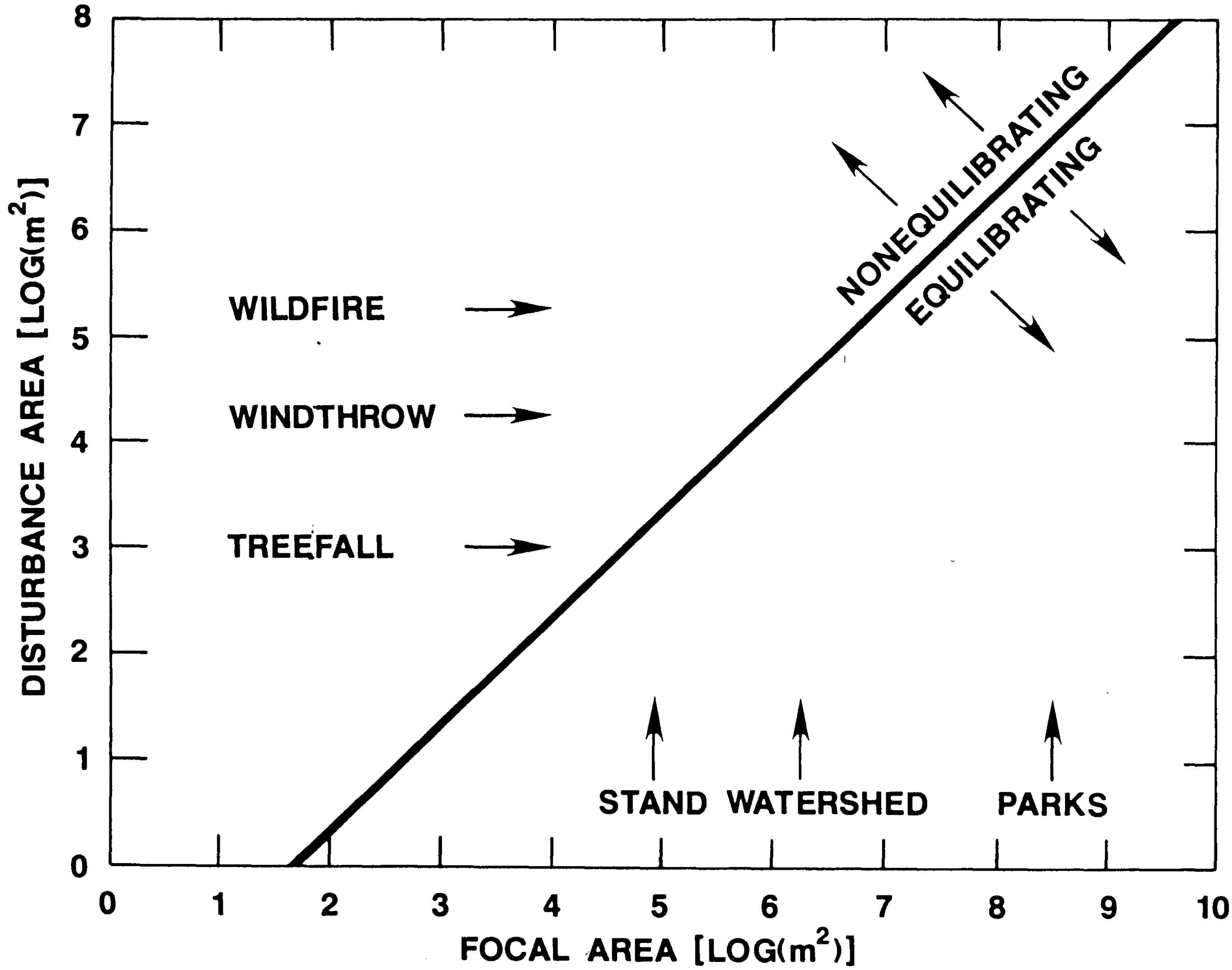

Figure 2. — Landscape equilibrium as a function of disturbance scale and containing area (from Urban et al. 1987). The diagonal, a 50:1 ratio which was found to be sufficient to statistically stabilize results from a forest succession model, is used to illustrate approximate transition from equilibrium to inherently nonequilibrium systems.

disturbance has a recurrence interval r and a spatial extent or size s. They normalized scaling parameters into two ratios: a temporal scaling parameter T (= r/t), and a spatial parameter S (= s/A, where A is total landscape area). This way, any disturbance regime can be normalized temporally (its recurrence interval relative to system recovery time) and spatially (its size relative to the containing area). In simulations of various disturbance regimes, they found that landscape dynamics fell into a few qualitative domains in the scaling parameter space (fig. 3):

(1) Systems with relatively small disturbances exhibited more-or-less equilibrium conditions regardless of disturbance frequency; the disturbance events were simply absorbed by the landscape.

(2) Systems driven by large, infrequent disturbances showed nonequilibrium dynamics wherein the landscape reflected each disturbance event as a perturbation.

(3) Systems driven by large, frequent disturbances exhibited a quasi-equilibrium in which the landscape was quite dynamic but remained within stable bounds.

Note that only in the restricted case of small, frequent disturbances are the conditions of the unit pattern met; in no other case is a stationary distribution of patch types expected. And note that this example is itself a simple case: a simple model with no topographic or edaphic complexities, and uniform disturbances.

Turner et al.'s model experiments offer some guidelines of what we might expect from a system, given the scaling parameters of its disturbance regime and successional dynamics. Clearly, for many systems the expectation is *not* a stationary landscape pattern. The recent Yellowstone fires underscore this conclusion for a system which has never shown a stationary configuration over the past several centuries (Romme 1982). Certainly, designs for a regulated forest become rather

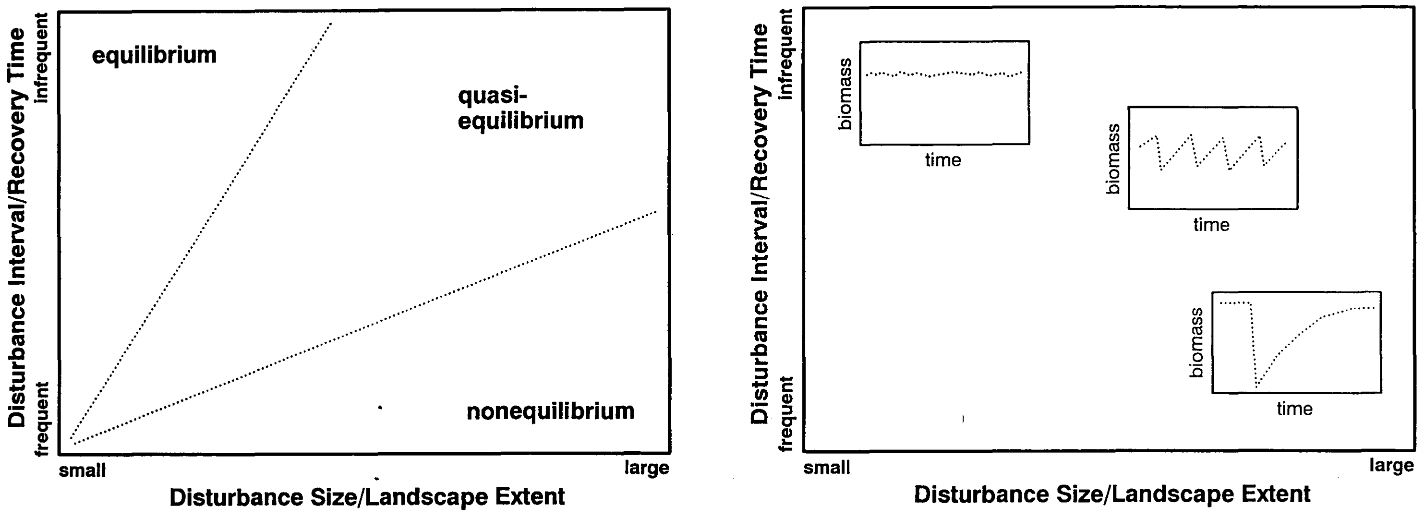

Figure 3. — Landscape dynamics under varying disturbance regimes normalized by temporal and spatial scale (modified from Turner et al. 1993). (a) Domains in which landscape dynamics can be viewed as equilibrium, quasi-equilibrium, and nonequilibrium. The dotted lines in the figure arbitrarily separate domains that grade into one another. (b) Typical dynamics one might expect from a system in each of these domains, i.e., their natural ranges of variability.

superfluous for landscapes driven by large, episodic disturbances such as hurricanes. Turner et al. (1993) discuss several natural and human-modified landscapes relative to their scaling parameters.

EXTENSIONS TO PATTERN MANAGEMENT

The pattern-based approach provides useful insights about the feasibility of maintaining a particular landscape in a steady-state condition. This conceptual framework can also be used to suggest guidelines for "rescaling" systems to effect their qualitative dynamics (Urban et al. 1987). Rescaling a fire regime via smaller, less intense, prescribed burns that might be sustained within a bounded region is a familiar example in current practice. The pattern-based approach can be extended further by considering explicitly the mechanisms that generate landscape pattern, and using these extensions as a further guide to managing complex landscapes.

Agents of Pattern Formation

Landscape pattern is generated by the interplay of three general agents: biotic processes, abiotic constraints, and disturbance. The first two are coupled inseparably in vegetation pattern, while disturbance can sometimes be decoupled and overlaid onto the system, depending on one's frame of reference (Allen and Starr 1982, Urban et al. 1987).

Biotic Processes

Biotic processes include plant demography (dispersal, establishment, growth, and mortality) and competition. It is important to note that even in a perfectly homogeneous environment, demographic processes over time would generate spatial heterogeneity. Indeed, the mechanism of pattern formation via plant growth and mortality was the basis of Watt's (1925, 1947) seminal ideas on pattern and process in plant communities.

Plant dispersal can act as an agent of pattern formation, particularly when coupled to differential rates of other population processes (especially mortality). This mechanism is addressed in a huge literature on diffusive instabilities in diffusion-reaction systems (e.g., Okubo 1980, Kareiva 1990).

Competition figures prominently in the generation of vegetation pattern. Because this differential success depends strictly on the environmental context of competition, it is necessary to consider these effects with reference to local patterns of abiotic constraints (see below).

Abiotic Constraint

Real landscapes are patterned by spatially heterogeneous features including soil catenas, topography, and other environmental gradients. Many of these aspects of landscape pattern are addressed in classical gradient analysis (Whittaker 1967, Gauch 1982). A long tradition of gradient analysis has identified two predominant axes of vegetation pattern on landscapes: temperature (often indexed as elevation) and relative moisture (often indexed as slope aspect or exposure, or soil depth). These features provide a template on which other pattern-forming processes act. Gosz (1992) has advocated using gradient analysis as a framework for exploring scenarios of landscape change.

Disturbance

Our thinking about disturbance has evolved considerably over the past few years, from earlier notions of disturbances as events from "outside the system" that disrupted things and were therefore "bad," to an acceptance of these events as a natural and integral component of the system (Pickett and White 1985).

A consideration of the spatial and temporal scaling of disturbance regimes has led to a further elaboration of disturbance as a component of ecosystems, in which a system can be referenced at two levels of organization (and hence, two scales). At a lower level, disturbances are "outside" and disruptive while at a higher level, they are incorporated into the system—they are "inside" and not disturbing at all (Allen and Starr 1982, O'Neill et al. 1986, Urban et al. 1987). This two-level depiction of disturbance lends itself nicely to an extension of the unit pattern concept from stands to landscapes. By this strict definition, a landscape has a stationary pattern at that spatial scale that can "average away" the perturbations associated with individual disturbance events.

Interactions and Feedbacks

One of the reasons landscapes are complex is that each of these agents of pattern formation interacts with the others. In particular, vegetation pattern cannot be interpreted without reference to demographic processes in the context of environmental gradients. Smith and Huston (1989, see also Huston and Smith 1987) used a simulation model to illustrate this interaction. Their model was an individual-based forest simulator (Shugart and West 1980, Huston et al. 1988) which was simplified to emphasize tree competition for light on sites along a soil moisture gradient. Smith and Huston proceeded from three premises about tree life-history traits, which reflect anatomical, morphological, and physiological trade-offs in plant strategies: (1) a species tolerant of low resource levels (e.g., shade, or low soil moisture) would have a lower maximum growth rate than intolerant forms (i.e., tolerance implies low maximum growth rate); (2) conversely, a species with a high maximum growth rate under favorable resource levels would have less tolerance to reduced resource levels (i.e., high maximum growth rate implies low tolerance) (fig. 4a), and (3) a species cannot simultaneously optimize for tolerance to reduced above- and below-ground resources, i.e., a shade-tolerant tree cannot also be drought-tolerant (although a shade-intolerant tree can be drought-intolerant as well). These three premises imply a species response space (fig. 4b) which Smith and Huston represented with 15 hypothetical species (fig. 5a). In simulations, interactions among these species were sufficient to generate classical successional patterns in species replacement as well as gradient response in space (fig. 5b). These patterns obtain as follows: On the most xeric site, only the most drought-tolerant species can survive, and it characterizes the sere at all ages. On less xeric sites, there is a classical pattern of species replacement, from the fastest-growing/most intolerant, to successively slower-growing but more tolerant species. On a mesic site, the species with the fastest growth rate dominates in early succession, while the most shade-tolerant species ultimately dominates old-growth ("climax") stands. In general, the succession is from the fastest-growing species for a given soil-moisture regime, to the next-fastest/next more tolerant for that site, and so on, to the most shade-tolerant species that can persist under that particular soil moisture regime. Because of life-history trade-offs (premise 3), seral patterns dictated by available light are related to spatial patterns in soil moisture. Thus, explanations of vegetation dynamics in time (succession) must be interpreted with respect to their position in space, along environmental gradients. Austin and Smith (1989) link these arguments more explicitly to classical gradient analysis.

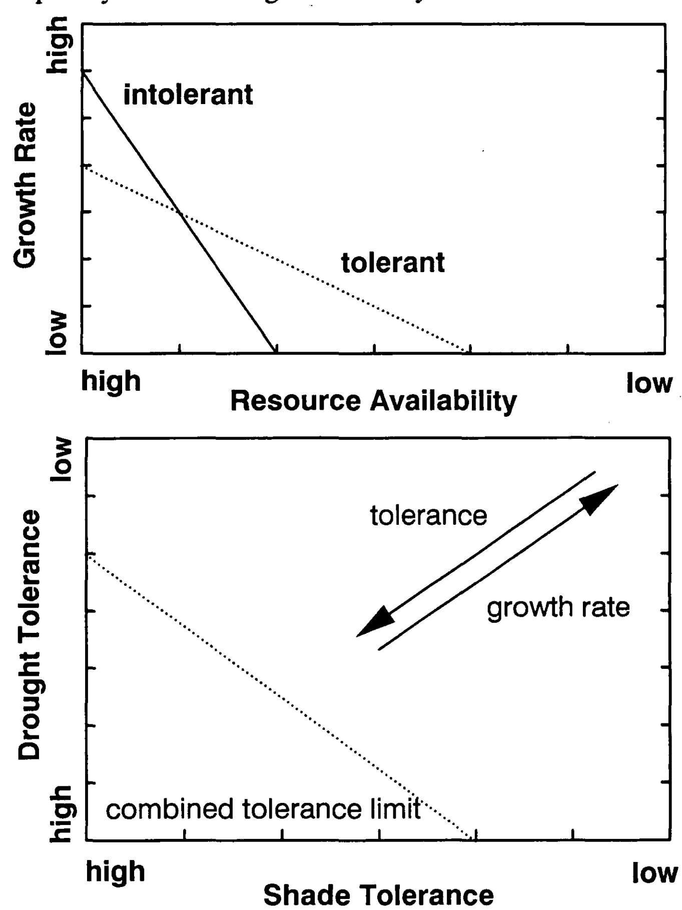

Figure 4. — Response space for tree species life-histories, relative to available light (shade tolerance) and soil moisture (drought tolerance). (a) For either resource, tolerance comes at the expense of reduced maximum growth rate. (b) These trade-offs arrayed along two axes: maximum growth rates increase toward the upper right, while tolerance increases to the opposite corners; no species can be very tolerant of drought and shade simultaneously and so the lower, left corner is devoid of species (after Smith and Huston 1989).

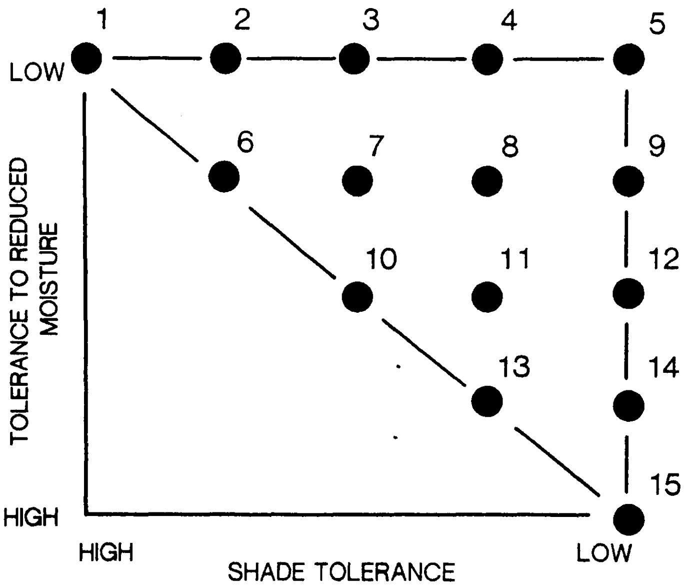

Figure 5(a). — Implications of life-history trade-offs for succession and gradient response (from Smith and Huston 1989). Hypothetical species invented to span the "triangle" implied by life-history trade-offs (see text and fig. 4b).

Disturbance, of course, interacts with the other agents of pattern formation. Fire is a familiar example of a spatially and temporally correlated disturbance regime: fires burn differentially with respect to topography, fuel type, fuel load (forest age or condition), and so on. Thus, disturbances interact with biotic processes and abiotic constraints, as well as with other disturbances (*e.g.*, Knight 1987).

Dispersal can generate pattern by itself, but it also may have a secondary effect as a local intensifier of patterns generated by other agents. Thus, dispersal may act as a positive feedback mechanism in pattern formation, reinforcing and amplifying initial pattern.

Implications of Agents of Pattern Formation

This discussion of pattern-generating agents in general, and the interplay of biotic processes with abiotic gradients in particular, has implications for managing landscapes for biodiversity. The simulations of Smith and Huston (1989) predicted that a sere includes more species on a mesic site than on a xeric site. This, in turn, suggests that for a landscape characterized by rather uniformly mesic sites, diversity would be maximized by managing for stands of varying ages because old stands tend to be dominated by the same species. Conversely, for a landscape of more heterogeneous (and mostly xeric) site conditions, older stands would likely be dominated by a greater variety of species because the seres would have various endpoints; diversity would be increased by maintaining a set of stands on different kinds of sites. Thus, the management prescription in the former case is for activity in the time domain; in the latter case, in the spatial domain. This prescription is

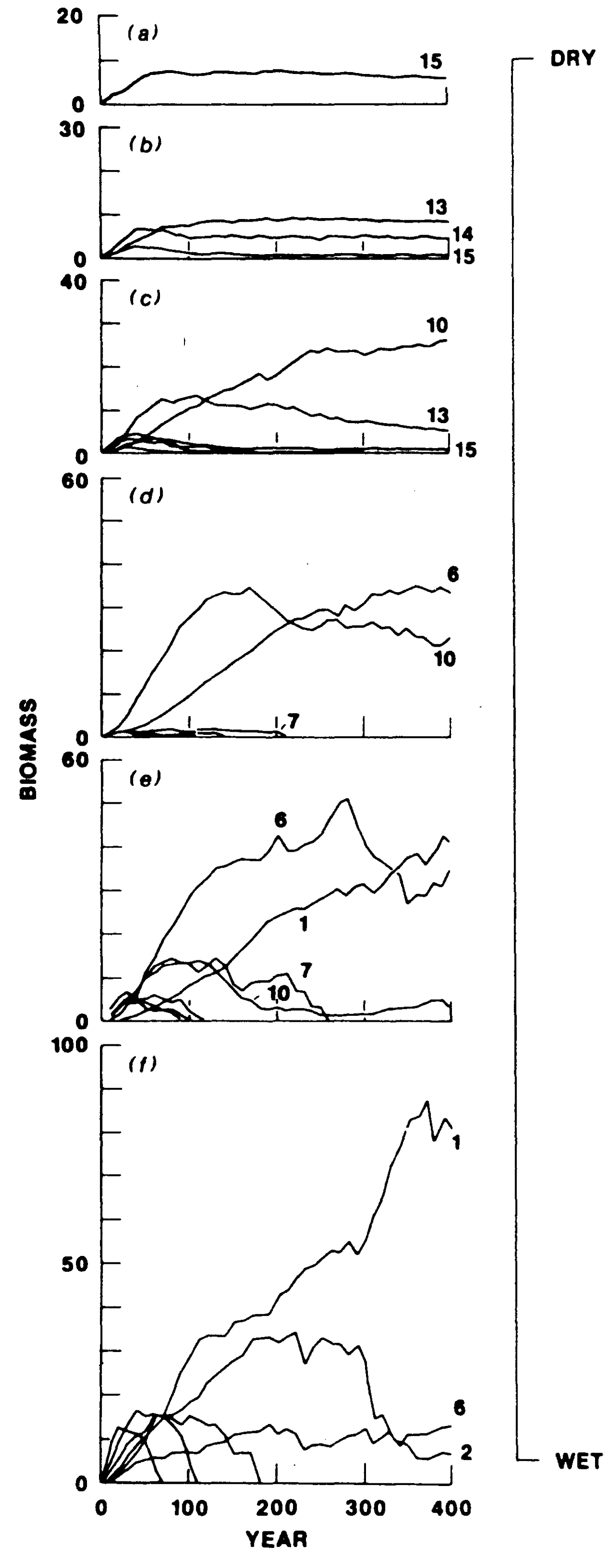

Figure 5(b). — Successional trends for these species, as simulated for sites along a soil moisture gradient. In general, succession proceeds from the most shade-intolerant/fastest growing, to slower-growing, more tolerant species that can persist under the soil moisture regime on each site, and so proceeds from right to left along the rows in (a).

borne of a consideration of how biotic and abiotic agents interact to generate patterns in species diversity (see also Gosz 1992 for similar conclusions).

This prescription is an example of how we might add an explicit consideration of pattern-generating mechanisms to enrich a management strategy based on pattern by itself. This extension is still amenable to the approach of predicting the qualitative dynamics of a reference area (management unit, forest, park) given the scaling parameters of its successional dynamics, abiotic template, and disturbance regime (following Turner *et al.* 1993). Likewise, this approach could be extended further to consider even more detailed (realistic) biotic processes, multiple environmental gradients, and various disturbances (including management), and so embrace more fully the agents of pattern formation on landscapes. In general, the approach remains the same but the simulations become more complicated. Because of this complexity, these extensions demand a new set of tools for researchers and managers alike.

MODELS IN ECOSYSTEM MANAGEMENT

Accounting for landscape pattern in space and time is an obvious challenge, and it seems equally obvious that models will play a crucial role in this approach. Behan (1990) has emphasized that the conceptual model, hence computerized tools, of multiple-use management are qualitatively different from that of sustained-yield approaches of single-commodity management. I would argue that the challenges of landscape ecology and ecosystem management will require still another generation of modeling tools.

Consider the sorts of models I've used here as illustrations: these range from fairly detailed, nonspatial simulators (Smith and Huston 1989) to simpler but spatial simulators (Turner *et al.* 1993). The current trend seems to be toward simulators that are spatially explicit and incorporate a wealth of ecological detail (Baker 1989, 1992; Sklar and Costanza 1991). These are new kinds of models, and we're only now learning how to use them cleverly; there are computational as well as ecological issues to resolve. Appropriately, there is a great diversity of approaches being pursued, which will ensure that a variety of useful and robust models will become more available to end-users.

These simulation models are used in a different way than optimization models used in planning (*e.g.*, FORPLAN). In planning models, an objective function is specified and the best solution is computed based on the specified constraints. By contrast, landscape simulators are used in an exploratory mode: *Is this scenario better/worse than this alternative scenario? Does this management prescription maintain more old-growth than the alternative? Will this cutting pattern generate more edge over the long run? If we do this, what will happen to wildlife habitat? Will water quality suffer? What might happen if we try this instead?* These are not really questions about optimization.

In general, ecosystem management is not an optimization problem. Part of the reason for this is implicit in the concept of "natural range of variability." The goal to maintain a system in some semblance of normalcy seems inconsistent with a simultaneous goal to maximize any particular aspect of the system. But a second reason problem with optimization is that we simply lack the tools: we do not have, and in the future we are not likely to have, modeling tools that reconcile disparate ecosystem attributes in a common currency. A model that provides useful predictions about wildlife habitat is not likely to have much to say about water quality; a stand yield model will likely be moot on butterfly diversity. While our policy goal may transcend "multiple use" to embrace the full complexity of ecosystems, our best models focus on single (or a very few) uses and will likely remain so.

The ultimate tool for ecosystem management might be some sort of marriage between geographic information systems (GIS), ecological simulators, and decision support models (*e.g.*, Covington *et al.* 1988, van Voris *et al.* 1993). Ecological models would provide a means to assess alternative management prescriptions or other dynamic scenarios (*e.g.*, climate change). A GIS would serve as a framework for data storage, manipulation, and display (*e.g.*, storing stand survey data and highlighting stands meeting user-specified criteria). A user interface incorporating decision support tools would allow a researcher or manager to move interactively among all components of the system. This goal implies new technological developments, and new training for resource managers and basic scientists as well. But ecosystem management seems to call for new tools and approaches, and we would do ourselves a disservice to ignore this challenge.

CONCLUSIONS

As with the nascent field of landscape ecology, ecosystem management is new and exciting. It is also uncertain, simply because we don't really know what we're doing; we have no historical precedent, and no real frame of reference by which to judge our success. Because of this uncertainty, we have to plan on learning as we go, using our management decisions as experiments from which to learn. With careful planning and the aid of models, these experiments can be as controlled and well-replicated as resources allow. Presumably, models will minimize the incidence of "unpleasant surprises" in management experiments, but we must also retain the flexibility to learn from our mistakes and take corrective action: this is the essence of adaptive management. But this is also the scientific method, and a partnership whereby managers helped perform experiments with researchers certainly would be to everyone's advantage.

A basic appreciation of landscape dynamics, and hence of ecosystem management as practiced on landscapes, leads to the conclusion that old-fashioned notions of "the constancy of nature" are not likely to apply to real landscapes (Botkin 1990).

This presents an educational challenge to ecosystem managers: we need to convince the public (and perhaps ourselves) that it is acceptable for nature to behave erratically, for landscapes not to look the same year after year. We have been reasonably successful in retraining the public about the role of fire in natural ecosystems, and so there is reason to be optimistic about the role of education in ecosystem management. But novelty is not always welcome, and so we must be aggressive in pursuing the change to the new, ecosystems perspective.

ACKNOWLEDGEMENTS

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Continuous and Discontinuous Change in a Southwestern Woodland

Donald A. Jameson[1]

Abstract — The traditional view of ecosystems stressed by man's activities is that a given stress will result in a given plant community equilibrium, i.e., there is a single equilibrium state for any stress regime, regardless of the previous state. This is the essence of a single steady state system, which allows for only "smooth" changes. Much has appeared in some ecological literature lately about alternative models that allow discontinuous changes. The simplest such model is a fold catastrophe. In this model, the response to some extreme value of a control variable is a single stable steady state. The response at the opposite extreme has another stable steady state. In between there is a zone where either state can be considered a stable steady state (along with some unstable steady states); the particular equilibrium state depends on the originating extreme state. The next most complex multiple steady state model is the cusp catastrophe. Whereas the fold model requires a "jump", i.e., an outside influence, to move from one steady state to another, the cusp model also allows a "smooth return" along one control axis that does not require an outside influence. The Southwestern pinyon-juniper woodland has many examples that appear to fit this last model — with both discontinuous and continuous change introduced by combinations of climate, fire, grazing and wood harvesting. The cusp model also may alleviate concerns of those who are unwilling to depart from the earlier paradigm that time and succession will cure all ills. The relationship of the appropriate equilibrium states to climatic stress and activities of mankind are a fruitful area of study.

SINGLE VERSUS MULTIPLE EQUILIBRIUM STATES

A traditional view of ecology is that a given stress to an ecological community will result in a given community equilibrium. In range management, for example, it has been assumed that there is a single equilibrium state (called range condition) for any grazing prescription, regardless of whether the previous condition was higher or lower. This is the essence of the concept of a single steady state system. When any outside stress is removed, the system migrates to a single stress-free condition, which may be known by some term such as "potential natural vegetation". If the single steady state concept is appropriate, random fluctuations may alter the current state but the system will return to the predetermined state for any given stress level.

The migration to a stress-free steady state, once the stress is removed, may take a longer time than can be allowed for study. Alternative study techniques may involve a space-for-time exchange, i.e., using ecological relationships displayed in space as a surrogate for ecological relationships that change in time. In fact, it may not be discernable whether a phenomena is occurring in space or in time. "When a moving ecological community reaches an observer" might be the same question as "when a moving observer reaches the ecological community". Although there are dangers in making assumptions about ecological patterns when the underlying processes are not observable or controllable (Cale et al. 1989), it is certainly tempting to do so when the time required to change exceeds the life of the observer.

[1] *Donald A. Jameson is Professor Emeritus, Colorado State University, and USDA Forest Service, retired. He currently resides at Sedona, AZ.*

In contrast to the traditional view of single steady states, multiple steady state concepts allow more than one state of the system to result from a given input or stress (Zeeman 1976). The resultant steady state for any stress may be determined by a previous state, rather than being independent of any previous state as in the single steady state concept. A classic example in range management is the California annual grassland/*Stipa pulchra*. Range managers have been forced to recognize that the *Stipa pulchra* communities of pre-European settlement will not be replaced through natural succession, and that the potential steady state, even if grazing is eliminated, is now an annual community dominated by introduced Mediterranean annuals. However, this has been commonly taught in range management classes as perhaps the single exception to a single steady state concept of range condition.

Recently there has been considerable attention in the literature (reviewed by Laycock 1991) of many interpretations of multiple steady states observed in natural vegetation systems, but explicit system models that are subject to control and statistical analyses are rare. However, Jameson (1991) reported experimental results that contained at least a partial test of a hypothesis that a cool- and warm-season grass community possessed multiple steady state properties.

Occam's razor teaches that the simplest usable model should be used to explain observed results, and the simplest model of multiple steady state systems is the fold catastrophe (Zeeman 1976). A drawing of a fold is simple, it is only the proof that it is the simplest multiple steady state model that is difficult. Even simplistic graphics programs can be induced to draw an appropriate fold. In the fold model, the response to some extreme value of a control variable has a single stable steady state. The response at the opposite extreme has another stable steady state. In between there is a zone where either state can be considered a stable steady state (along with some unstable steady states).

If we should sample for some measurement variable along the control axis (such as percent of vegetation made up of pre-European species), we would expect to find a low variance at the two extremes of the fold and a high variance in the middle zone, resulting in a cloud of observations in the multiple equilibrium region of the system. In fact, the high variance might obscure the fact that there are steady states of any kind, and we might assume that we have chaos.

In the annual grassland example, the percentage of vegetation made up of pre-European species is a simple index of community composition. In other ecosystems, other indices may be more appropriate. Some nominees for approaches to the index question include species richness indices, patch connectivity, diversity indices, fractal dimensions, phase transition parameters, discontinuity detection algorithms, edge detection algorithms, changes in spatial autocorrelations, and others.

An important consideration of the study of such systems is their equilibria, i.e., to what states does the system migrate because of its own properties.

ELEMENTARY CATASTROPHE THEORY

Some extremely helpful aids in addressing the concept of equilibria are found in the area of catastrophe theory within the definition:

> *Elementary catastrophe theory is the study of how the equilibria of a dynamic system changes as the control parameters change.*

We will now review various catastrophe manifolds. These manifolds are system response surfaces that show response of state variables x resulting from application of various controls u. Although the system dynamics as it returns to equilibrium has a time dimension, the time trace of the system is not explicitly shown. However, if the system is near the equilibria and the changes in controls are small, the trace of system dynamics is nearly the same as the equilibria manifold. The basic concepts of both adaptive management and mathematical catastrophes were developed in the 1960's. However, it was not until a decade later that these two concepts were sufficiently digested to be incorporated into studies of ecological systems (Zeeman 1976, Jones 1977, Bar-Shalom and Tse 1976), and were combined even later (Casti 1980).

The theoretical properties of systems that can be managed incrementally are well known (Bar-Shalom and Tse 1976) and have been discussed in terms of biological and ecological systems (Jameson 1986). It has been well documented that fixed schedule plans for management of ecological systems are not satisfactory, but again it was not until recently that there was sufficient understanding so that basic causes of failure of fixed schedule systems could be reasonably well discussed (Walters 1986). Traditional approaches to analyses of uncertain systems emphasize the importance of using a stochastic model. However, a stochastic model is not always necessary unless model equilibria demonstrate properties of certain of the catastrophe classes.

Several environmental systems have been modeled as fold catastrophes (McMurtrie and Wolf 1983, Noy-Meir 1982, Walker and Noy-Meir 1982, Walker et al. 1981). Jones (1977) presented a model of spruce budworm outbreaks as a cusp catastrophe. Loehle (1985) published a theoretical paper on application of catastrophe theory to grazing, but no concrete examples were included. Johnson and Parsons (1985) studied an example pasture system to collect data on a fold catastrophe response. Many earlier authors seemed tentative in suggesting the occurrence of a cusp, but Lockwood and Lockwood (1993) explicitly applied catastrophe theory to weather-driven grasshopper population dynamics with a detailed analysis of historical data that demonstrated the properties of a cusp. These published analyses have been a posteriori; except for the limited analyses of Jameson (1991) there has not yet been a published result that has examined the biological response of an environmental system to an applied treatment to examine a cusp catastrophe hypothesis.

A FOLD CATASTROPHE EXAMPLE

From elementary algebra, the equation

$$u = x^3 + x^2 + x \quad (1)$$

is a cubic function that yields a single value of u for each value of x. However, the equation

$$x^3 + x^2 + x = u \quad (2)$$

yields 3 values of x for each value of u. This equation is the simplest of the elementary catastrophes, or a fold catastrophe (Fig. 1). The second derivative or inflection point of this equation occurs at

$$6x + 2 = 0 \quad (3)$$

$$x = -1/3 \quad (4)$$

By subtracting the value of x of Equation (4) from the values of x in Equation (2), the manifold equation can be simplified to

$$-x^3 + x + u = 0 \quad (5)$$

Equation (5) is the simplest equation that can represent the discontinuous properties of a catastrophe. Setting the first derivative of Equation (5) to 0 yields x = 1/3; these points locate discontinuities that are one of the catastrophe properties. Between these points, there are three solution values of x; the upper and lower values represent stable equilibria and the intermediate value represents an unstable equilibrium. As control is increased or decreased across one of these discontinuity points, the equilibrium "jumps" to the alternative equilibrium.

The studies reviewed by Laycock (1991) could be perceived as a linked series of fold catastrophes, i.e., with discontinuities between pairs of several states of the system. However, the behavior of the system is thereby constrained by these discontinuities, and allows no "smooth return" to another state as would be necessary for many ecological situations.

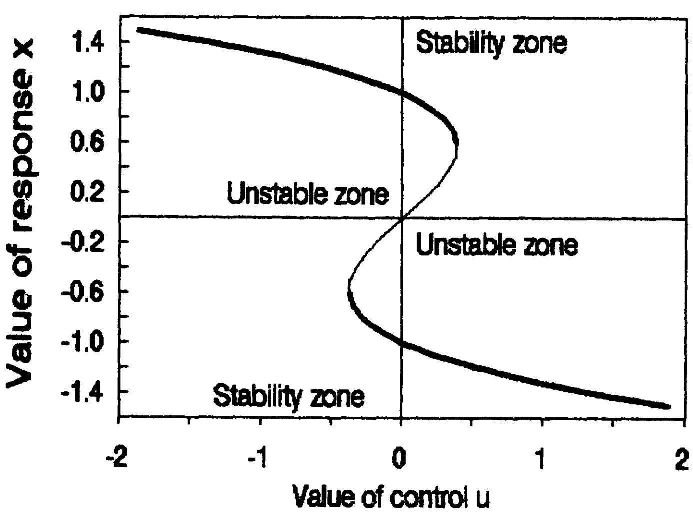

Figure 1. — A fold catastrophe to depict discontinuous changes in response x as a result of a management control u.

A CUSP CATASTROPHE EXAMPLE

The concept of cusp catastrophes in ecosystem management will be introduced by an example of a woodland - grassland transition. Typically, one would expect a shift toward grassland species and away from woodland species with fire or wood harvesting, and vice versa with heavy grazing. These reversals seem entirely reasonable and have been frequently observed (see Arnold, et al. (1964) and Jameson and Reid (1965) for fire and post-Columbian grazing effects, Samuels and Betancourt (1982) for prehistoric wood harvest effects).

Under xeric conditions, tree species may not become established even with grazing and cessation of fire (Cinnamon 1988). More mesic climatic conditions may be required for this shift to occur. On the other hand, the xeric climatic conditions and limestone soils typical of the southern Little Colorado River basin may not produce enough grass fuel for fire to be a factor (Clary and Jameson 1981). However, in areas with more grass fuel production, fire can be an effective deterrent to woody plant reproduction (Jameson 1962). Another situation in which fire suppression does not seem a likely factor in so-called invasions is where the sprouting *Juniperus deppeana* is the dominant tree species (Jameson and Johnsen 1964).

Betancourt (1987) stated "distributions of pinyon and juniper species (and their associations) should be considered ephemeral over the past two million years ... traditionally attributed to overgrazing and fire suppression, historic invasions could also mark the current progress of continued migration, climatic fluctuation, or recovery from historic and prehistoric woodcutting." These statement are supported with analysis of pack rat middens over the longer time periods (Van Devender et al. 1987), and fire scar and tree growth chronologies over a period of a few hundred years (Swetnam and Betancourt 1990). Jameson (1969) analyzed the summer and winter precipitation ratios across Arizona, but Webb and Betancourt (1992) have indicated that even the summer-winter ratios may be non-stationary because of variations in the Southern Oscillation (El Nino).

The complex situation of fire, grazing, wood cutting, and climatic shifts described in the preceding three paragraphs call for a complex model to deal with the combinations. Organizing information around any model has certain implications about how phenomena are expected to behave, and Figure 2 is an attempt to include all of these phenomena in a single model. The basic "fast" controls (u_1) considered will be fire, grazing and wood harvest. As demonstrated in Figure 2, there is also a second or "slow" control u_2 dealing with climatic pattern.

The front (folded) edge of Figure 2 represents four of five properties of a catastrophe model listed by Lockwood and Lockwood (1993):

1. Modality: the system tends to be either in a tree dominated state or in a grass dominated state; intermediate values cannot be reached directly along the grazing season axis and tend not to occur.

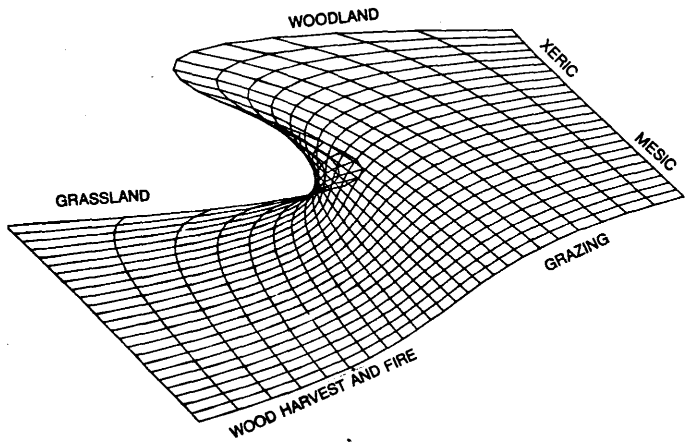

Figure 2. — A cusp catastrophe to depict a combined discontinuous change in a pinyon-juniper woodland. Taken from Jameson (1987).

2. Inaccessibility: the infolded region of the figure represents unstable equilibria which will not be reached by successional activity.
3. Jumps (discontinuity or catastrophe): as the controlling factor (fire versus grazing) moves toward either extreme, a point is reached where response can no longer move smoothly, the jump to a different level at this discontinuity is what gives catastrophe theory its name.
4. Hysteresis: The time path that the woodland/grassland response must make as grazing control moves to the right is different than the response as control moves to the left.

The fifth property of a cusp catastrophe, divergence, is represented in a movement from the front edge toward the back edge of Figure 2.

It should be emphasized that models described here, although postulated as reasonable for ecological observations, are only conceptual. In fact, the surface for Figure 2 was generated from:

$$-(x^3 + xu_1 + u_2) = 0 \qquad (6)$$

where x is the woodland/grassland response, u_1 is the "fast" control of the fire/grazing axis, and u_2 is the "slow" control along the climatic axis.

This equation does not represent ecological mechanism, but is the simplest mathematical form that will generate the desired surface (Jones 1977, Zeeman 1976). It should be emphasized that these model equations may contain parameters that are not readily identifiable with any known biological states or processes, and thus may not be experimentally determinable.

The algebraic model also has some other interesting properties in addition to those mentioned previously. Note that in the zone between the edges of the fold in Figure 1, the equation gives three solutions. However, for a given response and position along the response axis, there is a single input or control. Thus it may be possible to reconstruct from ecological evidence the causes leading to a particular response, but, because of the three solutions for a given input, it may not be possible to predict what the response will be without knowing the ecological history of the ecological system.

RISK AND CATASTROPHES

In Figure 3, the elementary fold catastrophe of Figure 1 is tilted so that the greatest benefits are shown near the upper discontinuity; this figure represents those environmental management situations where benefits can be increased, up to a point, by increasing the input control. If the system were deterministic, there would be no difficulty with such a management strategy. However, if the system is stochastic, a given control may result in either less than maximum benefits or a shift beyond the upper discontinuity to the lower stability zone. Once the system is in the lower stability zone, the original benefits cannot be restored by reversing the control.

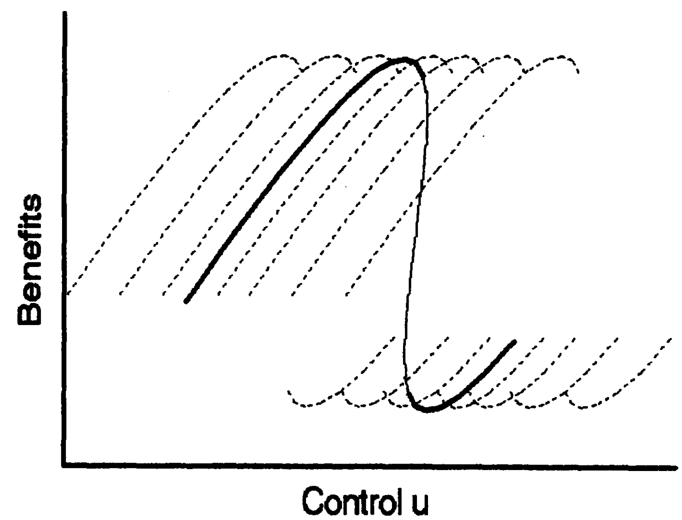

Figure 3. — A tilted fold catastrophe indicating the problems of managing for a maximum benefit with a discontinuous response surface. The heavy line represents the mean response with a given control u; the dashed lines indicate the response at individual sample years.

Operationally, there are three strategies appropriate for situations depicted by Figure 3:

(1) Choose a level of control u_2 sufficiently left of the optimal point such that benefits never fall to the lower stability level, or
(2) initiate a recovery action to lift the system performance from the lower stability zone to the upper stability zone, or
(3) initiate an alternative control (u_1 in Fig. 2) to move the system beyond the bifurcation point so that a smooth return to the higher benefit level can be restored.

Each of these strategies has its own cost to be considered in choosing the best action. The cost of strategy (1) is largely the cost of benefits foregone by operating at a conservative level of control. The cost of strategy (2) is the cost of the recovery action. The cost of strategy (3), initiation of slow control u_1, depends on waiting for the slow control to be effective, and largely results from discounting future benefits of the restored system.

In other cases, the best we can hope for until suitable techniques are developed is to depend on the experience and memory of human managers. Historically, human nature is such that memory serves only those that have experienced such catastrophes in their own life, and little wisdom is transferred to others.

FUTURE DIRECTIONS

Almost any natural resource system could be perceived and modeled as a single steady state system, or conversely could be modeled as multiple steady state. However, it is not always clear that the choice of models makes any practical difference. A useful approach is to model the system both ways, then use experiments on the models to determine if the multiple steady state properties of the model lead to management decisions that are different from those reached using single steady state assumptions.

An alternative approach is to directly examine the behavior of the natural system to see if responses are such that a multiple steady state approach must be used. For some systems, the single steady state approach may lead only to management inefficiencies rather than to "catastrophes"; for other systems the consequences are more severe than mere inefficiency. If multiple steady state properties can be found experimentally, it would clearly indicate that the system cannot be managed incrementally (i.e., with a passive adaptive approach) without catastrophic results. If catastrophe conditions are not found, then a simple incremental approach will at worst lead to inefficiencies. For natural resource management, a lack of catastrophe behavior means that satisfactory management corrections could be based on observations of ecosystem responses without leading to ecosystem destruction. If catastrophe conditions are found, then errors in management cannot be corrected merely by reversing management direction.

A handicap that we must face is that most of our history of natural resource research and management is based on concepts that we can subdivide the world and conduct experiments to determine what we need to know about system behavior. Walters (1986) has nicely documented some problems that arise when the system under study cannot be subdivided, but must be "probed" in order to learn the necessary aspects of system behavior in an active adaptive management scheme. Studies in global climatic change certainly will face problems of this kind: how does one probe the world?

The same history of thought also leads us to expect that we can perceive problems and manage resources as though the response surfaces do not contain discontinuities. Even those who insist that interesting models must be nonlinear seldom perceive that discontinuity and bifurcation is a more serious problem than the linear-in-a-neighborhood type of nonlinearity.

A new approach would require that we learn to conduct experiments and use appropriate analyses that escape the old thought limitations of subdivision and continuity. As an example, we might start with resource maps. Typically, such maps subdivide the world into "homogeneous" units. The nice thing about these homogeneous units is that our old models and concepts work well within the units. In fact, it is the boundaries between the map units that are the most interesting and contain the most challenges in management, as these areas are most likely to possess discontinuous responses or ecotones (Casey and Jameson 1988, Holland et al. 1991). A new resource mapping concept that emphasizes boundaries rather than homogeneous land units would at least indicate that we know were the challenging problems lie.

Displays of land area based on discontinuous responses or ecotones will lead to examination of the relationship of bifurcation and multiple stability zones of one "map cell" to behavior of another map cell. Are stability zones contagious? If the land area represented by one map cell shifts to a lower production stability zone, will this shift be absorbed by adjacent land areas, or will the shift spread to nearby land areas? The implications of these alternatives will cause us either to shrug off desertification and global climatic change, or to feel that desertification and global climatic collapse are inevitable.

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Unsustainability: The Shadow of Our Future

W.H. Moir and H. Todd Mowrer[1]

Abstract — Unsustainability is the counterpoint of sustainability. Sustainability is often predicted from ecosystem models. We examine the errors or uncertainties in a general assessment of different classes of predictive models. The uncertainties grow with model complexity and over longer periods of time and larger spatial dimensions. We conclude that assertions of sustainability derived from these models should be viewed cautiously. We do not really know what ecosystem conditions will exist in the future; and the more distant the future, the more the uncertainty. To the degree that humans are efficient predators, we may be forcing some ecosystems into chaotic behavior, although present evidence is not strong. Nevertheless, it is better for land managers and publics to manage for uncertainty than to assume sustainability. We suggest five management guidelines to resource managers who are looking into shadows of the future.

INTRODUCTION

One of the fundamentals in this new era of ecosystem management is the concept of sustainability. One fairly typical, proposed definition is complex (EMIT 1993): "The ability to **sustain** diversity, productivity, resilience to stress, health, renewability, and/or yields of desired values, resource uses, products, or services from an ecosystem while maintaining the integrity of the ecosystem over time". Often the definition of a commonly used word has become, when used as a technical word, a hodge-podge of vague nouns and adjectives. Efforts have been made to clarify the word, sustainability, by the context in which it is used (Norton 1991, Gale and Cordray 1991, Woodmansee 1992). For clarification we must ask the relevant questions (Maser 1992): what is to be sustained, at what level is "whatever" to be sustained, how long is it to be sustained, and for whom? Various attempts have been made to answer these questions (Gale and Cordray 1991, Costanza 1991, Toman 1992). In this paper, we attempt to limit the concept of ecological sustainability by drawing attention to its complementary side, unsustainability. In so doing, we try to reveal some difficulties when managing ecosystems for sustainability and to point out that the glib use of this term can incur a false complacency. We hope to show that the crystal ball of the future is clouded, and that we cast a long shadow in the direction of the "desired future condition".

In this paper we first discuss some of the reasons that limit predictions of sustainability. We explore limitations of several widely used kinds of prediction models. We ask whether ecosystems at given scales of time and space can become increasingly stochastic, thereby limiting prediction and perhaps subjecting affected populations to extinction (a clearly unsustainable result). We also ask whether ecosystems can behave chaotically. If so, they become very sensitive to differences in threshold variables that severely constrain predictions.

We pause here for an important clarification. Because we might not be able to closely predict future levels of goods or amenities, does not imply that ecosystems or their desired outputs will *be* unsustainable. Neither we nor the reader intend to fall into this trap. However, there are major implications for management when uncertainty shadows the future, and it is therefore essential that we understand the limits to prediction.

In confronting uncertainty we discuss why management should act conservatively, especially for ecosystems at or near high extractive levels or at risk of species extinctions. We suggest the use of the word, sustainability, is to be distrusted, and that a more appropriate approach to ecosystem management involves strategies addressed at future uncertainty. In conclusion several such strategies are proposed.

[1] *Research ecologist and research forester (mensuration), respectively, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.*

UNSUSTAINABILITY

Unsustainability may occur whenever we lack the knowledge to determine if the "what" of sustainability can meet the criteria of either "how much" or "how long". These criteria are often arrived at by projections of supply and demand at various scales of time and space. It is not our purpose to elucidate the signs of resource overexploitation. The number of threatened species in nearly all countries, whether developed or undeveloped (World Resources Inst. 1993), is but one sign of ecosystems in stress, where the "what?", "how much?" and "how long?" are disturbing questions. In addition to genetic losses, commodities such as timber, fuelwood, ocean fishing stocks, or clean water may project more to depletion than to sustainability (Ludwig et al. 1993). Similar forecasts can be applied to such ecosystem amenities as aesthetic appeal, wilderness solitude, critical habitats, sequestering of contaminants, clean air, and other declining levels of ecosystem services.

ECOSYSTEM MODELS

Depending on the ecosystem being modeled, its behavior must be approximated by a mix of deterministic (fixed cause and effect) and stochastic (random) relationships. Simpler components of an ecosystem are easily measured and are subject to minimal variation across the selected spatial and temporal measurement scale. Quite often this minimal variation can be ignored, and components of this sort are considered to be deterministic in their cause and effect relationships. However, the behavior of an ecosystem component is affected by sources of variation outside the selected scale of measure, or beyond our ability to understand their complexity. Consequences of these types of variation contribute to the stochastic or random portion of observed component behavior. Ecosystem components that behave deterministically at one level of spatial and temporal measure quite often appear to behave stochastically at other levels because of these factors. For example, the mean or average diameter growth of a forest stand can usually be predicted quite accurately, while the growth of an individual tree within that stand is subject to greater variation. It is difficult to understand, measure, and predict the behavior of various ecosystem components because they are subject to varying (and often unknown) combinations of these deterministic and stochastic effects. In recent years, stochastic effects have not, as a rule, been included in models of forest stand dynamics. It must be emphasized that just because stochastic effects are not included in a predictive model, does not mean they are not present. Under certain conditions, i.e., finer levels of model resolution and/or longer prediction horizons, stochastic effects may literally "swamp out" the environmental signal predicted by the model.

Stochastic effects are more difficult to quantify than the direct cause and effect relationships, because in addition to the most likely value for a response variable, one must predict the relative variation in the response behavior. Moreover, integration of stochastic variation into model predictions causes them to vary across a range of possible values, because different random variations affect each prediction. The result of a large number of predictions by a stochastic model would be a frequency histogram showing the relative frequencies of occurrence of various levels of response, such as those shown for two levels of variation in Figure 1.

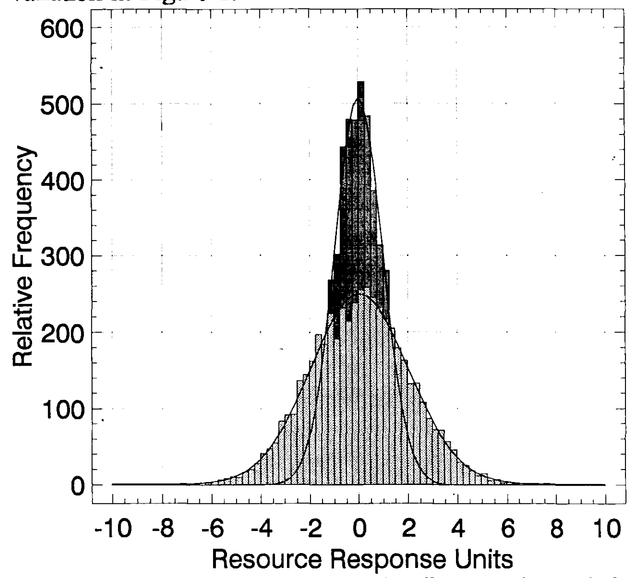

Figure 1. — Histograms of two normally distributed populations having the same mean and sample size, but different variances. The narrower distribution reflects less uncertainty in localized predictions and shorter time horizons. The wider distribution reflects more uncertainty in global predictions and longer time horizons.

The most well known shape that these frequency histograms assume is the bell-shaped or normal probability distribution function. Probability distributions are characterized by a measure of central tendency (mean) and relative dispersion (variance). In a well-behaved simulation model, i.e, not subject to chaotic behavior, the mean of the stochastic model (the highest peak of the histogram) would be identical to the single value predicted by the analogous deterministic model. As shown in Figure 1, the stochastic model provides additional information that is extremely important to the prediction of resource sustainability. This information is the relative dispersion of possible alternative future outcomes, particularly extreme values. While the mean value, or the single outcome from a deterministic model, provides the most likely result, in actuality it is likely to occur only a portion of the time! For example, the highest response level of the more peaked histogram in Figure 1 represents only about ten percent of the total responses represented by all bars of the histogram.

In the other 90 percent of the possible outcomes depicted by this histogram, other less likely but possible levels of the ecosystem response would be predicted. Again, it should be emphasized that just because a particular forest ecosystem model does not predict the range of possible outcomes, does not mean they will not occur. These uncertain outcomes are due to the range of data used to calibrate the model, inexact understanding of the true functional relationship being modeled, and uncertainties in each set of values used to initiate projections by the model (Mowrer 1991). These uncertainty components combine and propagate through the network of internal model calculations to affect the final values projected by the model. As the variance, or dispersion, increases in the distribution of possible outcomes, our uncertainty in the most likely response also increases. In the more broadly distributed histogram in Figure 1, the highest peak only occurs in about 5 percent of the total outcomes depicted. Because of the propagation of sources of variation through ecosystem models, increased uncertainty is always a factor over larger spatial and temporal scales (Mowrer 1989).

The above discussion relates to ecosystem components which can be modelled by classical probability theory. When chaotic behavior is present, future states often "bifurcate" or assume one of two extreme values, with no opportunity for intermediate values. Ecosystem components which display this type of behavior obviously function very differently from those displaying a continuum of probabilities between extremes. The possibilities of both classical statistical uncertainty and chaotic behavior in ecosystems cast an even longer shadow over assessments of resource sustainability.

Figures 2 and 3 illustrate possible mathematical approaches to ecological modelling. The kinds of equations and their calibrations shown in the 4 x 3 matrix are illustrated in figure 3. Additional modelling dimensions are time and scale, or spatiotemporal resolution (grain) and extent, as discussed by other speakers of this symposium (e.g., Urban, Wessman and Nel, Salwasser). The constraints to predictability that we have discussed apply to models at any spatiotemporal scale. In the past, risk reduction has driven modellers to reductionistic, but more reliable, models. Accuracy more than range was important (Leary 1992). Today, modellers are boldly addressing problems at global scales, and ballpark rather than exact outcomes are sought. Nevertheless, predictability can become severely limiting when models on the scale of years or decades are extended too far into the future, or when models at plot or stand levels are extended to landscapes at watershed or more extensive scales (Levin 1992).

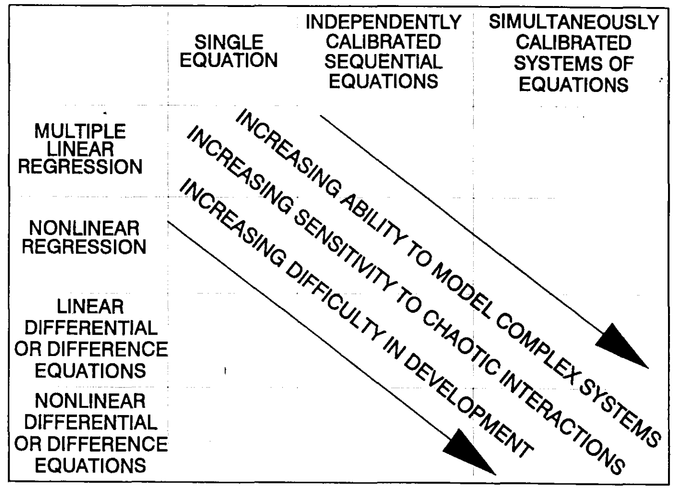

Figure 2. — The effect on model functionality of increasingly complex equation types (on the vertical axis) and calibration approaches (across the horizontal axis).

Multiple linear regression: linear in coefficients

$$y = b_0 + b_1x_1 + b_2x_2$$

Nonlinear regression: nonlinear in coefficients

$$y = b_0 + b_1x^{b_2} + b_3\exp[b_4x_4 + b_5x_5]$$

Sequential equations:

Separate calibration | Separate calibration

$$D_{10} = b_0 + b_1D_0 \qquad N_{10} = b_2 + b_3N_0$$

$$D_{20} = b_0 + b_1D_{10} \qquad N_{20} = b_2 + b_3N_{10}$$

Simultaneous calibration

Simultaneous differential equations:

$$\frac{dy_1}{dt} = a_{11}(t)x_1 + a_{12}(t)x_2 + \dots + a_{1n}(t)x_n + F_1(t)$$

$$\vdots$$

$$\frac{dy_n}{dt} = a_{n1}(t)x_1 + a_{n2}(t)x_2 + \dots + a_{nn}(t)x_n + F_n(t)$$

Figure 3. — Examples of some equation types and calibration approaches presently in use. Both the a and b coefficients are statistically calibrated. The coefficients are time period (t=1,...n) dependent.

LIMITS TO PREDICTION

The possibilities of countless population and environmental interactions apply to existing models of reduced or simplified ecosystems. Our survey of modelling limits suggests at least four constraints to predictions of sustainability as a consequence of ecosystem complexity.

Ecological surprises. One can trivialize this obstacle by asserting that surprise is nothing more than extreme but possible errors of prediction (tails of the frequency distribution). However, surprises are often considerably more in quality, magnitude and consequence than prediction error and can render models useless by invoking new rules (Loucks 1985, Hollings 1986). In other words, the model may address uncertainties within its limited framework, but the larger uncertainties are both hidden and potent (Wynne and Mayer 1993). It is possible for continuous environmental change to drive ecosystems past a critical threshold. Kay (1991) and Loucks (1985) use as illustration algal blooms erupting suddenly when a nutrient reaches a certain threshold. In addition, profound changes in ecosystems can occur, for example, when "accidents" occur. Introductions of alien biota such as cheatgrass, chestnut blight, or feral pigs into ecosystems are well known events (Gillis 1992). Another form of surprise is the occurrence of one or more highly improbable events usually not contemplated in models but whose effects "ripple" through the ecosystem as time progresses. Ecosystems affected by surprise may collapse, move to a new functional level, or return to a former condition. Examples are provided by Kay (1991), Savory (1988), and Muller-Dumbois (1987).

Critical thresholds/scaling effects. Thresholds, scaling effects (King et al., 1991; Hardin 1991; Levin 1992), ratchet effects (Ludwig et al. 1993), and surprises may be hard to differentiate. Whatever these effects are called, profound ecosystem change can occur at critical thresholds (Kaufman 1993). Environmental problems at a global scale are familiar examples. The change after some threshold condition is exceeded may be discontinuous and irreversible. An example of discontinuous change in pinyon-juniper woodland is given by Jameson (this volume). The irreversible change may also suggest a bifurcation point or fold (Kay 1991), and ecosystem behavior for all practical purposes may be indistinguishable from a chaotic ecosystem (discussed below).

Increasing stochasticity. Examples are found during periods of rapid climate change (Taylor et al., 1993), during social unrest and wars, and from ecosystems subject to heavy alteration. Environmental stochasticity acts upon populations and thus contributes to demographic stochasticity, as for example, upon metapopulations (Stacey and Taper 1992). One effect is that limiting factors for population survival are probably exceeded in some time interval, increasing the probability of local extinction (Menges 1992, Stacey and Taper 1992). In another example, insurance companies reduce stochasticity by limiting the range of coverage to as specific population classes as they have sufficient data for (for example, teenage automobile drivers in metropolitan Colorado) in order to better calculate accident or mortality probabilities and thereby determine costs of coverage policies. In general, the larger the scales of space and time, the greater the stochasticity.

Chaotic ecosystems. The possibility of ecosystems displaying chaotic behavior was suggested by May (1976). Chaotic ecosystems have limits to prediction, since small changes in initial conditions can produce largely discrepant outcomes for $t >> t_0$, and can cause the system to become unstable. Ecosystems stressed by predisposing factors, such as a period of unfavorable climate or other limiting conditions, may exhibit irruptions of insects or diseases, and have been suspected of chaotic behavior (Logan 1991). Chaotic behavior is a possibility when new conditions are imposed upon ecosystems, forcing them into population structures far from previously evolved stability conditions (Kay 1991, Ritchie 1992, Berryman 1991). A possible transition occurred during the rapid desertification of Southwestern desert grasslands coinciding with the introduction of livestock (Neilson 1986). Historic changes in the American Southwest at the end of the Little Ice Age were accompanied by soil loss and profound biotic reorganization to the point where new equilibria of desert shrubs, annuals, soil erosion pavements, and hydrological regimes were established (Neilson 1986, Dick-Peddie 1993).

Following Cambel (1993), we distinguish chaos as a condition from systems that are chaotic. The latter must satisfy numerous conditions, including having mathematical properties of non-linearity, non-equilibrium function, contain strange attractors, and have a Lyapunov exponent > zero (Cambel 1993).

There has been much interest in the possibility of chaotic ecosystems (Kaufman 1993). Ecosystems clearly satisfy some of the conditions for chaotic systems: they are dissipative, complex, non-linear, and have a mix of both deterministic and random properties. Evidence of chaotic behavior in ecosystems is arguable. There must be a long enough time of observation to discern strange attractor behavior and a significant positive Lyapunov exponent from analysis of population time series. Absent these special properties of chaotic systems, ecosystems can be interpreted to have high levels of stochastic randomness as an alternative (Ellner 1991, Pool 1989, Berryman and Millstein 1989).

At present, demonstrating chaos is mostly a matter of collecting sufficient evidence for or against a chaotic system, rather than a matter of proof (Turchin and Taylor 1992). There are a growing number of claims of populations that behave in accordance with chaos theory. This includes adding predation to food-limited prey populations (Ritchie 1992), outbreaks of forest insects (Turchin 1991), predator-prey systems in a eutrophic environment (Ritchie 1992), the spread of influenza and childhood measles (Olsen and Schaffer 1990).

Many assertions of ecological chaotic systems have alternative explanations (Pool 1989, Olsen and Schaffer 1990). However, we have indicated above that complex, nonlinear, multipopulation ecosystems have at least some of the properties of chaotic systems. These include feedback structures that are, in Berryman's (1991) words, the seeds of chaos. The possibility exists of pushing ecosystems into a chaotic domain as environmental changes approach magnitudes outside their natural evolutionary domains (Berryman 1991).

Affected populations appear to either adjust to a new equilibrium or become extinct. Modelling studies suggested that the former can happen if stressed populations are augmented by dispersal from metapopulations (Gonzales-Andujar and Perry 1993). Strong negative feedbacks have been suggested to induce chaotic instability. Berryman (1991) cites crab populations of northern California and the economics of fisheries as an example of a negative feedback. At a larger scale he also proposed that global warming, ozone depletion, deforestation of the tropics might be included as possible factors of system disequilibria. Activities that involve species introductions, long time-lags, rapid growth rates, or highly efficient predators or parasites could decrease stability and possibly lead into a chaotic environment (Berryman 1991). We return to these possibilities below.

CONFRONTING UNCERTAINTY

Ecosystem complexity is perhaps no better stated than by the humorous spiritual adept, Da Free John (1982):

"All things in themselves are the effects of the interests of independent living beings. Therefore, there is no great plan that we can depend on, because there are countless beings manufacturing effects, human beings, less than human beings, visible beings, invisible beings, big beings, little beings, beings in every plane within the hierarchical planes of manifestation. All beings are thinking and feeling and acting and desiring and creating effects. The summation of all of this chaotic desiring is the cosmos."

Given the evidence of ecosystem complexity and a rapidly changing global environment, assertions about managing for sustainability (of whatever, for how much, and how long) are suspect. One recommendation for resource managers offered by Ludwig et al. (1993) is to distrust claims of sustainability. This kind of distrust was expressed by Sachs (1989), who pointed out that most usage of the term "natural resources" was often equated with human exploitation that greatly exceeded the basic needs of human communities. Instead of claiming sustainability in management goals, it may be more useful and certainly more honest to address options to manage for uncertainty. One suggested principle is to manage ecosystems generally within the range of their "natural variation" (EMIT 1993). However, many ecosystems upon which we depend for future well-being are rapidly changing as a result of evolutionarily new technological impacts Berryman (1991). They may already be outside their range of natural variation (whatever that is!) or varying around new equilibrium conditions (Hollings 1986). We already cited two examples, the Southwestern desert grassland and pinyon-juniper woodland. The affected ecosystems may follow presently unknown rules of ecosystem reorganization and may also be in a period of instability and species displacement (in chaotic systems language, a bifurcation) while reorganization occurs. Moreover, *there may be no scientific consensus about the outcome of such systems.* This is the future's shadow, and the manager's dilemma (Ludwig et al., 1993). "We have no answers, so what should we do?"

Guidelines for resource allocation are not new where ignorance prevails. The decision to do nothing or maintain the status quo may be inappropriate. Common sense or best judgement underlain by a social or ethical imperative may have to substitute for scientific uncertainty (Maser 1992, Costanza 1991, Ludwig et al., 1993). Five itemized guidelines below may be helpful when "tough" decisions must be made.

1. Be as explicit as possible about responsibilities to future generations and discount the present resource values accordingly (Perrings 1991, Norgaard and Howarth 1991, Toman 1992). This usually means curtailing demand. In Hardin's (1991) words, every shortage of supply is equal to a longage of demand.
2. Analyze the effects of a proposed activity at all important space-time scales. In particular, probe and try to understand the larger uncertainties outside the domain of reductionist models. A decision to set aside resources without reducing high consumption of that resource in a global economy may increase the burden on other countries to provide that resource. Examples of local versus global analyses are given by Sachs (1989), Daily and Ehrlich (1992), and World Resources Institute (1993).
3. When the stakes at any scale are high (either now or in the future), then be cautious. The precautionary principle states that an action or non-action should be made before harm to the environment becomes visible (Perrings 1991, Wynn and Mayer 1993). High levels of natural variability and the reductionism of ecological models can hide overexploitation, surprise, and possible irreversible changes in non-equilibrium systems. The precautionary principle can also be used to shift to the user the burden of proof against harmful consequences at whatever scale. It also favors actions that are reversible (Bella and Overton 1972), and argues for high frequency monitoring (Savory 1988).
4. When environmental risks are high or when there is possibility of irreversible damage, then spread the effects unevenly over the land and maintain a high level of spatial diversity (Bella and Overton 1972). Odum (1969) argued for a mix of both production (early seral) and diversity (late seral) ecosystems in the landscape. However, such management is probably not sufficient. We realize now that "set asides" (primarily diversity oriented lands) do not harbor the needed diversity, nor are "multiple use" lands (primarily commodity oriented lands) sufficient to provide the necessary goods and services for current and future needs. Rather, natural resource lands and the ecosystems thereon must provide a mix of both commodity and diversity values (Franklin 1989ab; Roberts 1990). The resultant landscapes become diversified in shades of gray along spatial and temporal gradients rather than as a mosaic of black and white ecosystems.
5. To avoid instability or chaos, Berryman (1991) suggests taking actions that seek to avoid long time lags, large growth rates, and introductions of highly efficient predators or parasites. Essentially these are

measures of restraint, caution, and avoidance of heavy-handed actions that lead to overcompensatory feedbacks. The loggers versus owls conflict in the Pacific Northwest can be taken as an example of the kind of instability that resulted from failure to intervene early enough in a regional ecosystem (the old-growth forests) that had heavy impact and strong economic feedback at local, regional, and global scales.

SUMMARY

What constitutes a valid basis for determining if an ecosystem or natural resource is sustainable? We have attempted to be careful in defining our terms, and the extent to which these terms may be applied. While these limitations prevent any universal solution to the problem, they have allowed us to explore some limitations in the process of defining a set of ecosystem conditions or resources as sustainable. In so doing, we hope to have provided insight into reasons why sustainability is not an easy property to recognize or nurture. Recognizing the sources of uncertainty that create these difficulties may provide insight into the range of solutions and best approaches to ecosystems sustainability. An arguably appropriate response, voiced among scientists, conservationists, economists, and other concerned publics, but often opposed by industry and other resource "developers", is to take a precautionary, conservative approach to ecosystem management and resource exploitation. On the other hand, an overly conservative approach, often voiced by environmental groups, may fail to provide critical biological diversity or important needs of the human community as part of the functional ecosystem.

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Ecosystem Management: From Theory to Practice

Hal Salwasser and Robert D. Pfister[1]

Abstract — Ecosystem management (Robertson 1992) and sustainable development (UNCED 1992) have emerged in the early 1990s as major concepts and policies for the stewardship of human and biological communities in the United States. Both have a similar goal: the sustenance of desired conditions of lands, waters, biota, human communities, and the economic enterprises that depend on healthy, productive land and natural resources. Both have a similar compelling urgency: the human population is putting increasing pressures on the health and productivity of lands, waters, air, and resources, jeopardizing the ability to reach that goal (Silver and DeFries 1990). Ecosystem management and sustainable development are proposed as a prudent path to pursue. Both are already more than dreams; to some extent they are in practice or are being seriously tested. But they are also rapidly evolving. The purpose of this paper is to describe some principles and practices that we believe are crucial to the success of an ecosystem approach to land and resource stewardship that aims to sustain desired conditions of environmental quality as well as development of human communities and economies.

DEFINING ECOSYSTEMS

The ecosystem concept is central to the new era in land stewardship and resource conservation. Ecosystems are communities of organisms working together with their environments as integrated units (after Tansley 1935). They can occur from microscopic scales to the scale of the whole biosphere. For any plant or animal, including humans, an ecosystem is its home (Sahtouris 1989, Berry 1987, Rowe 1990).

All resources for life come from an ecosystem and all waste products eventually return to an ecosystem for recycling or storage. A rotting log is the ecosystem for a fungus. A pond is the ecosystem for a sunfish. A watershed is the seasonal ecosystem for a migratory ungulate. A whole mountain range is the ecosystem for a population of wolves. And the planet is now the ecosystem for the human population. In all cases, the organisms are integral parts of a complex of other organisms working together with their physical environments as a whole. The parts could not persist without the whole and its myriad of processes.

An ecosystem perspective on land and resource management means thinking about land—its soils, waters, air, plants, animals, and all their relationships—as whole units that occur in a hierarchy of nested places. The places—or ecosystems—are open to a constant flow of materials and energy in and out. They are constantly changing over time and much of the change is not precisely predictable by science (Botkin 1991). People are integral parts of ecosystems; both dependent on their resources and factors in affecting some of their changes.

Defining Ecosystem Management

Ecosystem management is variously defined by those who are shaping its course. Beginning with a standard dictionary definition, management is the process of taking skillful actions to produce desired outcomes. Combining this with the term ecosystem, ecosystem management is the process of seeking to produce (i.e., restore, sustain, or enhance) desired conditions, uses, and values of complex communities of organisms that work together with their environments as integrated units. This

[1] *Hal Salwasser is Boone and Crockett Professor of Wildlife Conservation. Robert D. Pfister is Associate Director of the Montana Forest and Conservation Experiment Station. Both are faculty members of the School of Forestry at The University of Montana.*

integrated or systems concept of land and resource management is broader than traditional approaches to the preservation of nature as historically practiced in national parks, wilderness areas, and nature reserves. It is also broader than traditional approaches to multiple-use land and resource management as often practiced on public lands. Certainly it is broader than intensive agriculture. Ecosystem management emphasizes the integration of ecological, social, and economic factors at different temporal and spatial scales to maintain a diversity of life forms, ecological processes, and human cultures. Traditional approaches to land or resource preservation attempt to either freeze ecological conditions at a desired state—which is not biologically possible—or allow natural forces to run without human interference—which is appropriate in some cases but not always socially or politically acceptable. Traditional approaches to multiple-use land and resource management tend to focus on sustaining yields of desired resources and uses in compatible blends such as timber, game, wildlife, water, livestock forage, fish, and recreation opportunities (Gale and Cordray 1991). Conflict among the various resources and their human constituencies is common in multiple use (Wondolleck 1988). The primary focus of agriculture is the sustained production of desired crops of plants or animals usually achieved through the simplification of ecosystems to guide net primary productivity into the desired crops.

An ecosystem perspective enlarges the focus of land management and resource conservation to whole ecosystems rather than selected parts or processes. It does not deny the importance of producing resources needed by people. Nor does it deny the need to protect certain places from certain kinds of human activities. But it focuses on sustaining desired ecosystem conditions of diversity, long-term productivity, and resilience, with yields of desired resources and uses being commensurate with the larger goal of sustaining those conditions. This is not what many practicing biologists, foresters, fisheries managers, or range conservationists were taught about management of their featured resource.

Some Key Ecosystem Characteristics

What might managers and citizens need to know about ecosystems to help guide their successful management? First, all ecosystems are dynamic. They change over space and time in response to inputs of energy, new species, natural events, internal growth and development processes, and how people treat the land (Botkin 1990, Burgess and Sharpe 1981, Waring and Schlesinger 1985). Ecosystems are always changing, whether people cause the change or not. But most ecosystems are now influenced in some way by human activities or human artifacts.

Second, the capabilities of any ecosystem to sustain desired conditions of diversity, ecological services, and resource uses and values are a result of climate, soils, topography, biota, natural processes, human influences, and how large the ecosystem is. In general, the larger the ecosystem the more diversity, resilience, and productivity it can sustain. Existing ecosystem capabilities determine what is possible in a human time frame, say a generation to a century. Any longer than that the basic capabilities of ecosystems may change and our ability to predict outcomes is rather poor. Thus, an inventory of current conditions and trends is useful in determining what is likely in the near future. But such an inventory is not likely to indicate ecosystem conditions beyond a century or two into the future.

Third, the collective needs and aspirations of the people who depend on ecosystems in a particular area for their well-being determine the desired current and future conditions of those ecosystems. Obviously the desired conditions cannot lie outside the bounds of what is there now or what is possible given existing ecological capabilities, financial resources, and technologies. The differences between existing and desired future conditions of ecosystems identify possible management objectives, that is, what it will take to sustain or restore ecosystem conditions to their possible and desired states and flows of resource uses and values.

Ecosystems in a Landscape Perspective

Landscapes are the working scale for ecosystem management. A landscape is a large area composed of many different kinds of ecosystems. It has repeatable patterns of habitats, physical features, and human influences (Forman and Godron 1986). Landscapes are large enough that it is possible to integrate the protection and management of ecosystems at site, stand, and watershed scales. Because of their large size, landscapes often involve multiple land tenures and multiple zones of different land-use classes. Thus, ecosystem management at landscape scales is invariably a cooperative endeavor.

Landscape patterns result from both enduring, slow-changing features of nature (e.g., soils, climate, and topography) and more dynamic patterns of biotic communities, ecological processes, and disturbances that shape short-term temporal and spatial change. When we look at the earth from an airplane we see a snapshot of a landscape at a point in time. If we had numerous snapshots, representing repeated fly-overs, spaced several decades apart, we would see that vegetation patterns and human influences on the landscape change from one photo to another, like a kaleidoscope.

Disturbances superimposed on long-term patterns and processes in ecosystems set the context for the temporal diversity of life and its changes over time. In some places the pattern is a patchwork of different kinds of ecosystems: in one example, stand-replacement fires create a mosaic of forests and openings at the landscape scale; in another example, the pattern might be a fairly continuous forest cover. But a view from inside a continuous forest might still reveal a diverse ecosystem containing trees varying in age from very young to very old, multiple canopy layers, and a profuse understory wherever gaps exist in the canopy. Ecosystem dynamics in such a forest might result not from fires but from winds or the actions of age and diseases on individual trees. Landscape patterns will also change when global or regional climate change is sufficient.

BIOLOGICAL DIVERSITY AND ECOSYSTEM MANAGEMENT

The variety of life—plants, animals, and various microorganisms and fungi—and its many processes in ecosystems determines ecological capabilities. This variety is known by the term biological diversity (The Keystone Center 1991). It includes variation and variability in genes, species, plant and animal communities, and the many processes through which they are all interconnected through space and time.

Biological diversity is a valuable characteristic of ecosystems for ecological, economic, educational, and aesthetic reasons. It is key to the productivity and sustainability of earth's basic life support systems. It provides numerous current and future resources for human well-being. It provides opportunities for better understanding the myriad of relationships between people and their sources of existence. And it contributes greatly to the beauty and wonder of the world we live in. Biological diversity also has an ethical element: how well we conserve biological diversity demonstrates our respect for other forms of life and our commitment to the well-being of future generations (Leopold 1949).

Scientists do not know all the ecological roles or potential values of biological diversity. Nor do they understand all the processes that keep ecosystems functioning. It is not likely that they ever will. But complete knowledge is not necessary to understand that retaining the natural parts and processes of biodiversity is important for the future health and productivity of all ecosystems (Leopold 1949). How to do this in the face of a growing human population is the challenge (UNCED 1992).

Incorporating the conservation of biological diversity into ecosystem management requires actions aimed to achieve specific objectives for species, biological communities, and ecosystem conditions. A strategic framework for such actions and objectives has been developed for U.S. federal lands through a national policy dialogue (The Keystone Center 1991). Though the recommendations of the Keystone dialogue are not Federal agency policy at the current time, they have been adapted here to offer land and resource managers and scientists a framework of specific and measurable goals. To guide on-the-ground actions, the following goals should be reflected in land-use allocations, standards in land and resource management plans, and working guidelines for project activities in specific places.

Threatened or Endangered Species

Listed species are the most vulnerable officially recognized elements of biodiversity. A net decline in the number of listed species in the area covered by a plan or program is an ideal goal to consider in ecosystem management. To accomplish this, management must protect existing populations and habitats of listed species and restore them if necessary.

Viable Populations of Native Plant and Animal Species

Species whose demographic or habitat trends are negative but not yet to the point of endangerment may be the next most vulnerable elements of biological diversity. Some such species may even be more vulnerable than officially listed species. The ideal goal is to secure the places and functions of all native species in regional ecosystems before they reach the point where formal listing as a threatened or endangered species calls into play the extreme measures of protecting species under crisis conditions (Salwasser 1991).

To sustain viable populations of native species, habitats, human activities and artifacts, and wild populations of plants and animals must be managed to assure that populations of native species are numerous and well-distributed throughout their geographic ranges. This requires a combination of actions to protect, restore, and enhance sufficient kinds, amounts, qualities, and distributions of sub-populations and habitats. Especially important in achieving population viability is the perpetuation of multiple, interconnected, demographically resilient local populations; the characteristic genetic variation of the entire species; and the full range of the species' roles in ecological processes. Principles of conservation biology (Soule and Wilcox 1980, Soule 1986, Soule 1987) and especially the population viability analysis and management process described by Marcot et al. (in press) are useful in this task.

Native Biological Communities and Ecosystems

Rare, unique, or sensitive biological communities or successional stages (often highly productive sites, riparian areas, and mature or old-growth successional stages) are likely to be vulnerable elements of biological diversity in certain landscapes. Lands, human activities and artifacts, and wildlife habitats must be managed to assure that a network of representative native biological communities and developmental stages of ecosystems is maintained across the landscape. This may involve ecological restoration in some cases. Especially important are communities or assemblages of species that are rare or imperiled in the region or nation (Jenkins 1988). The matrix conditions of a landscape should provide essential resources for all species to the degree this is possible, including conditions needed for normal movement of plants and animals throughout the landscape and for the full range of ecological processes characteristic to the area. Where this is not possible, a specific network of sites and connections between them may be needed. The sites and connections must be sufficiently large and diverse to accomplish their intended purposes.

Structural Diversity

Natural elements of structural diversity such as snags, caves, fallen trees, and seeps provide habitats for many species that would not occur in an area without them. These elements can be jeopardized by intensive human uses such as fuelwood gathering, heavy livestock grazing, clearcutting, and water diversions. Elements of structural diversity should be perpetuated in qualities, amounts, and distributions within patches and across landscapes to assure their roles in sustaining desired conditions of ecosystem diversity, productivity, and resilience from site to regional geographic scales (Franklin 1988).

Genetic Diversity

The genetic variation of intensively managed wild plant and animal populations can decline if sufficient attention is not paid to the effects of human selection for various traits. Species and habitats, especially those of high commercial value and thus intensively harvested, should be managed to sustain natural levels of genetic variation within and among populations and the genetic integrity of representative and extreme populations (Ledig 1986, Millar 1987).

Resources Needed for Human Well-Being

Human well-being ultimately depends on natural resources. People will obtain those resources from somewhere. The key is to produce them in ways that do not lead to undesired environmental effects at local, regional, or global scales. If resources can be produced in ways that reduce human pressures on biological diversity in other places then resource production zones can have a positive overall effect on biodiversity conservation. High productivity sites such as flat ground with deep loamy soils, and featured species such as pines, firs, oaks, elk, and trout should be managed with state-of-the-art efficiency to sustain the production of resources needed by people, thus meeting human needs with minimal impacts on more fragile sites and sensitive species.

Ecosystem Integrity—Soils, Waters, Biota, and Ecological Processes

Any human activity has some effect on lands, waters, or biota. Ideally, these effects can be minimized through sensitivity to ecosystem integrity. Actions that are known to degrade site conditions or long-term ecosystem diversity, productivity, or resilience should be avoided if possible or mitigated promptly when not. The natural restorative powers of ecosystems should be employed in resource management activities. Consider the kinds, amounts, and distribution of living and dead organic matter to be left in ecosystems for long-term diversity, productivity, and resilience following resource harvest along with how much biotic production of the system is to be removed for human uses. This is essentially a principle of treating the ecosystem as "capital" and the production as "interest" (Rowe 1992).

Degraded Ecosystems

Biological communities, waters, and soils that have been damaged by natural events or past human actions should be placed under restoration and renewal programs, embracing the concepts and methods of restoration ecology and management (Bonnicksen 1988, Cairns 1986, Jordan et al. 1987, Jordan 1988).

There is more to the conservation of biological diversity in ecosystem management than identified in this framework but these actions are a reasonable start on a comprehensive conservation program.

HUMAN DIMENSIONS AND NEW PERSPECTIVES

The reason an ecosystem perspective is needed for land and resource management is simple. Continued growth in human populations and increases in their production, use, and disposal of resources are not matched by corresponding growth in the land base available to meet those demands under traditional resource management approaches while sustaining desired levels of environmental quality (Silver and DeFries 1990). Managers of wildlands and natural resources throughout the world, thus face a dual challenge that grows in difficulty with each passing year: to provide people with the resources needed to sustain their lives and well-being while minimizing the impacts of resource production and uses on the diversity, productivity, and resilience of the ecosystems from which those resources are taken or used (LeMaster 1992, Reid et al. 1992, United Nations 1992).

Forests as a Case

Let us take forests and woodlands as just one example of this challenge. Forests and woodlands now cover an estimated 31 percent of the planet's terrestrial surface (4.1 billion hectares according to the World Resources Institute 1990). This is about 66 percent of the forested area that existed prior to the industrial and public health revolutions of several centuries ago. Meanwhile, the number of humans has grown by 11 times: from an estimated 500 million people to about 5.5 billion.

In per capita terms, each global citizen had an average of about 12 hectares of forest resource in 1750, while in 1990, each had only about 0.75 hectares. For the U.S., the corresponding statistics are: 45 hectares of forest per person in 1600 down to 1.2 hectares in 1990 (Salwasser et al. 1992). Meanwhile global use of wood from these forests has been increasing at an average 2 percent per year for the past 40 years (Haynes and Brooks 1991). And this does not take into account fuelwood use, which is nearly impossible to estimate on a global scale.

The U.S. is a major force in global wood use and its impacts on forests (U.S. citizens use about 33 percent of the world's annual production of industrial roundwood). This has been reflected in several events and issues during the 1980's that caused the USDA Forest Service to explore new perspectives for managing the complex lands and resources that comprise the National Forest System. Global issues included uncertainties associated with climate change, loss of biological diversity, and the growing human population. National issues included controversy over logging and forest regeneration methods (mostly clearcutting and even-aged forestry); declines in forest health due to pollution, drought, insects, fire suppression, and past management practices; controversy over subsidies to the development of public resources (including grazing fees and timber sales whose financial returns do not cover administration costs); loss of old-growth forests; a growing number of endangered species; poor conditions of public rangelands; rising demands of people for all natural resources; and declining domestic supplies of resources for which demand was creating increased foreign dependencies such as oil and timber. At local levels, concerns for soil productivity, aesthetics of land management practices, water quality, the vitality of communities that depend on public land resources for livelihoods and jobs, large and largely uncontrollable wildfires, and new and conflicting uses of public lands fueled the flames of change.

New Perspectives for the National Forest System

As an example of agency leadership on ecosystem management, the Forest Service translated these issues into four reasons for a program to explore new perspectives in land and resource management during 1990-92 (Overbay 1992):

A. people need and want a wider array of uses, values, products, and services from public lands than in the past, especially, but not limited to, the amenity values and environmental services of healthy, diverse lands and waters;
B. new information and a better understanding of ecological processes highlight the role of biological diversity as a factor in sustaining the health and productivity of ecosystems and the need for integrated ecological information at various spatial and temporal scales to improve management;
C. people outside the Forest Service want more direct involvement in the process of making decisions about public resources; and
D. the complexity and uncertainty of natural resources management call for stronger teamwork between scientists and resource managers than has heretofore been practiced.

The Forest Service chartered the New Perspectives program to do five things: (1) learn how to better sustain diverse and productive ecological systems; (2) better integrate the different aspects of land and resources management; (3) improve the effectiveness of public participation in resource decision-making; (4) continue building partnerships between forest users and forest managers; and (5) strengthen teamwork between researchers and managers.

Moving Towards Ecosystem Management in the Forest Service

In Summer 1992, the Forest Service announced its intent to develop ecosystem management as a strategic approach for sustaining desired conditions of ecosystem diversity, productivity, and resilience for the multiple uses and values of national forests and grasslands. Ecosystem management is a process. It is not a goal. The goals for ecosystem management come from ecological capabilities of the land together with legal mandates and public needs and aspirations.

The Forest Service has a Congressional mandate to manage lands and resources entrusted to its care under the concepts of multiple use and sustained yield, without impairing the long-term productivity of the land (MUSY 1960). The Forest Service also has a legal mandate to conserve threatened or endangered species (ESA 1973, as amended) and to "provide diversity of plant and animal communities ... to meet overall multiple-use objectives" (NFMA 1976).

Research has shown that biological diversity is important to the long-term productivity and resilience of ecosystems, i.e., the land. Combining this knowledge with the agency's legal mandate, the Chief of the Forest Service, defined ecosystem management as follows: "an ecological approach will be used to achieve the multiple-use management of the national forests and grasslands by blending the needs of people and environmental values in such a way that the national forests and grasslands represent diverse, healthy, productive, and sustainable ecosystems" (Robertson 1992). Other federal and state land management agencies are implementing similar ecosystem management policies.

WORKING GUIDELINES FOR ECOSYSTEM MANAGEMENT

Field managers and scientists are now implementing ecosystem management through changes in agency regulations, national program direction, amendments to integrated land use plans, field projects that carry out the direction in those plans, research programs, and cooperative endeavors with conservation partners in universities, other government agencies, and the private sector.

At this point, some working guidelines for ecosystem management have evolved from New Perspectives projects. These guidelines remain open for refinement and are presented here to show the state-of-the-art in the early 1990s.

1. **Work Within the Scope of Natural Processes that Shape Landscape and Ecosystem Conditions.** Work within the ecological capabilities and natural processes of different ecosystems, maintaining as much diversity as possible and minimizing the energy costs of management to sustain or restore desired ecosystem conditions and functions.

Natural disturbances such as fires, floods, droughts, and storms are major forces which shape ecosystems and landscape patterns. These processes are the context within which long-term management strategies to sustain desired conditions of ecosystem diversity and productivity must be developed.

Be especially careful with soils and waters, particularly in sensitive areas such as wetlands, riparian zones, fragile sites, and rare species' habitats.

Always think about scale effects, both spatial and temporal, at least one scale higher and one scale lower than what you're working on and at least several generations into the future, more and longer if possible.

Think complex, model simple, and maintain options.

2. **Focus on End Results—Desired Future Ecological and Social Conditions** and the land-use classes and management actions that will best attain them. Use landscapes as a basic unit for planning and managing lands to meet specific objectives for conditions that will yield both desired future ecological conditions and desired economic and social goals while reconciling conflicts between competing uses and values.

3. **Coordinate Strategies for Conservation of Shared Resources.** Many natural resource issues and concerns cross jurisdictional lines. Examples include migratory fish and wildlife, wide-ranging endangered species, long-term regional timber supplies, air quality, and water flows. Regional-scale ecosystems are logical units in which to coordinate land uses and management actions to achieve desired conditions regarding these resources.

Complementary roles for different land tenures, including the legitimate rights of private land holders, may be blended by using existing authorities (Salwasser et al. 1987) or concepts such as Biosphere Reserves (Gregg and McGean 1985) and landscape linkages (Harris and Gallagher 1989).

4. **Get People Involved** in all aspects of public resource decision-making so that managers will know their needs and views; so that people will understand their personal responsibilities, what is possible, and what the relative tradeoffs are; and to obtain informed consent on the course of action selected.

Use consensus building and negotiated problem solving (Wondolleck 1988) as primary approaches to conflict management. People who are affected by resource management and conservation strategies must feel a strong commitment to being part of the solution.

5. **Integrate Information and Technology,** such as ecological classifications, inventories, data management systems, and predictive models, and use them routinely in landscape-scale analyses and conservation strategies. Agencies and affected interest groups and enterprises should contribute to common inventories of the basic conditions of soils, waters, and biota and share data and other information as appropriate to their missions and property rights. Inventories of biological diversity in the U.S. should build from the foundation of state Heritage Programs (Jenkins 1988) and multi-resource inventories conducted by various state and federal agencies. They should allow prudent choices to be made based on realistic assessments of needs and priorities for investment and protection actions (e.g., Scott et al. 1987, Scott et al. 1991).

6. **Integrate Management and Research** to continually improve the scientific basis of ecosystem management. Agencies, universities, and affected interest groups and enterprises should cooperate in long-term, interdisciplinary ecosystem research and development. Managers need practical tools and methods for planning and evaluating the expected effects of management options. They also need expanded choices for sustainable harvest and management of resources.

7. **Revitalize Conservation Education and Interpretation.** Agencies, universities, and affected interest groups and enterprises should cooperate in comprehensive programs of interpretation, education, and demonstration of ecosystem management. The result should be a better understanding among the citizenry about the effects of personal actions in sustaining desired ecosystem conditions and better support for the complementary roles played by different agencies and ownerships in overall conservation strategies.
8. **Develop, Monitor, and Evaluate Vital Signs of Ecosystem Health.** Agencies, universities, and affected interest groups and enterprises should cooperate in identifying and monitoring carefully selected indicators of ecosystem health and diversity, including conditions and trends of valued resources. Monitoring should be guided by the use of decision analysis tools (Maguire 1988, 1991) to ensure that the most vital information is being collected in useable quality and in a timely fashion for the specific purpose of adapting management based on new information (Holling 1978, Walters 1986).

A Bigger Role for Research

Research has a significant role in ecosystem management, including the use of scientific methods in understanding the basic capabilities of different ecosystems; discerning the needs and wants of people; setting ecologically, economically, and socially sound management goals; and designing monitoring systems to allow for periodic adaptation to new knowledge (National Research Council 1990, Lubchenko et al. 1991). However, scientists are not the only source of information for solutions to difficult political and social choices. For example, there are not unique or scientifically perfect answers for how a balance of goals and practices for ecosystem management should be struck. People's values, preferences, and aspirations are crucial factors in policy making.

The role of science in ecosystem management is to help define what is possible and what is desired: to shed light on how to best attain a desired set of conditions or benefits and help people understand the estimated costs, benefits, and consequences of alternative courses. To fulfill this role effectively, social, biological, and physical sciences must be integrated to reflect the complexity of how ecosystems actually function.

LANDSCAPE SCALE ECOSYSTEM MANAGEMENT: PRACTICAL EXAMPLES

Land and Resource Management

Landscape-scale ecological planning was being attempted in some areas in the U.S. during the 1970's, stimulated by the book Design with Nature (McHarg 1969). But environmental conflicts of the 1970's led to new laws and regulations which caused many landscape level planning activities to fall by the wayside. The "gridlock" caused by current applications of narrowly focused environmental laws, regulations, and "rights" has caused natural resource professionals to seek new ways of accomplishing the vision of McHarg (1969) and others who passionately believe that we should be able to develop harmonious ways to live with each other within a healthy environment.

Scientists provide knowledge, principles, and methods. Agencies and organizations provide leadership and establish policy. But, the practical application of knowledge and technology to implement policy is clearly in the hands of the professional at the field level. The practice of ecosystem management must include the art of applying science in the intelligent, responsible planning of ecosystem futures.

Many of the recent "experiments" in landscape level analysis, evaluation and planning were stimulated through the USDA Forest Service's New Perspectives Program. Establishment of the Landscape Ecology Research Work Unit at Rhinelander, Wisconsin has provided important information in methods of evaluating "natural" and "managed" landscape ecosystems. An application phase of the program involves land management planning by an interdisciplinary, multi-ownership cooperative team effort.

One of the earliest examples of ecosystem management in the national forests was the Shasta Costa Project on the Siskyou National Forest in Oregon. This case study illustrated the basic principles of ecosystem management identified previously in this chapter. It had limited success due to divergent expectations of the participants and the political uncertainty of public land use policies in the northwest (Salwasser 1992). Stumbling blocks were more social and political than scientific, though some observers erroneously perceived how scientific information was being applied to be the major barrier to successful implementation of the new principles (Frissell et al. 1992, Lawrence and Murphy 1992).

In the Northern Region of the Forest Service, the "Trail Creek Supplemental Information Report" (USDA Forest Service 1991) provided an immediate opportunity for a regional task force to explore ecosystem management principles for conflict resolution. Two other pilot projects have recently been analyzed and compared by a team including outside participants (O'Hara et al. 1993).

A recent publication from Oregon (Diaz and Apostol 1992) offers a process for developing and implementing land management objectives for landscape patterns. A unique aspect of the process is the evaluation of flows of animals and human uses across the landscape. A major concern has been expressed that landscape-level analysis has the dangerous (inefficient) potential of adding yet another cumbersome level of planning to government projects. However, landscape analysis can be a very efficient exercise when sufficient inventory information and GIS technology are in place.

Applications of ecosystem management at the landscape level are not limited to the USDA Forest Service. During 1992, Potlatch Forest Industries in Lewiston, Idaho (Steve Smith, Lewiston, ID Personal communication) began exploring the application of landscape-scale ecosystem management in support of their Forest Stewardship Program. With operational state-of-the art remote sensing and GIS technology in place, they are projecting the future conditions that will result from current land management activities. They can then evaluate expected future conditions against their Stewardship Goals, i.e., desired future conditions. The first pilot demonstration stimulated considerable internal professional discussion. A second pilot project is being undertaken in cooperation with adjacent public landowners who share checkerboard ownership in a 30,000 acre watershed. This represents progressive practice of ecosystem management by private and public sector parties and a great opportunity for "fishbowl visibility" by public organizations with contrasting management objectives but common ecosystem management concerns.

Education

Land and resource management is only one venue for the emergence of ecosystem management in practice. Undergraduate, graduate, and professional education are also adopting the concepts. The University of Montana, as one example among many, has been offering an annual continuing education program, "Ecology and Management of Forest Landscapes", for the past five years. Part of the training includes a landscape planning exercise for a 3,200 acre landscape within Lubrecht Experimental Forest. (This same area is used for a senior-level, conventional, integrated resource management planning exercise.) For the landscape shortcourse, students go through six basic steps:

1. Familiarity with the area through displays of information available in the GIS (topography, stand types, soils, vegetation habitat types, roads, wildlife distribution, etc.)
2. Establishment of five alternative management directions:
   A. No Treatment and Fire Control
   B. No Treatment and Natural Fire Allowance
   C. Optimize Biological Diversity (Using Silviculture & Fire)
   D. Mimic Natural Stand Conditions (Using Silviculture & Fire)
   E. Optimize Intensive Timber Production within Old-growth and silvicultural constraints.
3. Student teams develop written landscape prescriptions to implement a specific management direction (specification of **what** treatments, **when** to implement, and **where** to implement).
4. Utilize simple succession/treatment algorithms to move the stands through time and display future landscape maps at 30 year intervals for 120 years into the future. General summaries of multi-resource production and values can be generated to accompany the map displays.
5. Group evaluation of each of the alternative directions and prescriptions against a standard set of criteria that relate specific concerns addressed in current definitions of ecosystem management.
6. Students play the role of the public in advising Lubrecht Experimental Forest relative to our general management directions for that part of the Forest. Rather than rank the five alternatives, we ask them to identify one preferred alternative and rate the others as "unacceptable" or "acceptable".

The strength of this exercise for practical consideration is that it is simple, rapid, efficient, and easy to communicate to a general public audience. Simplified algorithms for predicting succession are adequate for the major questions being addressed. A long-term perspective on the future and demonstration of natural stand dynamics become self evident. Major issues of forest health, biodiversity, old-growth preservation, role of fire, silvicultural systems, sustainability of various items and human activities can be addressed. Major tradeoffs involved with single-purpose objectives become transparently obvious. Individuals have the opportunity to express and defend their personal priorities in a comfortable group setting. Since it is a classroom exercise, consensus is less threatening.

During the evaluation, we find it very difficult to address many stated "ecosystem principles" because they are difficult to quantify or measure. For other attributes, we recognize that certain inventory information would be crucial before making final decisions. However, the process is valuable in sending students out, inspired with ecosystem management thinking and awareness of new technology to tackle the difficult, almost **impossible** task of being a leader in the practice of ecosystem management. ("The impossible just takes a little longer." —Author unknown)

As theory concepts, principles and methodologies are debated at symposiums and workshops, we must have faith in the army of professionals waiting to practice ecosystem management, if we can continue to provide knowledge, methods and support for their job at hand. We can only hope that doors will remain open in the U.S.A. for dedicated professionals to continue to practice their honorable, selfless, profession for the "greatest good for the greatest number in the long run". "Those who say it can't be done need to get out of the way for those who are already doing it!" (Lee Iacocca?)

CLOSING THOUGHTS

The need for new perspectives in land and resource management gave people inside and outside the Forest Service a chance to try some new and some old thinking. Five themes have emerged:

A. Sustain diverse and productive ecological systems;
B. Integrate the different aspects of land and resource management, research, and conservation;
C. Improve the effectiveness of public participation in resource decision-making;
D. Build partnerships between resource users and resource managers; and
E. Strengthen teamwork between researchers and resource managers.

Sustaining desired ecological, economic, and social conditions in ecosystems that are managed for multiple purposes, such as the National Forest System and other public lands in the U.S., is a big challenge. But it is not an impossible task if people realize that no single objective can dominate ecosystem management at all geographic scales or even at the same site for all times. Success in sustaining desired ecosystem conditions will depend on having scientifically sound, economically feasible, and socially acceptable strategies for achieving combinations of ecological and social goals. For example, it will depend on meeting specific objectives for viability of native species and biological communities such as spotted owls, grizzly bears, elk, and tall-grass prairies. It will depend on meeting specific objectives for the characteristics of landscapes such as habitat conditions that are aesthetically pleasing and allow for free movement of plants and animals over time. It will depend on coordination among people responsible for species or resources that transcend administrative boundaries such as the spotted owl, large predators, eagles, migratory birds and mammals, and timber and mineral resources. It will depend on practical management standards for the desired characteristics of distinct patches in a landscape, such as diversity of species, structures, and functions provided by snags, fallen trees, riparian areas, and prairie barrens. It will also depend on better integration of research with management, especially in monitoring conditions pertinent to objectives.

These and other actions to sustain desired ecosystem conditions in multiple-use lands are changing traditional approaches to both multiple-use management and nature preservation. Some of these changes may result in higher management costs or to less access to resources for people who have high dependencies on those resources for subsistence or economic well-being. On the other hand, they may also result in lower management costs or in resource management being carried out in areas previously considered to be "off-limits" to human use of resources. Only site-specific assessments can show what the nature of the changes might be. In any case, changes in public land uses are often difficult to establish and implement both politically and logistically. But they are tractable if people decide they are willing to make them. This will require dedicated professional leadership and commitment, an eagerness to share in the excitement and potential of ecosystem management with publics, and a stable social setting that supports long-term common good.

The learning process on ecosystem management is far from complete. In fact, ecosystem management as a process for sustaining diverse, healthy, and productive land has just begun. To date, we have some principles, guidelines, tools, research and development programs, and several hundred practical demonstrations to draw from. These will expand in our continuing pursuit of new knowledge and technologies.

An ecosystem perspective on sustaining desired conditions of diversity and productivity in multiple-use lands is the right way to go. But it is insufficient by itself to sustain a harmonious relationship between people, land, and resources. Regardless of how well ecosystem management works, other actions will be needed to bring people and land into a better harmony.

Needed: A Globally Responsible Conservation Ethic

Foremost among these actions, Americans must become more conservative in how they (we) produce, use, and dispose of natural resource products. American behaviors regarding consumption and waste of resources are major forces of change in the global ecosystem (Silver and DeFries 1990). Americans need to renew their conservation ethic to bring balance to the complementary roles of managing ecosystems, producing resources, and conserving resources (Postel and Ryan 1991). A recent public opinion survey found that, despite an economic recession, the American public is prepared to make the commitments such an ethic entails (The Roper Organization 1992).

Sustaining diverse, productive, and resilient ecosystems in the U.S. is important. But it must be balanced by a commensurate change in how and where Americans get and dispose of their resources. The potential off-site effects of protecting ecosystems in one place, such as old-growth forests or wildlife refuges in certain regions of the U.S., while continuing profligate use of resources that are produced in other places, such as oil from the middle east or timber from Canada, are not well understood by many people.

All of this planet's ecosystems are ultimately interconnected. The potential for interregional and international transfers of the economic, ecological, or social effects of where resources are produced and where they are used highlight the veracity of Garrett Hardin's (1985) comment that "it is not possible to do only one thing in an ecosystem." U.S. resource policy needs to start paying as much attention to the off-site effects of actions or inactions we take to protect nature or produce resources as we do to the on-site effects. And we need to think more about long-term dynamics in ecosystems. Things do not stay fixed or in the same place over time. Again we turn to Garrett Hardin (1985). His key ecological question, "and then what?" is a clue to how citizens and ecosystem managers must think when they think they are closing in on a simple solution to a complex problem.

Needed: A New Model for Conservation Science

Finally, to shape ecosystem perspectives on land and resource management, especially on the linkages between ecological, economic, and social factors that an ecosystem view implies, social, biological, and physical sciences must become better integrated (National Research Council 1990, Lubchenko et al. 1991). We will not learn how to sustain diverse, healthy, and productive ecosystems if we continue to pursue only traditional disciplinary sciences and education whether they be oriented to biological, physical, or social goals.

Ecosystem perspectives on sustainable resources management have the capability to bring forth a new model for developing the scientific basis of conservation: interdisciplinary teams of researchers working hand-in-hand with managers, educators, and citizens to address both short and long-term dynamics in the many dimensions of relationships between people and the land. The working principles for such a model, known as adaptive management, have been evolving for nearly 20 years now (Holling 1978, Walters 1986). It is time to make the adaptive management model standard procedure for sustaining diverse, healthy, and productive ecosystems.

ECOSYSTEM MANAGEMENT: A PROCESS FOR SUSTAINING DESIRED CONDITIONS OF ENVIRONMENTS, COMMUNITIES, AND ECONOMIES

Ecosystem management employs a full spectrum of land-use classes and resource management practices—ranging from preservation to sustainable production to restore and sustain diverse, healthy, productive ecosystems. Four principles guide the practical development of ecosystem management (adapted from Robertson 1992):

1. **Protect the land** by restoring and sustaining the integrity of its soils, air, waters, biological diversity, and ecological processes, thereby sustaining what Aldo Leopold (1949) called the land community and what we now call ecosystems.
2. **Meet the needs of people** who depend on resources of the land for food, fuel, shelter, livelihood, and inspirational experiences.
3. **Improve the well-being of communities, regions, and nations** through efficient and environmentally sensitive production and conservation of natural resources such as wood, water, minerals, energy, forage for domestic animals, and recreation opportunities.
4. **Seek balance and harmony between people and land** with equity between interests, across regions, and through generations, meeting this generation's resource needs while maintaining options for future generations to also meet their needs.

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Incorporating Landscape Ecology Concepts in Forest Management: Forest Landscape Analysis and Design

Nancy M. Diaz and Dean Apostol[1]

Abstract — A fundamental aspect of Ecosystem Management is that sustainability of ecosystem structures and processes is ensured in planning, designing and implementing land management strategies. The authors have developed a process for describing sustainable "target future landscapes" for National Forests, utilizing the concepts and terminology of landscape ecology. The process consists of Analysis and Design phases. The Analysis phase involves compiling information regarding 1) existing and potential landscape elements; 2) landscape flows or processes; 3) characteristics of natural disturbance processes and succession, and their influence on landscape patterns; and 4) linkages outside the landscape. The resulting information is used, along with existing legal and policy constraints on landscape patterns and on values expressed by the public, in a Design phase to develop narrative objectives regarding the kinds and arrangement of landscape elements desired to sustain landscape ecosystem function, and also to meet public expectations of aesthetic, recreational and economic benefits. A spatially explicit "master plan" is then developed, utilizing landscape architecture design techniques to give concrete form to the narrative objectives. This paper describes the application of the Forest Landscape Analysis and Design process to an 8800-acre area on the Clackamas Ranger District of the Mt. Hood National Forest in western Oregon.

INTRODUCTION

Using an "ecological approach" to forest land management requires that the basic structures and processes of ecological systems be sustained, such that both their intrinsic value for biological diversity and their utility to humans are protected (Znerold and others 1992). Alteration of landscape patterns through human activities (logging, fire suppression and conversion of forests to agricultural and urban land uses) has resulted in serious questions regarding the ability to sustain character and function of Pacific Northwest forest ecosystems. In the belief that the USDA Forest Service's Ecosystem Management initiative offers an opportunity to formulate solutions, we have proposed a process for developing sustainable landscape patterns, that protect and perpetuate the structures and processes of landscape ecosystems. This paper briefly describes the process of Forest Landscape Analysis and Design (FLAD) and how it was applied to an 8800-acre area ("Leoland") on the Clackamas Ranger District of the Mt. Hood National Forest. A more expanded version of this discussion can be found in Diaz and Apostol (1992).

[1] *USDA Forest Service, Mt. Hood National Forest, Gresham, Oregon.*

Objectives of Forest Landscape Analysis and Design

In Ecosystem Management, we are as concerned with the condition in which we LEAVE the land, as in what we TAKE away (commodities and benefits). Thus, the intent of the FLAD process is to allow for National Forest landscape patterns (what we LEAVE) to be developed in a purposeful manner that is informed by an understanding of and commitment to the landscape as an ecological system. This means that clear objectives about the kinds and arrangements of structural elements (matrix, patches, corridors) of landscapes must be formulated and tested against what is known about the sustainability of different patterns and the processes that interact with them.

We also wanted the FLAD process to facilitate integration of physical, biological and social factors. We feel it is important to understand BOTH how landscape ecosystems function in an undisturbed state (because this generally provides the best framework for understanding sustainability) AND how they function with the introduction of humans (because this is the real world).

Finally, we hoped to create a process that most land managers were capable of implementing in a realistic amount of time, using available resources. The process relies heavily on information that is already collected for other purposes, and emphasizes qualitative analyses.

About Leoland

The landscape used as an example in this paper is referred to as "Leoland". It is a roughly rectangular area of approximately 8800 acres on the west flank of the Cascade Mountains in northwestern Oregon, on the Mt. Hood National Forest. Within a 1-1/2-hour drive of most of the Portland metropolitan area, Leoland is popular with hunters, campers, picnickers and hikers. Logging has occurred in Leoland since the 1940's, when a mill was located within the area. The mill site has since been converted to the Timber Lake Job Corps Center. Other settlements within Leoland include the Ripplebrook Ranger Station and associated residences, the Oak Grove Work Center, and Three Lynx, a small community associated with a hydroelectric power generating facility. Access to Leoland is provided by Oregon Hwy. 224, along the Clackamas River

The southwest 1/3 of Leoland is an earthflow with rolling topography and several seeps and wetlands. The landforms rise steeply to a high elevation plateau that stretches some distance to the north and east of Leoland. The northwestern portion of Leoland is occupied by the Cripple Creek drainage, a tributary of the Clackamas River. Bisecting the Leoland landscape (from northwest to southeast) is a pipeline that feeds the hydropower plant at Three Lynx.

THE FOREST LANDSCAPE ANALYSIS AND DESIGN PROCESS

Figure 1 shows the eight steps of the FLAD process. There are two basic parts to the process, an ANALYSIS phase (Steps 1 through 5) used to gather information about the structures, processes and interrelationships of the landscape, and a DESIGN phase (Steps 6 through 8) in which landscape pattern objectives are developed and spatially represented.

Analysis Phase

This first phase of the FLAD process uses the basic structure/function terminology of ecosystems science in combination with the concepts of landscape elements and flows of Forman and Godron (1986) to organize information about existing landscape patterns and processes (Steps 1 through 3). In addition, "natural" agents of change and their effects are described (Step 4) to provide a picture of the "range of natural conditions" that might occur in a sustainable landscape. Finally, a larger-scale view (Step 5) is taken, to describe the context of the surrounding landscape and processes that occur over a larger area.

Step 1: Landscape Elements.— In this first step, structural units of the landscape are described as matrix, patches and corridors. The process is basically one of delineating areas based on plant community or vegetation type, successional stage, within-patch structural characteristics and ecological capability or productivity. The intent is to describe those elements of the landscape that interact differentially with important landscape processes or flows.

For Leoland, the following landscape elements were mapped:

- Matrix (late successional conifer forest)
  - Large timber stands (DBH 21")
  - Small timber stands (DBH 11-21")
- Patches
  - Shelterwood harvest units
  - Closed sapling/pole plantations
  - Open sapling/pole plantations
  - Clearcuts - shrub/forb successional stage
  - Talus/rock outcroppings with shrubs, forbs, grasses
  - Talus/rock outcroppings with scattered trees
  - Fens/bogs with shrubs, forbs, grasses
  - Red alder swamps
  - Rock quarries
  - Altered wetlands (e.g., pasture)
  - Pipeline corridor
  - Developed sites
  - Lakes
- Corridors
  - Roads
  - Trails
  - Pipeline
  - Forested riparian corridor (Cripple Creek)

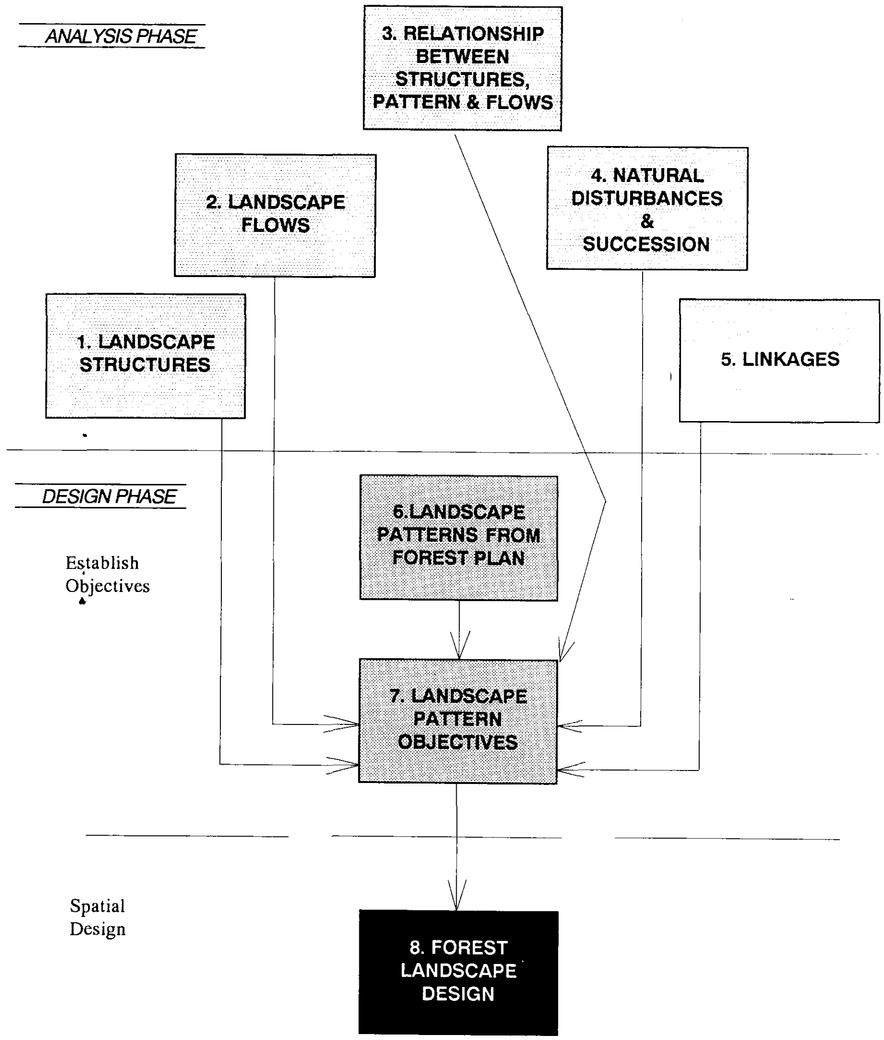

Figure 1. — The Forest Landscape Analysis and Design process.

Step 2: Landscape Flows.— Landscape flows are those phenomena that move across or through landscapes. They can be energy or matter, living or non-living. Examples are water, wind, people, grazing animals, seasonal wildlife migrations, etc. Our approach is to focus on those flows that are important to the character or function of the landscape ecosystem, or that are highly sensitive to human manipulation of landscape patterns. In this step it is important to document the location, pathway, direction and timing of landscape flows, and the extent to which they are dependent on certain landscape elements or patterns (e.g., are disrupted by fragmentation).

In the Leoland example, we described four landscape flows:

- Elk - In summer occupy the upper plateau area, in winter migrate into the wetland portions of the earthflow. The combination of abundant forage in wetlands and other openings, and adjacent late successional forest for cover makes this particularly high quality winter range.
- Deer - Very abundant throughout Leoland. Use ridges in northern part of Leoland to migrate between summer and winter range. Good forage in early successional clearcuts.
- Water - Flow is primarily northeast to southwest, into either Cripple Creek or earthflow area. Controls the pattern of wetlands in earthflow. Rain-on-snow events may occur at mid-elevations.
- People - Dispersed recreation use focussed on main road systems and trails leading to roadless area to northeast. Hunting is especially popular in winter range area, partly due to dense network of old roads (now passable by foot only).

Step 3: Relation between Landscape Structures and Flows.— In this Step, a systematic assessment of the interactions between landscape structures and flows or processes is conducted. The central question here is: How do the individual landscape elements (matrix, patches, corridors) and their arrangement affect (foster, inhibit, facilitate, direct, etc.) various landscape flows. The purpose is to produce a view of the functional character of the landscape. This Step can be as simple as producing a table with landscape elements as rows and flows as columns. Table 1 illustrates an abbreviated portion of the table that was developed for the Leoland landscape.

Table 1. — Interaction between selected structures and flows - Leoland landscape.

LANDSCAPE ELEMENTS	LANDSCAPE FLOWS	
	Deer	People
Large timber	Optimal cover, important late & early season habitat; forage in canopy openings	Visually "forested", hiking, commercial uses
Closed sapling/pole	Some thermal cover	Little commercial or recreational value
Rock patches	Good forage (shrubs)	Natural visual & vegetative diversity, view points, fall color
Road corridors	Harassment when open, travelways where closed or light use	Essential travel corridor

Step 4: Natural Disturbances and Succession.— At the conclusion of Step 3, the qualitative view of the character of the existing landscape is relatively complete. But in order to fully understand the landscape as an ecological system, processes that produce changes in that character must be addressed. Step 4 poses the following questions:

What agents of change would have existed in the unmodified (by humans) landscape?

What would their effect have been on the arrangement, composition, size and shape of patches, connectivity and characteristics of the matrix?

How might "natural" landscape patterns have influenced the behavior of natural disturbance phenomena?

The purpose of this Step is to frame the "possibilities" of the landscape. If natural landscapes provide a model of sustainability, then objectives regarding landscape patterns must relate to conditions within the natural range (USDA Forest Service, Northern Region 1992). Thus, the information generated in this Step is fundamental to creating target landscape patterns in the Design Phase.

In the Leoland example, fires and earthflow events are considered to be the mechanisms that have the greatest influence on natural landscape patterns. To gain a view of the effects of fire, historic photographs that predate logging, road building and fire suppression were analyzed. It was readily apparent that landforms exerted strong control on how fire created different combinations and arrangements of matrix and patches. On the earthflow and lower slopes, fires caused a diverse, patchy pattern of stands of various ages as they "meandered" across the rolling topography. Recent fires appeared to have been rather low in intensity, resulting in patches of many different sizes, curvilinear edges and abundant residual live trees. In contrast, on the high elevation plateau, fires appeared to be more intense, creating large areas of evenaged stands with relatively uniform structure. On the steeply sloping midslope portion of Leoland, fires appear to have burned perpendicular to the contour, most intense on exposed ridges. This resulted in a "finger" pattern of rather linear stands in protected drainages, with open ridges in between.

The effects of earthflow on landscape patterns in Leoland are more subtle than those of fire, but nevertheless have made a major contribution to the biological diversity of the landscape. The topography of the earthflow is rolling, and numerous wetlands of varying sizes occupy sites of concave relief. In addition, rock outcroppings and talus slopes are found throughout the area. The variety of these features and the habitats they create add greatly to the diversity of plant and animal species found in Leoland.

Step 5: Linkages.— The final Step of the Analysis Phase addresses the context of the landscape within the surrounding environment. No landscape delineation can completely circumscribe all the processes and flows present. Therefore, this Step poses the question:

What landscape flows or processes cross the borders, and what is the role of the various landscape elements in this transfer?

In the Leoland example, we analyzed external linkages for the four flow phenomena described in Step 2 - deer, elk, water and people. For example, we found that seasonal elk migration occurs by two major routes from two different summer range areas, and that it is facilitated by a combination of landscape elements that provide both forage and cover. Mapping these linkages uncovered strong ecological ties between Leoland and the Shellrock Creek drainage to the east, and the Oak Grove Fork drainage to the south.

Design Phase

The Design Phase of the FLAD process is a very different task from the Analysis Phase. It requires answering the question "what SHOULD be?", while the Analysis Phase focussed on "what is?", "what was?", and "what could be?". The essential difference is that while the Analysis Phase is primarily an enumeration of facts about the landscape, the Design Phase introduces human values and requires resolution of conflicts and expectations. It is frustrating that landscape pattern objectives do not spring forth from the Analysis Phase, apparent and agreed upon by all. We therefore developed the Design Phase, in an attempt to systematize the steps of negotiating a solution for sustainable landscapes, based on data from the Analysis Phase. Subjectivity cannot be entirely avoided, however, and since values must be part of the discussion, the Design Phase provides a logical place for active participation by members of the public.

The basic sequence of the Design Phase is to first document objectives for landscape patterns that are already formulated in existing plans and policies (Step 6), then to develop narrative objectives for landscape patterns based on a combination of the Analysis Phase, Step 6, public views about local resource issues, and reports from individual resource specialists (Step 7), and finally to spatially represent the narrative objectives on the real-life three-dimensional landforms present in the landscape (Step 8).

Step 6: Landscape Patterns from the Forest Plan.— For National Forests, Forest Land Management Plans provide direction for carrying out various management activities. Since the focus of the FLAD process is landscape pattern, the interest here is in Forest Plan direction that refers specifically to such items as size of openings, adjacency constraints, proportion of watersheds allowed to be in an open canopy condition, fragmentation, and so on. At the time most Forest Plans in the Pacific Northwest were written, the importance of landscape patterns was not as well appreciated as it is now, therefore Forest Plans vary considerably in the amount of attention given to this subject.

The Mt. Hood National Forest Land Management Plan allocates various portions of Leoland to five different Management Area categories: Clackamas River Wild and Scenic River Corridor, Earthflow/Winter Range, Scenic Viewshed, Special Interest-Scenic and Timber Emphasis. Each of these categories has specific management direction that sets a "theme" for the Management Area (for example the theme of the Scenic Viewshed category is to provide a natural-appearing landscape view from Hwy. 224), standards and guidelines for management activities and a statement of "desired future condition". In some cases this direction is specific to landscape patterns, for example:

Scenic Viewshed - should appear primarily forested, may have openings that appear natural and are in harmony with the landforms

Earthflow/Winter Range - A matrix of mature and young forests with scattered small openings. No more than 10% of the area in an open condition at one time. 25% or more of the area should be in large timber, in blocks of 30 acres or larger. Openings restricted to 10 acres or less.

Timber Emphasis - Fragmentation should be minimized. Openings should vary in size from 20 to 40 acres. The pattern will be patchy, with a mosaic that contains a full range of successional stages.

In other cases, the Forest Plan direction is vague with respect to landscape pattern, for example:

Wild and Scenic River Corridor - Evidence of human activities should not dominate the landscape.

It may become apparent in this Step that a different allocation of lands to Management Area categories does a better job of ensuring sustainability. In such cases the FLAD process provides good information with which to pursue adjustments to existing Forest Plans.

Step 7: Landscape Pattern Objectives (Narrative).— This Step poses the question: "What kinds, sizes, shapes and arrangements of matrix/patches/corridors are desirable in the landscape, based on the processes and functions we hope to sustain?" Answering this question requires a strong tie to the information developed via the Analysis Phase (which addresses the "kinds, sizes, shapes and arrangements" that will sustain "processes and functions"), but also involves a high degree of subjectivity (the "we hope to" part of the question). Sources of information for this Step include not only Step 6 and data from the Analysis Phase, but also information on social values, public views on resource issues, and any additional resource data from specialists' reports.

The following questions may be useful in getting to specific landscape pattern objectives:

Are there rare, unusual, critical or unique landscape elements that should be protected or enhanced (e.g., wetlands, migration corridors, old growth stands, etc.)?

Where in the landscape is connectivity desired?

To what extent and where is it desirable to mimic natural patterns, or restore natural processes?

Are there places where fragmentation should be minimized, or where a high degree of edge and contrast is desirable?

Is there a desirable proportion of various successional stages within the landscape?

In the Leoland example, we used the following sources to answer the above questions: Forest Plan direction from Step 6; the Analysis Phase, particularly Steps 3 (Relationship between Landscape Structures and Flows) and 4 (Natural Disturbances and Succession); a variety of maps, reports and personal observations of resource specialists; and a report on significant resource issues that had been developed earlier from public comments. Some examples of our resulting narrative landscape pattern objectives are below (the examples are a small subset of the total list, which can be found in Diaz and Apostol 1992):

Special Interest/Scenic Mgt. Area: A diverse and highly textured pattern of "fingers" and patches of forest interspersed with irregular rocky openings on the steep midslope area.

Wild and Scenic River Corridor: A forested corridor, emphasizing old growth stand characteristics.

Scenic Viewshed: In the western portion of Cripple Creek, a forested matrix with a few small openings that emulate natural rock outcroppings. North slope of the drainage retains closed canopy forest (thinnings or harvest of small groups or trees is allowed). In upper portion of Cripple Creek, larger openings may be made within the forest matrix, contoured along landforms.

Earthflow: Irregularly shaped small openings (less than 10 acres), emulating patterns from natural fires in this area. Wetlands surrounded and connected by late successional forest stands.

Step 8: Forest Landscape Design. — This Step gives spatial form to the narrative objectives, in the context of the actual landforms. Design techniques borrowed from the discipline of landscape architecture (similar to "master planning") are utilized in the FLAD process to 1) convert the narrative objectives into "design elements" and 2) position the design elements on three-dimensional topographic features. It must be emphasized that the purpose of this step is NOT merely to make the forest "look pretty". While visual esthetics may or may not be a primary concern in a particular landscape, the intent here is to produce a spatially-explicit design that meets the intent of Step 7, and the underlying goal of landscape ecosystem sustainability.

This Step involves first preparing a "landform analysis" which assesses the dominance of various geomorphic features, and produces a skeleton upon which the design elements may be "draped". The design elements consist of areas within which a common landscape pattern is desired. Next a map of "opportunities and constraints" is prepared, which reflects the "givens", or important landscape flows or structures that are not flexible as to location (e.g., location of an interior old growth forest block, or an important migration route). Building further, the target matrix area is sketched in, and then various configurations of patches are used to fill in the non-matrix areas. Through successive iterations, a map emerges that allocates various sectors of the landscape to design blocks that have a particular landscape pattern. As a final step, a rough sketch may be made that depicts what the landscape might look like when the target landscape is implemented. It is highly desirable, when doing Step 8, to work back and forth between map (two dimensions) and perspective (three dimensions) views. The two views present very different kinds of information, both of which are essential in testing how well various flows and processes are sustained by the target pattern, as well as giving a visual impression of how it will "look".

For Leoland, five design blocks were developed from the Step 7 narrative objectives, and placed on the landforms via the procedures described above. The Cripple Creek drainage, parts of the earthflow/winter range and the Special Interest/Scenic portion of the high elevation plateau are characterized as "unfragmented forest with old growth characteristics"; this constitutes the matrix of the future landscape (note: this is not an unmanaged reserve; thinnings and small openings are allowed). In the balance of the earthflow area, and on the midslopes, the prescribed pattern is "patchy forest with five to ten acre openings, 60% closed canopy forest". On the Timber Emphasis portion of the high plateau, the target pattern is "patchy forest with larger openings managed for huckleberries". Smaller midslope areas dominated by natural rock openings and talus are to retain their natural character, and restoring fire to maintain their diversity will be considered. Finally, the developed areas within Leoland will retain their character, but natural vegetative diversity will be restored where possible. The design blocks were configured to fit the landform features, and to take advantage of existing landscape elements or patterns that already conform to the target conditions.

CONCLUSION

The product of the FLAD process, the "target landscape", may be used for a variety of purposes. The main intent has been to provide a framework for initiating projects and activities that protect, restore or enhance sustainability of landscape ecosystems, and that produce benefits to humans. Use of the FLAD process to produce target landscape patterns provides a good forum for negotiating among various interests about what is desired from our National Forest landscapes. The primary limitation is lack of information about landscape processes and their relationships to landscape patterns. Particularly, the role of disturbances such as fire and insect infestations, and the range of conditions that are sustainable, seems critical to our understanding of the function of landscapes. Our hope is that the FLAD process, or at least its central logic, will lead to an appreciation of the need to develop this understanding, and ultimately to better protection of the diversity and health of National Forest landscapes.

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A Distant Perspective: Approaching Sustainability in a Regional Context

Carol A. Wessman[1] and Elizabeth M. Nel[2]

Abstract — Ecosystem dynamics are influenced by the landscape mosaic of which they are a part, as well as by the regional context of the landscape itself. The interconnection of ecosystems through mechanisms such as atmospheric and hydrologic transport makes them susceptible to broadscale neighborhood influences. The large-scale perspective provided by remote sensing promotes the understanding of such regional influences and, hence, the management of ecological systems. Understanding pattern state and dynamics can assist in identifying and monitoring anthropogenic perturbations that alter ecological processes and render ecosystems unsustainable. Such monitoring and change detection is facilitated by the repetitive measurement capability of satellite sensors. If circumstances that threaten the sustainability of ecosystems are to be recognized, knowledge of key ecosystem processes operating across the landscape is vital. Satellite data can be coupled with ecosystem models that calculate variables such as photosynthesis, evapotranspiration, respiration, decomposition, and biogeochemical cycling. Remote sensing of canopy chemistry can shed further light on natural ecological gradients such as soil fertility and nutrient availability across the landscape. The nature of remotely sensed data generates a new body of theory that requires a reevaluation or an expansion of ecological understanding. Successful ecosystem management will require large scale perspectives incorporating remote sensing technology and ecological theory.

INTRODUCTION

A new landscape-based approach to land management and sustainability is gradually developing within the U.S. Forest Service. "Ecosystem management", as opposed to the past timber-based strategy, has been proposed to facilitate the use of public lands in a sustainable manner (see Salwasser 1993, in these proceedings; Robertson 1992, memo to Regional Foresters and Station Directors). Even the "keystone species" concept in the preservation of biodiversity is being reconsidered in the context of the landscape; ecosystem/landscape-level approaches may supplement or in some cases replace single-species management (Franklin 1993; see Urban 1993, Diaz and Apostol 1993 in these proceedings). Landscape ecology takes a broader view of management by extending beyond the narrow boundaries of a forest stand, for example, to include the surrounding matrix. While certain natural resources are presently managed to promote sustainability (e.g. forestry and agriculture), current research efforts and management practices are inadequate to deal with ecological systems involving multiple resources and multiple ecosystems at large spatial scales (Lubchenco et al. 1991). Multi-use management based on an understanding of the structure, functioning and resiliency of natural systems across spatial and temporal scales is required to assure sustainability. Analysis of ecological patterns is possible with geographic information system (GIS) technology and remote sensing and will be crucial in developing objectives and prescriptions for management of ecosystem sustainability.

[1] *Carol A. Wessman is an Assistant Professor with the Environmental, Population, and Organismic Department and the Cooperative Institute for Research in Environmental Science at the University of Colorado, Boulder.*

[2] *Elizabeth M. Nel is a Professional Research Assistant in the Center for the Study of Earth from Space, Cooperative Institute for Research in Environmental Science at the University of Colorado, Boulder.*

The contribution of remote sensing to ecosystem studies ranges from empirically-based classification and mapping of land cover types to quantitative characterization of radiative transfer and energy balance. Statistical classification of digital imagery is used to describe spatial patterns in land cover types, their location, area, and change over time. Process-level questions require explicit linkages between the ecosystem function under study and the structure of the landscape in space and time. Quantitative remote sensing of parameters that represent such links provides information on dynamics at spatial and temporal scales previously inaccessible to study.

This paper reviews ecosystem parameters that are currently and potentially retrievable from remote sensing data (Table 1). The role of remote sensing in describing ecological structure, function, and change is discussed in the context of sustainability.

Table 1. — Ecosystem parameters sensible from space. Attributes of landscapes demonstrated to be sensible from space or, from limited studies, show strong potential for direct observation.

Plant Ps/Respiration	Carbon Storage Vegetation and Soil	Decomposition (Soil respiration)	Trace Gases
Photosynthetic capacity	Biomass	Litter input	Land cover type
Leaf area index	Land cover type	Foliar chemistry	Photo-synthesis
Greenness	Vegetation height		
APAR	Vegetation spatial distribution		

REMOTE SENSING OF ECOSYSTEM STRUCTURE

Managing for sustainability is predicated upon knowledge of the baseline structural parameters within the landscape. Multiscale studies of landscape pattern provide a powerful means for developing regional understanding of the processes that define a landscape's spatial characteristics and the factors that bring about their change. Definition of dissimilar patches within a landscape provides information on surface cover types, their spatial interdependency, and the changing mosaic over time. While landscape patterns will not always be uniquely related to particular ecosystem processes, they can in many instances be indicative of the dynamic interaction of biotic and abiotic factors, natural or anthropogenic in origin. Structural characteristics of the vegetation canopy, such as leaf area and gap frequency, can also be indicative of the state and health of an ecosystem.

Remote sensing of vegetation structure and function is largely based on the theory that plant growth is related to the fraction of incident radiation absorbed by the canopy and the dry matter:radiation quotient (an "efficiency" coefficient defining the carbon fixed per radiation intercepted) (Monteith 1972, 1977). Radiation interception properties of plants are strongly influenced by chlorophyll; its unique absorption of energy in the red (R) spectral region relative to the highly reflected near-infrared (NIR) region distinguishes live vegetation from soil and other non-photosynthetic materials. The spectral reflectance features of vegetation are controlled largely by leaf pigments, leaf cell structure, and leaf water content (Figure 1).

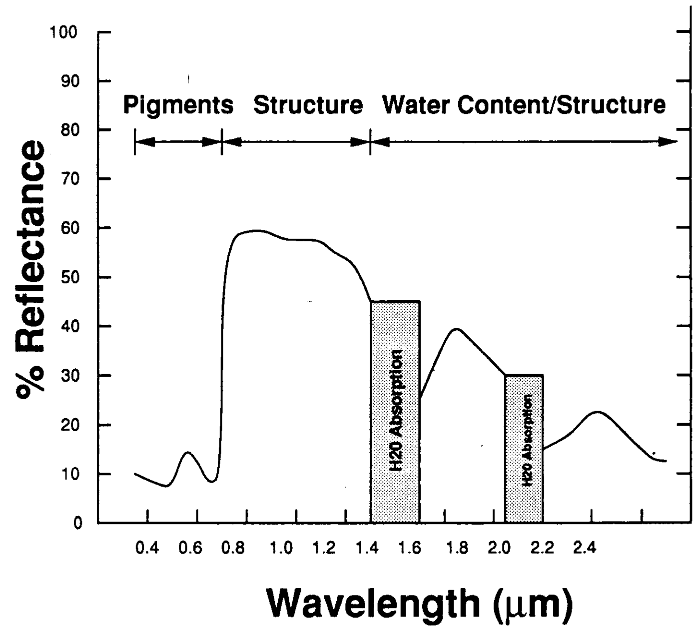

Figure 1. — A typical reflectance curve for healthy vegetation shows absorption features at 0.48 and 0.68 um, points of strong chlorophyll absorption. The reflectance at 0.52-0.60 um indicates the green portion of visible light which is not absorbed. The contrast between chlorophyll absorption and the strong reflectance feature extending from approximately 0.75 to 1.3 um characterizes healthy leaf tissue. Atmospheric water vapor absorption occurs at 1.4 and 1.9 um.

Early field studies investigated the near-linear relationships between spectral reflectance indices based on measurements of red and NIR reflectance (e.g., a simple ratio NIR/R or the Normalized Difference Vegetation Index (NDVI = (NIR-R)/(NIR+R)) and standard measurements of the canopy properties of biomass, leaf area and photosynthetically active radiation (PAR) absorbed by the canopy (Tucker 1979, Hipps et al. 1983, Asrar et al. 1984, Hatfield et al. 1984). Remote sensing of the amount of leaf surface area available for gas and moisture exchange (described by leaf area per ground area — leaf area index (LAI)) is of particular interest to the ecological community. Vegetation indexes (VI) are asymptotic with respect to LAI as the signal saturates (Asrar et al. 1989, Peterson and Running 1989), but linearity can extend from LAIs of 2 to 6 for crop and grassland canopies (Tucker 1977, Asrar et al. 1984, Ripple 1985) and up to approximately 8 for coniferous forests (Peterson et al. 1987, Running et al. 1989). Ground measurements of canopy transmitted light have gained in importance for rapid characterization of canopy leaf area and architecture (Norman and Campbell 1991). These measurements greatly enhance capabilities to acquire adequate ground calibrations for satellite measurements (Pierce and Running 1988, Gower and Norman 1991, Lathrop and Pierce 1991).

The synoptic coverage provided by satellite sensors has been proven to be useful for determination of areal extent, distribution, and change in land cover types over time. Single and multidate Landsat data have been used routinely to classify vegetation community types with accuracies on the order of 70 to 90% (e.g. Botkin et al. 1984; Franklin et al. 1986; Bolstad and Lillesand 1992). Information on canopy and landscape structure can be derived from studies of the texture (i.e. spatial variation in reflectance) within an image (e.g. Otterman 1981, Franklin and Peddle 1990, Briggs and Nellis 1991). Variance in shade versus illuminated vegetation has been used to quantify the number and spacing of forest trees (Franklin et al. 1986; Li and Strahler 1986, 1988) which, when monitored over time, could be used to track forest stand dynamics such as gap formation and regrowth. Other digital image processing techniques such as principal component analysis and image ratioing enhance spectral differences between materials and can be used to identify substrates with a particular characteristic (Sabins, 1987). Spectral mixture analysis provides a means to estimate the spatial cover of vegetation in a sparse community, independent of the spectral characteristic of the substrate (Ustin et al. 1986; Smith et al. 1990a, 1990b).

MEASURES OF ECOLOGICAL FUNCTION

Biophysical Processes of Photosynthesis and Transpiration

Familiarity with ecosystem processes operating across the landscape is vital if events that threaten the sustainabilitiy of the ecosystem are to be recognized. Since observations in red and near-infrared spectral regions are indicative of factors related to chlorophyll density and indirectly to carbon fixation rates, these observations should provide information on photosynthetic capacity (Tucker and Sellers 1986). In this context, photosynthetic capacity specifies the upper limit of the photosynthetic rate for a given PAR flux; i.e. the gross photosynthetic rate that occurs under no environmental stress. Rates of transpiration can be derived from this value of photosynthetic capacity since water vapor diffuses out of leaves via the stomatal pores which open for the influx of atmospheric carbon dioxide.

Strong relationships have been demonstrated between time integrals of satellite-derived VIs and net primary production (NPP) (Goward et al. 1985, Fung et al. 1987), the geography and seasonality of vegetative cover (Justice et al. 1985, Tucker et al. 1985), and simulated photosynthesis and transpiration (Running and Nemani 1988). Theoretical analyses by Sellers (Sellers 1985, 1987, Sellers et al. 1992) examined the links between spectral vegetation indexes and canopy properties of LAI, absorbed PAR, photosynthetic capacity, and canopy resistance to water vapor efflux. A mechanistic basis for the observed correlations (given a horizontally uniform canopy) was demonstrated with a two-stream approximation model of radiative transfer and simple leaf and canopy models of photosynthesis and stomatal resistance. These results suggest that indices such as the simple ratio and NDVI are indicative of instantaneous biophysical rates of photosynthesis and conductance, but are not reliable estimators for any state (leaf area, biomass) associated with vegetation. Furthermore, they are related to the maximum photosynthetic output of the vegetation; the actual rates being determined by the PAR flux and environmental factors. Conditions constraining the predictive powers of vegetation indices include those that affect the photosynthesis/PAR relationship such as environmental stress and different photosynthetic pathways (C_3, C_4), and conditions that may influence spectral estimates of absorbed PAR such as contributions from background soil and litter reflectance. Biological processes and their respective sensitivity to VIs and environmental variables must be considered for different vegetation types (Bartlett et al. 1990). For example, land cover should be stratified according to ecosystem or biome type before relationships are established between PAR and a vegetation index. Fung et al. (1987) determined global net primary production from NDVI using an empirically-derived scaling factor that essentially accounted for Monteith's conversion efficiency for each biome type. Prince (1991) has cited efficiency factors converting annual APAR energy in megajoules (MJ) to NPP in grams for different biome types.

The relationships between NDVI, absorbed PAR and photosynthetic capacity are highly linear in spatially heterogeneous (but physiologically uniform) canopies (Asrar et al. 1992) and under circumstances when background reflectance (soil, rocks, litter) is minimal (Sellers 1987, Sellers et al. 1992). However, measurements in two spectral bands may provide an ambiguous measure of vegetation when background reflectance is a significant component of the total surface reflectance. Confounding influences from background variation, atmospheric attenuation and off-nadir viewing cannot all be accounted for using a two-band ratio such as NDVI (Choudhury 1987, Huete and Jackson 1988, Baret and Guyot 1991, Goward et al. 1991, Middleton 1991). Modifications to NDVI have been suggested to account for first-order soil-vegetation interactions (i.e. soil brightness effects) (Huete 1988, Baret et al. 1989). However, secondary soil variations due to soil optical properties can only be addressed using multiple spectral bands through either factor-analytic inversion models which allow composite plant-soil mixtures to be separated into component spectra (Huete 1986, Huete and Escadafal 1991), or selection of spectral regions where soils reflectance varies linearly.

Data from high spectral resolution instruments such as NASA's AVIRIS (Airborne Visible and Infrared Imaging Spectrometer) may yield more information on biophysical and biochemical processes than do current operational broad-based sensors such as the Landsat TM (Thematic Mapper). Variables of spectral shape such as width, depth, skewness, and symmetry of absorption features are more directly indicative of biochemical state and canopy physiology than broad-band averages (Wessman 1990). Studies relating chlorophyll content with the location of the inflection point of the long wavelength edge of the absorption feature have met with varied success (Schutt et al. 1984, Rock et al. 1988, Milton and Mouat 1989, Curran et al. 1990, Miller et al. 1991). It appears that the wavelength of the inflection point in the red-edge region is less dependent on soil optical properties, atmospheric effects and irradiance conditions than are broad band VIs (Baret et al. 1992). Pigments other than chlorophyll have been found to be more directly indicative of actual photosynthetic rates (as opposed to photosynthetic capacity) (Demmig-Adams 1990). Light-induced changes in a xanthophyll pigment assumed to be closely linked to changes in photosynthetic activity have been related to spectral changes in green reflectance at 531 nm (Gamon et al. 1990, 1992). Such wavelength-specific absorption differences among the variety of photosynthetic pigments may permit quantification of their concentrations through spectral mixture analysis (Adams et al. 1989; Smith et al. 1990a, 1990b; Ustin et al. 1992) and derivative spectroscopy (Wessman 1990, Demetriades-Shah et al. 1990). Second derivatives of high spectral resolution reflectance data in the visible and near infrared regions appear to be strongly related to absorbed PAR and relatively insensitive to the reflectance of non-photosynthetically active materials such as litter and soils (Hall et al. 1990). However, derivative techniques are likely to be problematic due to their sensitivity to noise.

Biogeochemical Cycles

Remote sensing of photosynthesis, as described above, can provide substantial information for modeling aboveground carbon pools and other element cycles, and contribute substantially to understanding of regional ecosystem functioning. Some of the terms used to calculate carbon turnover time, nutrient availability and soil respiration may be provided by new techniques in imaging spectrometry that offer the possibility for determining the chemical composition of vegetation canopies (Waring et al. 1986, Peterson et al. 1988). These ecosystem processes are intimately linked with rates of decomposition, which are strongly regulated by the chemical quality of the organic matter (Melillo et al. 1982, Meentemeyer and Berg 1986, Aber et al. 1990). Remotely sensed estimations of lignin (the most recalcitrant material in litter), canopy nitrogen, or other constituents related to C:N ratios may serve to constrain decomposition submodels in ecosystem simulations, thus stabilizing model inversions (Aber et al. 1990, Schimel et al. 1991).

Analytical spectroscopy of organic mixtures in the shortwave infrared region (0.7 to 2.5 mm) is a well established technique for biochemical analyses in agricultural forage assessment and the food industry (e.g. Barton and Burdick 1979, Shenk et al. 1981, Wetzel 1983, Marten et al. 1985). Spectroscopy applications to analyses of foliar biochemistry of native species has strengthened sampling strategies for ecosystem studies; the rapidity of the method enables processing of large numbers of samples (Wessman et al. 1988a, McLellan et al. 1991). Knowledge of major leaf constituent (e.g. cellulose, lignin, protein) absorption characteristics may permit remote assessment of canopy level concentrations if high spectral resolution reflectance information is acquired (Wessman 1990). Application of these techniques to imaging spectrometer data over temperate forests yielded strong relationships with ground measurements of canopy lignin concentrations that in turn allowed the mapping of nitrogen mineralization for the study site (Wessman et al. 1988b, 1989). Significant correlations have also been noted between imaging spectrometer data and canopy nitrogen content across a range of coniferous forest stands in Oregon (Peterson and Running 1989) and fertilization plots of Douglas-fir (*Pseudotsuga menzeisii*) in New Mexico (Swanberg and Matson 1987). Gao and Goetz (1990) demonstrated that canopy water content can be retrieved, using spectral curve fitting techniques, from canopy reflectance acquired by imaging spectrometers. Further studies on the question of remote sensing of canopy chemistry are currently being pursued (Goetz et al. 1990, Martin and Aber 1990, Curran et al. 1992).

The application of analytical spectroscopy to remotely sensed data is still early in its development. Detection of minor absorption characteristics will rely on high spectral resolution sampling, sufficient characterization of atmospheric conditions, and high signal-to-noise sensors. Integrating spectrometry studies at the leaf, canopy and landscape levels will enhance our understanding of vegetation optical properties and the transfer of spectral information with increasing scale and landscape complexity. These investigations into the question of canopy chemistry have led us to consider the use of remote sensing in extrapolation models of biogeochemistry. For this purpose we must rely on surrogates since belowground processes significant to biogeochemical cycling are invisible to the sensor (Wessman 1991). This amplifies our need to better understand how properties such as plant physiology and biochemistry reflect the balance between factors limiting to the system (Aber et al. 1990, Schimel et al. 1991).

STRUCTURAL AND FUNCTIONAL CHANGE IN ECOLOGICAL SYSTEMS

The capability to make repetitive measurements with remote sensing allows for the detection of landscape changes that may contribute to the unsustainability of component systems. By monitoring ecosystem structure and dynamics over time, potentially harmful changes in the landscape can be evaluated from an ecological perspective.

Regional biogeochemical flux estimates and atmosphere-biosphere interactions are significantly influenced by the type and successional stage of ecosystems within a landscape. The rapid rate of land-use changes occurring in many parts of the world, including encroachment of urban areas on natural ecosystems, contribute directly to perturbations in flux and matter dynamics. Successional patterns reflect local variations in resource availability and linked carbon and nitrogen cycles. Effects of climate change or human disturbance will, in turn, be modified by the stage and pattern of succession within the landscape (Pastor and Post 1986). Large-scale spatial heterogeneity and long-term patterns of successional dynamics have prevented past extrapolations of ecosystem research from local to regional scales (Hall et al. 1991). Remote sensing and ground-based evaluations provide the most promising tools for compiling geographical information on the stage and condition of ecosystems over time.

Detection of long-term change in ecosystems requires knowledge of the static situation, e.g. health, structure and seasonal productivity (Hobbs 1990). Several remotely sensible variables, when monitored over time, will lead us to deeper insights on ecosystem functioning. Seasonally integrated vegetation indices and canopy chemistry are variables that will be affected by and respond to environmental change.

The role of remote sensing in monitoring change detection is well illustrated in a study where regional estimates of the extent and severity of damage due to acid deposition in spruce-fir forests of the northeastern United States have been made with an index combining reflectance in the near- and mid-infrared spectral regions (Vogelmann and Rock 1986). Such studies can be extended over time to monitor rates of damage or the success of abatement efforts. Remote sensing has confirmed predictions of near-exponential increases in the rate of tropical deforestation (Tucker et al. 1984, Malingreau and Tucker 1987) and has aided research into effects of deforestation such as changing trace gas flux and desertification (Matson et al. 1990, Kaushalya 1992). Secondary succession patterns within Minnesota boreal forests have been studied using Landsat Multispectral Scanner data over a ten-year period (Hall et al. 1991). Transition rates from one successional stage to another were generated for each landscape component. The ten-year observations indicated considerable change within landscape components in a region that has been relatively stable over several centuries. In particular, wilderness areas were less heterogeneous and dynamic than managed areas.

SUGGESTIONS FOR PRACTICAL APPLICATIONS

The first step in developing an integrated regional management plan for a particular area is to conduct a quantitative analysis of the spatial pattern within the landscape. Information on the landscape structure can be useful in future enhancement and restoration of structural features, in evaluating the ecological integrity of the landscape, and in determining the functional importance of the observed patterns (Mladenoff et al. 1993). Quantitative knowledge of the spatial characteristics of an undisturbed landscape can also be helpful in setting objectives for management or restoration of similar patterns and flows in a disturbed landscape. Vegetation types can be classified or mapped using aerial photography or other remotely sensed data and integrated into a GIS. These maps can be used to describe patch type, number, area, size class distribution and importance. Indices of landscape diversity and dominance (Tuner and Ruscher 1988) can further quantify the landscape structure. Mladenoff et al. (1993) used fractal analysis to describe the complexity of patch size and shape relationships. In addition to vegetation and land cover maps, appropriately processed remotely sensed data can describe structural (e.g. LAI) and functional (e.g. photosynthesis) variables important to ecological management. Other information such as soil type, topography, land use, land ownership, and disturbance history can be digitized and stored as registered layers within the GIS.

Acquisition of such large amounts of data will necessitate interaction among scientific disciplines and government agencies and may even require a separate body charged with assembling relevant spatial data for public lands. For example, the International Geosphere-Biosphere Program (IGBP) has designated a Data and Information Systems group (IGBP-DIS) to coordinate acquisition, storage and management of data for general access by the scientific community interested in global change research (IGBP 1990). Spatial data, used in conjunction with ecosystem simulation models such as FOREST-BGC (Running and Coughlan 1988), can be used to evaluate the impacts of various potential management decisions on a landscape scale. Whether region-specific or inclusive of all public lands, geographically-referenced databases within the Forest Service will provide the foundation for successful ecosystem management. This will, of course, be contingent upon dedicated resources drawn from adequate federal funding. Although such an integrated management approach based on the existence of a comprehensive GIS database and ongoing research on ecosystem function and modeling may take years to implement, it will facilitate relatively rapid management decisions rooted firmly in ecological principles.

CONCLUDING REMARKS

In order to pursue economic growth and development in a sustainable manner, we must understand the biological and physical world of which we are a part and adjust our behavior in recognition of the innate limits of our environment. As our influence on the environment increases in intensity and extent, we must expand our analytical capabilities to understand the potential response of the systems we affect. Given the complexity of demands being placed on our natural resources, it is important that management practices be founded on ecological principles. Sustainable practices rely on a well-developed understanding of the structure and functioning of ecological systems and their change over time. Remote sensing provides the synoptic views needed to study landscape structure and functioning at regional scales that are important to multi-use management. Landscape structure and biophysical attributes sensible from space will determine how widely ecological extrapolations can be implemented. Reflectance data from vegetation are indicative of the biophysical processes of photosynthesis and transpiration, and have been shown to correlate with biomass and productivity. Imaging spectrometry offers improved characterization of reflectance properties, leading to spectral models that aid in better defining landscape complexity. Successful estimation of parameters such as absorbed photosynthetically active radiation (APAR), canopy chemistry, and canopy water content will provide insights on changes in seasonal photosynthesis and photosynthate allocation, the susceptibility of plants to disease, rate of litter decomposition, and other changes related to environmental stress.

While remote sensing is not the panacea for large-scale questions, as was suggested early in its development, its utility is unsurpassed in producing a consistent data base at spatial and temporal resolutions useful for resource monitoring and management. When coupled with other data bases through the use of information systems, it has the potential to alter our models, our methods of analysis, and, in essence, influence if not change our paradigms.

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Multicultural Dimensions in Ecosystem Sustainability

Celedonio Aguirre-Bravo[1]

Abstract — Ecosystem sustainability has a human component of multicultural dimension. In each culture, the dimensions of ecosystem sustainability are not only material, but also intangible or metaphysical. Man itself is more than a biological organism, it is a complex phenomenon, unpredictable, and naturally endowed for making choices. Societies are the sum of all cultural complexities and expressions of the phenomenon of man. Because of these complexities, ecosystem sustainability has a different appeal and significance to people of contrasting cultural backgrounds, sociohistory, and geographic location. In scientific cultures, for example, ecosystem sustainability may imply a logical process grounded on a mechanistic vision of the world, a condition void of metaphysical interpretations and meaning. For traditional cultures, ecosystem sustainability may be viewed as an implicit condition of a general cosmic-holistic principle hidden in the physics of every natural thing. To these cultures, this condition is an important part of their **modus vivendis-operandis** which ensures the long-term survival of all. Critical to ecosystem sustainability is the **convergence integration process** of these multicultural views of nature. This paper attempts to provide insight into the issue that ecosystem sustainability has a dynamic component of multicultural dimension, and that its integration into the planning process of ecosystem management is critical for making this management vision operational.

INTRODUCTION

Nowhere on Earth do the questions to be discussed in this conference on " Sustainable Ecological Systems" find cognitive significance but in Man's sociosphere. Ecosystem sustainability, given a human-populated biosphere, cannot be conceived apart from a multicultural perspective. Human activities in the biosphere influence ecological systems in a number of complex ways. Earth's biosphere has an anthropogenic context which is dynamic, diverse, and it is linked to ecological systems at all levels of scale. Through time, sociosphere linkages with biosphere systems have not only been material but also metaphysical, and culturally related in all cases. These relationships, the **human software interface** which keeps " all things" linked together, are critical to the scientific and operational foundations of ecosystem sustainability. As a process, ecosystem sustainability has a **human interface dimension**. People from different cultural backgrounds are the primary actors in this process. Through their diversity of cultural expressions, people bring alternative views to the conceptual and operational significance of ecosystem sustainability.

Ecosystem sustainability, like many other terms and expressions, has multiple definitions and interpretations (Allen and Hoekstra, 1992), each reflecting the cultural complexities of Man and society. Let's take for example the question of " is it sustainable?" to use it as an ethical benchmark to discern among alternative management strategies. We will find, however, that the question has a different appeal and significance to people of contrasting cultural backgrounds, sociohistory, and geographic location. The question, in itself, is embedded in physical and metaphysical meaning, a situation which reflects the cultural complexities of society. Any attempt to separate this duality from a management perspective will conflict with the cultural values and perceptions of social institutions.

[1] *USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, Colorado, USA.*

In a multicultural society, ethical considerations of the relationships between man and nature bring purpose and direction to the operational dimensions of ecosystem sustainability. In scientific cultures, for example, ecosystem sustainability may imply a logical process grounded on a mechanistic vision of the world, a condition void of metaphysical interpretations and meaning. For traditional cultures, ecosystem sustainability may be viewed as an implicit condition of a general cosmic-holistic principle hidden in the physics of every natural thing. To these cultures, this condition is an important part of their **modus vivendis-operandis** which ensures the long-term survival of all things. These divergent multicultural views of nature, like the physical processes responsible for the biodiversity and complexity of natural ecosystems, are necessary conditions for assuring and maintaining the sustainability of ecological systems.

This paper attempts to provide insight into the issue that ecosystem sustainability has a dynamic component of multicultural dimension, and that its integration into the planning process of ecosystem management is critical for making this management vision operational.

CONVERGENCE INTEGRATION OF CULTURAL COMPLEXITY

Sustainable Ecological Systems encompass the whole spectrum of material systems perceived and known by humans. In the evolutionary trend of all material systems, biological evolution and human evolution might be considered as two asynchronous phases of a single general process. Through billions of years, the properties of the evolving material has undergone radical changes with a clear tendency towards **"complexification"** (de Chardin, 1965) of the natural world. Generally, this process is characterized by increasingly elaborate organizational complexity, as manifested in the passage from subatomic units to atoms, from atoms to inorganic and later to organic molecules, thence to the first subcellular living units or self-replicating assemblages of molecules, and then to cells, to multicellular individuals, to cephalized metazoa with brains, to populations-communities-ecosystems-landscapes-biomes-biosphere, to primitive man, and now to civilized societies. Ecosystem complexity, under the influence of this recent anthropogenic context, became far more complex.

Humans, in this evolutionary process, came to provide a new direction to the development of ecological ecosystems. In this process, humans turned out to be an additional primary factor of ecological change. This does not mean that ecological systems prior to the advent of man were already in a state of stable equilibrium. Ecosystems in a state of stable equilibrium, since they were established in the early Paleozoic, may have never existed due to a history of global changes in environmental conditions Earth has experienced through time (Behrensmeyer et al. 1992). Most of terrestrial ecosystems, under an anthropogenic context, have been forced to provide for the needs of humans, a force which has driven them into a condition of successional changes, but not necessarily into unhealthy ecosystems (Allen and Hoekstra, 1992), in most cases. From the perspective of an anthropogenic biosphere, ecosystem sustainability can be seen as a human-driven ecological process.

Complexification has brought man to be part of and have presence in all material systems. Man's linkages with the ecosystems are not only material, but also conceptual and abstract, through the human soft system **(the mind)**. Through this system, the brain being the physical manifestation, humans have the potential to link themselves with the material systems at any physical and metaphysical level. In our anthropogenic biosphere, therefore, the **convergence integration** of this **complexification** manifested in the phenomenon of man provides foundation to an infinite number of cultural expressions that characterize our culturally complex societies. The brain alone, in this **convergence integration process**, is a "piece of biological nonsense", "as meaningless as an isolated human individual" (de Chardin, 1965). Teilhard suggests that "mind is generated by or in complex organizations of living matter, capable of receiving information of many qualities of modalities about events both in the outer world and in itself, of synthesizing and processing that information in various organized forms, and of utilizing it to direct present and future actions." Humans and ecosystems, from a sustainable perspective, can not be viewed as separate material system components, but as highly integrated systems undergoing **complexification**.

HUMAN-ECOSYSTEM INTERACTIONS THROUGH TIME

A basic understanding of Man and his development is fundamental to the analysis of Man's relations to natural ecosystems. Early humans (**Homo erectus**) are believed to have appeared at the end of the last glacial periods, some 500 to 700 thousand years ago. By this time, man's skills, tools, and weapons were still too primitive to produce significant impacts on the environment. Many believe that the first widespread impact of humans on the natural environment occurred when they learned to control and use fire. It is not known when this happened precisely, but there are indications in China that this may have occurred 300 to 400 thousand years ago. Human settlement of temperate latitudes really took hold during the Great Interglacial period (400 to 200 thousand years ago) as small bands of hunters exploited the rich game populations of European river valleys. Fires caused by these bands shaped the natural landscape and dense forests were converted to savanna-forests and ultimately to grasslands. Other than the impact of broadcast burning, and despite fairly sophisticated social institutions and technology, early humans did not greatly disturb the ecological balance. They were an integral part of the natural environment with large home territories and a population regulated by food supply. Food and shelter needs were easily met by these small bands of hunter-gatherers without adversely impacting ecosystem carrying capacity.

With the advent of **Homo sapiens** came a radical change. This new species of Man began to exploit territories never inhabited before. Still hunter-gatherers, they began to exploit the western Russian plains, the Siberian tundra, and the Far East. Japan was already occupied by the late Pleistocene, and Australia perhaps more than 30,000 years ago. The North American continent was also settled by late Pleistocene hunters. Small bands of hunter-gatherers from Siberia and Northeast Asia appear to have been the first humans to reach the New World. The exact date of this migration is still unknown, but it may have occurred during the last glaciation (27 to 8 thousand years ago). During this time period, many big game species became extinct. Nowhere were the extinctions so drastic as in the New World. It has been estimated that about three quarters of the mammalian genera there abruptly disappeared at the end of the Pleistocene (Fagan, 1977). In addition to climatic changes, hunters might have been the final variable which accelerated these extinctions and the loss of more species than might otherwise have occurred.

Hunting and gathering were critical to the survival of early humans. To this end, they developed complex toolkits and sophisticated techniques. Initially, humanity lived in ecological balance with the natural environment. Early people, like other animals, did not greatly disturb the natural systems, for their numbers were strictly controlled by available food. Later hunter-gatherers, however, were characterized by the following behavioral conditions: (1) they had become the dominant animals in every ecosystem they occupied, (2) they eliminated competition from other predators by hunting them as well, and (3) they had some influence over which animal and vegetable communities lived in their territory (Fagan, 1977). As human activities became more specialized, new socio-economic conditions developed which led humans to compete for the same limited resources. For the first time, humans laid down the conditions which ever since have resulted in significant environmental impacts.

CULTURAL COMPLEXIFICATION

Though the **anthropogenic biosphere** is a recent phenomenon in Earth's geological history, its cultural **complexification** is still an on-going process. From the time the Earth was formed (4.5 billion years ago), to the time when ecosystems were organized and assembled, and thereafter for about half a billion years, the biosphere remained void of conscious entities capable of understanding the complexities of the natural world. It was not until relatively recently, between 2.5 to 5 million years ago, that the complex human creature appeared and began to be a significant driving force of global ecological change. This human creature, the **phenomenon of Man** (de Chardin, 1965), represents the highest level of **complexification** of all life forms on Earth. In contrast to other animals, it took Man just a few thousand years to populate most of the landscapes of the world. Unlike the evolution of terrestrial ecosystems, the process of how humans populated the world was rapid and continuous. It was a complex sociohistorical process, strongly constrained by geographic conditions, which resulted in a continuum of culturally diverse populations distributed throughout the world.

Ever since their origins, human cultures have displayed geographically uneven patterns of development. Today in our modern society we ask the question of what exactly is culture. It is a distinctively human attribute. Fagan (1977) defines it as "historically created designs for living, explicit and implicit, rational and irrational, and non-rational, which exist at any given time as potential guides for the behavior of man." Human cultures are never simple or static, nor unidirectional, they are always adjusting to both internal and external influences of material systems as well as to the influences of their particular socioeconomic systems. Like terrestrial ecosystems, human cultures have been undergoing a dynamic process of **complexification**. In itself, each culture is a complex system of multiple interrelated expressions of the phenomenon of man.

While some cultures have based their sociohistory on a traditional cosmic-holistic view of nature, others have discovered the mechanical principles of the natural world, always eager to find a physical explanation for the unknown. In many ways, these two extremes of cultural expression, traditional vs. scientific, have interacted with and modified differently the ecosystems. Within these two cultural extremes, there is a continuum of cultural views of nature, all of which are fundamental to the process of ecosystem sustainability. Every society or nation has a number of ways of expressing its cultural dimensionality. In every case, this condition is not culturally uniform, but complex and diverse, and to some extent unpredictable. The human component of the biosphere is nonlinear in cultural expressions and behaviors. It is a changing condition, not only at the individual level, but also at the societal level. Human cultures are means by which context is provided to terrestrial ecosystems at all levels of scale.

COLLISIONS WITHIN THE CULTURAL COMPLEXITY

Eventually, people began to shift from a hunting and gathering way of life to a more specialized agriculture-based economy. The cultures resulting from this process displayed great diversity geographically. These "village cultures" or "traditional cultures", as they are often denoted by historians and geographers, not only developed particular sets of site-dependent relations with their environment, but also distinctive philosophical views for interacting with their physical environment. In most of earth's bioregions, a significant number of these cultures (tribes or clans) dominated the landscape, each with a unique sociological and ecological history. Generally, these cultures had considerable knowledge of natural and technological processes but little systematic study of nature, few traces of a scientific tradition, and no scientific institutions. In these cultures, the bond between Man and nature was a fundamental condition for sustaining their natural environment.

Several factors and conditions may have accelerated or precipitated the socioeconomic transition from village cultures to intensive food producing societies. Though still controversial, recent evidence (Dorn, 1991) suggests that this change was neither the direct result of an agricultural invention nor any fundamental revision of man's relationship with nature; it was sparked instead by a realignment of ecological variables, primarily increasing population density and diminishing availability of collectible plants and large-bodied animals. As these emerging societies developed more complex economies, their administration became more centralized, along with which came an aggressive process of assimilation of local cultures spelling the loss of their ecological knowledge. As a result, a collision of visions about approaches to interacting with natural ecosystems took place. It is speculated that this human tragedy influenced the collapse of ancient kingdoms in Egypt, Mesopotamia, India, China, and the New World. Continuing until today, these cultural collisions have been a matter of continuous political struggle between people and governments, and ultimately between Man and nature.

Since every society sees civilization in its own way, it is difficult to determine the exact moment at which civilization first appears in world history. Most definitions of "civilization" reflect ethnocentrism or a value judgement. If civilization is defined by literacy and a preference for urban life, then its origins go back to the beginnings of towns and city-states in Egypt and Mesopotamia. The consequences of urban life most important to human history were in politics and social history. As urban societies appeared, the Man-nature separation process accelerated, and for the first time Man began to change significantly the natural world. Urbanization brought about a deep misunderstanding between "city cultures" and "nature cultures", a problem deeply rooted in the history of collisions of cultural visions with respect to natural ecosystem management. Similar cultural conflicts have taken place in many other regions of the world.

Critical to the development of the great ancient kingdoms was the organization and centralization of political decision making power. In contrast to the social structure of village cultures, the governments of these kingdoms exerted strong influence and control on the relationships between Man and nature. Science and technology, as developed by local cultures, were forced by centralized governments to change towards a utilitarian approach. Moreover, though with less emphasis in classical Greece, the governments of these kingdoms financed science and bureaucratized its pursuit. Generally, science and technology control was centralized to sustain agricultural development, and as such was of considerable social importance and of central interest to the government authorities. It has been suggested that the intensification of irrigation agriculture, which encompassed the development of an extensive network of hydraulic infrastructure and intensive labor, could not have been possible without a centralized authority. Most of the ancient major states started from a hydraulized agriculture, which cannot be a mere historical coincidence. On the social foundation of irrigated agriculture, unilateral approaches to land use management were imposed on the local cultures. Natural forest ecosystems, under the dominance of these centralized monocultural visions, began to be changed at alarming rates.

Traditional cultures were not necessarily more noble or wiser than the great centralized cultures in the ways they interacted with the natural environment. Each culture's sociohistory is driven by material and ethical considerations. Some cultures have chosen to be more material while others have been more contemplative and mystical about the marvels of nature. These extremes of cultural behavior evolved into a broad array of societies with a different vision of the natural world. Few of these societies chose to be scientifically and technologically advanced while many others decided to live according to their traditional knowledge and culture. Whereas village cultures were forced to adapt to their surroundings in order to survive, the great civilizations had the power and technology to impose a singular vision over vast areas with no adaptation for local cultural and ecological factors.

Sociohistorical analyses have often been approached from a disjointed or unrelated perspective. Most analyses, far from considering the ecology-economy interplay of these ancient societies, which could provide a holistic explanation of their socio-ecology history, have tended to emphasize one factor at a time. Historians, particularly those dominated by a materialistic philosophical view, have concentrated on "modes of production". This taxonomy of social systems, in contrast to holistic analyses, has a utilitarian tendency which emphasizes an historical determinism of social development and clearly masks the fact that Man and society are an integral part of natural ecosystems.

Generally, the socio-history of ancient cultures reveals that the centralization of land use policies accelerated the process of Man's separation from nature and limited Mans' freedom for developing a genuine philosophy of nature. Cultural conflicts, and competition for nature control, precipitated the decline of many great cultures. In each case, the decline was correlated with the decline of agriculture. As most records suggest; "because the fertility of the land was decreased, the kings who followed were no longer of such consequence as those who went before" (Dorn, 1992). Whatever the exact circumstances, the rise and decline of ancient cultures shows a strong correlation with the cultural separation of Man from nature, a process induced by the centralization of social and political institutions.

In many ways, the sociopolitical and cultural history of ancient hydraulic civilizations influenced profoundly the sociospheres and biospheres of today's world. The socio-history of ancient cultures reveals a distinctive process of Man-nature separation induced by centralized government policies. In most cases, the process is framed by sociopolitical conditions which are remarkably similar not only to other ancient cultures, but also to modern societies. Man-nature separation has been accelerated by the utilitarian emphasis of science and its political centralization. Humans and ecosystems are part of a **convergence integration process** that works at all levels of scale. Deviations from this process, mostly driven by cultural and material differences, must be understood and minimized in order to provide significance and direction to the process of ecosystem sustainability.

CONVERGENCE INTEGRATION OF PEOPLE AND ECOSYSTEMS

At any level of scale, the concept of sustainability provides an opportunity for integrating multicultural perspectives into the management of ecosystems. For any given space-time dimension of the ecosystem, it is not just biological populations that interact with the biosphere, but living human beings capable of making economic and political choices (the "sociosphere"). While ecosystems have been evolving for millions of years, the sociosphere came to be a part of the natural scenario only very recently. In addition, natural ecosystems are continuous units linked at different levels of geographic scale. Sociosphere systems, on the other hand, are encompassed by a variety of geopolitical units (nations, peoples, cultures, institutions), each with different perceptions, values, and cultural expressions.

Like natural ecosystems, societies all over the world are not independent units, but continuous systems connected by historical, cultural, and socioeconomic processes. They all are part of and live in the very same anthropogenic biosphere of planet Earth. As any other biological population, the sociosphere elements (people) are ecosystem components, but their activities cause significant ecological disturbance which, in many cases result in species habitat destruction and species extinction. At all levels, the sustainability concept implies that human interactions on the biosphere must be managed first before attempting to manage ecosystems. In making this concept operational, the differences in perceptions and values of people must be seriously taken into account, otherwise the practice of ecosystem sustainability is bound to be just **"business as usual"**. Ecosystem sustainability is emerging as a framework for people to interact wisely with the biosphere. In most cultures, ecosystem sustainability is something appealing to people, and is gaining widespread political support for using it in management applications. Ecosystem sustainability, therefore, appears to be a new approach for confronting the natural resource and environmental problems society faces today. Within the USDA Forest Service, ecosystem management is beginning to mean "using an ecological approach to achieve the multiple-use management of national forests and grasslands by blending the needs of people and environmental values in such a way the national forests and grasslands represent diverse, healthy, productive, and **sustainable ecosystems**" (Gerlach and Bengston, 1993). The recognition that people are a critical part of the equation to achieve ecosystem sustainability is a necessary condition for making this approach operational. In contrast to traditional forest management views, the ecosystem sustainability approach provides limitless possibilities for integrating the complex nonlinearities of people and their cultures into the framework of natural ecosystems management, at all levels of scale.

An example of this evolution in thought has begun also to take place in Mexico. The Mexican government recently passed a sweeping set of natural resource management and environmental protection laws which call, among other things, for integrated, multiple use management of the nation's forests under sustainable ecological principles (SARH, 1992). Current Mexican forest policy is promoting the integration of local cultural perspectives into forest ecosystem management. Several examples now exist in which forest ecosystems are being managed by integrating local cultural views. In this process, landowners and forest ejidos are required by law to include multiple use and multiple resource criteria in their integrated forest management plans. Ecosystem sustainability, in most cases, is being implemented as a learn-as-you-go incremental process of adaptive management. Community participation, and the formation of partnerships as a means to minimize conflicting cultural views, have been fundamental in making ecosystem sustainability operational. The use of this approach to forest resource management provides decision makers a framework for understanding that decisions which are made at the local level are bound to have regional, and even global, consequences.

Ecosystem sustainability is meaningless if the various scales of its geographic linkages are ignored in making it operational. The ecosystems of North America, such as the Colorado-Rio Grande Rivers, the Chihuahuan and Sonoran Desserts, or the mountain forests in Mexico which are critical for migratory bird populations, provide clear examples of the continental and global complexities of ecosystem sustainability. In addition, these ecosystems are the ground on which complex sociospheres systems interact, according to their respective perceptions and values. Across the landscape, whether at a regional or global scale, there are numerous local cultures whose particular ecological histories and values could be shared with the rest of the global community and integrated into the framework of ecosystem sustainability (Kidd and Pimentel, 1992). Multicultural participation at all levels ensures drawing on a wider array of knowledge than previously and a better use of the adaptive experience of local cultures for making ecosystem sustainability operational.

The global dimension of ecosystem sustainability calls for international cooperation and participation. Today, after two hundred years of industrial revolution and scientific accomplishments, society is beginning to realize that the world is changing very rapidly and is becoming more different. The global environmental problems confronting all societies cannot be blamed on science and technology, but on the dominant role of social and political institutions for implementing monocultural visions of the human-nature interplay. Rather than go back to the time when humans were hunters and gatherers, the sociohistory of humanity has to be redirected towards higher levels of **convergence integration** with the biosphere systems. Allen and Hoekstra (1992) discuss the significance of people integration into the management of ecological systems.

The global complexities of human actions on natural ecosystems have been recognized at the highest level of government and by international government organizations. In the USSR, former President Mikhail Gorbachev in discussing ecological security stated that: "for all the contradictions of the present-day world, for all the diversity of social and political systems in it, and for all the different choices made by the nations in different times, this world is nevertheless one whole. We are all passengers aboard one ship, the earth, and we must not allow it to be wrecked. There will be no second Noah's Ark." Likewise, former president George Bush declared: "No line drawn on a map can stop the advance of pollution. Threats to our biosphere systems have become international problems. Nations must participate in developing an international approach to urgent environmental issues." It is implicit in the above statements that ecosystem sustainability has multicultural and global linkages. It is also clear that solutions to ecological problems cannot be found only in the corridors of Washington or of any other institution, but in the convergence integration process of truly incorporating the multicultural dimensions of people into the planning framework of ecosystem sustainability.

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SUSTAINABLE ECOSYSTEMS AND FOREST HEALTH

Session Summary

Michael R. Wagner and Jill L. Wilson[1], Chairs

The symposium organizers challenged session chairs and speakers to bring into focus the widely professed idea of management for sustainable ecological systems. Our particular symposium was challenged to address the interactions between forest health and sustainable ecosystems. This proved to be a formidable task and readers are encouraged to judge for themselves whether or not this task has been accomplished. Both of the concepts " sustainable" and " health" remain without clear definition and consequently there is no consensus on the procedures needed to achieve " healthy sustainable ecosystems." Much of this ambiguity arises because the root terms of health and sustainability were developed in the context of forest commodities and usually in the context of individual trees or small stands. When these terms are applied to the much larger watershed or landscape scale their meaning could be applied more broadly, increasing what are already ambiguous concepts. Wagner suggested other concerns with historical definitions and proposed some modifications.

While precise definitions may not have emerged, there was considerable consensus regarding the need to focus on the role insects and diseases play in basic ecological processes. Schowalter suggested that insects and diseases may contribute to maintenance or recovery of forest functional equilibrium through pruning, thinning, nutrient cycling, etc. He further maintains that these organisms may be instrumental in maintaining ecosystems. Clancy likewise emphasized the role of defoliating insects on forest biomass production, bioelemental transfer, and nutrient cycling. Clancy suggested research approaches that might be used to address these issues for an important western defoliator, the western spruce budworm (*Choristoneura occidentalis*). Various discussions and questions throughout the session indicate that all speakers share the view that insects and disease are essential elements of the ecosystem and greater appreciation and research on this role is justified.

Hagle and Byler discussed how ecosystems respond to the introduction of exotic organisms. They observed that introductions greatly altered some insect and disease regimes but they did not observe the loss or addition of any ecological function. This novel view could change considerably how impacts of insects and diseases are interpreted. This conclusion contrasts somewhat to Wilson and Tkacz who suggest that changes in incidence of insects and diseases may reflect changes to the underlying structure and function of ecosystems. Additional analysis, debate and discussion is needed to clarify whether population outbreaks of insects and diseases are indicators of unhealthy conditions or are healthy feedback processes in ecosystems that function to restore forest equilibrium. Under either of the above scenarios it is clearly justified to examine long range trends in insects and diseases to allow for interpretation of overall forest condition.

Finally, Liebhold discussed the value of spatially explicit models to assist in tracking trends in insect populations. Analytical approaches to address more landscape level effects appear in place and will likely replace the historic modeling approaches of systems models and process models. Spatially explicit models are clearly the appropriate approach for ecosystem level examination of the role of insects and diseases.

As land management philosophies evolve from sustained yield management through multiresource management toward ecosystem management, the concepts of healthy forests and sustainable ecosystems will change. All advocates of these concepts need to recognize that consensus does not exist on the meaning of these concepts nor on the likelihood of achieving these objectives. Thoughtful managers, scientists, and the public should continue to demand more clarity and specificity of these ideas and encourage continuing research to better understand what is a healthy sustainable ecosystem.

[1] *Professor, School of Forestry, Northern Arizona University, Flagstaff, AZ and Pest Management Specialist, USDA Forest Service, State and Private Forestry, Flagstaff, AZ.*

The Healthy Multiple-Use Forest Ecosystem: An Impossible Dream

Michael R. Wagner[1]

Abstract — Forest health is a widely used term that is viewed by many managers and users as a desirable future condition for the nation's forests. However, the definition of a healthy forest is subjective, a function of objectives, and highly dependent on whether the forest is viewed from the utilitarian or ecosystem centered view. The forest health paradox is created because forest health is both a future desired condition and dependent on the future desired condition. The nature of the forest health paradox is described. Alternatives for the resolution of this paradox may require deviation from the multiple-use and explicitly multiresource concept of forest land management. Some approaches to resolve the forest health paradox are presented.

INTRODUCTION

Forest health is a concept that is currently widely used in the context of a desirable future condition for forests. The notion of maintaining a " healthy forest" is currently popular and enjoys near unanimous approval by all forest users. The question arises as to why forest health has such wide support in an environment filled with controversy about the appropriate management direction for the public forests of America.

The purpose of this paper is to discuss the concept of forest health, explain why forest health has risen to popular status as a land management objective and to discuss the forest health paradox that creates conflict between land management philosophies such as multiple use, multiresource, ecosystem management and forest health.

DEFINITION OF FOREST HEALTH

To understand the concept of a healthy forest we first need to examine the divergent views that individuals have regarding what forests provide society. Forests are viewed from a continuum that ranges from the product oriented "utilitarian view" to the "ecosystem centered" view. The utilitarian view is that forests contribute to human welfare by providing commodities to harvest such as timber, fiber, water, forage, and wildlife. The ecosystem centered view is that a variety of basic ecological processes occur within the forest, such as decomposition, nutrient cycling, etc. that should be sustained. The latter view is that commodities can only be harvested to the extent that they do not negatively impact basic ecological processes.

This spectrum of forest uses has lead to divergent views on how forest health is defined. An example of a utilitarian definition is provided by the USDA Forest Service (1993):

> *"Forest health is a condition where biotic and abiotic influences on the forest (that is pests, silvicultural treatments harvesting practices) do not threaten resource management objectives now or in the future."*

This view follows closely the long standing notion of a forest pest as described by Barbosa and Wagner (1989):

> *"A ...... species is considered a pest when it interferes with the intended use of a tree, forest or forest product. The relationship between intended use and type of injury determines the significance of inflicted damage and the appropriate strategy for control".*

Clearly the essential element in the utilitarian definition of forest health is non-interference with land management objectives; essentially a healthy forest is one without pests.

In contrast, the ecosystem centered view of a healthy forest tends to focus on ecological processes and sustainability rather than commodities. This view is expressed in all of the following definitions:

[1] *Professor, School of Forestry, Northern Arizona University, Flagstaff, AZ.*

"A healthy forest is one that is resilient to changes and characterized by tree species and landscape diversity that provides sustained habitat for fish, wildlife and humans" (Joseph et al. 1991).

"In the broadest sense, a healthy forest is a description of a productive, resilient, and diverse ecosystem; a forest with a future." (Wilson 1991, cited in USDA Forest service 1993).

"A forest could be classified as healthy if various biological and physical influences do not threaten present or future management objectives. A forest in good health is a fully functional community of plants and animals and their physical environment. A healthy forest is an ecosystem in balance". (Monning and Byler 1992).

"Healthy is the capacity of the land for self renewal". (Leopold 1949).

The ecosystem centered definitions of forest health attempt to shift the focus of the desirable condition away from commodities to focus more on ecological issues such as resilience, diversity, ecological balance, and sustaining ecological processes such as decomposition, nutrient cycling etc. The notion of a healthy forest as a forest that meets objectives remains as a basic part of the definition.

An issue raised by these divergent views is how to measure forest health. A commodity driven view of a healthy forest allows for relatively easy assessment of the level of output. Land managers generally know how to measure wood volume, forage production, wildlife populations and so on. However, methods for measuring resilience, ecological balance, and sustainability remain largely unexplored. The desired level of either a commodity or an ecological process still needs to be specified in order to judge whether we have met our objectives. A common default position for measuring the level of ecological processes is to estimate the pre-European settlement conditions. There are some problems with this approach in that it is not always possible to estimate pre-settlement conditions nor are those conditions necessarily the most desirable to achieve management objectives.

BASIS FOR POPULARITY OF FOREST HEALTH

I have observed over the past 2 years broad scale support for the notion of forest health. Many professional forest pest management specialists in Canada and the US now refer to themselves as forest health specialists. This level of support for forest health is remarkable given that there are few other land management issues about which there is any significant agreement between user groups. What is the basis for this support of forest health and why has it taken so long for managers to realize this? The reason for the broad support of management for forest health is the divergent views of what constitutes a healthy forest. If you do not have a clear concept of an idea you tend to define that idea in terms you understand. The notion of forest health is popular because everyone defines it in terms of their own personal view of what is desirable from a forest. The wood processing industry supports healthy forests because they are productive and produce large quantities of fiber. Wildlife biologists also want healthy forests because they produce a diversity of wildlife species. Forest health has unanimous support because it is defined in terms of meeting objectives and not in terms of specific commodity outputs. Who would argue that meeting your objective is bad?

THE FOREST HEALTH PARADOX

The above discussion leads us to the forest health paradox. A paradox is a seemingly contradictory statement that is nonetheless true. The forest health paradox is that a healthy forest is both a future desired condition and dependent on the future desired condition (objective) for the forest. Basically all of the definitions of forest health include the element of meeting objectives. Because forest health is dependent on objectives, then managing for forest health is a paradox.

Many forest pest management specialists have, perhaps unknowingly, encountered the forest health paradox. The typical scenario is that a land manager asks for assistance in recommendations to reduce incidence of an insect or disease, for example, dwarf mistletoe (*Arceuthobium* spp). The pest specialist usually responds by asking what is the long term objective for the stand. If the manager suggests their objective is to reduce populations of dwarf mistletoe the pest specialist usually responds by saying reducing dwarf mistletoes is only reasonable in the context of a specific objective. Because dwarf mistletoes play a variety of important ecological roles as habitat and food for wildlife the particular objective might call for increasing populations of dwarf mistletoes. In this scenario the land manager has to either specify objectives or disregard the input of the pest specialist.

FOREST HEALTH AND LAND MANAGEMENT PHILOSOPHY

The land management philosophy being applied to a particular land area greatly influences the likelihood of achieving the condition of a forest considered to be "healthy". Under conditions of a singular objective the achievement of forest health is straightforward. However, the absolute condition of "healthy" will depend on the singular objective. In other words what is considered healthy will vary dramatically depending on which singular objective is established. However, under multiple-use or multiresource (simultaneous multiple use Behan 1990) there is no single measure of health that can possibly measure the degree to which we have achieved the multiple objectives dictated by these management philosophies. Consequently, under current definitions of forest health, a healthy multiple use forest is an impossible dream. If multiple use management is achieved through the aggregation of single uses across the landscape then it might be possible to achieve forest health in the aggregate. In this scenario each stand or single-use area would likely have a different but equally healthy condition. Multiresource management, managing for all outputs on the same land base, can never lead to a singular healthy forest.

The emerging land management philosophy of ecosystem management emphasizes using an ecological approach to achieve multiple-use management by blending the needs of people and environmental values in such a way as to produce diverse, healthy, productive, and sustainable ecosystems. Measuring the degree to which land treatments are consistent with ecosystem management is even more difficult than for other land management philosophies. The healthy forest under ecosystem management can only be defined when the terms diverse, productive and sustainable have far more precise meanings than they do now. The current tendency to define management philosophy in increasingly more subjective terms like sustainable, productive, and diverse only complicates the determination of what constitutes a healthy forest.

SOLVING THE FOREST HEALTH PARADOX

The previous discussion has raised some of the problems associated with the current usage of the term forest health and the forest health paradox. To address these problems I propose two solutions: modification of the notion of forest health to delete emphasis on achieving objectives and/or modification of land management toward a balanced allocation of land to categories of similar uses.

The forest health paradox is created by defining health in terms of achievement of objectives. For the concept of forest health to serve a more useful function it must be defined more precisely in terms of absolute levels of commodities or specific levels of a particular ecological process. An example of a suitable measure of health from a commodity view might be that a stand produce an annual increment no less than 8O% of the increment for similar managed stands. A suitable measure of a healthy forest from the ecosystem management view might be a forest in which forest floor decomposition rates do not vary more than plus or minus 50% from unmanaged stands or perhaps where decomposition rates are within the range of decomposition of managed and unmanaged stands. Defining the levels of any process that falls within the healthy range would require considerable research effort in understanding the role of various agents such as insects and diseases under widely variable stand conditions. Many individuals have viewed variation in populations of insects and diseases as indicators of unhealthy conditions. Determining if a particular population level was unhealthy would be entirely dependent on the natural variation which is indeed enormous for many insects. Other papers in this volume will address the role of insects and diseases in forest ecosystems and how we might define what are the appropriate levels that could be defined as healthy.

The second solution is to adopt a land management philosophy in which the land base is divided into perhaps 3 or 4 groups with distinct objectives. For example, use groups might include a high yield economic zone to maximize wood fiber yield, natural reserves to preserve ecological processes, and wildlife emphasis areas to maximize total species diversity. Just such a land allocation strategy was proposed by Seymour and Hunter (1992) for Maine. I refer to this approach as a balanced land allocation system (BLAST). BLAST would result in different objectives for each land use category and a more realistic measure of forest health (degree to which objectives have been met or specific level of an ecological process) would be achieved. For this scenario forest health would remain clearly a function of the management objective and therefore would vary widely. A major advantage is that it would be considerably easier to arrive at the future desired condition than is currently possible. This second solution attempts to address what is perhaps the fundamental issue — For what purpose should the land be managed? Many of the new approaches to forest land management do not address this issue of lack of consensus on how forests should be used. Until there is agreement on how forests should be managed there will be little agreement on what constitutes a healthy forest.

CONCLUSIONS

In this paper I have discussed the utilitarian and ecosystem centered views of a healthy forest. Definitions to date generally have "achievement of objectives" as an important theme. This component leads to the forest health paradox which is created when a healthy forest is both a future desired condition and dependent on the future desired condition of a forest.

Considerable rethinking of the definition of a healthy forest or of land management strategies is needed. In addition, a greater focus on understanding the role of all components in forest ecosystems, including the physical and biological components (ie., microbes, insects, diseases, animals, plants etc.), is needed. Adopting the goal of managing for healthy forests does not simplify the task for forest managers.

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An Ecosystem-Centered View of Insect and Disease Effects on Forest Health

T.D. Schowalter[1]

Abstract — Phytophagous insects and pathogens traditionally have been blamed for declines in forest health. Accumulating evidence, however, supports an ecosystem-centered view that these organisms respond to changes in forest condition in ways that contribute to maintenance or recovery of forest functional equilibrium, i.e., forest health. Populations of phytophagous insects and pathogens grow on abundant and/or susceptible host species. Pruning and thinning reduce competition, enhance productivity of survivors, and promote non-host plant species. Turnover of plant parts through herbivory, mortality and decomposition maintains nutrient cycling processes essential to soil fertility and permits reallocation of resources from inefficient plant parts to younger tissues. Accumulated fuel increases the likelihood of regular, low-intensity fires that mineralize litter and maintain forest structure. Because tree species are adapted to different conditions following disturbances, increased diversity promotes functional stability and recovery of the forest ecosystem. Few studies have addressed integrated or long-term effects. Contributions to the health and stability of forest ecosystems should be addressed for balanced assessment of impact and need for suppression of insects and pathogens.

INTRODUCTION

Phytophagous insects and pathogens are major components of forest ecosystems, representing most of the biological diversity and affecting virtually all forest processes and uses. They have been viewed as detrimental to forest health and commercial production of forest products and have been targets of suppression efforts. However, accumulating evidence indicates that many "pests" may be instrumental in maintaining ecosystem processes critical to forest health.

Despite theoretical consideration of insect and pathogen contributions to ecosystem stability through feedback effects on ecosystem processes (Mattson and Addy 1975; Schowalter et al. 1981, 1986; Seastedt and Crossley 1984), few experimental studies have evaluated insect and pathogen roles, especially in forests. Advances in this area require an ecosystem framework for experiments, with randomly replicated insect or pathogen abundances, designed to evaluate effects on integrated ecosystem processes. Narrowly-focused studies of effects on commercial tree establishment, growth and survival cannot address integrative and longer-term effects on ecosystem processes that contribute to forest health; studies of integrated ecosystems that do not monitor or manipulate insects and pathogens cannot provide insight into feedback effects. Although non-confounding experimental manipulation of insect or pathogen abundance in mature forests is difficult, techniques have been developed for manipulation of bark beetles (e.g., Schowalter and Turchin 1993). Defoliation often has been simulated by artificial clipping of foliage, but this technique does not simulate all effects of natural defoliation (Schowalter et al. 1986). Adequate replication of randomly assigned treatment plots in integrated ecosystems requires improved cooperation between scientists and resource managers.

This paper describes an ecosystem-centered view of forest insects and pathogens, not as "pests" but as indicators of forest condition (health) and regulators of forest function. Although some insect and pathogen effects may continue to interfere with some forest management goals, consideration of their potential role in maintaining health is essential to balanced assessments of impacts and need for suppression of these organisms and to diagnosis and treatment of forest condition.

[1] *Tim Schowalter is professor of forest entomology and ecosystem ecology in the Entomology Department at Oregon State University, Corvallis.*

ASSESSMENT OF FOREST HEALTH

Discussion of forest health requires definition and appropriate measures of forest health. I will use an ecosystem-based definition of forest health, i.e., the ability to maintain or recover long-term functional equilibrium. Functional equilibrium represents a dynamic balance between dissipative forces and ecosystem processes that maintain suitable conditions for sustained productivity (fig. 1).

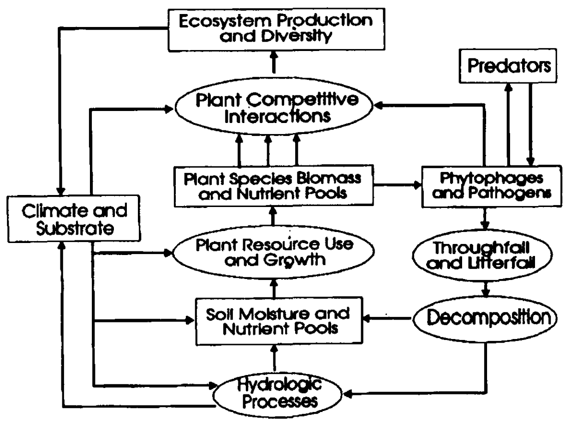

Figure 1. — Simplified ecosystem model showing pools (boxes) and mediating processes (ovals), with arrows showing direction of effect. Positive and negative feedbacks maintain functional equilibrium and modify abiotic conditions.

Ecosystem development reflects the cumulative ability of the community to modify environmental conditions. For example, interception of incoming solar radiation, precipitation, and air currents by vegetation reduces surface temperature, erosion, and wind speed. These processes maintain moderate temperatures and relative humidities, and facilitate acquisition, retention and uptake of resources (e.g., Hobbie 1992, Lucas et al. 1993, McCune and Boyce 1992). The massive structures characterizing forests exemplify ecosystem regulation of climate and nutrient fluxes (Dickinson 1987) and may buffer forests against significant change in external conditions (Franklin et al. 1992).

Forest health depends on replacement of weak or intolerant organisms by more tolerant organisms and on turnover of resources to prevent bottlenecks in fluxes of critical resources (processes accelerated by insects and pathogens) as environmental conditions change (fig. 2). At the same time, species critical to recovery of internal environment and to nutrient retention following disturbances depend on sufficiently large canopy gaps (often created by insects and pathogens) for survival. Accordingly, the shifting mosaic of successional communities that compose the forest landscape represents a healthy forest ecosystem in functional equilibrium with abiotic conditions. Forest health can be represented by multiple equilibrium states reflecting tradeoffs among various regulatory

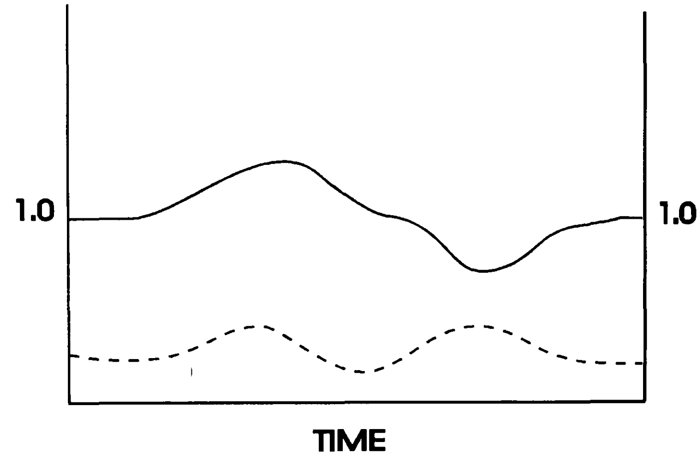

Figure 2. — Hypothetical relationship between resource demand:resource supply ratio (solid line) and insect/pathogen abundance (dashed line). Stress resulting from insufficient resources (demand/supply >1.0) triggers insect and pathogen responses that suppress hosts (reduce demand); nutrient subsidy resulting from demand/supply <1.0 stimulates productivity and tolerance to insect/pathogen-enhanced turnover; balanced demand/supply (= 1.0) limits resources for insects and pathogens.

(feedback) processes. This view differs from a commercial, site or stand based view that emphasizes persistence and maximum growth of a particular forest community.

Impaired health reflects functional degradation, often indicated by insect or pathogen responses to host stress resulting from extreme climate fluctuation (or change), increased crowding, and/or substrate deterioration (Lorio et al. 1993, Mattson and Haack 1987). Stressed plants alter resource allocation between growth, defense, and other metabolic pathways, often becoming more susceptible to phytophagous insects and pathogens (Bazzaz et al. 1987, Lorio et al. 1993). Rapidly growing plants also can become vulnerable as a result of phenological or physiological processes that limit expression of defensive ability (Lorio et al. 1993).

Closely spaced hosts are likely to trigger outbreaks of insects and pathogens. In diverse forests, potential hosts can be "hidden" among non-host vegetation; even vulnerable trees may be relatively resistant to small numbers of insects or pathogen propagules that find their way through surrounding non-hosts (Hunter and Arssen 1988, Waring and Pitman 1983). Tree turnover will be low and continuous in such forests. Conversely, in monocultures tree defenses can be surmounted quickly by larger numbers of insects or pathogens dispersing from surrounding conspecific trees, especially during vulnerable periods. (Schowalter and Turchin 1993, Waring and Pitman 1983). Outbreaks of phytophagous insects and pathogens abruptly reduce dense host populations (to levels incapable of sustaining the outbreak) and promote resource turnover and non-host productivity.

Insects and pathogens (along with fire) traditionally have been considered to impair forest health. However, moderate pruning, thinning and litter mineralization resulting from interaction among insects, pathogens, and fire in unmanaged forests are important processes that facilitate nutrient turnover (especially in arid regions) and maintain vegetation structure and diversity (e.g., Schowalter et al. 1981). Outbreaks and catastrophic fire result from impaired litter decomposition and nutrient cycling in dense managed forests protected from frequent, low intensity fires (Hagle and Schmitz 1993, Schowalter et al. 1981).

Maintenance or restoration of forest health will require attention to ecosystem processes and natural regulatory mechanisms. Measures of forest health include a) balanced resource accumulation in biotic sinks vs. resource supply through input and turnover processes, as this balance affects forest productivity, b) community ability (through species interactions) to regulate nutrient flow rates and lag times and thereby minimize variation, and c) community regulation of internal climate and substrate conditions essential for continuous resource turnover and availability. Bottlenecks in biogeochemical cycling result from excessive tree density and resource accumulation in slow turnover sinks such as wood and from inhibition of critical control processes (such as nitrogen fixation and establishment of species that maintain key processes following disturbances). Insect and pathogen outbreaks can be viewed as triggered responses that indicate and alleviate imbalances in nutrient turnover or other processes (fig. 2).

INSECT AND PATHOGEN EFFECTS ON FOREST FUNCTION

Phytophagous and saprophagous invertebrates and pathogens are capable of rapid responses to changing conditions and can affect vegetation composition and turnover processes dramatically. In an ecosystem (cybernetic) sense, these organisms potentially function to regulate ecosystem processes, including the timing and rate of plant growth, hydrology, carbon and nutrient fluxes, and vegetation composition (Mattson and Addy 1975, Seastedt and Crossley 1984).

Elevated insect or pathogen activity on stressed vegetation reduces growth and hastens host decline and replacement. However, surviving trees may show compensatory growth if defoliation alleviates stressful conditions (Schowalter et al. 1986, Trumble et al. 1993). Wickman (1980) and Alfaro and MacDonald (1988) found that, following the expected short-term growth depression during the period of conifer defoliation, defoliated trees grew faster during the next 2-3 decades, more than replacing the lost growth (fig. 3). In fact, Alfaro and MacDonald (1988) found that the magnitude of this compensatory growth following defoliation was inversely proportional to the severity of defoliation. Schowalter et al. (1991) reported that manipulated levels of defoliation (up to 20%) by lepidopteran larvae did not reduce growth or nutrient content of young Douglas-fir. All saplings doubled in size over the 3-year period, indicating compensation by the defoliated saplings. Compensatory growth may reflect improved water or nutrient conditions, as described below.

Insect and pathogen effects on canopy structure affect interception of precipitation and evapotranspiration. Reduced canopy coverage increases precipitation penetration through the canopy and reduces evapotranspiration (Klock and Wickman 1978, Leuschner and Berck 1985, Schowalter et al. 1991, Swank et al. 1981). Schowalter et al. (1991) reported that 20% foliage removal by native defoliators doubled the amount of precipitation reaching the forest floor under Douglas-fir saplings during the relatively dry spring and summer in western Oregon. Increased soil temperature and moisture, as well as nutrients and, perhaps, herbivore products, improves the litter environment for saprophagous organisms, especially during drier

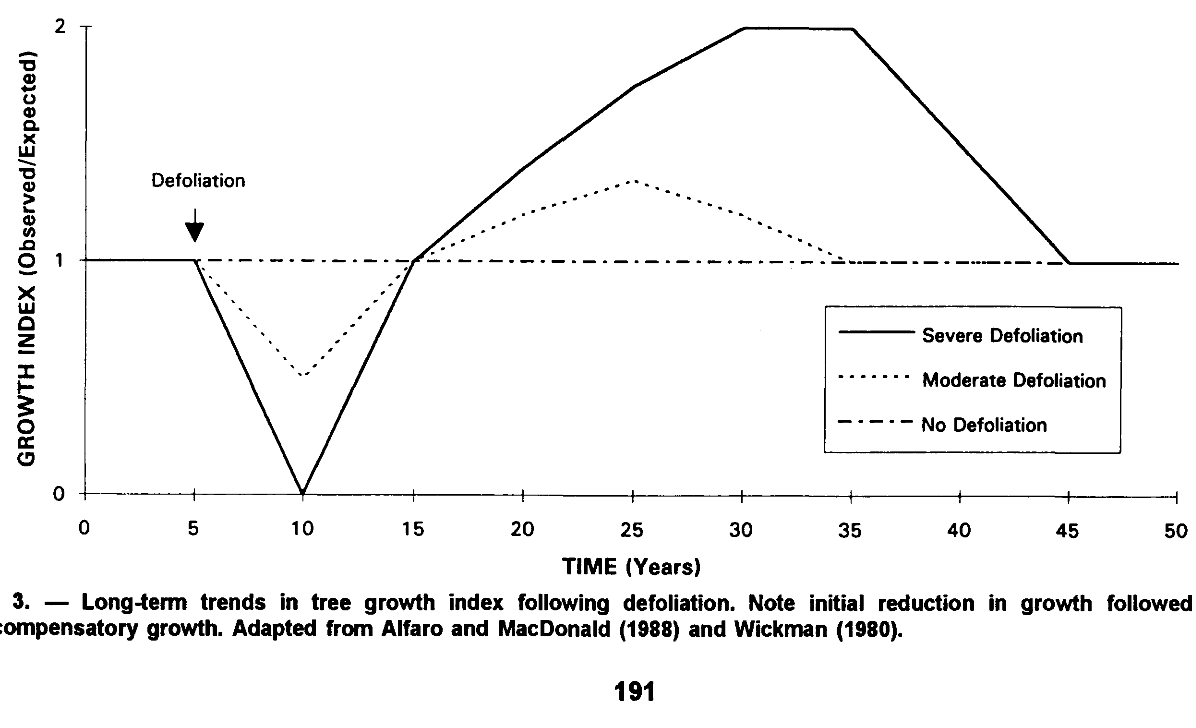

Figure 3. — Long-term trends in tree growth index following defoliation. Note initial reduction in growth followed by long-term compensatory growth. Adapted from Alfaro and MacDonald (1988) and Wickman (1980).

periods (Schowalter and Sabin 1991, Seastedt and Crossley 1983). These organisms are critical to litter decomposition and to porosity (water storage) of woody litter and soil. Improved water balance enhances plant survival during drought.

Carbon flux is affected by changes in canopy structure and plant metabolism, such as caused by insects or pathogens. Oaks, maples and birches showed increased carbon dioxide assimilation by residual and regrowth foliage following artificial defoliation (Heichel and Turner 1983, Prudhomme 1983). Defoliation can mobilize carbon from starch reserves in older foliage and wood for production of new foliage (Webb 1980). Canopy opening increases soil temperature and moisture, conditions that promote decomposition and carbon dioxide flux to the atmosphere. Effects on carbon flux influence carbon transformation and turnover processes, hence ecosystem energetics.

Phytophagous insects and pathogens stimulate nutrient cycling in several ways. These organisms can concentrate major cations several orders of magnitude over plant and soil/litter concentrations (Cromack et al. 1975, Schowalter and Crossley 1983). For example, defoliators are particularly rich sources of potassium, calcium and magnesium (Schowalter and Crossley 1983). The elemental pools represented by these organisms are normally small relative to plant and soil/litter pools but could become important short-turnover pools during outbreaks (Schowalter and Crossley 1983).

Insects and pathogens are major regulators of nutrient turnover from plant biomass. Pruning and/or thinning stimulate plant growth by reducing competition for limited plant resources (Velazquez-Martinez et al. 1992). Folivorous insects and pathogens typically remove less than 10% of foliage and shoots, apparently functioning as natural pruning agents (Schowalter et al. 1986). Removal of these plant parts reduces plant metabolic demands and facilitates reallocation of plant resources.

Turnover of plant parts throughout the growing season provides more constant nutrient input to litter, compared to seasonal litterfall (fig. 4), thereby contributing to forest floor processes and soil fertility (Risley 1993). Kimmins (1972) reported that experimentally elevated sawfly populations increased cesium-134 turnover from young red pine, primarily through leaching from chewed leaf surfaces. Schowalter et al. (1991) and Seastedt et al. (1983), manipulated folivore abundance in young coniferous forest and deciduous forest, respectively, and found that phytophagous arthropods significantly increased turnover of biomass, nitrogen, phosphorus and potassium from foliage to litter (fig. 4). Schowalter et al. (1991), but not Seastedt et al. (1983), also found that phytophagous arthropods significantly increased calcium turnover from young conifers. Calcium generally is considered a relatively immobile element, but enhanced turnover to the acidic soils under conifers could promote soil fertility and biological activity. Insects and pathogens can improve quality of litter detoxified during digestion (Zlotin and Khodashova 1980) but may reduce quality of residual and regrowth foliage with high content of induced inhibitory compounds (Rhoades 1983, Schultz and Baldwin 1982). Defoliation also can stimulate nitrogen fixation and nitrification processes on the forest floor, reflected in increased export by streams (Swank et al. 1981).

Xylophagous insects and root pathogens are instrumental in initiating decomposition and nutrient turnover from dying trees and woody litter. Beetles, especially, penetrate bark and inoculate wood with saprophytic and nitrogen-fixing microorganisms

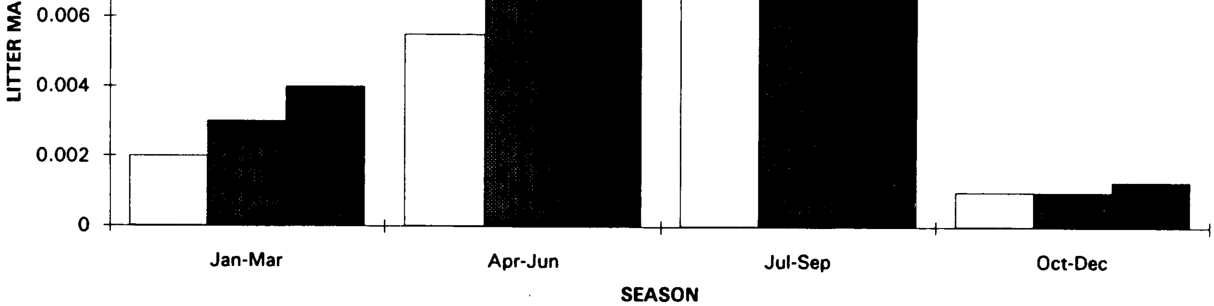

Figure 4. — Seasonal litterfall of young Douglas-fir in western Oregon during 1983-1986, as affected by sap-sucking insects feeding April-September, defoliating insects feeding September-June (peak in April-June), and insect exclusion. Defoliation significantly (P < 0.0005) increased litterfall and turnover of nitrogen, potassium and calcium during April-June, compared to other treatments. Data from Schowalter et al. (1991).

(Ausmus 1977, Bridges 1981, Dowding 1984, Schowalter et al. 1992). The winding galleries of xylophagous beetles and termites ensure rapid inoculation of microorganisms throughout logs (Dowding 1984, Schowalter et al. 1992). Basidiomycete fungi (including facultative or obligate pathogens) typically are the major degraders of lignin and cellulose, but a variety of ascomycete and deuteromycete fungi and bacteria provide vitamins and fixed nitrogen essential to fuel wood decay and, in turn, further transform breakdown products of lignin and cellulose (Blanchette and Shaw 1978).

Nitrogen fixation and nutrient accumulation in decomposing wood create nutrient "hot spots" that facilitate germination of some trees (Schowalter 1992). Soil under decomposing logs may receive considerably greater nutrient input than does soil under leaf litter. Accordingly, mycorrhizal fungi and tree roots infuse decomposing logs, transporting essential nutrients to living trees. In nitrogen-limited forests, termite colonies in living trees might provide nitrogen to the host trees.

The process of ecosystem recovery from disturbance, as affected by insects and pathogens, also contributes to nutrient balance in forest ecosystems. Nutrients, especially nitrogen, are more available in canopy gaps as a result of reduced uptake and storage in tree tissues and increased turnover and mineralization, as above (Schowalter et al. 1992, Waring et al. 1987). Recovery of ecosystem function within the "gap" is essential to prevent loss of sediment and resources.

Recovery is facilitated by fast-growing early successional species that incorporate nutrients into biomass. Nitrogen-fixation during this stage is particularly important to succeeding forest stages that may largely depend on stored nitrogen. Pruning, thinning and enhanced nutrient turnover by phytophagous insects and pathogens may initially stimulate rapid growth by hosts flourishing under optimal resource conditions. As these species grow and later successional species become established, increasing biomass leads to competitive stress, eventually triggering insect and pathogen outbreaks. Functional equilibrium (but not appearance) is maintained by a rapid successional transition to more tolerant, nutrient-conserving species. This transition is facilitated by the successive colonization of predisposed hosts by insect and pathogen species that accelerate host decline and replacement. Self-thinning might eventually produce this transition but at increased risk to critical ecosystem processes.

Unfortunately, rapidly-growing early-successional trees most valued for commercial timber and fiber production also are most likely to suffer from resource limitation and insect/pathogen response. Recognizing insects and pathogens as indicators of forest health will facilitate development of management practices that remedy the underlying imbalances, rather than simply treating symptoms. In forests managed for sustainable uses, consideration of insect and pathogen roles in integrated ecosystems is essential to balanced assessment of impacts and protection of natural mechanisms for maintaining functional equilibrium (health).

CONCLUSIONS

Forest health, defined as maintenance of functional equilibrium, can be evaluated as the degree to which the forest maintains balance between vegetative demand for resources and long-term resource availability and maintains moderate internal environmental conditions suitable for survival of critical functional elements. Impaired health is indicated by species decline, resource bottlenecks, and insect or pathogen responses to host stress. This view of forest health requires greater attention to ecosystem processes underlying forest condition.

Accumulating data suggest that forest insects and pathogens not only respond to changing host condition, but may represent regulatory mechanisms for controlling dominance by intolerant vegetation and alleviating bottlenecks in biogeochemical cycling processes fundamental to forest health. These roles appear to promote functional equilibrium and capacity to recover functional equilibrium following disturbances. Accordingly, insect and pathogen effects may become more pronounced as ecosystems respond to global change. The limited evidence for insect and pathogen contributions to forest health should encourage a broader experimental approach to studying and managing these organisms. Longer-term studies of integrated effects of insects and pathogens on ecosystem function are necessary to quantify the importance of these roles and to provide more balanced assessments of impacts and need for suppression of these organisms.

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Status of Insects and Diseases in the Southwest: Implications for Forest Health

J.L. Wilson and B.M. Tkacz[1]

Abstract — Most insects and pathogens affecting forests in the Southwest are naturally occurring components of Southwestern ecosystems and play an important role in their dynamic processes. They provide food and habitat for animals, affect short and long term vegetative structural diversity, and contribute to the biological diversity of the system. These organisms, along with fire, are among the major disturbance causing agents affecting vegetative change in the Southwest. Direct and indirect evidence suggests that the incidence of bark beetles, western spruce budworm, dwarf mistletoes, and root diseases has changed in recent years. These shifts in insect and disease activity are thought to reflect changes to the underlying structure and function of forest ecosystems in the Southwest. The introduction of exotic agents, such as white pine blister rust, will also have significant impacts on southwestern forest ecosystems. The long term success of Ecosystem Management in the Southwest will depend, at least in part, upon how well we understand, and incorporate into our management, the effects of insects and diseases on the landscape now and into the future.

INTRODUCTION

Most insects and pathogens affecting forests in the Southwest are naturally occurring components of Southwestern ecosystems and play an important role in their dynamic processes. They provide food and habitat for animals, affect short and long term vegetative structural diversity, and contribute to the biological diversity of the system. These organisms, along with fire, are among the major disturbance causing agents affecting vegetative change in the Southwest. Outbreaks of forest insects and diseases can result in shifts in vegetative species composition and structure. The degree of perturbation depends upon the particular insect or disease and on the condition of the ecosystem affected.

Evidence suggests that the incidence of some of these agents has changed in recent years. These shifts in insect and disease activity are thought to reflect changes to the underlying structure and function of forest ecosystems in the Southwest. In other cases, there is no clear evidence of changes in insect and disease activity, however, evidence suggests that forest conditions have changed dramatically in the Southwest. Some of these conditions may favor outbreaks of native forest insects and diseases. Recent discovery of the non-native pathogen causing white pine blister rust here in the Southwest will also have a major impact on southwestern forests. We will examine the status of some of these agents at the present time and in the recent past, speculate about their prospects for the future, and discuss implications for forest health.

WHAT IS A HEALTHY FOREST?

Forest health is a concept developed in recent times. It has been defined in a number of ways. Knauer et al. (1988) defined a desired state of forest health as a condition where biotic and abiotic influences on the forest (ie, insects, diseases, atmospheric deposition, silvicultural treatments, harvesting practices) do not threaten management objectives for a given forest unit now or in the future. Further they added that a healthy forest can be described by many standards, each related to a management objective for the forest. No single standard or definition covers all objectives. Monnig and Byler (1992) expanded this definition. They added the component of ecosystem function. They stated that a forest in good health is a fully functional community of plants and animals and their physical environment. A healthy forest is an ecosystem in balance. They

[1] *U.S.D.A Forest Service, Southwestern Region, State and Private Forestry and Forest Pest Management, Flagstaff, Arizona, USA.*

also recognized that this balance was not static, and that to focus on forest health means to focus on forest processes. They conclude that the health of a forest is best measured by comparing its current patterns and rates of change with historic patterns, recognizing, of course, that historic patterns were not completely static either. They chose to measure forest health as a combination of ecosystem function and management objectives, where management objectives reflect ecosystem limitations. We will use their definition as our standard for assessing implications of insects and diseases for forest health.

STATUS OF INSECTS AND DISEASES

Bark Beetles

Numerous species of bark beetles affect forests in the Southwest, attacking all species of coniferous trees. Most are fairly host specific, and are confined to primarily one tree species. Some of the more important ones include: *Dendroctonus rufipennis* Kirby, the spruce beetle, in Engelmann spruce; *D. pseudotsugae* Hopkins, the Douglas-fir beetle, in Douglas-fir; *Dryocoetes confusus* Swaine, the western balsam bark beetle, in subalpine fir; *Scolytus ventralis* LeConte, the fir engraver, in white fir; and the mountain pine beetle, *Dendroctonus ponderosae* Hopkins, western pine beetle, *D. brevicomis* LeConte, roundheaded pine beetle, *D. adjunctus* Blandford, pine engraver, *Ips pini* (Say), and the Arizona Five spined Ips, *I. lecontei* Swaine, in ponderosa pine (Wood, 1982).

The direct effects of bark beetle infestation include tree mortality, and top-killing (Stark, 1982). Outbreaks can result in changes in forest density, structure and composition (Schmid and Frye, 1977). Bark beetles can affect plant and animal species composition and abundance both directly and indirectly by their activity. They affect animals directly by providing food and habitat, and indirectly by modifying environmental conditions that may favor some species over others. Bark beetle outbreaks have also been reported to predispose affected areas to more extreme fire events (Stark, 1982).

Historically, large bark beetle outbreaks have probably always been a significant type of natural disturbance in certain forest types. In the spruce-fir type, Baker and Veblen (1990) report that historic photos and tree dendrochronology data indicate that spruce beetle has been a major disturbance agent from central New Mexico to north central Colorado since at least the 19th century. Their paper reports that spruce beetle outbreaks have been a significant type of natural disturbance in these forests, perhaps comparable to fire. Sizeable outbreaks probably have occurred periodically whenever favorable stand conditions developed. Schmid and Hinds (1974) describes a hypothetical process of succession in the spruce-fir type involving the spruce beetle. Spruce beetle outbreaks resulted in shifts in species composition, favoring sub-alpine fir over spruce. Over time, fir, a shorter lived species dies out and spruce again predominates, matures, and again is ripe for another spruce beetle outbreak.

Two major spruce beetle outbreaks occurred during the 1980's in the Southwest. An outbreak occurred in the White Mountains in Arizona, primarily on the Fort Apache Indian Reservation. This outbreak covered approximately 20,000 acres between 1981 and 1984, killing approximately 400,000 spruce (Linnane 1985). A second major outbreak of spruce beetle occurred in the Pecos Wilderness of northern New Mexico between 1982 and 1985, killing an estimated 30,000 spruce over approximately 7,000 acres according to aerial detection survey records. These outbreaks probably do not represent a change in status of spruce beetle. This insect has caused large outbreaks in the recent past and will probably continue to do so. A large portion of the spruce-fir type in the Southwest is densely stocked with large diameter spruce which are in excess of 150 years old and in a state of declining vigor due to competition. These stands are very susceptible to spruce beetle once an outbreak is initiated.

Similarly within the ponderosa pine type on the Kaibab plateau in northern Arizona, there is evidence of numerous mountain pine beetle outbreaks dating back to 1837. Blackman (1931) reports on a large outbreak that occurred between 1917 and 1926, and was caused primarily by the mountain pine beetle, referred to then as the Black Hills beetle, (Blackman 1931). In this outbreak, he reports that 12 percent of the ponderosa pines were killed on the plateau both on the Kaibab National Forest and Grand Canyon National Park. Using increment cores and evidence of pitch pockets (from unsuccessful attacks), he also found evidence of older outbreaks: 1837-1846, 1853-1864, 1878-1882, 1886-1892, and 1906-1910. He reports that the largest of these outbreaks was the one occurring during 1886-1892. Since 1926, only three localized and low level outbreaks have occurred on the plateau, 1935-1938, 1950, and 1973-1977 (Parker and Stevens 1979). Though no outbreak has occurred in recent years, stand conditions in many areas on the Kaibab Plateau are very favorable for development of mountain pine beetle outbreaks, consisting of dense stands of ponderosa pine larger than 10 inches in diameter. It is probably only a matter of time before another large outbreak occurs.

Elsewhere in the Southwest, reports of historic beetle activity, particularly outbreaks, are scarce. This may be due, at least in part, to the extensive early logging activity that occurred across much of the pine type, outside of the Kaibab plateau, in the Southwest, where some of these other bark beetles are important. Hopkins (1909) reports that in general the amount of tree mortality caused by what he called the Black Hills beetle, now the mountain pine beetle, was less in New Mexico and Arizona and southern Colorado than in the Black Hills. He also reports that the western pine beetle, which was then considered to occur along the Pacific coast from California to Washington and Idaho, caused more mortality than what he called the southwestern pine beetle, now synonomized, occurring in the Southwest. Pearson (1950) reports bark beetles to be among the four main causes of mortality and notes that they are a major cause of death in reserve stands, causing about one third of the mortality reported by all causes. In virgin stands monitored between 1925 and 1940, he notes that they accounted for 1.6 percent of the mortality to trees overall, with somewhat higher rates for trees in the larger diameter classes. Meanwhile in cutover stands they accounted for 0.3 percent of the mortality to trees.

In recent times, however, some large outbreaks have occurred. Perhaps the most notable case is in the Sacramento Mountains of southeastern New Mexico. Prior to the 1970's, outbreaks in the Sacramentos were small, a few thousand acres at most (Massey et al 1977). However, since 1971 two large outbreaks have occurred. In one in the early 70's, an estimated 400,000 trees were killed on over 150,000 acres (Massey et al 1977). This outbreak, which occurred in second growth ponderosa pine, resulted in mortality to between approximately 11 and 54 percent of ponderosa pines in sampled stands (Stevens and Flake 1974). Meanwhile, the average diameter of ponderosa pine remained about the same. Overall the outbreak resulted in a shift from ponderosa pine to other species such as Douglas-fir, white fir, southwestern white pine, pinyon, juniper, oak and aspen. Between 1990 and 1992, another outbreak involving both the roundheaded pine beetle and the western pine beetle killed an estimated 100,000 trees over some 87,000 acres. Around the same time, two smaller yet significant outbreaks of roundheaded pine beetle have occurred in the Pinaleno Mountains of southeastern Arizona (Flake 1970, Wilson 1993). We are not aware of any sizeable outbreaks prior to this time in the Pinalenos. We expect that this trend will continue and may extend to other areas in the southwest, primarily due to increasing tree densities as compared to prevailing conditions present prior to european settlement.

Western Spruce Budworm

The western spruce budworm (WSB), *Choristoneura occidentalis* Freeman, feeds on foliage of true firs, Douglas-fir and spruce throughout the western US. In the Southwest, its principal hosts are white fir and Douglas-fir (Linnane 1986). Larvae feed primarily in buds and on foliage of the current year. Complete defoliation can occur when outbreaks persist for several years. Sustained heavy defoliation can result in decreased growth, tree deformity, top-killing, and death. Stand level effects include changes in stand structure and composition. Tree mortality is generally more prevalent in the smaller, suppressed, understory trees so outbreaks can result in fewer understory trees and increases in the average diameter. Species composition may be shifted toward nonhost species or less susceptible species.

Like bark beetles, WSB affects plant succession. In mixed species stands on true fir, spruce, and Douglas-fir habitat types, WSB outbreaks favor seral trees (Wulf and Cates 1987). Meanwhile in pure stands of climax hosts, outbreaks have been likened to a thin from below. Fire exclusion, grazing, and past logging have had a great effect on southwestern forests. These have in turn changed the nature of southwestern forests. The effects of these practices have also changed the nature of WSB outbreaks.

Changes in WSB populations and stand structure and composition affect other animals as well. Twenty six species of birds have been found to feed on the western spruce budworm in the northwest. Some species consume a considerable number of budworm larvae and pupae and increase enormously on sites experiencing budworm outbreaks (Garton, 1987).

An excellent record of the history of western spruce budworm outbreaks for southern Colorado and northern New Mexico has been put together by Swetnam and Lynch (1989). Western spruce budworm outbreaks have occurred in this region at irregular intervals in mixed conifer forests for at least the last 300 years. At least nine outbreaks have been identified in the mixed conifer stands of the Colorado Front Range and Sangre de Cristo Mountains between 1700 and 1983 based on tree ring studies. The results of this study indicate that a change has occurred in the incidence of budworm outbreaks during this century. Though the frequency of moderate to severe outbreaks during this century is not clearly more or less than during previous centuries, the spatial and temporal pattern of occurrence has changed. Outbreaks in the latter half of this century have become more synchronous over the host type. This suggests that recent outbreaks have become more extensive than previous outbreaks (Swetnam and Lynch 1989). There was also evidence suggesting that the most recent outbreak may have been more severe than past ones. The authors hypothesize that this change is due to changes in age structure and species composition, which have favored western spruce budworm, involving widespread establishment of younger multi-storied, shade-tolerant, budworm host susceptible trees resulting from management practices around the turn of the century.

Dwarf Mistletoes

The dwarf mistletoes (*Arceuthobium* spp.) are highly specialized dicotyledonous parasites of conifers throughout North America that have evolved with their hosts, as evidenced by fossil records dating back to the Pleistocene epoch (Hawksworth and Wiens 1972). More than 2 million acres of National Forest lands in Arizona and New Mexico are infested with dwarf mistletoes (Johnson and Hawksworth 1985). Most southwestern conifers are parasitized by species of *Arceuthobium*, however, the most significant damage occurs to ponderosa pine infected with the southwestern dwarf mistletoe (SWDM), *A. vaginatum* subsp. *cryptopodum* (Engelm.) Hawksw. & Wiens) and Douglas-fir infected with the Douglas-fir dwarf mistletoe, *A. douglasii* Engelm.

The distribution and rate of increase in dwarf mistletoe populations are affected by numerous host, stand and environmental factors including: site quality, host vigor, host age, stand density, stand structure, stand composition, and stand history (Parmeter 1978). As parasites, dwarf mistletoes cause significant changes in physiological processes and structural characteristics of infected trees which result in changes in the structure and function of forest communities (Parmeter 1978, Tinnin 1984). Infected host trees are slowly weakened and eventually killed as the dwarf mistletoes drain them of water and nutrients (Tocher et al. 1984). Survival of ponderosa pine is influenced by the severity of dwarf mistletoe infection. Hawksworth and Geils (1990) determined 32-year survival rates of tagged ponderosa pines with various intensities of dwarf mistletoe at Grand Canyon National Park, AZ. While more than 90% of the uninfected and lightly infected trees survived the 32-year period, only 5% of the heavily infected trees over 9 in. dbh and none of those in the 4 - 9 in. class survived.

Dwarf mistletoes are natural components of many forest ecosystems in the West. They provide forage for many species of birds and mammals, and the witches' brooms they cause can serve as nest sites. As groups of trees are killed by dwarf mistletoes, the microclimate and vegetation composition of the openings are affected. Some of the changes in forest structure brought about by dwarf mistletoes can be beneficial to some wildlife species. The abundance of birds and species richness of bird communities were enhanced by dwarf mistletoe infection in central Colorado (Bennetts 1991).

Wildfires are one of the primary ecological factors in determining the distribution and intensity of dwarf mistletoes in unmanaged coniferous forests (Alexander and Hawksworth 1976). Relatively complete burns tend to have a sanitizing effect on infected stands, while partial burns can lead to rapid infection of regeneration if scattered infected trees remain following the fire. Harrington and Hawksworth (1990) evaluated the interactions of fire and dwarf mistletoe on mortality of ponderosa pine following prescribed burning in Grand Canyon National Park. They found that infected trees suffered more crown scorch than healthy trees because they had flammable witches brooms and lower crowns. Moreover, given equal amounts of crown scorch within the 38% to 87% range, heavily infected trees had less than half the probability of survival that uninfected trees have.

Fire scar chronologies from southwestern forests for the period from 1700 to 1900 indicate mean fire intervals of 4 to 5 years for ponderosa pine sites and 6 to 10 years for mixed conifer sites (Swetnam 1990). Since severe dwarf mistletoe infection leads to accumulations of dead trees, witches' brooms, and other fuels, the frequent low-intensity fires common in pre-european settlement forests probably reduced dwarf mistletoe in many areas (Parmeter 1978). Surveys of the ponderosa pine forests in Arizona and New Mexico conducted in the 1950's and 1980's (Maffei and Beatty 1988) suggest that the incidence of southwestern dwarf mistletoe may have increased due to human activities such as inappropriate harvesting practices and suppression of wildfires.

Although the basics of dwarf mistletoe control have been known for a long time (Koristian and Long 1922, Pearson 1950), past cutting practices may have exacerbated SWDM infection in southwestern ponderosa pine stands. Light improvement selection cutting was extensively practiced throughout the Southwest until the 1980's (Heidmann 1983). Under this system, mortality losses in virgin stands were reduced by harvesting merchantable trees that were dying or expected to die during the following 20-year cutting cycle. A long-term study of silvicultural control of SWDM on the Fort Valley Experimental Forest, near Flagstaff, AZ (Heidmann 1983) compared the effects of light improvement selection, limited control, and complete control in heavily infected mature ponderosa pine stands. After 27 years, the only effective silvicultural control method was complete removal of infected overstory and understory trees.

Root Diseases

Root diseases are common in many mixed conifer, spruce-fir and some pine stands throughout the Southwest. A survey of commercial timber-producing lands on six National Forests in Arizona and New Mexico indicated that root diseases and associated pests were responsible for about 34 percent of the trees killed (Wood 1983). Root diseases are caused by decay fungi, such as *Armillaria* spp.(Fr.:Fr.)Staude, *Heterobasidion annosum* (Fr.) Bref., and *Inonotus tomentosus* (Fr.:Fr.) S. Teng. They injure trees by decaying and killing roots. These fungi can survive for decades in the roots of stumps and snags and can infect susceptible regeneration through root contact (Shaw and Kile 1991, Otrosina and Cobb 1989, Tkacz and Baker 1991). Infection by *H. annosum* can also occur by windborne spores that infect freshly cut stumps or basal wounds (Otrosina and Cobb 1989). *Armillaria* spp. produce rhizomorphs that grow through the soil and can penetrate root bark (Shaw and Kile 1991). Immediate regeneration of susceptible species on sites infested with root disease perpetuates and may increase the disease in subsequent rotations (Tkacz and Hansen 1982).

Some root diseases can kill young trees rapidly, but others slowly decay the roots and rob the trees of water, nutrients and structural support. Infection by root diseases results in reduced growth, increased mortality (often by bark beetle attack), altered stand structure, and sometimes, large openings in forests. *Armillaria* root disease was found to be the most frequent cause of canopy gaps in white spruce in the Black Hills of South Dakota (Lundquist 1993). Surveys of Engelmann spruce stands in Utah (Tkacz 1987) and Arizona (Fairweather and Wilson 1991) indicated that infection centers can occupy up to one third of the total stand area.

After a half century of fire exclusion and selective harvesting, the incidence of root disease is suspected to have increased in mixed conifer stands in the Intermountain Northwest as ponderosa pine, a species that is relatively insect and disease resistant, is replaced by Douglas-fir and true firs, species that are much more prone to infection by root diseases (Hagle and Goheen 1988). Similar shifts in species composition are occurring in southwestern mixed conifer stands (Swetnam and Lynch 1989). Even though direct comparisons of root disease incidence in pre-european settlement and present times are not possible, these diseases have probably increased in southwestern forests due to the greater abundance of susceptible hosts and inoculum created by harvesting.

White Pine Blister Rust

White pine blister rust is caused by the exotic fungus *Cronartium ribicola* J.C. Fisch. This fungus has spread throughout virtually the entire range of western white pine since its introduction to British Columbia in 1910. It was discovered in the Southwest for the first time in March 1990 on southwestern white pine in the Sacramento Mountains near Cloudcroft, New Mexico (Hawksworth 1990). Surveys indicate that this disease is now present throughout most of the range of southwestern white pine on the Lincoln National Forest and the adjacent Mescalero Indian Reservation (Hawksworth and Conklin 1990). The fungus has caused seedling and sapling mortality, as well as extensive branch mortality on all size classes of southwestern white pine in the affected areas. This disease will likely have a major impact on the white pine population in the Sacramento Mountains. Young trees will suffer more damage than larger, older trees since they are more prone to girdling cankers. The gradual decline in southwestern white pine regeneration will have significant impacts on the species diversity of the mixed conifer forests in the Sacramento Mountains. This disease can also be spread, either by man's activities or by windblown spores, to other areas of southwestern white pine, limber pine, and bristlecone pine in the Southwest.

IMPLICATIONS FOR FOREST HEALTH

Prior to european settlement, disturbance events were infrequent and high intensity in some forest types, such as the spruce-fir type (Covington and Moore 1992). Bark beetles and fire were among the major disturbance agents. The same pattern exists today and has many repercussions for management.

In other forests types, such as the ponderosa pine type and lower elevation mixed conifer type, disturbance events were frequent and low intensity (Covington and Moore 1992), predominated by the effects of fire. Since european settlement the patterns of disturbance have changed and have resulted in different forest and insect and disease conditions (Covington and Moore 1992, Swetnam and Lynch 1989).

In the ponderosa pine type, these changes have resulted in higher densities, and canopy closures of ponderosa pine (Covington and Moore 1992). At the same time fire frequencies have decreased and fire size has increased. Both of these changes, along with past management practices, may have resulted in an increase in dwarf mistletoe incidence and severity. Many of these conditions also favor bark beetle outbreaks, and similar to fire, beetle outbreaks in the future may become larger and more intense.

In the mixed conifer cover type, dense, multi-storied, second growth stands predominated by Douglas-fir and white fir, are believed to have replaced more open stands composed of these species and a significant component of ponderosa pine, as a result of selective harvesting, and fire exclusion (Swetnam and Lynch 1989). These stands are now very susceptible to both western spruce budworm as well as root disease.

The introduction of the exotic white pine blister rust fungus into southern New Mexico will have significant impacts on the biodiversity of mixed conifer forests since southwestern white pine shows little innate resistance to this pathogen. Introductions of other exotic pests may have similar disastrous consequences.

Recently the Southwestern Region of the US Forest Service adopted a new management philosophy called Ecosystem Management. This change reflects the Region's desire to take a more holistic approach to management of forests in the Southwest, one that is ecologically based. The focus of this new strategy will be on desired future conditions of the land and its human communities at multiple scales, always striving to maintain a balance between sustaining (1) the resource, (2) lifestyle or social goals, and (3) economic goals. This new strategy emphasizes sustainability, multi-resource management, integrated inventories and analytical tools, and (wherever possible) ecosystem management over single species management.

The Management Recommendations for the Northern Goshawk (Reynolds et al. 1992) represent one of the Region's first attempts at ecosystem management for a Forest Service Sensitive Species. The hallmark of this strategy is that it manages for habitat for both the goshawk and its prey species, thus it is a multi-species approach to management for this sensitive species.

The long term success of these new strategies will depend, at least in part, upon how well we understand, and incorporate into our management, the effects of insects and diseases on the landscape now and into the future. This will require new knowledge and new analytical tools. New knowledge will be required about insects and diseases, how they affect and have affected ecosystems and ecosystem processes, and how our new management strategies will affect insects and diseases and in turn the landscape. In order to assess implications for ecosystem management we will need to understand these effects at different temporal and spatial scales. Accomplishing this will require the forging of better ties between entomologists, pathologists and specialists and researchers in other disciplines.

We have many excellent analytical tools that have served us well over the years, however, many have been geared to analyzing effects of insects and diseases on the timber resource outputs. For example many bark beetle risk or hazard rating systems display outputs in terms of low, medium, or high risk, however these terms mean little to a wildlife biologist. To implement ecosystem management we need to be able to describe potential effects to the vegetation. What does high risk mean? Does it mean that there is a great likelihood that a goshawk territory may be compromised? These kinds of tools are still useful today for planning, however they need to provide answers that all resource managers can use in terms they understand.

Models are another tool that can help us better understand and display effects of insects and diseases. A number of models have been developed by Forest Service Research and Forest Pest Management's Methods Application Group and are available for Forest Service managers. These are attached to the Forest Service's forest vegetation simulator (FVS) and can be used to display effects of a number of forest insects and diseases. The sophistication of these tools has been increasing in recent years. One of these models can simulate effects of two pests. Another one in development will be able to display in a spatial format the progress of a bark beetle outbreak across a landscape. Implementing ecosystem management will require more of these tools, ones that will simulate more than one insect and disease across a landscape and will display those effects spatially.

CONCLUSIONS

Forest insects and diseases have and will continue to be dominant agents of change in many forest ecosystems. A better understanding of how these agents affect ecosystem functions, processes and linkages is vital to the success of long term ecosystem management.

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Use and Abuse of Insect and Disease Models in Forest Pest Management: Past, Present, and Future

A.M. Liebhold[1]

Abstract — Most forest pest populations fluctuate through large ranges of density over relatively short periods. This eruptive nature of pest populations is problematic to their management because impacts are difficult to predict. In the late 1950s and early 1960s, "life tables" and associated analytical techniques were developed as methods for summarizing population parameters such as natality, dispersal, and mortality due to specific agents. Ecologists used these data to develop "process models" that simulated these numerical relationships. By the late 1960's and early 1970's, these process models had expanded into highly complex systems models, many of which encompassed 50 or more population processes. Results from large systems models were mixed; they were useful for predicting and understanding the impacts of pests on timber production, but they were of little use either for understanding or predicting pest dynamics. Pest population models most successfully used for prediction in forest pest management have been simple models that are derived statistically instead of being process-based. There has recently been considerable effort in the development of expert systems or decision-support systems that aid pest management decision-making by providing information from rule bases or simple simulation models. Although these systems may have great potential for increasing the efficiency of pest management, developers need to avoid the excessive complexity that historically crippled the application of process models. Simulation of pest population processes over space is the new frontier in the development of models. While the complexity of spatial relationships has previously prevented the development of spatially explicit models, the advent of geographical information systems and spatial statistical procedures provides new opportunities. Incorporation of ecological interactions at various spatial scales ultimately will lead to management of pest populations at the landscape level and ultimately should lead to more ecologically sound practices.

INTRODUCTION

Of the thousands of insect species that inhabit North American forests, most remain at relatively innocuous densities (Price et al. 1990). However a small fraction of forest insects are "eruptive"; they occasionally reach epidemic densities that have spectacular effects on their habitat. Pest outbreaks can have devastating effects on forest resources, and therefore, considerable effort often is expended to reduce their impacts. Over one billion pounds of pesticides are used annually in the United States (Pimentel et al. 1978). Though forest usage is a small fraction of this total, pesticide use can be extensive. For example, more than 1 million ha of forest land in the Northeast have been sprayed to suppress gypsy moth outbreaks during the past decade as part of the U.S. Cooperative Forestry Assistance Act of 1978 (USDA 1985).

[1] *Research Entomologist, Northeastern Forest Experiment Station, USDA Forest Service, Morgantown, WV.*

With some exceptions, chemical pesticides were not widely used for pest control until the 1940's. Prior to that time, management of forest pests was accomplished mainly through the use of silvicultural and biological control tactics. With maturity of aviation technology following World War II and the discovery of DDT and other synthetic pesticides, chemical control of pests through aerial application increased to a position of dominance in agriculture as well as in forestry. In the 1960's Rachel Carson (1962) and others exposed the damages caused by the application of DDT to control forest pests and since then there has been growing pressure to reduce pesticide usage in forestry while economic and sociological pressures to reduce pest impacts have persisted. The concept of integrated pest management (IPM) was developed in the late 1960's in response to growing concerns about excessive use of pesticides. Initially the concept of IPM meant simply the use of a diversity of control strategies to manage pests (i.e., biological, chemical, and silvicultural control). Since its inception, IPM has evolved to an approach to pest management that emphasizes integration of information during the decision-making process; pest control activities are minimized by using information on pest levels, habitat susceptibility, and expectations of pest impact to evaluate the necessity of action (Waters and Stark 1980).

An ability to forecast when and where pest outbreaks will occur as well as their impacts is an essential component of pest management. Both in primitive pest management programs and in more complex integrated management programs, the timing of intervention measures necessitates some forecast of future pest impacts. Unfortunately, the ecological relationships that determine fluctuations in forest pest population levels typically are complex and these fluctuations can be difficult to predict (Logan 1991). Over the last 50 years there have been numerous attempts to use mathematical models to make these predictions. Below I have outlined the historical development of this work, recent developments, and finally a glimpse into the future to see new developments that are likely.

THE PAST

The patterns and mechanisms of insect population fluctuations have been the subject of considerable research. Forest insects are desirable systems for population ecology studies because human-caused perturbation of their habitats is rare. Most of the "milestone" studies of insect population dynamics have focused on forest insects. These studies include Morris's (1963) study of the spruce budworm, *Choristoneura fumiferana*, Varley and Gradwell's (1968) classical work with the winter moth, *Operophtera brunata*, Wellington's (1964) work with the western tent caterpillar, *Malacosoma pluviale*, and many other studies.

A great deal of population dynamics research has focused on understanding specific population processes that affect the population dynamics of herbivorous insects. These processes include predator-prey relationships (e.g. Holling 1959), host-pathogen relationships (e.g. Anderson and May 1980), dispersal processes (e.g. Greenbank et al. 1980), host plant relationships (e.g. Schultz and Baldwin 1982), and competition (Strong 1984). These studies have revealed that the processes that determine the dynamics of any herbivorous insect are complex, often encompassing several trophic levels. From complexity arose a need for mathematical models as a method for interpreting and understanding these complex relationships. Initial attempts at modeling began some time ago with simple, theoretical approaches, such as the exponential growth model (Malthus 1798), the logistic model (Verhulst 1838), Nicholson-Bailey (1935) and Lotka-Voltera (Lotka 1925) models, which summarized intraspecific competition or interactions between a population and one of its predators using one or two differential equations.

In the late 1950's and early 1960's, life-tables and associated analytical techniques were developed as methods for summarizing population processes such as natality, dispersal, and mortality due to specific agents (e.g. Morris 1963). With these data in hand, entomologists were able to develop what have become known as "process models." These models typically predicted changes in population density as a function of a handful of variables, such as host or natural enemy population densities, and exogenous variables, such as temperature and precipitation. Independent variables included in these models usually were selected based on a analyses of life tables, such as "key factor analysis" (Morris 1963), that indicated the importance of that variable in predicting population changes. Models often were fit using linear regression. Again, forest defoliators were the focus of much of this research. Process models were developed for important forest pests such as the spruce budworm (Morris 1963), the gypsy moth (Campbell 1967), the winter moth (Varley and Gradwell 1968), and the western pine beetle (Stark and Dahlsten 1970). The purpose of these modeling efforts was to serve both as aids to understanding the complexities of pest population dynamics (Morris 1963) and for actual prediction of future population levels for use in pest management (Campbell 1973). These objectives were not fully met. The methods used to identify "key factors" in life table analysis are now considered deficient and this approach has essentially been abandoned. The models developed during this era generally are not being used to forecast population levels as part of pest management programs, partially because the variables needed to make forecasts are not usually available (e.g. natural enemy densities) or because the models did not perform well.

In the late 1960's and early 1970's concepts referred to as "information theory", "cybernetics", and "systems analysis" began to emerge in several of scientific disciplines. The essential element of these concepts was that any complex system could be understood and predicted by breaking it down into its component pieces (Watt 1966). Scientists adopted this approach by expanding the simple life table based process models into highly complex "systems models". Under the systems approach a pest population model was composed of submodels that represented the numerous specific interactions with natural enemies and host trees (Waters and Ewing 1976). Many of these models encompassed 50 or more population processes (e.g. several specific mortality agents). In many cases, actual life-table data were not available to quantify specific interactions so "educated guesses" were used to estimate parameters for specific processes. Workshops commonly were held to seek input from panels of scientists on the design and parameterization of the various submodels. Modelers adopted a rationale derived from the field of systems analysis that even though specific relationships may not be correctly parameterized, sensitivity analysis would indicate which processes were important and needed further study. Thus, one of the goals of these modeling efforts was to prioritize various research topics (Waters and Ewing 1976). The other two goals of this research were 1) to develop an understanding of the emergent properties of the entire population system, and 2) to forecast future population trajectories (Waters and Ewing 1976, Waters and Stark 1980).

Considerable effort and millions of dollars were spent developing these models for a variety of insect pests, both in forestry and in agroecosystems. Funding for development of systems models for forest pests came largely from the USDA Forest Service. A list of some of the major efforts is given in Table 1. The Forest Service is continuing the development of systems models. Through funding from the Methods Application Group of Forest Pest Management, State and Private Forestry, the Forest Service is proceeding with development of a generalized western bark beetle model, a root disease model and a dwarf mistletoe model. The Northeastern Forest Experiment Station is nearing completion of the gypsy moth life system model.

Table 1. — A list of some of the major systems models of forest pests in the U.S.

Model Name	Pest Simulated	Reference
TAMBEETLE	Southern Pine Beetle, *Dendroctonus frontalis*	Coulson et al. 1989a
SPBMODEL	Southern Pine Beetle, *Dendroctonus frontalis*	Stephen and Lih 1985
Douglas-fir Tussock Moth Outbreak Model	Douglas-fir Tussock Moth, *Orgyia pseudotsugata*	Overton and Colbert 1978
Western Budworm Model	Western Spruce Budworm, *Choristoneura occidentalis*	Sheehan et al. 1989
Western Root Disease Model	Armillaria spp. and *Phellinus weirii*	Stage et al. 1990
Gypsy Moth Life Systems Model	Gypsy Moth, *Lymantria dispar*	Colbert and Racin 1990
Mountain Pine Beetle Model	Mountain Pine Beetle, *Dendroctonus ponderosae*	Crookston 1979

A common feature to all of these systems models is that they are composed of a pest population model and a stand growth model. The pest population models are composed of numerous submodels that simulate interactions with various biotic (e.g. natural enemies) and abiotic (e.g. weather) agents and thereby simulate changes in pest population levels and provide damage levels as inputs to forest growth simulators. Numerous forest growth models are available, representing an evolution from simple yield tables to detailed, physiologically-based individual-tree growth models (Sharpe 1990). To date, most of the systems models of forest pest dynamics have used some variant of the "prognosis model" (Stage 1973), which increments tree diameters using equations derived from local tree growth data using linear regression. In many cases, a separate version of the model is available in which the pest population model is disabled and the user specifies damage scenarios as inputs to the stand growth model.

Given the enormous effort that has gone into development of these systems models over the last 20 years, it is valuable to look back at the effort and evaluate whether it has been a useful endeavor. There is no question that these models have provided important tools for quantifying the impact of pests on timber resources. Before the advent of these models, impacts were typically quantified by simply estimating volume losses during an outbreak and multiplying these losses by a stumpage value to estimate monetary loss. The fact that trees grow between the time that damages occur and when they are harvested affects these estimates in two ways. First, trees may compensate for growth loss and mortality by increased growth following outbreaks. Second, mortality may result in loss of future growth. Stand growth models can mimic these effects in order to derive more realistic estimates of impacts. The use of damage scenarios with stand growth models to quantify pest impacts has been successfully demonstrated for a variety of pests (Leuschner et al. 1978, Cole and McGregor 1983, Liebhold et al. 1986). Over the last 10 years, these models have gained some acceptance as tools for guiding forest management.

The other part of these systems models, pest population models, appear to have been less successful in meeting their goals. Due to the complexity of the ecological interactions that affect pest population levels, these models often have become unwieldy (Lee 1975, Berryman 1990). Numerous parameters exist in these models for which there is little or no supporting experimental data. The complexity of the models, along with the lack of realism, appears to have precluded their contribution of any major insight into population processes. These models have been further criticized as not yielding accurate predictions of population trajectories (Berryman et al. 1990; Fleming 1990). Even if these models faithfully predicted density changes, the data required to initialize them typically are not available (e.g. densities of natural enemies). Thus, there is little evidence that the systems models have significantly contributed to solving either applied or theoretical problems.

THE PRESENT

After 25 years of attempts to model forest-pest interactions, the results have been only partially successful. While attempts to use models to evaluate the impacts of pest damages on timber values have provided analytical insight as well as useful levels of prediction, attempts to use complex models of pest populations have been less valuable. Some individuals may conclude that no model will ever be useful for prediction of pest levels. Logan (1991) recently proposed that the complex nonlinear dynamics of forest/pest interactions may cause prediction of pest population trends to be impossible. I do not believe that this is the case. Though we rarely consider it, predictive models, albeit simple ones, are used routinely in forest pest management. Models are used to answer the question of *where* impacts will occur; statistically derived "risk-rating" models that predict the susceptibility of forest stands from vegetation and physiographic measurements have been extremely useful in forest management (Hedden 1981, Hicks et al. 1987). Models are also used to answer the question of *when* impacts will occur: for a variety of forest pests, the decision to treat a stand is based upon censuses of pest population levels. Evaluation of whether pest densities exceed predetermined threshold levels is a common approach to decision making in pest management (Speight and Wainhouse 1989, Ravlin 1991). These threshold densities are derived from empirical data that relate densities to damage levels and they represent a class of simple predictive models that are useful in pest management decision making.

Several models that predict the phenological development of insect pests and their host trees have been developed over the last 25 years (Valentine 1983, Regniere 1987, Sheehan 1992). Because insects are poikilothermic, their development is almost entirely dependent on ambient temperatures. Consequently, it is possible to develop models that predict insect development with a useful level of prediction. Some of this work was initially begun as components of larger systems models but many of these phenology models have been split-off as stand-alone units. These types of models have been adapted for use in the timing of a variety of pest management activities such as sampling, aerial spraying, and mass-trapping.

One area that recently has received considerable attention is the development of expert systems or decision support systems (DSS) to aid in the pest management decision process. The advent of this activity is in many ways analogous to that of the advent of systems modeling 25 years ago. Just as systems analysis was a new and promising technology, artificial intelligence (AI) is a new technology that has been successfully applied to a variety of scientific and technical problems. With the advent of this technology, there has been considerable interest in applying the AI approach to forest pest management (Coulson and Saunders 1987, Ravlin 1991). Systems models were initially designed as tools to overcome the enormous complexities and incomplete knowledge of forest pest dynamics in order to provide useful predictions of future pest impacts. DSS share a similar goal in that they are designed to overcome the biological complexity and incomplete knowledge that exists when pest management decisions are made. DSS have been developed or are under progress for the southern pine beetle (Coulson et al. 1989b), the Jack Pine Budworm (Loh et al. 1991), and the gypsy moth (Twery et al. 1990). As the application of AI to forest pest management continues to evolve, it appears that the most useful systems concentrate on providing data to humans in a useful fashion (ie. DSS), rather than using those data to make decisions (ie. expert systems) (Ravlin 1991). In forest pest management, the decision-making process is often complex and difficult to decompose into a set of rules. Most of the problems associated with forest pest management are not due to a lack of "experts" but they are related to a lack of useful data. Predictive models will probably be incorporated as part of DSS systems in the future. Integration of models that predict where, when, and how much damage will occur should yield decisions that are more sound than those that treat each question separately, as has been the approach in the past.

THE FUTURE

Over the last 10 years there has been a renaissance of relatively simple mathematical models of population processes (e.g. Royama 1977, Anderson and May 1980) and new concepts from non-linear dynamics have been useful for understanding certain phenomenon (Allen 1988, Loehle 1989, Berryman et al. 1984, Logan 1991). Though systems models largely failed to elucidate complex population processes, there have been real successes in using simple models to accomplish this goal (Turchin 1990). Unfortunately, the second goal (prediction of future population trajectories), which is ultimately one of the most important goals of all population dynamics research, has received little attention.

The most useful approach to developing models for predicting future pest impacts is to start with the very simple models and build from there. Density thresholds are widely used for triggering action in forest pest management and their use represents the application of very simple population models. Sometimes these thresholds are derived from empirical data that relates pest densities with subsequent damage but often their origins are not clear. The statistical relationships between pest densities and damage need to be more clearly delineated. There is also a need to more clearly define the probabilistic nature of the relationship between measures of density and subsequent damage and incorporate sampling error into these probabilistic models.

One feature that is common to nearly all of the models that were previously developed for forecasting outbreaks is that they use data and make predictions for specific locations; they largely ignore regional spatial information. One of the reasons for limiting models to purely temporal dimensions has been that inclusion of spatial information adds considerable complexity. The other reason for not including spatially stratified data is that large, spatially stratified data have rarely been available. The latter situation is rapidly changing; in many systems, especially in forestry, a regional approach is being taken toward pest management (Liebhold et al. 1993, Coulson et al.1993). One of the key developments that has facilitated the regional pest management approach is the advent of geographical information systems (GIS). A GIS is a system of computer programs that builds a standardized database in which each data element identifies a spatial aspect comprised of points, lines, or areas. Similar data elements, such as defoliation patches, can be overlaid to form a data theme, coverage, or layer. Many of these themes, such as defoliation, forest type, and elevation, can be combined to form a full GIS. The system serves as a mechanism for analyzing interactions among data themes across a large, heterogeneous landscape. There is currently a rapid proliferation of GIS designed to track pest populations across large regions (Johnson and Wrobec 1988, Gage et al. 1989, Reardon et al. 1977, Coulson et al. 1989c). These systems can manage a variety of complex spatially referenced databases and extract useful summary information in a manner that heretofore has not been possible.

The advent of GIS as a tool in resource management has highlighted a clear gap in the family of models available for forecasting insect population dynamics (Liebhold 1993). Models are needed that use spatially stratified pest population and landscape data, such as is increasingly available in a GIS, to forecast future pest population levels and damages. Intuitively, most entomologists realize that population conditions in nearby areas represent useful information relevant to predicting future conditions in a specific area. However, even today, there still is no rational and statistically sound modeling procedure that incorporates this information. Perhaps the main challenge to pest modelers is the construction of these types of models.

Despite various failed attempts there have been major successes over the last 30 years in development of models that provide answers to the questions of where, when, and how much pest impacts will occur. Given recent developments in computer hardware and software for handling spatial data, it is likely that useful new models will continue to be developed. Perhaps the most important lesson we can learn from the past is that "big is not always better". Considerable research and development resources were channeled into the development of large systems models and similar patterns are emerging today with decision support systems. R&D planning has often focused on very large efforts to develop models that would be useful for both the scientific understanding of systems as well as for prediction of dynamics and impacts. Instead of focusing on "the" model that can be used for all possible purposes, we need to recognize that different types of models are appropriate for different types of problems.

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Research Approaches to Understanding the Roles of Insect Defoliators in Forest Ecosystems

Karen M. Clancy[1]

Abstract — Forest insect defoliators have traditionally been viewed as pests because they damage their host plants, causing reduced growth and reproduction or even mortality. However, many forest insect defoliators are endemic species; they have coevolved with their hosts over thousands of years, and they are important components of the forest ecosystem. We need to develop a better understanding of the roles that insect herbivores play as recyclers of nutrients, agents of disturbance, members of food chains, and regulators of the productivity, diversity and density of plants. I review some of the empirical approaches that have been used by other scientists to investigate the roles of insect defoliators as recyclers of nutrients and regulators of primary production. These include: 1) Simulating the effects of herbivory on forest biomass production; 2) Estimating bioelement transfers by insect herbivores; and 3) Testing the effects of herbivore density on primary production, nutrient turnover, and litter decomposition in forest ecosystems. I also present some ideas I have on using greenhouse experiments to investigate these roles for the western spruce budworm (*Choristoneura occidentalis*)/Douglas-fir (*Psuedotsuga menziesii*) model system I am working with. The strengths and weaknesses of the various research approaches are compared.

INTRODUCTION

The new Forest Service philosophy of ecosystem management requires that we use an ecological approach to managing our National Forests and Grasslands, so that they represent diverse, healthy, productive, and sustainable ecosystems. Accordingly, there is a growing recognition of the need to understand the roles that insects and diseases play in these ecological systems (e.g., see Seastedt and Crossley 1984, Schowalter et al. 1986, Schowalter 1988, Wickman 1992, Haack and Byler 1993, Schowalter 1993, USDA Forest Service 1993).

Many forest insects and diseases have traditionally been viewed as pests because they damage their host plants, causing reduced growth and reproduction or even mortality. It is generally accepted that current and recent destructive outbreaks of some native forest insects and diseases are largely due to past forest management activities that created forest conditions favoring the survival or growth of these "pests" (USDA Forest Service 1993). Thus, recent epidemics of native pests should be viewed as symptoms rather than causes of "unhealthy" forests (Wickman 1992).

Although current epidemics of native forest insect defoliators, such as western spruce budworm (*Choristoneura occidentalis*), are symptoms of previous management practices, Wickman (1992) noted that presettlement natural forest ecosystems also suffered major pest outbreaks, implying that long-term stable states for forest communities may be unnatural. The point here is that native forest insect defoliators have coevolved with their host trees over thousands of years. They are undoubtedly important components of the forest ecosystem, functioning as recyclers of nutrients, agents of disturbance, members of food chains, and regulators of the productivity, diversity, and density of plants. Thus, we need to develop a better understanding of the roles that insect herbivores play in forest ecosystems in order to use an ecological approach to forest management.

[1] *Karen M. Clancy is a Research Entomologist and Acting Project Leader, Rocky Mountain Forest and Range Experiment Station, USDA Forest Service Research, 2500 S. Pine Knoll Dr., Flagstaff, AZ 86001*

Generation of this new knowledge through research presents interesting opportunities and significant challenges. I will review some of the empirical approaches that have been used by other scientists to investigate the roles of insect defoliators as recyclers of nutrients and regulators of primary production. I will also present some ideas I have on how to address these questions for the western spruce budworm/Douglas-fir (*Pseudotsuga menziesii*) model system I am working with.

Haack and Byler (1993) review information on the additional roles of defoliators as agents of disturbance (and drivers of forest succession), regulators of the diversity and density of plants, and members of food chains. I will not discuss research approaches to quantifying and understanding these roles. Most of what we know to date is based on observation of historical patterns (e.g., vegetation changes following "natural experiments"), and associations between densities and distributions of insect defoliators and insectivorous birds (e.g., see papers in Dickson et al. [1979] and Morrison et al. [1990]). Although a lot of research has documented a prominent role for birds as predators of forest insect herbivores (Holmes 1990), the importance of herbivores as a food source that regulates the population dynamics of birds (or insectivorous mammals, reptiles, etc.) is largely unknown.

RESEARCH APPROACHES

The underlying concept of the role that insect herbivores play as recyclers of nutrients and regulators of primary production is illustrated in Figure 1, redrawn from Berryman (1986). This shows the growth cycle of a forest stand and the role of insects in thinning the stand and recycling nutrients. When plants first get established and during their maximum growth phase, water, nutrients, and light are not limiting. However, when maximum biomass is reached, these resources limit plant growth, and the rate of increase of plant biomass declines. During the biomass reduction phase, insect herbivores thin weakened trees from the stand and recycle nutrients that were tied up in foliar and woody biomass. This enables a growth recovery phase, where nutrients, water and light are no longer limiting, which leads to a second maximum biomass phase, which will be followed by another biomass reduction, and so on.

Simulating Effects of Herbivory on Forest Biomass Production

In their seminal 1975 paper in Science, Mattson and Addy simulated the effects of herbivory on forest biomass production. They used empirically-based simulations to quantify and compare annual biomass production for aspen (*Populus tremuloides*) and spruce and fir (*Picea* spp. and *Abies balsamea*) forests with and without defoliation from one of their major insect herbivores (forest tent caterpillars [*Malacosoma disstria*] for aspen, eastern spruce budworm [*C. fumiferana*] for

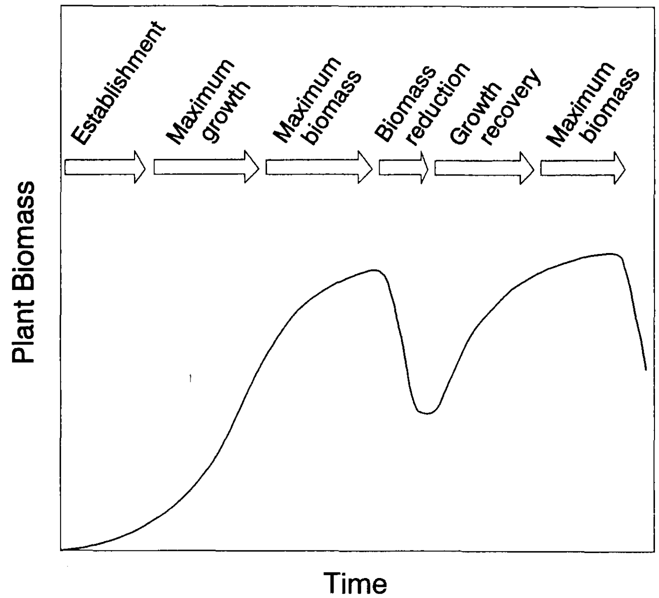

Figure 1. — The growth cycle (plant biomass) of a forest stand over time, and the role of insect defoliators in thinning the stand and recycling foliar nutrients (i.e., biomass reduction phase). See text for details. Redrawn from Berryman (1986).

spruce-fir). Wood growth was calculated using equations for stand diameter growth, plus height growth curves. Likewise, data from the scientific literature were used to estimate foliage and caterpillar production, plus the effects of the insect defoliators on wood production. This calculated information on annual biomass production was manipulated to map the insect-plant interactions in a periodic coordinate system, as shown in Figure 2 for the aspen-forest tent caterpillar system. The "reference zero" (R0) circle represents null interactions where systems with and without insects have equivalent biomass production (insects plus vegetation). The area outside the circle indicates cases where the system with insects has more total

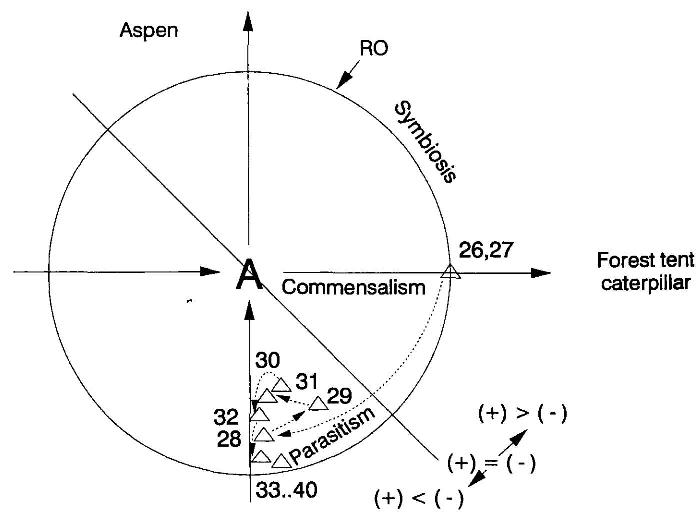

Figure 2. — Periodic coordinate system map of interactions between aspen forests and the forest tent caterpillar, redrawn from Mattson and Addy (1975). See text for details of how to interpret the insect-plant interactions.

production (i.e., there is symbiosis), whereas the area inside the circle shows less total production with insects (i.e., the insects parasitize the trees). The numbers near the triangles show the age of the aspen stand.

The aspen-forest tent caterpillar interaction was commensalistic at ages 26 and 27, when caterpillars were present in the infested stand at low densities (Mattson and Addy 1975). However, it moved into parasitic coaction space at age 28, as the herbivore population increased. At ages 29 to 31, the interactions moved inward to a maximum "parasitism" depth; this was associated with outbreak levels of the forest tent caterpillar in the infested stand: The interaction intensity declined after the outbreak subsided (moved back towards R0), but it remained in parasitic coaction space for ages 32 to 40. The mapping showed that the forest tent caterpillar affected forest production most severely in the fifth and sixth years, but after this the effect gradually diminished to become nearly zero. Mattson and Addy (1975) noted that their forest tent caterpillar-aspen example is typical for such outbreaks, which usually last for 2 to 3 years and then subside. Few if any trees die from such defoliation, except for suppressed individuals.

When simulating the interactions between spruce budworms and their host forests, Mattson and Addy (1975) included the understory response to overstory defoliation (Fig. 3). All vegetative biomass production data was based on stemwood increments. Figure 3 shows that whereas the budworm outbreak destroyed most of the overstory (note overstory after budworm curve), large numbers of understory seedlings and saplings survived after defoliation and grew (see understory released curve). By the 15th year following the outbreak, understory wood production in the defoliated (released) forest exceeded that in the undefoliated (no budworm) forest. Also note that the overstory trees in the undisturbed (no budworm) forest had become avigorous and inefficient producers due to old age and disease.

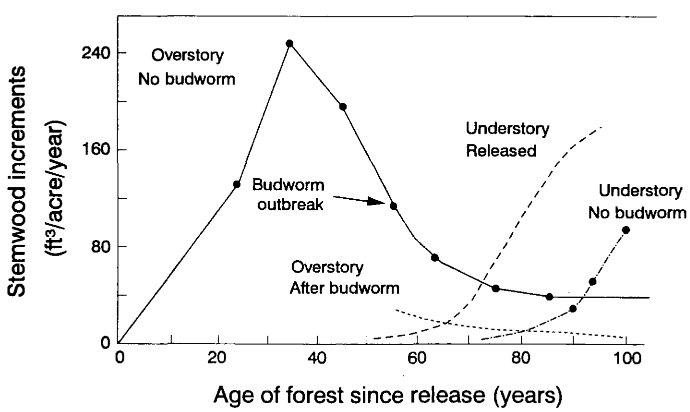

Figure 3. — Stemwood production by overstory and understory balsam fir with and without spruce budworm outbreaks, redrawn from Mattson and Addy (1975). See text for details.

The spruce budworm and spruce-fir coaction mapping demonstrates this positive releasing effect that defoliation had on understory biomass production (Fig. 4). At stand age 50-55 (the initial years of budworm population buildup), the budworm-balsam fir forest interaction was commensalistic. At age 55-60, during the peak of the outbreak, the interaction was strongly parasitic, and remained so for another 10 years after the outbreak subsided. But, by age 70-75 (the 15th year after the outbreak), with understory release, the interaction moved from parasitism to symbiosis. In other words, wood production in the defoliated forest exceeded that in the undefoliated forest in the long run. This implies insect-plant relations may be mutualistic in the long term, despite temporary parasitic coactions (Mattson and Addy 1975).

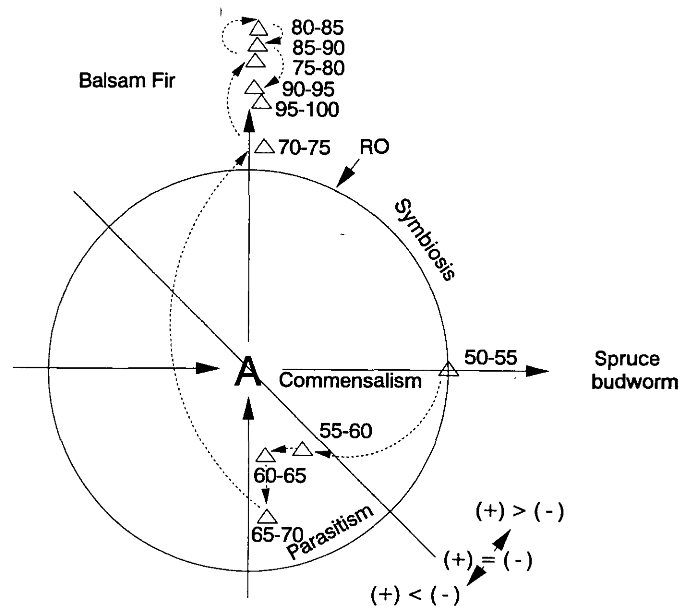

Figure 4. — Periodic coordinate system map of interactions between balsam fir forests and the spruce budworm, redrawn from Mattson and Addy (1975). See text for details of how to interpret the insect-plant interactions.

Schowalter (1993) showed data from studies by Wickman (1980) and Alfaro and MacDonald (1988) that provide direct empirical support for such compensatory growth following defoliation by forest insect herbivores. Figure 3 from Schowalter (1993) illustrates trends in the growth index of conifer trees subsequent to defoliation; the initial reduction in growth caused by herbivory was followed by greater long-term incremental growth for defoliated trees relative to non-defoliated trees.

Estimation of Bioelement Transfers by Insect Herbivores

Another way to examine the ecological roles of insect herbivores in recycling nutrients is to estimate the transfer of bioelements from the canopy to the soil caused by insect feeding. Larsson and Tenow (1980) used this approach to describe the process of consumption by needle-eating insects in a mature (ca. 120 years old) stand of Scots pine (*Pinus sylvestris*) in central Sweden.

Larsson and Tenow (1980) made observations throughout the season of the available needle biomass, and different age-classes of needles, plus the abundance (i.e., number of larvae present) of different insect groups, the grazing damage they caused, and their production of feces and green litter (needle litter cut off by the larvae). Needle biomass and insect abundance and grazing were measured from samples of canopy foliage taken from a mobile skylift (the plot had a low density of trees and a level ground surface). Litter-traps on the ground were used to sample feces and green litter production. The feces data were used in combination with information on specific assimilation efficiencies for each group of insect herbivores to make indirect estimates of needle biomass consumption. In other words, they conducted feeding studies in the laboratory to measure how much frass the larvae produced when eating a known amount of needle tissue. This allowed them to predict that if they collected *x* amount of frass in their litter-trap, this means that *y* biomass of needles was consumed. They also measured the concentrations of bioelements (including N, P, and K) in the needles, the needle litter, the green litter, and insect feces. This yielded calculations of bioelement fluxes from insect feces and green litter.

Figure 5 (redrawn from Larsson and Tenow 1980) is a schematic representation of needle biomass and transfers of dry matter (on an annual basis) estimated from this study for 1974; the bold numbers show the biomass measurements in kilograms dry weight per hectare. For example, there were 626 kg of current-year needles, 771 kg of 1-year-old needles, and 804 kg of 2-year-old or older needles present in the canopy; 61 plus 674 kg of the 1-year-old or older needles were dropped and became 735 kg of needle litter. At the end of the season insect grazing had removed 15.5 kg of needle biomass, which was 0.7 percent of total needle biomass or 2.5 percent of current-year needle production. Of the 15.5 kg removed by grazing, 1.5 kg was green litter, or needles that were cut off by larvae but not consumed; 14 kg was consumed, with 11 kg (79 percent) being returned to the litter (soil) as insect feces.

The bioelement transfers of N, P, and K are shown in italics, and are in grams dry weight per hectare (Fig. 5). When green litter and feces inputs are combined, the input to the soil was 92 g of N, 10 g of P, and 48 g of K. Larsson and Tenow (1980) concluded that in 1974, feces plus green litter transferred about 1 percent of the carbon, calcium, and sodium, 2 percent of the nitrogen, phosphorus, magnesium and sulfur, and 4 percent of the potassium carried annually to the forest floor by total pine litter. Thus, a part of the bioelement content of this ecosystem is circulated through the insect herbivore consumer chain, although Larsson and Tenow (1980) noted that the effect of these bioelement transfers on soil processes are unknown.

Testing the Effects of Herbivore Density on Primary Production, Nutrient Turnover, and Litter Decomposition in Forest Ecosystems

Schowalter (1993) emphasized the need to use an ecosystem framework for experiments that are designed to evaluate the effects of insect defoliators on integrated forest ecosystem processes. Figure 6, which is redrawn from Seastedt and Crossley (1984), illustrates a simplified model of elemental cycling in a terrestrial ecosystem, where the roles of arthropod consumers (e.g., insect defoliators) are emphasized. Indirect

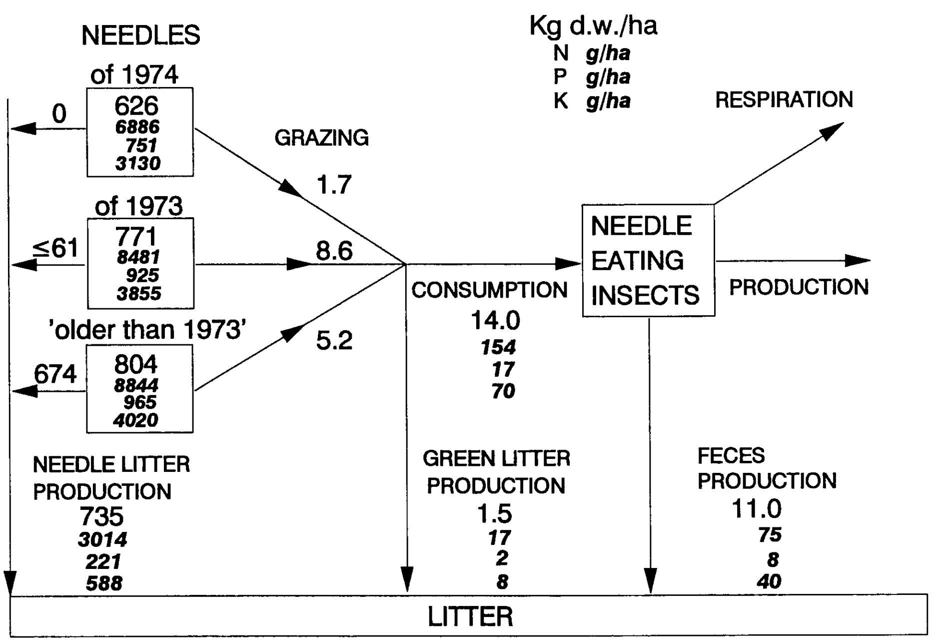

Figure 5. — Schematic representation of needle biomass in October and annual transfers of dry matter and the bioelements N, P, and K, due to grazing of needle-eating insects and normal needle litter fall in a Scots pine forest in Sweden in 1974. Redrawn from Larsson and Tenow (1980). See text for details.

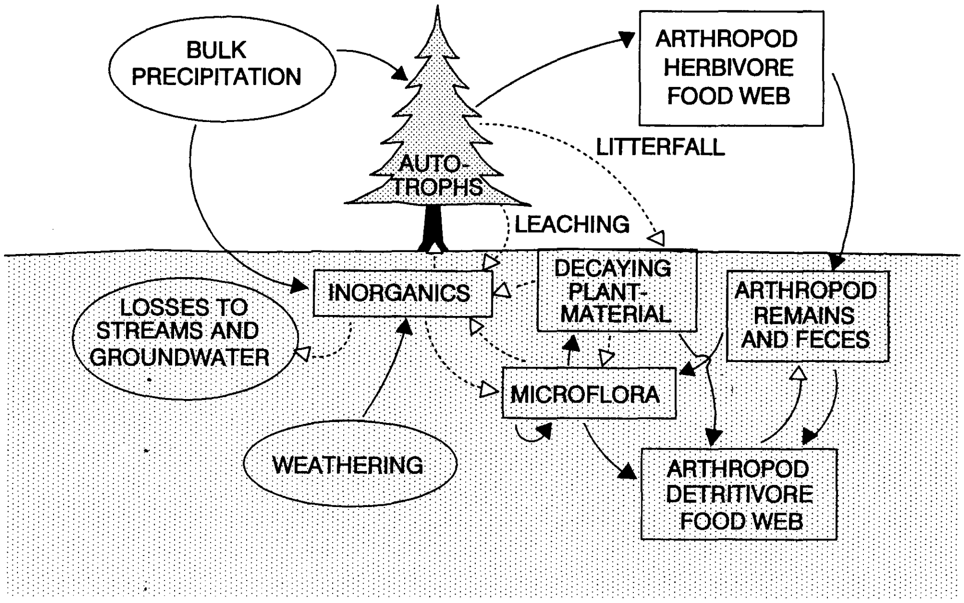

Figure 6. — A simplified model of elemental cycling in a terrestrial ecosystem, where the roles of arthropod consumers (e.g., insect defoliators) are emphasized. Indirect regulation of elemental flows by arthropods are indicated by dashed lines and open arrows. Redrawn from Seastedt and Crossley (1984). See text for details.

regulation of elemental flows by arthropods are indicated by dashed lines and open arrows. Insect herbivores remove foliage that contains bioelements from their host trees, but they also return much of this material to the soil through their feces, molted exoskeletons, and dead bodies. Nitrogen and other minerals are more concentrated in this insect-derived material than in the senescent leaves and needles that trees normally drop. This provides increased nutrients to arthropod detritivores and microflora, which could stimulate the activity of decomposer organisms, and enhance rates of decomposition of plant material. The green litter (partially consumed or clipped leaves and needles) that results from herbivore feeding is also a richer source of bioelements than normal senescent litterfall. Furthermore, accelerated leaching (or throughfall) of nutrients from grazed foliage may make important contributions to the inorganic mineral pool in the soil, where they could be reassimilated by roots.

Crossley et al. (1988) summarized conclusions from studies at Coweeta that were centered on determining the impact of canopy arthropods on forest nutrient cycling. They found that a partial defoliation in the Coweeta basin by the fall cankerworm (*Alsophila pometaria*) resulted in marked changes in nutrient cycling within the affected watersheds. Nitrate concentrations in streams increased during the defoliation, there was a net increase in net primary production, and increases in litterfall, nutrient inputs from frass and canopy throughfall, and soil nitrogen pools and associated microflora.

Schowalter et al. (1991) have also tested the effects of herbivore density on primary production, nutrient turnover, and litter decomposition of young (8 years old) Douglas-fir in western Oregon. The defoliator they used was the silver-spotted tiger moth (*Lophocampa argentata*), which feeds only on previous years' foliage. Target densities of the caterpillars were maintained at the 1 ha study site by manually adding or removing larvae from individual trees, based on biweekly counts during the feeding period. Twenty trees were used per defoliation treatment (low and high defoliator abundance), with equivalent numbers used for controls. Each tree received the same treatment for 3 years. Effects on primary production were measured by estimating foliage and total plant mass from regressions based on trunk diameters at the litter surface. Small (1 g) samples of current and 1-year-old needles were collected from each tree in June and analyzed for N, K, and Ca. Proportional sampler pans were used to collect throughfall/stemflow precipitation and litterfall from 10% of the canopy of each tree. The throughfall was shunted via plastic tubing to big jugs for storage. Mesh screens in the collectors retained particulate matter. The throughfall in the jugs and litterfall on the screens were collected and measured twice a week, and composite samples were analyzed for N, K, and Ca content. Finally, litter decomposition rate was measured as mass loss of 10 litter samples under each tree, using litterbags filled with Douglas-fir needle litter. The N, K, and Ca content of the litter samples was also determined at the start of the experiment, and after 3-27 months in the field.

Based on their experimental results, Schowalter et al. (1991) concluded that defoliation by the silver-spotted tiger moth did not affect Douglas-fir growth or foliar nutrient content, suggesting compensatory growth and replacement of lost nutrients. The decomposition rate of Douglas-fir needle litter was also unaffected, implying that herbivory does not "prime" decomposition via throughfall or litter enhancement. However, the mass of litterfall and the volume and nutrient content of the throughfall were positively related to defoliator abundance during the early growing season. Turnover of N, K, and Ca were also enhanced by the defoliation treatments.

Greenhouse Experiments with Western Spruce Budworm and Douglas-fir

Since 1985, I have been working on a project designed to determine physiological mechanisms of Douglas-fir resistance to western spruce budworm defoliation. An important result of this work has been the identification of 24 pairs of mature Douglas-fir trees that are phenotypically "resistant" versus "susceptible" to western spruce budworm damage; the resistance is associated with foliar nutritional chemistry, vigor of growth, and phenology of budburst (Clancy 1991a, Clancy et al. 1993).

We are now in the process of vegetatively propagating cuttings from these 48 genotypes through grafting. This will provide a pool of ontogenetically mature Douglas-fir "trees" in pots that can be readily manipulated in greenhouse experiments to evaluate the role of budworm defoliation in changing plant physiology and chemistry, and in recycling foliar nutrients. I also maintain a laboratory culture of non-diapausing western spruce budworm (Clancy 1991b), which gives me a continuous supply of budworm larvae to achieve prescribed levels of defoliation on these potted trees. Moreover, these trees can be manipulated through changing their exposure to day length and temperatures so that 2 or 3 annual growth and defoliation cycles can be compressed into a single year. This will enable much more rapid determination of the long-term cumulative effects that defoliation has on Douglas-fir physiology and productivity.

Another advantage of using a greenhouse experimental approach will be the ability to manipulate nutrient inputs from frass, green litter, and throughfall leaching. Screen barriers could be placed around the base of the plant to intercept the frass and litterfall, yet allow throughfall. Or, by not using any overhead watering system, I could eliminate throughfall. I could also add frass and green litter from defoliated plants to undefoliated plants. Furthermore, since I have a diversity of Douglas-fir genotypes with different physiological characteristics to use, I can examine variation in responses to defoliation. It seems possible that some genotypes are better adapted to tolerate and compensate for defoliation than others, and this may be an important component of the resistance I have observed in the field (Clancy et al. 1993).

Finally, underground components of the ecosystem could presumably be manipulated as well. For example, mycorrhizal associations could be enhanced via inoculations or reduced by using fungicides. Similarly, soil dwelling detritivore insects could be added at different densities. Diverse soil types and nutrient regimes could be created, or soil pH could be varied.

SUMMARY AND CONCLUSIONS

Several research approaches have been used successfully to increase our understanding of the roles that insect defoliators play in forest ecosystems with regard to regulating primary productivity and recycling nutrients. Mattson and Addy (1975) simulated the effects of defoliators on biomass production of aspen and spruce-fir forests, using empirical data from the literature. This was a powerful approach because it allowed looking at long-term effects of herbivory on primary production, but it was a simulation rather than direct observation or an actual experiment. Subsequent studies by Wickman (1980) and Alfaro and MacDonald (1988) were based on direct observation of the effects of different levels of defoliation on growth indices of trees; the results supported Mattson and Addy's (1975) hypothesis that insect-plant relations are mutualistic in the long term. A limitation of all these studies was that the actual mechanism for the compensatory growth following defoliation was not investigated. Recycling of nutrients through defoliation was suggested, but not proven.

Larsson and Tenow (1980) estimated annual nutrient transfers from defoliators in a Scots pine stand, based on empirical observations of foliar biomass, herbivore abundance and grazing damage, and the amount of frass and green litter produced by defoliators. This study demonstrated that insect defoliators do circulate a part of the bioelement content of the Scots pine ecosystem, but it did not identify the long-term impacts of these nutrient transfers on soil processes or primary productivity.

Very few experimental studies have actually tested the effects of manipulated defoliator densities on primary production, nutrient turnover, and litter decomposition. Schowalter et al. (1991) did this with young Douglas-fir trees, and they found that defoliation enhanced turnover of N, K, and Ca by means of increased litterfall and throughfall. However, because their study only lasted for 3 years, they could not address longer term effects of defoliation on ecosystem processes.

I have proposed that greenhouse experiments could be used to investigate the role of western spruce budworm defoliation in recycling nutrients and regulating primary productivity of Douglas-fir. The strength of this approach is that many of the system inputs could be readily manipulated; the weakness is that a grafted Douglas-fir tree in a pot may not respond the same way a mature tree in the forest would. Also, it would not be possible to recreate all the ecosystem components and larger scale effects in a greenhouse environment. Nonetheless, I think the strongly experimental approach that is possible using potted plants and budworm larvae from a laboratory culture could yield valuable information that would be very helpful in terms of identifying the key processes to monitor in large scale ecosystem studies in the field.

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Modern Forest Management: It's About Opening Up, Not Locking Up

W. Bruce Shepard[1]

Abstract — Ecosystem management, if it is to succeed, must involve more than the application of improved scientific understandings; it must embrace the political responsibilities of the land manager. The idea of forest management as applied science was a highly successful recipe for political success during earlier decades. This was an inadequate response to the challenges that emerged in the 1970's. Today, the land manager must apply an approach to forest management that encompasses scientific, economic, sociological, and political understandings. This will be difficult: several lines of reasoning lead to the conclusion that the issues that land managers are likely to face will be emotionally charged and political "no win" situations. There are also two challenging questions that must be resolved in designing a modern and more effective approach to forest management: how does one blend national, regional, and local considerations? and how does one integrate scientific, economic, sociological, and political analyses in decision making? Elements of answers to these questions are considered.

INTRODUCTION

It is not just the nation's forests and its forest policy that are undergoing critical reassessments. The profession of forestry itself is showing inclinations toward change. There are roundtables on the ethical dimensions — not of forest management *per se* — but of the forestry profession itself (Banzhaf 1993). Issues of the *Journal of Forestry* are devoted to assessing the adequacy of curricula for preparing foresters. And, there is increased attention given to issues of diversity within the profession and within natural resource agencies (Bembry 1990; Kennedy 1991; Pytel 1991).

That the profession is seeking reorientation seems clear. That there is full appreciation of the depth of change required is less clear. More murky still — even among those calling for change — are the details of how a new forestry profession might operate.

Part of the change reflects an exhilarating interjection of emerging biological understandings, including a fuller and scientifically grounded understanding of the uncertainties and limits that attach to scientific knowledge. This appears, at least to a fuzzy thinking social scientist, to be very heady stuff. I have sat in seminar rooms where scientists share with practioneers the latest findings on soil microorganisms and there is electricity in the air. "Paradigm shift" is a much overused term. But, in the questions being asked and being answered, I believe that term, in its Kuhnian sense, may appropriately describe what is happening in the applied science of forestry (Kuhn 1962).

Forestry is much more than applied science. The very idea of a forest as a natural resource cannot exist apart from a society that values the forest. Without values, forests may be forests but they cannot be resources, whether as sources of wood fiber or wonderment. As Shannon (1992) put it:

> *What we call natural resources are those connections made manifest by social values and realized through available, often changing, technologies. Taking this perspective, natural resources represent the primary organization of a society. Zimmerman's "resources are not, they are becoming" remains the classic statement of this continuously co-creative relationship.*

Understanding the "soft" sides of forestry can be particularly difficult within a profession that has largely defined forest management in terms of applied science. That approach, traceable to Pinchot, has a long history of considerable success. However, the blinders it introduced left the profession ill-equipped for the turmoil that began in the early '70's when the need was for better politics, not better science (Shepard 1990,

[1] *Assistant Vice President for Undergraduate Studies and Associate Professor of Political Science, Office of Academic Affairs, Oregon State University.*

1993). The need to deal explicitly with contending social values goes against the grain of long professional tradition. With all the excitement — and controversy — about emerging scientific understandings, one may lose sight of the fact that it is people, not spotted owls, that are posing the most significant professional challenges.

Today, there may be wider understanding of the need to pay attention to the political responsibilities of the natural resource manager. However, as Ellefson (1993) has pointed out, even in modern university curricula, the focus tends to be upon teaching policy analysis but not the origins of policy in incremental processes characterized by widespread bargaining and the use of non-economic criteria for policy selection. I will push beyond the policy-analysis level of dealing with the political.

Some wag has observed that we should never look too closely at the making of sausages or laws. I am going to get into some of the sausage making. I will approach the subject by considering three questions:

1. How do political matters end up on the desks of natural resource managers?
2. What are the roles of planning in politics?
3. What are the roles of politics in planning?

In a final section, my attention will be upon two as yet inadequately answered questions that must be successfully addressed in order to design a modern and more effective approach to forest management:

1. How does one blend national, regional, and local considerations?
2. How does one integrate scientific, economic, sociological, and political analyses in decision making?

HOW DO POLITICAL QUESTIONS END UP ON THE DESKS OF NATURAL RESOURCE MANAGERS?

In my experience, students preparing to work for natural resource management agencies are almost always motivated by a desire to be close to the resource being managed: to be on the land. Yet, former students report that they spend all their time dealing with people and paper. They are often involved deeply — and resentfully — in political issues. How can this be?

In part, the answer is simple: to recognize the false dichotomy between policy and administration. One dominant feature of American political culture is to hide or disguise the political elements of decisions — to expect that decisions about who is going to win and who is going to lose can be answered as technical matters through three shelf-feet of EIS statements, benefit cost analyses, and the like. We evolved institutions — the city manager is an example as is the professional forest manager — to try to maintain the fiction of a separation between making policy and implementing policy. This belief that management as applied science could eliminate politics has long and strong roots in the area of natural resources (Hays 1959). However, the dichotomy is false. Policy is made through its application, and policy implementation is bound to have a political element.

The role of politics in policy implementation is accentuated in the political systems of the United States because of a dependence on interest groups as the vehicle for linking citizens to government. In many developed democracies, political parties perform the linkage function by offering coherent programs and carrying out those programs when in power. In the United States and for a variety of structural reasons, political parties are only vehicles for winning elections; they are not, as Bill Clinton is discovering, mechanisms for governing. Interest groups, almost by default, become the mechanism for translating citizen preferences into policy. Bargaining among interest groups is the major mechanism for the development of public policy and that bargaining extends from early stages of policy formation through to policy implementation.

Two models of the policy process yield predictions about the types of questions that are likely to be encountered by natural resource managers. In a model found to have wide utility, Salisbury and Heinz (1970), building upon the work of Lowi (1964), present a policy typology that relates types of public policies to the nature of the politics — demands and supports — surrounding the matter. Their basic distinction is between allocative and structural policies: allocative policies allocate — they deliver the goods (or services); structural policies set up or designate structures and processes for subsequent allocations. The National Forest Management Act (NFMA) is a structural policy. It sets up rules and procedures — page after page — but never decides the harvest controversies that caused it. Legislation banning (or dictating) clear cutting would be an allocative policy.

Structural and allocative policies are further broken down as shown in Figure 1. Allocative policies may be distributive if they confer benefits to all active participants; "pork barrel" and "log rolling" are terms typically applied to distributive policies. Allocative policies become redistributive if there are both winners and losers and if the losers understand that others have benefited at their expense. Structural polices are self-regulatory if decision making is left largely in the hands of the regulated; professional licensing boards — e.g., for doctors, lawyers, realtors, even worm growers in Illinois — are common examples. Regulatory policies redistribute power and authority, taking power previously held by individuals or groups and vesting that power in different arrangements.

Distinctions in types of policy depend upon differences in the structure of political demand — is it split among many groups or integrated? — and the net political costs to decision makers. If the political benefits of making an allocation are high compared with costs, then legislators will make the allocation and reap the political benefits. If political benefits are low relative to costs — a common occurrence in a pluralist, interest group based politics — then structural policies result. Put more simply: the buck is passed. NFMA is a classic example.

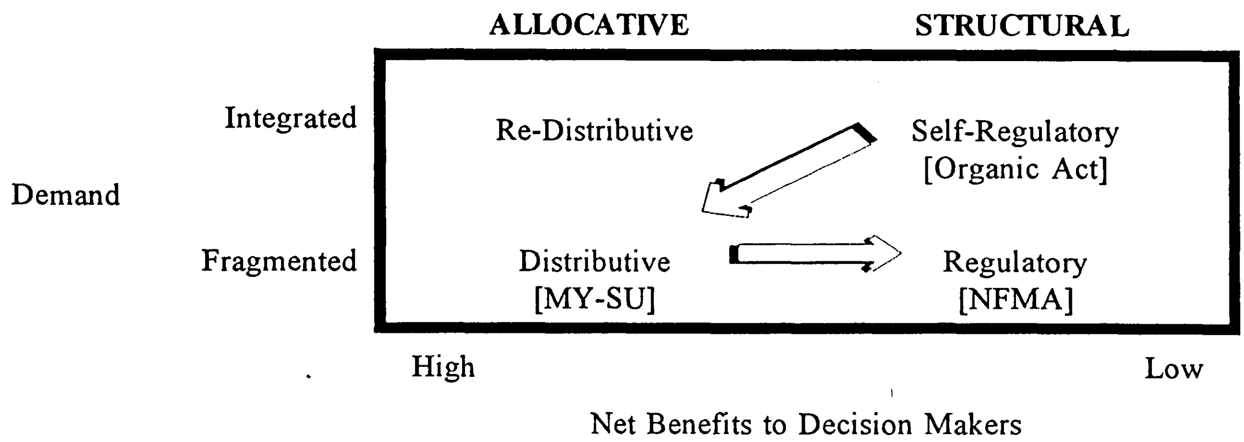

Figure 1. — Types of public policies.

The arrows in Figure 1 identify a pattern in the evolution of public policy that Salisbury and Heinz describe as common and that seems to fit the area of forest policy. Policies start out as self-regulatory: demands are limited and those interested in a resource decide its use. As demands increase, distributive policies result, each interest getting some of what it wants, as suggested by the language of the Multiple Use Sustained Yield Act. Eventually, demands cannot all be met, and policy is forced to become regulatory.

It is structural policies that end up on the desks of natural resource managers. These are politically difficult situations, issues for which any particular solution (any particular allocation) is almost certain to upset more people than will be pleased. If there were some solution that had a positive political payoff, then the legislature would have adopted it and claimed the credit.

Murray Edelman (1967) took a different approach to linking types of policies with types of politics. Edelman studied many examples of regulatory policies, and he was interested in explaining a phenomenon called "regulatory capture" in which the interests being regulated end up heavily influencing those regulating them. He distinguished between material goods — e.g., money, timber, power — and symbolic goods. Symbolic goods are actions designed to provide the unorganized but anxious with psychological reassurance. His studies lead him to conclude that — often in the same piece of legislation — organized groups get material goods in proportion to their bargaining strength while the unorganized get symbolic actions that provide reassurance without conferring material benefits.

Edelman's analysis went on to observe that symbolic reassurance kept the anxious but unorganized from getting organized and going after material goods. This fit an earlier period when studying such matters as the regulation of railroads or airlines. Edelman's analysis can be expanded to recognize that individuals care very much about symbols. Symbolic policies involve identity, morality, and status. Capital punishment, prayer in schools, gays in the military, and abortion are examples of policy issues that have major symbolic components. In contrast to what Edelman found, battles over symbols can be the basis for effective political organization. The civil rights movement and the Equal Rights Amendment are cases where early efforts focused on largely symbolic matters that, nevertheless, became the basis for groups to organize and to later demand material goods. Another symbol would be the clear cut. There is herbicide spraying. And, of course, the spotted owl.

When we add Lowi, Salisbury and Heinz, and Edelman together, we get explanations of how political matters end up on the desks of natural resource managers. We get more. We get the prediction that the types of political questions that natural resource managers are forced to deal with will be emotionally highly charged, political "no win" situations. But, then, who needs political science to come to that conclusion?

What are the Roles of Planning in Politics?

Politics is about who gets what, when, and how. It is about the authoritative allocation of things people value. It is about winners and losers. Think about the National Forest Management Act of 1976. That legislation had its roots in harvest controversies on the Bitteroot and Monongahela National Forests. When groups successfully used the language of the Organic Act of 1897 to halt clear cutting, Congress was forced to act. It responded with a piece of planning legislation. Why? What are the political uses of planning? What role does planning play in politics?

There are many possible political uses of planning. In particular cases, it may be used to open decision making processes to wider involvement or it may be used to centralize decision making authority. In more general terms, planning legislation can be politically attractive for three reasons:

1. Almost by definition, planning legislation is a structural policy. It allows legislators to avoid the political pain of making an allocative decision by passing responsibility for the decision to another body.

2. Planning legislation provides symbolic reassurance to the anxious but unorganized. There is the appearance of having done something even if none of the underlying controversies have been addressed.
3. Planning approaches can improve the legitimacy of the decisions that eventually result.

The last point requires elaboration. Governments worry enormously about their legitimacy. Policies viewed as legitimate will be obeyed. If policies do not have legitimacy, governments can soon exhaust their resources in trying to enforce policies. Controversial policies will be accepted as legitimate by many if the processes that produced the policies are accepted as legitimate. Consequently, governments invest great effort in designing processes that are accepted as legitimate when controversial decisions are anticipated.

To figure out what will make processes legitimate, one must look at a country's political culture. In the United States, processes that provide opportunities for participation have increased legitimacy. Since it is extremely difficult for individuals to trace the influence of their participation through to final policy outcomes, it is the opportunity for participation, and not its effectiveness, that is politically relevant. Second, our political culture places great emphasis on decision making that incorporates non-political, scientific, rational, quantitative, technical analysis. That questions of winners and losers cannot be settled on scientific grounds is irrelevant to the political cover provided by such processes. Planning processes, of course, can appeal to both key values. This is the greatest political attraction of planning, and planning legislation has been the institutional response to many allocation controversies. Politically, the attraction of planning is not that it might lead to making better decisions; rather, it makes decisions look better.

What are the Roles of Politics in Planning?

Whatever the role of politics may be in planning, it will involve interest groups. In the United States, it is interest groups that provide governments with political information. For simplicity, the examples and illustrations that follow sometimes refer to individuals. Please keep in mind that, in the real world, individuals are usually represented politically by the groups that they belong to and support. There are both advantages and disadvantages to such a system for linking citizens to government. It is, though, a political reality that natural resource managers must recognize.

In planning, politics provides information about what people want. More important, it conveys information about how badly people and their groups want things. It is this last type of information — known to political scientists as "salience" — that is the central contribution of politics to planning processes. Planning processes must incorporate information about the salience of preferences as well as analyses of what people prefer. There are both empirical (why it ***is*** that way) and normative (why it ***ought*** to be that way) reasons.

Political decision makers — and this means natural resource managers as well as elected officials — must pay attention to salience if they wish to hang onto their jobs. Who is going to follow what an official does and reward or punish that official at the next election — or the next budget hearing? It will be those for whom the issue is salient. This principle explains, for example, why we do not have stronger hand gun control legislation even though poll after poll shows that 80 percent or more of U.S. citizens would prefer such legislation. There is a minority for whom the issue is very salient while, for the majority, the issue is not salient.

The empirical "that-is-the-way-that-it-is" reason for paying attention to salience may be initially offensive. It rubs against our democratic sensibilities for it means that some preferences count more than other preferences. It is an unavoidable characteristic of our political system: to the extent that democratic institutions require that government officials be responsive, the mechanisms force responsiveness to those who participate. However, there is also an argument that political systems ought to pay attention to salience. This is the normative rationale, and it is the same as is used to justify a marketplace as a mechanism for distributing goods and services. Briefly put, the logic begins with the assumption that goods should go to those who derive the most reward from them. If you prefer fine Bordeaux wines and I am happy with Ripple, it makes no sense to allocate the grape crop equally among us. Instead, the salience of our preferences is measured by our willingness to pay at the liquor store check-out counter. Note the logical leap here: we have gone from a premise that goods should be distributed based upon their utility to people to a conclusion that goods should be distributed based upon prices people will pay. So long as we are comfortable with that leap — namely, with differences in people's ability to pay — then the justification for markets, and for paying attention to salience, is tight. Just as in the private marketplace, so too with some government goods and services, it would be wasteful to give the same attention to my preferences as to yours if the issue has very little value to me while the matter is of great importance to you.

Governments rely upon two major techniques for measuring salience: making participation costly and logrolling. By making participation costly, one obtains the preferences of those for whom the issue is important enough to pay the costs of participation. That, for example, is why elected officials give more attention to mail counts than to opinion polls. An opinion poll is, in a sense, a form of participation in which the people conducting the poll have subsidized the costs of participation. Writing a letter requires effort. A member of Congress who believes the polls on hand gun control legislation instead of mail counts will not be responding to salience and risks losing office.

Polls, testimony at public meetings, and the like, suffer from the "strategic lying" problem. If participants believe someone else will pay the costs of what they want, then they have an incentive to overstate its value. If people believe they may be forced to pay entirely for what they desire, then they may understate its value. The history of the testimony of grazing interests on user fees for grazing on public lands is replete with examples of the strategic lying problem. People cannot, though, lie with their behavior. They either pay the costs of participating by participating. Or, they don't. So, at a public meeting, it is not what the people who show up say that counts; it is who shows up that conveys the relevant political information on salience.

Logrolling is the other technique for obtaining information on salience. Logrolling involves bartering away one's support on something of less importance to obtain another's support on something of greater importance. It often occurs in legislatures. Imagine a senator from North Carolina and a senator from Oregon getting together to discuss two bills: a bill the Oregon senator wants that would promote replanting of clear cuts in the Pacific Northwest and a bill to provide tobacco price supports introduced by the solon from North Carolina. Though the North Carolina senator may be opposed to helping economic rivals in the Pacific Northwest with a timber subsidy and even though the Oregon senator may oppose using her constituents' tax dollars to help tobacco farmers produce a poison, each senator may agree to swap votes and support both bills. This can occur if the "no" vote that each would give up is worth less than the "yes" vote that each would obtain in exchange. Salience is revealed in the trades that people are willing to make. (Note, again, it is not what people say, but their actual behavior — just as in a private marketplace — that reveals information about salience.)

Logrolling need not be restricted to legislatures. You and your family probably engaged in logrolling to figure out this summer's vacation plans. (You certainly did if, like me, you had four teenagers to deal with.) A set of alternative forest plans can be thought of as a group of possible logrolling packages designed to offer various trading opportunities. Successful logrolling packages make distinctions based upon salience; they offer people opportunities to obtain something of greater importance to them by giving up something of less importance.

The extension of logrolling and making participation costly to the goods and services that governments provide may not be obvious. Consider an example. Imagine an extremely simple national forest on which there are only two uses: snowmobiling and cross country skiing. As this is a greatly simplified illustration, I will call this the Dan Qualye Nationale Forest. Further imagine that, since these two categories of users do not get along well, that the management of the DQNF have subdivided the forest into many different areas, each of which will be allocated to one or the other use. Your problem, as a manager, is to decide how each area will be designated: for exclusive cross country skiing or for exclusive snowmobiling.

If we did not rely upon salience and instead conducted an opinion poll, then the outcome would depend upon which of the two groups of users were most numerous. Imagine that a slight majority of the potential users are snowmobilers. Then, if an opinion poll is the only basis for allocating areas, every area will go to snowmobilers. That would not only be inequitable — what about the poor skiers? — but would also be highly inefficient. Even areas that are worthless for snowmobiling but highly prized by cross country skiers would be closed to skiing.

Imagine two other approaches. In the logrolling approach, managers could estimate which areas are most important to snowmobilers and which areas are of greatest importance to cross country skiers and offer a forest plan that takes advantage of such trading opportunities. Or, participation could be made costly. Imagine something as simple as an area-by-area election where the choices are "snowmobile only" or "cross country ski only." Snowmobilers, because we have assumed that they are more numerous, may win the first few elections. Eventually, though, they become at least somewhat sated while the cross country skiers are becoming panicked. Or, areas of little value for snowmobiling but of great value for cross country skiing may come up for a vote. Participation by snowmobilers drops while participation by skiers increases. Cross country skiers start winning elections. By this very simple device of making participation costly — and not by listening to what people say things are worth to them — resources are allocated in a way that is more equitable and more efficient than would have been the case if an opinion poll — everybody's preference counting equally — had been used.

Providing information about salience is a key role for politics in natural resource planning. However, the two common techniques for obtaining this information — logrolling and making participation costly — have imbedded within them a guarantee that, if they alone are relied upon, they will fail. Academics who study public choice identify logrolling as a useful technique whereby legislative bodies can achieve the provision of optimal levels of public goods. Yet, newspapers treat logrolling as an unseemly and disreputable practice. In part, this may be because logrolling requires that representatives vote against the interests of their constituents on matters of less importance. It is also true that terms like "pork barrel" are frequently attached to the results of logrolling. Logrolling works in theory if all affected individuals can participate in the decision. However, if the costs of providing a public good can be transferred to non-participants, then logrolling can lead to over provision. Legislation laden with ridiculously inefficient water projects would be a typical example. Various interests in assorted districts trade support for each other's projects, and the practice fails on efficiency grounds because the costs of the projects are transferred to the general taxpayer who is not involved in the trading.

Logrolling fails when costs can be transferred to nonparticipants. There will always be strong incentives to make the transfer, to make somebody else pay. Note that the second means for measuring salience — making participation costly — assures that there will be nonparticipants. *One technique for measuring salience guarantees conditions that will lead the other to fail.* Economists build a rationale for governmental involvement because "market failures" lead to the under supply of public goods. However, "government failures" can lead to the over supply of publicly provided goods. This realization is important in natural resource management for it suggests partial answers to questions that I wish to explore in conclusion.

CONCLUSION

Challenges for Modern Forest Resource Management

A forestry that approaches resource management and allocation as largely a technical problem amenable to technical solutions — bigger and more sophisticated versions of FORPLAN — is inadequate (Alston and Iverson 1987). It simply follows the trajectory of the past in which forest management was defined in terms of applied science. Forestry today is about values, cultures, communities, and also about politics, about winners and losers. That recognition is, itself, a significant step forward in developing modern forest resource management. Programs like "New Perspectives" are attempting to bring together such significant changes: a shift from emphasis upon the production of wood fiber toward protecting ecosystem vitality; the introduction of emerging scientific understandings; and, a recognition of the social and political responsibilities of the forest manager.

What does it mean to recognize the political elements in resource management? This paper has explored some elementary implications. Paying attention to the political means recognizing that it is the politically most difficult and most emotionally charged issues that are likely to end up being assigned to public natural resource management agencies. It means understanding the important role of planning processes in lending legitimacy to unavoidably controversial decisions. It means that, in managing natural resources, one must pay attention not only to what people want but also to how badly people want things. It does not mean finding the final, politically correct solution; rather, it involves assisting a society in a constant process of reevaluating and redefining what constitutes appropriate uses of the nations's forests. It does not mean locking up, it means opening up.

I see two major procedural questions that must be addressed before there is a system of modern forest management adequate to the tasks ahead. The questions are: how does one blend national, regional, and local considerations?, and how does one integrate scientific, economic, sociological, and political analyses in natural resource decision making? As in any flexible and evolving human system, we will never find "the right answer" to these questions. Rather, we will experiment, fail, and experiment again.

Integrating National, Regional, and Local Concerns

How are natural resource management policies — say, a forest plan — to integrate national, regional, and local constraints and concerns? As Behan (1972) pointed out twenty years ago, one may start by interacting with local interest groups. However, it is dangerous to assume that local politics are a microcosm of regional or national politics. Different groups have different degrees of influence depending upon the level of government involved. Even within a single sector, say in wood products, national firms with headquarters out-of-state are not going to be able to compete as effectively with local firms if decisions are being made at multiple local sites.

A great deal of political bargaining can be involved in simply establishing the level at which decisions will be reached, and groups will seek to force decisions to that level at which they are most influential. Federalism is not a tight structure of clearly defined governmental authorities. Rather, the pattern of intergovernmental arrangements that exist at any particular moment is a dynamic and changing bargain among various interests concerning the appropriate arrangement of responsibilities. Not a layer cake but a marble cake that never quite makes it to the oven.

From the perspective of a ranger or a forest supervisor, a workable model might be one in which statute, "Washington Office," and regional policies establish the boundaries within which "on the ground" policies and decisions are developed. This seems to be the model implicit in the organization of the *Forest Service Manual* and *Handbook*. The approach has some utility. Laws reflecting interests articulated at the national level do establish constraints: say, in the area of protecting cultural heritage, in the procedures for environmental impact statements, and the like. Consider, though, the Endangered Species Act. This is as unambiguous a piece of federal legislation as is ever likely to emerge from Congress; it lacks the usual weasel words and delegations of responsibility and is, consequently, allocative rather than structural. The statute is a statement of interests that have been effectively articulated at the national level. But, that has not lead to much clarity concerning just what are the *real* national political constraints that must be recognized in local decision making. Instead, there has been paralysis.

There are several difficulties with a model of hierarchically arranged levels of legal and policy constraints as a way to integrate national concerns with local decisions. For reasons discussed earlier, structural and symbolic legislation adopted at the national level will frequently and intentionally avoid providing specific guidance on the controversies that gave rise to the act. Witness NFMA. More important, national institutions are increasingly incapable of providing any kind of guidance. James Madison designed for us a political system that multiplied points of access to reduce the possibility of rule by a "majority faction." We ended up getting government by "interest group veto." Neither Congress nor the executive nor the bureaucracy nor the courts have the capacity or the inclination to exert leadership. Instead, publics and their leaders focus upon largely symbolic entertainments, pale imitations of governing. Frustrations grow. This is the fundamental predicament for the "on-the-ground" natural resource manager: not only is there inadequate guidance from above, but "higher ups" may very well be issuing contradictory demands — e.g., congressional requirements to protect sustainability while keeping the ASQ's up — and solutions developed locally may be vetoed on up the line. The delicate warp and weave in the fabric of federalism woven by James Madison is tearing.

At least part of the answer is to start the reweaving process at the local level. This means interest group politics at the local level: messy sausage making involving bargaining, logrolling, interests, influence, and salience. It means exploring ad hoc and quasi-governmental arrangements outside the ordinary way of approaching problems. It means involving "outsiders" throughout resource management processes and procedures. It means experimentation, which means failing. And, it means local officials sensitively understanding national and regional interests and, in so doing, providing the leadership that our national political institutions may be structurally incapable of delivering.

Integrating Scientific, Social, Political, and Economic Analyses

Whether it is a simple draft environmental impact statement or a complex, interagency regional management plan, many types of information must be brought together. Figuring out how to accomplish this integration is, perhaps, the single greatest challenge for the development of a modern forest management. I will approach the subject in two parts, first identifying the complementarities among social, political, and economic analyses, and then exploring how these analyses are integrated with scientific understandings of the physical and biological resources being managed.

Political information — for example, the results of public involvement processes — economic analyses, and social impact assessments are frequently confused. However, they serve quite different but complementary purposes. The differences in purposes need to be understood if appropriate designs for each analysis are to be employed (Shepard 1981). Natural resource management and planning boil down, at its most basic level, to answering two questions: where do we want to be tomorrow? and where will today's decisions (or lack of decisions) leave us tomorrow? Political analyses address the first question: "where do we want to be?" Social impact assessments and economic analyses do not. Rather, those efforts help answer the second question: "what are the consequences tomorrow of the options we are considering today?"

Social impact assessments provide information on how people will be affected. Political analyses reveal information about what people want, about demand. Economic analyses provide both types of information — impacts and demands — but are complementary rather than duplicative. Consequences identified through economic analyses add to the broader range of effects identified through social impact assessments. Inferences about economic demand are quite different from the information on demand provided by political assessments. They address the demands of different audiences. There is a more fundamental difference. While economic analyses can provide useful insights into overall efficiencies, demands for goods including public goods, net social benefits, and the like, it is individuals — not societies — that pay costs and receive benefits. Economic analysis cannot, generally, provide conclusions about whether any particular *distribution* of costs and benefits is preferable. Who will win and who will lose? Societies have governments — including natural resource managers — to handle such distributional questions, and that is where political information becomes germane.

The complementarities go further. Earlier, I introduced the notion of "salience" as an important dimension of political information. That presentation ended with a significant dilemma: mechanisms that are available to assess salience will, if solely relied upon, guarantee inefficient resource allocations. The problem arises because political analyses of salience must, necessarily, be restricted to identifying the interests of those for whom the issues are important. Many, indeed most people, are left out of any particular analysis. Yet, decisions may have consequences for them. Large groups, each of whom have a small stake in the outcome of a natural resource decision will be particularly disadvantaged by political participation mechanisms. Well designed social impact assessments and economic analyses provide decision makers with information about these consequences. To the extent that political analyses allow decision making latitude, the natural resource manager can use this information to move decisions toward greater efficiency and greater equity than would emerge were the salient interests of the most involved participants the only guide.

Social impact assessments and economic analyses can also shape public political involvement. The results of such analyses about policy consequences can change people's perceptions of various options, thereby shifting their political demands. Education can occur. However, this possibility may often be overstated. The politically involved generally already have above average levels of information and, more important, have strongly held opinions and beliefs. Discordant messages are unlikely to penetrate these perceptual barriers. Social impact assessments and economic analyses may, however, identify logrolling options that were not apparent to the participants. Such analyses may also change the mix of participants by providing information that provokes other interests to realize that they have a stake in decisions.

Social, economic, and political analyses provide needed, complementary information. How is that information to be integrated with other assessments, in particular, with scientific understandings about the natural resource being managed? One finds prominent ecologists emphasizing the role of the social sciences (Hardin 1968). And, there are social scientists who reject that position (Crowe 1969) or who would leave the ball in the ecologist's court (Caldwell 1987). Sort of an interdisciplinary tennis match.

Whatever the type of analysis — social or biological — several principles seem to apply:

- Information is not understanding. Understanding is the result of a creative act by the observer and "accepted" understandings — scientific or political — are the result of ongoing social processes.
- Facts have no meanings until humans interpret them. No amount of data collection is going to obviate the need for judgement.
- Science does not make decisions. Societies do. Whether society should take an action that will lead to the extermination of a species is a political, not a scientific, question.
- Information and understandings will always be incomplete, tentative, subject to change, and possibly wrong. During a long life, one will mistakenly reject the null hypothesis at the .05 level 1 in 20 times. One will never know which of the times were the errors. Risks must be taken. This does suggest weighing the consequences of various types of errors in establishing when to be particularly cautious.
- Professional judgement and ethics are crucial ingredients of decision making processes. Professional ethics are a source for criteria that will be used to evaluate options. Professional standards are also the basis for determining the reliability and the validity of the various assessments that are used in making decisions.

The basic view taken in this section can be summarized diagrammatically. Starting at the left side of Figure 2, decisions are the result of evaluations of the consequences of various options. Alternatives are judged based upon how close they get us to where we want to be. The values to be applied to the consequences come from political analyses, professional ethics, and interpretations of applicable laws and policies. The consequences to be evaluated result from various assessments and professional judgments as to the soundness of those assessments.

Figure 2 represents one type of answer to the question of integrating different sources of information in natural resource decision making. It is also possible to approach the question as a challenge in human relationships. In natural resource management, the interdisciplinary team is an example of this approach. Rather than try to precisely define uses for each type of analysis, a sociological — and political — institution is designed with the expectation that it will achieve the desired

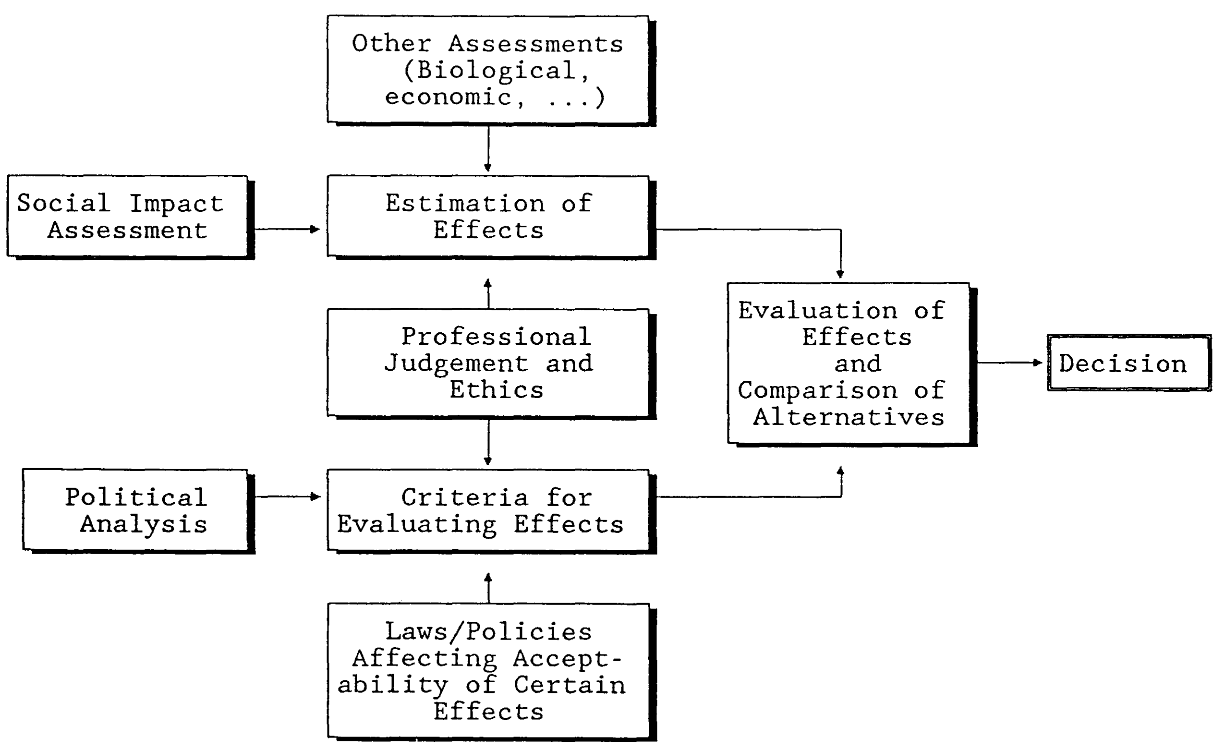

Figure 2. — Integrating political, social, economic, and biological information in natural resource decision making.

integration of understandings. On-the-ground "New Perspectives" experimentation with structures that incorporate "non-traditional" publics provides other examples (Lichen 1993). Wondolleck's (1988) advocacy of conflict management techniques and Brown and Peterson's (1993) suggested use of "citizen juries" are other examples of the more general strategy: to integrate various understandings by focusing upon the structuring of interpersonal communication and social and political relationships rather than upon the flow charting of steps in a process. There are many ways in which one might restructure decision making to achieve an improved forest management although, being nontraditional, the variations require imagination to conceive, organizational flexibility for implementation, and a willingness to risk failure.

With today's controversies and challenges, it is easy to lose track of how far natural resource management has come in "opening up" to publics, to performing the political act of allocating resources based, in part, on assessments of what people want and how badly they want things. Twenty years ago, lack of responsiveness to publics lead to harvest controversies and then to the National Forest Management Plan (Weitzman 1977). Fifteen years ago, euphemisms like "institutional analysis" still had to be used to refer to political responsibilities (Shepard 1980). Even when such euphemisms were used, personnel in agencies like the Forest Service denied that their jobs entailed such responsibilities, asserted that such matters were for "higher ups" to take care of, and the "much higher ups" were reluctant to provide line personnel with increased understanding of means for being politically responsive to publics. Undertakings like "New Perspectives" and "ecosystem management" represent a dramatic change from that earlier orientation because they acknowledge that social and political responsiveness is a legitimate aspect of a forest manager's job, and this is found both in agency policy (Salwasser 1990; Robertson 1992; Kessler 1992; Overbay 1992) and practioneers' beliefs (Clark 1991). "Being responsive" and "opening up" sound great. However, as this paper has explored, the political aspects of natural resource management are complex, challenging, and may require uncomfortable confrontations with conventional assumptions about the uses of both political and scientific information in decision making.

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A Political-Economic Perspective on Sustained Ecosystem Management

Thomas C. Brown and George L. Peterson[1]

Abstract.—Our political system places great stock in the value judgments of the citizenry. However, in some decision making situations, involving difficult questions about public resource allocation, carefully considered value judgments are a scarce commodity. Especially in decisions about the rate at which irreplaceable natural resources should be used, we lack procedures that allow for well-informed citizen input about values. This paper presents a value framework of functional, held, and assigned values that defends the role of citizens as the appropriate source of value judgments, and proposes the citizen jury as a source of such value judgments.

INTRODUCTION

As René Dubos (1976) recounts, when the city of Chicago held a World's Fair in 1933, the fair's guidebook contained a section titled "Science discovers, Industry applies, man conforms," with text proclaiming that "Individuals, groups, entire races of men fall into step with ... science and technology." While science and technology have in the ensuing 60 years become pervasive, few people today would suggest that humankind should meekly follow the lead of science and technology. Rather, as Dubos notes, the present view is that scientific technology must be managed with a strong concern for the long-range consequences of human interventions into nature.

A similar shift in view has occurred in public land management. In 1910, Gifford Pinchot declared: "The first principle of conservation is development, the use of the natural resources now existing on this continent for the benefit of the people who live here now" (p. 43). In contrast, the newly published mission of the U. S. Forest Service includes "advocating a conservation ethic in promoting the health, productivity, diversity, and beauty of forests and associated lands." As the supply of pristine natural areas has dwindled and as our understanding of ecosystems, and the various services they perform, have improved, the values we assign to public lands have certainly changed.

In response to changing values, emphasizing concern for our environment and for our descendants, managers are trying to operationalize new concepts such as *ecosystem management*, *sustainability*, and *biological diversity* so that they affect everyday decisions. We hope to shed some light on this process of reflecting values in actions, by considering three questions: (1) Are ecosystem management, sustainability, and biological diversity policies or values? (2) If values, are such values merely preference-based, or do they derive from some deeper foundation? (3) What is the role of the public in this new environmental era, and how can that role be facilitated? In answer to the last question, we propose the use of citizen juries as a source of value judgments for difficult, value-laden public natural resource decisions. The common thread uniting these three questions is that wise land management depends on an understanding of the public's values.

POLICY OR VALUE?

Ecosystem management, *sustainability*, and *biological diversity* are increasingly popular buzzwords in environmental management circles. The U. S. Forest Service, for example, recently adopted ecosystem management as its modus operandi. After much discussion, however, there is little agreement about how to implement these concepts in environmental policy and management. Different academic backgrounds have spawned special interests that bring narrow interpretations to the conference table. Because the buzzwords do not have operational definitions, they mean different things to different people and nothing to some people.

[1] *Thomas C. Brown is an Economist and George L. Peterson is a General Supervisory Engineer, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, located at Fort Collins, Colorado.*

Public land management, given scarcity, is essentially a matter of choosing among conflicting objectives or states of the resource. For a management policy to be useful, it must provide some criteria for choosing or compromising among conflicting goals. For example, a policy may (1) state categorically that vehicle use is prohibited in officially designated wilderness areas, (2) declare that wildfires will be extinguished *if* certain risk criteria are exceeded, or (3) state that timber will be sold *if* the bid price exceeds agency costs. These three examples of policies each reflect a consideration of various goals or values. The wilderness access policy addresses a conflict between wilderness preservation and user accessibility. Similarly, the fire policy attempts to resolve a conflict between ecological process and resource protection, and the harvest policy addresses a conflict between lumber availability and industry jobs on the one hand and taxpayer relief on the other.

Unlike these three policies, ecosystem management, sustainability, and biological diversity fail to provide the manager with a basis for choosing or compromising among conflicting goals. These three concepts are policies only in the loosest sense of the word, because they are currently so poorly defined that they provide only vague guidance to managers. Rather than policies, we suggest these three concepts are more like goals or values, to be balanced with other goals or values in the course of reaching useable policies.

Ecosystem management is an overriding philosophy management of ourselves as a component of the greater whole (the ecosystem), management of our interactions with and impacts on the whole, and management of the whole so as to maintain, enhance, and sustain long-term improvement in the quality of human life. Use of the word "system" in "ecosystem" implies that the whole is a complex interaction of many things, with it's overall behavior and condition being greater than the sum of its parts. Because it is a very complex system that we understand poorly, it is difficult or impossible for us to predict its behavior in response to many of the things we do. Ecosystem management is therefore a goal toward which we strive—a concept of the good or the preferable—not an operational way of doing things. It suggests, simply, that sustained improvement in human welfare depends on a harmonious relationship with the whole ecosystem of which we have become a dominant part.

Sustainability is also a goal or value toward which we strive. Our biological mandate is to survive and improve our ability to survive as a species. Having outrun most of our predators and opposing forces, developed a technological culture, and multiplied in unprecedented numbers, we have begun to dominate the larger ecosystem and mine its capital. However, the ecosystem is a depreciable and depletable asset, not a bottomless pit into which we can continually dump the consequences of our existence. Sustainability means that humans are living in symbiosis with the greater whole so as to maintain or increase environmental capital, relative to human welfare. However, as discussed in more detail later, sustainability is sometimes a goal that must be compromised with other goals.

Biological Diversity is a goal or value embodying a cautious and conservative approach to environmental management in the face of incomplete information, as well perhaps as a felt obligation towards other species.[3] It is another concept of the good or preferable, whether as an instrument in meeting some other good or as an end in itself.

To consider these three goals or values as policies only leads to confusion at decision time, because they fail to resolve conflicts or provide specific management direction. But if values rather than policies, how can these values play a role in public land management decisions? What is the source and role of value in such decisions? To begin to answer these questions, let us first consider in some detail the way we use the word "value."

THE SOURCE OF VALUE

Perhaps the most fundamental distinction among the ways we use "value" is between those uses of "value" that rely on human preference and those that do not. Nonpreference-related uses of "value" include the mathematical and the functional.[4] In the mathematical sense, "value" means "magnitude." We may say, for example, that the values of n in the expression $n^2=4$ are 2 and -2. In the functional sense, value refers to the physical or biological relationships of one entity to another. For example, we speak of the value of cover for elk habitat, the value of nitrogen for wheat production, or the nutritional value of vitamin C. These values are input-output relationships. They exist whether or not humans prefer them or are even aware of them — they are discoverable, but exist no matter what we prefer.

Preference-related values include *held* values and *assigned* values. A held value is an "enduring conception of the preferable that influences choice or action" (Brown 1984:232). Such conceptions of the preferable could concern modes of conduct (e.g., honesty, fairness), end-states (e.g., happiness, wisdom), or qualities (e.g., friendliness, intelligence). Environmentally oriented values include beauty, sustainability, productivity, naturalness, and diversity.

A person's held values may conflict, as do generosity and frugality, or duty and pleasure, and thus usually do not individually provide specific direction for action. Similarly, an agency's goals often conflict, as productivity and beauty or development and diversity sometimes do, and the conflicting goals must somehow be balanced in the course of developing specific policy.

An assigned value is the "expressed relative importance or worth of an object to an individual or group in a given context" (Brown 1984:233). An assigned value is thus the standing of an object relative to other objects, where an "object" signifies whatever can be preferred to something else (including physical things, persons, emotions, images, thoughts, symbols, etc.). A person's choice or purchase indicates an individual assigned value. A majority vote or a market price is an example of a group assigned value. An assigned value results from preference relationships between a person (or many people) and an object, given the person's held values and the context of the valuation. Thus, an assigned value is the result of held values and often incorporates a context-specific resolution of conflicting held values.

Functional values have a role to play in the world of preference-based values, for knowledge of functional values can affect preference-based values. If someone values health, for example, and knows of the functional value of vitamin C to health, valuing health may lead the person to value vitamin C. However, the value assigned to vitamin C is preference-based because that value follows from the value assigned to health.

Clearly, assigned values are the result of preference. But are held values, upon which assigned values rely, also based in preference? While the reasoning seems circular, held values are indeed also preference-based. That is, held values are "objects" that are ordered via preference relationships. A held value may be considered a "preference of the first order," but nevertheless a preference.

It is perhaps obvious by now that we are not using "preference" in a trivial sense, such as to simply indicate what one "likes." Rather, preference is being used in its broadest sense, as the basis for human choice. That is, preference allows for or gives rise to choice. We are making no claims about the source of preference or about the nature of free will as it applies to choice. It may be that some of our preferences are heavily influenced hereditarily or by early childhood experiences so that we have incomplete control over some of our choices, and it may be that individuals differ considerably in the amount of personal control they have over their choices. These complex and fascinating issues are beyond, and, we argue, irrelevant to, this discussion. We are taking preferences as given as a property of the individual.

This value framework rests on two philosophical bases. First, personal freedom, or sovereignty, is central to the political, philosophical, and economic definitions of value in public policy and is the foundation from which preference becomes the practical justification of value. In the political sense, the "right to decide" rests in the sovereign power. In a democratic society, sovereignty resides in the individual citizen, i.e., "government of the people, by the people, and for the people," with government "deriving its just powers from the consent of the governed," and with the powers of government limited by the rights of its citizens. In harmony with this political justification of value and derived from the same underlying political philosophy, market-based economic theory anchors value in consumer sovereignty, as the product of the choices consumers make among their budget-constrained alternatives. Thus, except in the realm of abstract philosophy, value is defined, assigned, and justified by sovereignty as an act of human choice.

Second, value (except in the functional or mathematical senses) is absent without the valuer. As stated by Santayana (1896:18), ". . .there is no value apart from some appreciation of it, and no good apart from some preference of it before its absence or opposite. . . Or, as Spinoza clearly expresses it, we desire nothing because it is good, but it is good only because we desire it."

Within the constraints imposed by law and public policy, individuals express and assign value by exercising their sovereignty through individual preferences and choices, and those same individuals collectively express and assign social values and impose legal constraints upon their choices by exercising sovereignty through due process of law by social choice. Philosophers can state beliefs and posit ethical maxims and prophets can cry repentance from the walls of the city, but sovereignty rules the land.

Given this value framework, when biologists argue for an environmental policy that aims to restore an area to its "natural" condition, they are in essence stating a preference for one held value (naturalness) over others (e.g., utility, refinement).[5] And when philosophers claim that a species or ecosystem has intrinsic value, they are in essence stating a preference for one held value (preservation) over another (e.g., serviceability).

Some statements of biologists and even philosophers seem to claim that the source of such concepts as naturalness or biological diversity is deeper than mere human preference. For example, Rolston (1982:145) argues that "intrinsic natural value recognizes value inherent in some natural occasions, without human reference." Rolston's claim appears to be that natural entities, like species and ecological processes, have value that transcends (does not derive from) human preference. The intrinsic value claim seems to suggest a non-preferential truth that can and has been discovered by the author. However, we would argue that the claim actually reflects either (1) a faith that nonhuman entities of nature have inalienable rights, or (2) a functional value (a physical relationship). In the first case, the claim is a held value. While the assignment of value to natural entities just because they exist (not for any service they render to humans) is certainly reasonable, we wish to note that someone *chose* to assign such value. We must unavoidably regard the claim as preference based, no matter how keenly it is felt.[6] In the second case, the attribution of intrinsic value to some entity simply indicates that the entity is an essential input in a physical input-output relationship. For example, accepting the value of biological diversity to ecosystem robustness as a functional value might lead one to say that biological diversity is inherently or intrinsically valuable; however, it is essential to recognize that the attribution of intrinsic value to biological diversity is dependent on assigning value to ecosystem robustness.

Although held and assigned values are preference-based, people do not necessarily prefer that which will increase their welfare. While avoiding a specific definition of welfare, we suggest that it is generally accepted that preferences depend on knowledge, and that lack of knowledge may lead to preferences that do not enhance welfare.[7] In other words, well informed preferences may differ from poorly informed preferences, and welfare may differ depending on one's preferences. For example, our welfare may depend on certain natural processes (on certain functional values), whether or not we are aware of them; that is, certain natural processes may be "of value" (be important to our welfare) even though we do not assign value to them because we are unaware of them. The dependence of preference on knowledge certainly suggests that science has a role in resource management and that the public whose values help determine resource allocation should be well informed of scientific findings. However, in a democratic society, resource allocations derive from peoples' values, even if its citizens are ill-informed.

The realization that our held values are, as far as we can be sure, a matter of preference (are "up to us") does not trivialize those values or make them an invalid guide for public land management. Our values may flow from the highest reverence for nature, even from a conception of nature as the key to the mystery of God.[8] Neither does the rooting of value in preference suggest that all assigned values are equally valid for a given decision. Reliance on preference does not negate the concept that there is some truth to which we may aspire, or negate the dependence of our future welfare on the values that our current actions reflect. But our social/political decisions must rely on some amalgam of the plurality of values of society, not on what a minority consider to be the truth.

This principle, that resource allocation should be based on the values of the full constituency affected by the allocation, has important implications for value measurement when long range commitments of public resources are being considered. We now turn to the measurement of values that appropriately represent the relevant constituency.

APPROPRIATE VALUES

Recognizing that policy is based on human values, in combination with what physical and biological relationships (functional values) we can determine and effectively convey to people, we must proceed to decide how best to measure human values and incorporate them into policy decisions.[9] While in a representative democratic system we rely heavily on the judgments of our elected governmental representatives, those representatives, as well as the resource managers who articulate and execute policy, must by law or by good sense rely on the values of the citizenry. Of most use would be a system of assigned value measurement that counterbalanced the political system's heavy reliance on or susceptibility to pressure groups. Among various more broadly based frameworks for measuring public values, a prominent approach is economic valuation of relevant costs and benefits, as an input to the information system we call benefit-cost analysis.[10]

A common complaint with the economic approach to incorporating values into policy decisions is that the difficulty of measuring the values of nonmarket goods tends to depreciate them relative to market commodities, so that the economic analysis suggests an inefficient solution biased towards resource development. For example, without adequately valuing the full costs of a timber harvest, whether they be to downstream water users because of erosion caused by the harvest, or to persons who value the old growth that will be lost, the harvest may seem more advantageous than it is. In response to this valid complaint, economists, especially in the last 20 years, have worked to improve methods for estimating the economic value of nonmarket goods. The early efforts focused on private-like goods such as recreation opportunities, but more recent work has focused on public goods such as species preservation. While there is still much controversy about the ability of nonmarket valuation methods to estimate values comparable to those available for market goods, let us assume for the moment that such methods exist, and thus that we can reliably estimate what people really would pay (or accept in compensation) for all relevant goods. We propose this assumption in order to focus on a more fundamental sustainability issue — the relative value of resources over time (i.e., the discount rate).

First consider a short-term decision about a proposed reallocation of resources, one that affects only the current generation and one where the discount rate is of little importance. If a benefit-cost analysis shows that benefits exceed costs of the proposed reallocation when the values of all relevant resources are measured, the reallocation is considered to be "efficient" (i.e., it satisfies the Potential Pareto Improvement criterion, such that the gainers could pay the losers for their loss and still be better off). This proposed reallocation would clearly then be a candidate for serious consideration. However, efficiency is only half of the economic picture, for the efficiency determination ignores the equity of the proposed reallocation. There are two major equity concerns. First, the losers may not actually be reimbursed, so some people may gain at others' expense. Second, monetary values depend on the existing income distribution, since those with more income essentially have more "votes" in determining the values. If a different income distribution would have produced different monetary values, a different benefit-cost determination might have resulted. If a fairness criterion is accepted, any income distribution less fair than the current one that would result in different monetary values can be ignored; however, if an income distribution more fair than the current one would produce different monetary values, the equity concern is a serious one. Resolution of equity concerns is a political matter; all an economist or other analyst can do is articulate the distribution of costs and benefits associated with the proposed resource reallocation.

Accepting the existing income distribution as given is generally considered to be a reasonable course, since it is assumed that economic values are not very sensitive to the income distribution, and the existing distribution is the outcome of the distribution of sovereign power. The more important equity concern for short term decisions is whether losers will accept their loss. In practice, the losers are typically the general taxpayers while the gainers are a specific group, often allowing the action to move forward with limited opposition, especially when the action is generally regarded as having social merit, such as with public education.

However, now consider a decision that has implications for future generations, and again assume that comparable economic values for all relevant goods can be estimated reliably for the current generation. Further, assume that preferences remain constant over all generations and that any real price changes are accounted for in the analysis. Based on the commonly observed propensity to discount the future, the typical procedure in benefit-cost analysis is to discount future costs and benefits at some market-based rate of interest. When such a rate is used for very long range decisions, however, the implications can be dramatic, because a positive net benefit (an "efficient" outcome) is more likely the sooner the benefits happen and the later the costs are incurred. Clearly, the greater the discount rate (i.e., the more the current generation discounts the future), the more likely a proposed resource reallocation that benefits the current generation at the expense of the future will be declared "efficient." Equity issues thus become extremely important if the costs and benefits are not equally distributed over time.[11]

The intertemporal equity concern can be looked at from the perspective of the initial distribution of income or from the perspective of gainers and losers. From the first perspective, consider that the actual values used in the benefit-cost computation are the current values times the discounting factor. The further into the future a cost or benefit occurs, the less is the present value of that cost or benefit. The effect of discounting is therefore to lower the income (the votes) of future generations, and this unequal distribution of income can have a compelling effect on the results of the analysis when costs and benefits are not equally distributed over time. Using the fairness criterion mentioned above, it may be difficult to argue that a discount rate based on the rate of time preference of the current generation, as established in the market, produces for the analysis a fair distribution of income across generations. From the second perspective, it is easily seen that the effect of discounting when the benefits occur sooner than the costs is that the current generation gains and future generations lose. From either perspective, an important constituency of the decision—future generations—has been poorly represented.

In contrast to the standard discounting approach, Rawls (1971) has suggested that decisions affecting the future should be made with decision makers placed behind a "veil of ignorance" about which generation they belong to. This impartiality criterion suggests equal use of irreplaceable resources across generations, implying a zero discount rate. But with a zero discount rate, if enough generations are involved, use of nonrenewable resources (such as oil) approaches zero for any given generation. Likewise, irreversible development (such as building a dam in a unique natural area) is essentially precluded. Furthermore, a zero discount rate may foreclose future options by undervaluing investments that produce wealth and new technology that would be of great value to future generations.

Clearly, some compromise is needed between a zero discount rate, which would preclude many resource uses and perhaps prevent valuable investments, and a typical market rate that reflects only the atomistic time preferences of the current generation. This compromise has been called a social rate of discount. Marglin (1963) presents three arguments for a social discount rate, which we will reduce to two. The "authoritarian" argument follows from Pigou who regarded the government as the actual representative of not only the current generation but of all future generations as well.[12] The argument is that the government in this role should consider the wishes (the values) of both current and future generations. Because the welfare of future generations depends on current consumption patterns, the government should assure protection of future welfare by policies that force sufficient saving (e.g., resource protection). In essence, the government would proclaim what it deemed to be an appropriate discount rate.

The other argument takes a more democratic approach, realizing that the government is run by and for the current generation, such that any saving for the future must rely on the values of the current generation. The basis of this argument is that most citizens have a set of held values that include a concern for the larger group (including the future) as well as concern for self. As Etzioni (1988:83) states, "people do not seek to maximize their pleasure, but to balance the service of two major purposes — to advance their well-being and to act morally." If people do value the welfare of the future, then what is needed is a way for that value to be expressed and measured—a way that avoids the atomistic context of the market place.

People acting alone have little power to affect the future, just as they have little power to affect air quality or provide for public defense, because others, while applauding a person's altruism (for they too value the future or clean air or public defense), may remain free-riders. The welfare of the future, like air quality and public defense, is a public (i.e., nonrival and nonexcludable) good. Typically with a public good, the benefit to person A that flows from A's act in behalf of the public good is less than the cost to A. However, not only A benefits from A's magnanimous act; rather everyone benefits, and with enough people the aggregate benefit from A's act is greater than A's cost, even if the benefit to each individual is small. While A has no personal incentive to incur the cost, everyone else would like A to. In fact, each individual is better off if everyone else incurs the cost. Thus, the logical solution is collective (e.g., government) action—what Hardin (1968:1247) calls "mutual coercion mutually agreed upon"—to assure that everyone shares the cost. To simplify Marglin's presentation (ignoring, for example, intragenerational utility interdependency), the total cost, to be shared by all, should increase to the point where the marginal social benefit equals the marginal social cost. In the context of decisions affecting future generations, the future (a public good) is properly valued in benefit-cost analysis by choosing a social discount rate that reflects this comparison of marginal social benefit and marginal social cost, a comparison that results from the values of the current generation.

While recognizing the public good nature of the welfare of the future theoretically allows for the determination of a social discount rate, measuring that rate is not simply a matter of observation of human behavior. Choice of the social rate of discount requires resort to human judgments, hopefully made in light of a rich understanding of the issues. The rate is essentially an assigned value, a value assigned by the present that reflects the value the present places on the welfare of the future. The value will reflect an unavoidable compromise between our standard of living and that of our descendants.[13]

A vehicle we suggest for making the choice is the citizen jury (Tonn et al. 1993). Such a jury would represent the citizenry just as a judicial jury does, and be compensated for its time. It would be chosen randomly, subject to certain selection criteria, from within a geographical area appropriate for the decision at hand. The selection criteria would assure two things: that jury members do not have exceptional monetary interests in the outcome[14], and that they satisfy certain minimum standards of intellectual ability (because of the complex issues that jury members would be exposed to within a short time). Jury members would become fully informed of the issues, the pros and cons of various rates of social time preference, during the deliberation process. Unlike trial juries (but not unlike grand juries), members would be able to ask questions of those presenting the technical and evaluative information. And the jury would attempt to reach consensus about the rate that should be used in decisions affecting irreversible allocation of resources. Separate rates may be determined for decisions involving different kinds of resources, such as mineral energy deposits, unique ecosystems, species survival, or archeological sites. The choice may also reflect a judgment of the likelihood that technology will alleviate shortage, which is more likely for some resources (e.g., mineral energy deposits) than for others (e.g., unique environments).

Three important advantages of a citizen jury are that it is representative, well-informed, and formal. First, the citizen jury, unlike a task force or advisory panel, is randomly chosen and thus represents society, not specific interest groups. Its frame of reference is the larger society; thus the welfare of future generations would be considered by the jury to the extent that the jury members considered the welfare of the future to be a concern of the current members of society. Second, unlike a survey (such as a contingent valuation survey used by economists), the citizen jury becomes well informed about the issue it is deliberating in the course of listening to and asking questions about the information that is presented by well-informed advocates of opposing viewpoints. Third, the citizen jury process is formal and systematic, following rules that would need to be established at the national level before citizen juries could routinely be used. National level sanction plus the history of jury use in the United States should lend authority and weight to citizen jury deliberations.

The essential difference between a citizen jury for the consideration of resource valuation and a judicial jury is that the former only recommends assigned values, while the latter proclaims a verdict. Nonetheless, properly constituted and utilized, a citizen jury's decision could carry considerable weight among the ultimate decision makers in a democratic society like the United States, where use of juries is widespread.[15] Indeed, this may be the reason such juries have not been used thus far, for to provide the "silent majority" with a voice in public resource management would tend to dilute the force of special interests as well as of the bureaucracy. A citizen jury would give new meaning to public involvement, which too often is an exercise in diffusing public criticism of an agency's preferred path.

To this point we have been assuming that the economic values of all relevant goods and services could be estimated from the choices of current citizens. However, some goods and services may be beyond the purview of economic valuation, such as the value of a subsistence way of life (see Brown and Burch 1992). In this case, it may be reasonable to remove such goods and services (certain resources) from the economic analysis to consider them as constraints in the analysis and focus the analysis on the remaining, more easily valued, goods and services. In essence, this approach assigns infinite value to the removed resource. The decision to remove a good from the analysis may result from a law that constrains decisions (such as the endangered species act). In other cases, a citizen jury could be consulted to suggest whether a good should be considered beyond the purview of economic valuation.

However, benefit-cost analysis may be a poor tool for assisting analysis of a proposed resource reallocation, even if an appropriate discount rate has been estimated. If so, we are left wanting some procedure that fills the gap left by the exit of benefit-cost analysis. Principally, that gap is the lack of a procedure that attempts to consider the preferences of the larger society (that responds to other than the wants of special interests). To fill this gap, we would again suggest the citizen jury in decisions that hinge on an articulation of assigned value. Such a jury could address specific proposals for resource reallocation, just as it could address the issue of the social discount rate. The jury would take the time to become fully informed about the proposal. While its decision would be advisory, it would represent the only officially sanctioned impartial judgment that is likely to be available.

CHANGING VALUES

As the examples in the introduction of this paper illustrate, values change considerably over time. The common assumption of benefit-cost analysis, that the values of those currently living (perhaps adjusted for past trends in real prices) can be accepted as adequate representations of the values of our descendants, is obviously heroic. Changing resource supplies and growing scientific knowledge will certainly influence the values of the future. One scenario, advanced by Krutilla (1967), is that technological advance will continue to lower the real cost of commodities, while increasing incomes will continue to raise the value of amenities, so that resource development that consumes irreplaceable amenities, such as a unique environment, should be seriously questioned. To this argument we would add that as we continue to learn of the intricacies of the natural world and gain a fuller understanding of the relationship of human welfare to well-functioning ecosystems, we are likely to increase the value we place on such ecosystems. The implication of this scenario is that we should tend to avoid resource extraction where it conflicts with protection of unique amenities.

Another scenario is that as oil and natural gas supplies dwindle and population continues to grow, technology will fail to save the day with a new clean source of energy, leading to declining real incomes and increases in the real prices of many commodities. The implication of this scenario is that we should lower our current consumption of basic natural resources.

Other scenarios could be posited, but perhaps the main lesson of such scenarios is that there is great uncertainty about future conditions and values. Given this uncertainty, intergenerational fairness suggests that we should strive now to maintain options for our descendants by both protecting unique amenities and by lowering our consumption of basic resource stocks.

FINAL COMMENTS

In this paper we have taken peoples' held values as given, and focused on ways to measure assigned values that are relevant to public resource management. But we do not wish to leave the impression that there is no role for public land management agencies in educating the public about values. We are all bombarded with information that tends to affect our values (for example, commercial interests, in an effort to sell their products, in essence sell lifestyles that have implications for one's propensity to save for the future). It behooves agencies, which have been given the job of managing resources, to enrich the public's understanding of those resources and the role of those resources in human welfare. That enrichment certainly should include new information about functional values, but can also include preference-based information, such as that which would engender a respect or reverence for the natural world.

From the human perspective we see the ecosystem as having a purpose. That purpose is to support and sustain human welfare as defined by the values and preferences we hold. But, because human life depends on the condition of the ecosystem as a whole, the higher purpose is to sustain the ecosystem such that it is capable of sustaining human life, and not only human life, but quality human life and hope of continued improvement into the future. This purpose lifts us out of the ecosystem, so to speak, into a role of stewardship. Because we have emerged as the species in control (some would say "out of control") but capable of reasoned action and self control, we must accept the responsibility to manage the behavior of the whole ecosystem by managing ourselves within the system as well as managing and nurturing the system with which we interact.

While we chided those who claim that natural entities have intrinsic value, we think there is an important place in the public discourse about resource management for held values that reflect a reverence for nature, a reverence of the kind that Wordsworth wrote of:

To me the meanest flower that blows can give
Thoughts that do often lie too deep for tears.

Such feelings suggest an ethical relationship to nature that is often lacking in our modern world. We would simply urge that the public discourse center on our role as guardians of our environment rather than on the polarizing notion that humans are only the problem, not the solution; and on an appreciation of human values rather than on "truths" that cannot be supported in the world of political conflict resolution.

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ENDNOTES

[3] *Biological diversity also serves as an indicator of the robustness and health of the greater system.*

[4] *Other nonpreference-related uses of "value" include its use in art to indicate luminosity or brilliance, and its use in music to indicate the length or duration of a note.*

[5] *We must be careful with our use of the word "natural." Some who hold the aforementioned value that prefers "naturalness" seem to be espousing a dualistic definition of nature that defines homo sapiens as outside "nature." This view perhaps reflects an Augustinian interpretation of Genesis, whereby humans fell from grace and from their natural place with the choices of Adam and Eve (see Pagels 1988). However, as with any species, we see the human species as a natural phenomenon. The problem is that we have gotten out of balance with the rest of nature, and we don't want to continue to destroy those aspects of nature that are of value to us.*

[6] *As Aristotle (ca B.C. 320) so wisely perceived, "As every knowledge and moral purpose aspires to some good, what is in our view the good at which the political science aims, and what is the highest of all practical goods? As to its name there is, I may say, a general agreement. The masses and the cultured classes agree in calling it happiness, and conceive that 'to live well' or "to do well' is the same thing as "to be happy.' But as to the nature of happiness they do not agree ..."*

[7] *There is much disagreement about what constitutes welfare, from the pure libertarians who believe that welfare is enhanced whenever people are free to choose, no matter what within the law they choose, to those who argue that welfare is enhanced only when equality is enhanced. However, the claim that welfare may not be maximized if preferences suffer from lack of knowledge is probably accepted by most welfare theorists, no matter how they specifically define welfare, except perhaps by the pure libertarians who simply equate welfare with free choice.*

[8] *The following excerpt from W. H. Carruth's poem "Each in his own tongue" expresses this notion beautifully:*

A haze on the far horizon,
The infinite, tender sky,
The ripe, rich tint of the cornfields,
And the wild geese sailing high;
And all over upland and lowland,
The charm of the golden-rod,
Some of us call it Autumn,
And others call it God.

[9] *Any system relying on current values of the citizens will fail to assure sustainability if basic needs are not met. Only once those basic survival needs are met can we save for the future. We are assuming in this discussion that basic needs are met. Thus we are assuming population has not outstretched the limits imposed by resources and technology.*

[10] *Benefit-cost analysis is an information system, not a sovereign decision criterion, unless the law defines it as such. As an information system, the responsibility is to fully inform all stakeholders in a given social choice, so that each stakeholder makes an informed expression of personal values.*

[11] *The point is that while valuing all relevant resources (not just marketed goods) helps to accurately determine whether a proposed reallocation is efficient, it still yields an analysis with strong equity implications if a discount rate based on the atomistic decisions of the current generation is used. This point was recently made by Howarth and Norgaard (1992:473), who state, "incorporating environmental values per se in decision-making will not bring about sustainability unless each generation is committed to transferring to the next sufficient natural resources and capital assets to make development sustainable."*

[12] *Pigou (1932) wrote "It is the clear duty of Government, which is the trustee for unborn generations as well as for its present citizens, to watch over, and if need be, by legislative enactment, to defend, the exhaustible natural resources of the country from rash and reckless spoliation. ...there is a valid case for some artificial encouragement to investment, particularly to investments the return from which will only begin to appear after the lapse of many years."*

[13] *Such a compromise must be mindful of the grim implications of the Second Law of Thermodynamics. As Georgescu-Roegen (1980, p. 58) concludes:*

Every time we produce a Cadillac, we irrevocably destroy an amount of low entropy that could otherwise be used for producing a plow or a spade. In other words, every time we produce a Cadillac, we do it at the cost of decreasing the number of human lives in the future. Economic development through industrial abundance may be a blessing for us now and for those who will be able to enjoy it in the near future, but it is definitely against the interest of the human species as a whole, if its interest is to have a lifespan as long as is compatible with its dowry of low entropy. In this paradox of economic development we can see the price man has to pay for the unique privilege of being able to go beyond the biological limits in his struggle for life....It is as if the human species were determined to have a short but exciting life. Let the less ambitious species have a long but uneventful existence.

[14] *Nearly all citizens will have some monetary interest if the outcome affects taxes or prices. Exceptional interest, however, refers to those who stand to gain or lose more than usual. For example, if the issue had to do with whether to allow development of a ski resort on a national forest, ski resort owners or adjacent private property owners would not be appropriate jury members.*

[15] *Tens of thousands of jury trials, and 80% of all jury trials, are held each year in the United States (Hans and Vidmar 1986).*

The Human Dimensions of National Forest Ecosystem Management

Greg Super[1] and Gary Elsner[2]

Abstract — "People are part of ecosystems and human conditions are shaped by, and in turn, shape ecosystems." This is a clear concept — but one that is challenging to operationalize in both day-to-day forest management and in strategic planning. To successfully incorporate the concerns of humans into ecosystem management means we will give equal weight to societal values and expectations along with the physical and biological dimensions. Recognition of the human dimensions is, of course, the first step and these are complex ranging from the spiritual, ethical, cultural, historic, aesthetic, economic and social. Planning for the human dimensions requires good information on how people use the forests and what benefits they strive to achieve. We have found that there are interrelated roles for both managers and researchers in shaping the way we incorporate the human dimension. This paper describes a conceptual model that helps social, natural and biological scientists and managers understand and incorporate the human dimension in forest ecosystem management. This paper utilizies the work of three interrelated projects which are described in the credits section.

INTRODUCTION

The Chief of the USDA Forest Service wrote in a June 4, 1992 letter that " An ecological approach will be used to achieve the multiple-use management of the National Forests and Grasslands. It means we must blend the needs of people and environment values in such a way that the National Forests and Grasslands represent diverse, healthy, productive and sustainable ecosystems." (Chief's June 4, 1992 letter).

The Forest Service has been in the ecosystem management business since it was created in 1906. Since that time it has managed the National Forests and grasslands to reflect the values held by the American public as best the Agency was able to interpret them. Public values have continued to evolve, sometimes in dramatic ways, during the last few years and the Agency is changing how it operates to meet what it perceives the public now wants.

The Chief's letter articulates clearly that greater attention will be given to assuring ecosystem sustainability than was in the past. It should be noted that the National Forests and Grasslands will still be managed to productively provide for the multiple-uses demanded by the public while sustaining basic ecosystems. Thus, the Agency is evolving by shifting the balance of what it produces and by applying ecosystem management principles in order to leave the basic components of ecosystems intact and capable of being sustained over the long run. A production shift is implied that the Agency will continue to move from a " traditional commodity orientation to one where other values receive more attention (e.g., sustainable ecosystems, recreation, wildlife species preservation, aesthetics, cultural and spiritual values). Commodity production will continue as an important output, but it will not always be the dominant purpose for managing the land.

In the past, amenity values were emphasized where no commodity values existed and management of amenity values was largely limited to support of commodity production. Ecosystem management allows a broader look at all values and a context in which to measure their importance on their own merits. Human perceptions are very important in this contextual study, since it will be human perceptions that place a higher or lower importance on the various resource conditions or outputs.

[1] *Greg Super is National Team Leader, Human Dimensions of Ecosystem Management, Ecosystem Management Staff, USDA Forest Service, Washington, DC.*

[2] *Gary Elsner is Assistant Director, Recreation, Heritatge, and Wilderness Management Staff, USDA Forest Service, Washington, DC.*

Responsible resource allocation decisions must be made knowing the full physical, biological, and human consequences of that decision.

The Human Dimension Task Team has proposed a human dimension principle of ecosystem management for inclusion in the formal statement of Ecosystem Management principles:

> *"People are part of ecosystems and, as such, humans shape ecosystems, and in turn, are shaped by ecosystems. People value or desire a broad spectrum of benefits (including survival) from ecosystems. To make effective ecosystem management decisions the Forest Service must have a scientifically sound and integrated understanding of the physical, biological and human dimensions of ecosystems. The human dimension of ecosystem management must include information about people's traditional and changing perceptions, beliefs, attitudes, behaviors, needs, values, and the past, present and possible future influences of humans on ecosystems."*

The importance of the human dimension in ecosystem management has also been articulated by Hal Salwasser (former Director, Ecosystem Management for the Forest Service):

> *"It is not whether human dimensions will be integrated with the biological and physical dimensions of ecosystem management, but whether the biological and physical sciences and technologies can be sufficiently redirected to serve the human purposes that cause institutions like the Forest Service to exist in the first place. It is people and their needs, values, and aspirations that define not only what ecosystems management is but what it is for and how it will be pursued."*

Ecosystems have three interrelated dimensions: the physical (landforms, minerals, geology etc.), the biological (plants, animals) and the human dimension (social, economic, spiritual, cultural, historic, etc.). The essence of sustainable ecosystem management is the balancing of all three dimensions to produce what people want while not preempting the options of future generations needlessly. This conceptualization of ecosystem management is shown in figure 1.

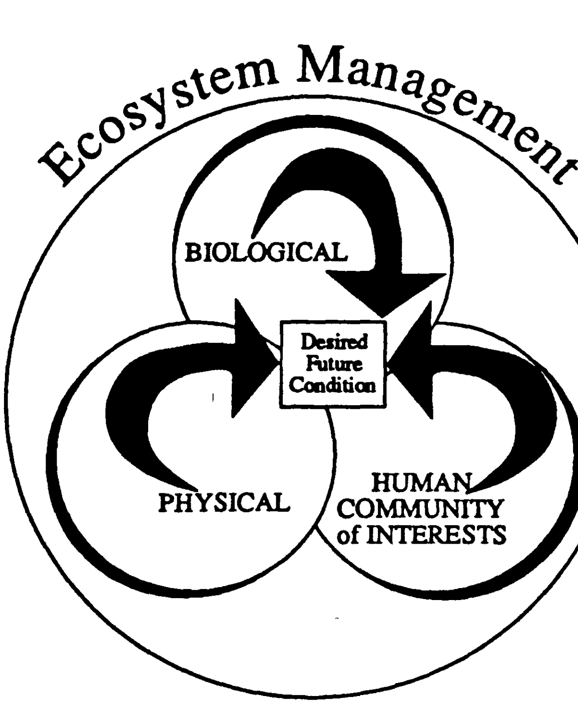

Figure 1. — National Forest Ecosystem Management Model

The Forest Service has considerable expertise in all three dimensions of the ecosystem management concept. However, while there are thousands of people working on the physical and biological dimensions, considerably fewer are working on the human dimension. Those involved in the "hard sciences" have traditionally viewed the social, cultural, spiritual, economic, ethics, and other components of the human dimension with some skepticism. The public, however, increasingly emphasizes the human dimensions as clear expressions of ecosystem values.

These growing public concerns, along with legislation requiring public involvement in resource allocation and management processes led the Forest Service to adopt a demands-driven planning process that lists "identification of public issues and concerns" as the first step. This contrasts sharply with the old "scientific model of management (adopted early by Gifford Pinchot) which relied primarily, to totally, on the technical judgment of professionally trained resource management (which is commonly identified as "the German school of forestry"). This model worked well in the early years of the Forest Service when agency goals were oriented mainly toward watershed protection, forest fire prevention, provision of lumber, and maintenance of grasslands for domestic stock. But as demands for amenity uses both grew and expanded, and as public concern about environmental protection increased, the public wanted to be involved in this "scientific management," by natural, physical, and biological scientists who understood much about technical production functions (e.g., how to grow trees, grass, fish, and game animals and how to prevent fires and stabilize watersheds but who often knew too little about managing to meet human demands for so called noncommodity products (e.g., amenity goods and services).

These demands for public involvement greatly increased the difficulty of management. A particular problem is knowing how to accommodate conflicting public demands. Another is how to respond to poor public information. For example, it is difficult to understand the potential difference between human perception (in this case, public perception) and "scientific" facts on which management decisions have been traditionally based. The publics' perceptions are based on many different things ranging from scientific findings, to media reports, and to personal experiences that reflect philosophical, religious, cultural and other orientations. These perceptions must be recognized, considered and reflected in policy and management decisions if ecosystem management is to be truly ecologically and socially sustainable. But this is not easy even for people professionally trained in the social sciences.

We must recognize that amenity values cannot be considered as secondary to commodity values or the "hard sciences". It is also important to understand that both ecological systems and human systems are in reality the same system; they are interdependent, dynamic and evolving together, as is the state of the knowledge about each. Incorporating spatial scales and temporal variation into the characterization of the human and physical/biological components of ecosystems is crucial for advancing understanding of ecosystem management.

This paper describes in some detail the human dimension components and how they might be understood and applied in Ecosystem Management. An important point of logic—and not value—must be understood in this regard. It is: Humans manage and sustain natural ecosystems to meet human needs, including the needs of humans to know they have been good stewards. The central questions are: What human needs can be met while still sustaining the basic natural ecosystems? Understanding these needs—or the reasons we manage and sustain natural ecosystems—and how these needs can be met through sustainable ecosystem development are the basic questions that face a multiple-use (as contrasted with a preservation-oriented) public agency such as the Forest Service. Put simply, since humans manage and sustain natural ecosystems to meet human needs, knowledge is needed both about these needs and how to sustain the natural ecosystems.

THE CONTENT AND METHODS OF HUMAN DIMENSION INVENTORY AND ANALYSIS

Natural systems, human habitats and social lifestyles are closely intertwined and evolving together. They cannot be compartmentalized and analyzed separately. This leads to the obvious conclusion that any approach to describing, studying, evaluating and managing ecosystems must give equal consideration to human dimensions, along with biological and physical dimensions. Indeed, human dimension expertise must be a full partner at the ecosystem management table and throughout the process for all dimensions to be effectively incorporated in the decision process.

The Human Dimension of EM and Levels of Motivational Needs

Human demands and needs from ecosystems vary from survival to personal well-being. What we seem to be seeing in the 1990's is a respect for the complete spectrum of motivational needs. Maslow's hierarchy of motivational needs, while developed for individuals, can also be interpreted for the family and community levels. The National Forests provide products and services at each level of human need as shown in figure 2.

The Forest Service influences the human experience at each level of the pyramid:

- Survival — National Forests provide vast amounts of clean water and air as well as many basic commodities such as food and fiber.
- Security — Many people are employed in the extraction and amenity resource areas associated with National Forests.
- Belonging — Many of the recreation facilities (trails, campgrounds etc) provide places and experiences for people to come together as families and groups and feel part of a bigger whole.
- Self esteem — Many of the recreation and other amenity opportunities available on the National Forests are challenging and require skills to be developed and mastered.
- Self-Actualization — Many National Forest experiences provide opportunities for easing tensions of modern life and renewing the human spirit.

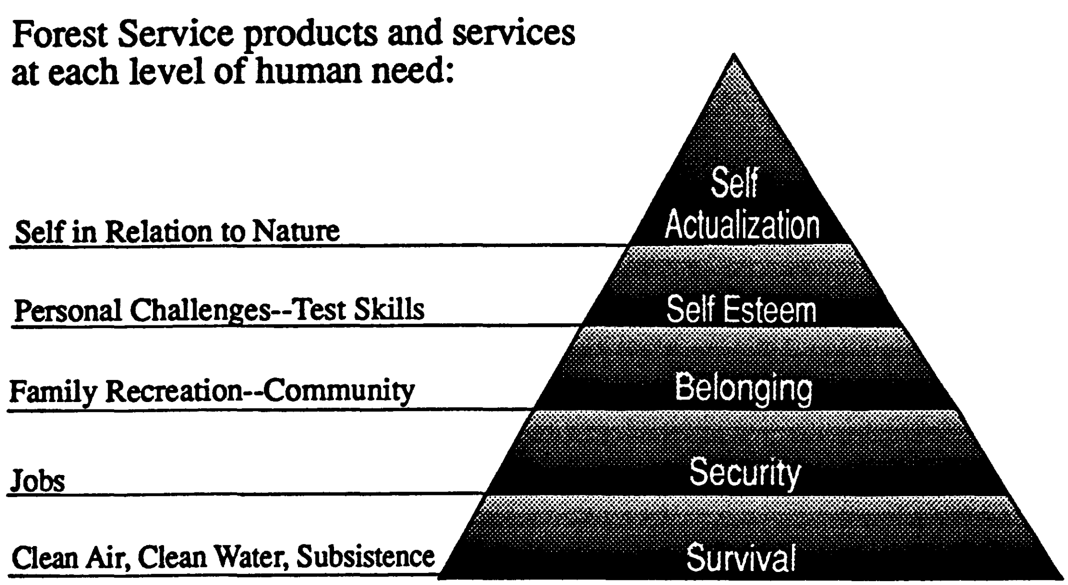

Figure 2. — A hierarchy of needs related to forest ecosystem management.

There are no values placed on these levels—they are all real and they are all part of the human dimension of ecosystem management.

HUMAN DIMENSION CLASSIFICATIONS

The human dimension of ecosystem management must include information about people's traditional and changing perceptions, beliefs, attitudes, behaviors, needs, and values and the past, present, and possible future influences of human concerns and motivations with the biological and physical fabric of ecosystem management in order to have a credible professional decision making process.

The human dimensions of ecosystems may be easy to identify, but they are not so easily defined and made an operational part of National Forest ecosystem management. At the current stage of thinking we are proposing that the human dimensions may be represented in two broad categories: first, Demands and Needs and second, Contextual Considerations.

It is important to note that each of these categories includes both objective and subjective factors. Both types of data are needed for a complete representation of the human dimension. The history of forest planning and management has shown that it is easier to include the objective than the subjective factors. However, if we are going to include people in ecosystem management we have to consider people's experiences as part of the ecosystem. And since the United States is quickly becoming a truly multicultural society the challenge of understanding both current cultures and the emerging cultures will be an extraordinary challenge.

Category I. Demands and Needs

Generally demands and needs include commodity, non-commodity, and appreciative demands and needs. While some of these demands are expressed through economic markets, many others are expressed in more indirect ways, such as political structures, on-site use patterns, organizational memberships, environmental education, viewing scenery, and many other avenues. Demands are derived from the values, traditions, and dependencies that people attach to natural resources and the ecosystems they compose.

It is our belief that land management planning is demand driven and that the factors listed as demands and needs cover the demands for goods and services from National Forests and Grasslands that create the underlying needs for the different levels of planning, including RPA, Forest and Project planning. If other relevant demands and needs have been overlooked and thus omitted, they should be added.

Category II. Contextual Considerations

While the driving forces for ecosystems planning are human demands and needs, including the needs for humans to know that they are being good stewards of the land (i.e., are practicing sustainable management), there are other types of social data that the planner/analyst must consider and/or be aware of to assure that effective, responsive, responsible, and efficient plans will be developed and implemented. These types of HD factors that must be considered are labeled " Contextual Considerations" and cover the five remaining categories of human dimensions factors shown. These five information factors in Category II provide a context for management decisions. That context may be social, physical or managerial. These contextual factors supplement, and in some instances define, the "demand and needs" factors listed in Category I.These five types of contextual factors are:

A. History and Heritage Sites [Spatial/Matrix]
B. Descriptive Landscape Characteristics [Spatial/Matrix]
C. Descriptive Social and Economic Considerations [Spatial/Matrix]
D. Forest Service Culture and Modus Operandi [Nonspatial/Narrative]
E. Other Physical Environmental Factors [Nonspatial/Narrative]

A. History and Heritage Sites:

These factors are important for gaining perspective on past uses and human effects on the ecosystem (or how human settlement has influenced the evolution of the ecosystem) and traditional uses and expectations of the ecosystem.

B. Landscape Characteristics:

These factors are attributes of the ecosystem that effect how people are attracted to the landscape, what physical features we have put on the landscape, or what physical or climatic features affect what we put on or do on the landscape. (Note: 1) See also "Other Physical Environmental Factors, and 2) We have not included a category of landscape characteristics that draw people such as mineral deposits or timber supplies.)

C. Social and Economic Considerations:

These factors describe the social and economic forces and directions that drive or resist uses of National Forest System land and resources.

D. Agency's Institutional Culture and Operation

The Forest Service's Multiple-Use mission leaves considerable room for management interpretation. The proper "mix" of goods and services to be provided by the National Forests is, largely, at the discretion of the agency. How the agency interacts with the public, the degree of public trust, public attitudes, public perception of the issues and the agency, etc. are all partially shaped by institutional culture and methods of operation. Because of this, it is important to consider Forest Service culture, philosophy, and management policy as you analyze human wants and needs relating to your particular situation.

E. "Other" Physical Factors of the Environment

There are many aspects of the physical environment that affect people; or are effected by people. In the contextual segment of the document some landscape factors are presented. It is also important to be aware of the effects climate, topography, vegetation, etc. have on people and vice versa. For example, lots of people who live in Phoenix head for the mountains around Flagstaff primarily for climatic relief. Once here, they tend to camp in relatively flat sites and in shady locations. Thus, their actions are influenced by climate, topography, and vegetation. Another example is the close link vegetation has on human lifestyles and employment. Ranching is an important source of income and lifestyle in grassland settings. Logging and it's associated industries occur in close proximity to productive timberlands. These associations are obvious. However, our management can, and has, influenced the vegetation, in some places, to the point of altering the ability for these human enterprises to continue. Most information of this type is gathered during your "normal" analysis process. The important thing to remember is that some of your most important, but easily overlooked, analysis factors for the human dimension are closely connected to physical factors like the ones discussed in this paragraph. We have assumed that information on these factors will be gathered as a part of planning analysis needed to define the natural ecosystems, so a list is not proposed for these factors.

To gain further insights on the definitions of these dimensions consider the following details shown below for both categories. The intent here is not to be comprehensive, but to suggest an initial list of data for consideration.

Category I. Demands and Needs

- Recreation activities
- Subsistence uses (fuelwood, food, and others)
- Recreation settings (urban, rural, roaded natural, semi-primitive non-motorized, semiprimitive motorized, primitive)
- Recreation facilities and services (trailheads, trails, campgrounds, ski areas, and others)
- Uses of and preferences for cultural-heritage resources
- Special places (eg, tourism destinations and high use areas)
- Scenic resources (Scenic Byways, National Recreation Trails, and others)
- Wildlife and fish and related uses (hunting, fishing, watchable wildlife, et al.)
- Access for people with disabilities
- Access to public lands (through private lands)
- Infrastructure needed for eco-tourism and heritage tourism (safe road, water supply, access to resources, et al.)
- Infrastructure needed for local community to support its desired future (wood supply to mill, trail system town to Forest)
- Stability and sustainability of local rural communities lifestyles and values
- Individual lifestyle preferences and expectations
- Promote/stabilize rural economic livelihood
- Infrastructure needed for region (State, etc) to support its desires future (eg, transmission line, highway)
- Commodity product uses & preferences other than subsistence & amenity uses above) (lumber, minerals, grazing, oil and gas, et al.)
- Special uses (rights-of-way, all types of permits and special uses)
- Legacy and options for the future
- Global survivability
- Spiritual, religious and other cultural values and uses
- Uses of land for scientific and educational purposes
- Preferences and expectations not shown above

Category II. Contextual Factors

History and Heritage Sites

- Traditional cultural properties — American Indian
- Historic cultural uses — non-American Indian, Hispanic, cowboys, miners, ranchers, loggers
- Prehistoric landscapes
- Past land use history (settlement history, settlement pattern, changes)
- National and local heritage sites (includes prehistory & historic districts)
- History of agency management

Landscape Characteristics

- Landscape variety
- Landscape distance zones
- Landscape sensitivity
- Existing visual condition
- Visual character and quality (VQO)
- Existing human-made structures (railroads, roads, buildings, towers, airports, dams)
- Location of streams and lakes if not in GIS
- Risk factors (floodplains, wildfire, etc.)
- Accessibility challenge level
- Access (right of access to public land)
- Recreation Opportunity Spectrum (ROS) classification
- Unique-outstanding land forms and natural features (geographic, scenic)
- Unique-outstanding recreation opportunity (rock climbing site, glider launch, etc)

Social and Economic Considerations

- Social institutions and infrastructure (schools, colleges, cultural center, etc)
- Resource dependent communities and industries
- Economic diversity (industrial mix or structure)
- Transportation facilities/systems
- Market areas for forest products
- Primary centers of trade
- Spatial locations of major sub-cultures
- Tourist destination areas (magnets)
- Local social and information networks
- Community political organization
- Interest groups and/or active individuals
- Demographic statistics & trends (age, sex, income distribution, education, race & ethnicity, unemployment, employment type)
- Population trends (immigrants, emigration, birth/death rates, etc) of relevant units (cities, counties, states, region)
- Adjacent land ownership
- Cultural characteristics (ranching, religion, subsistence, hunting, spiritual, etc)
- Community cohesion (degree of unity and cooperation)
- Community stability (ability to absorb and manage change)

Agency's Institutional Culture and Operation

- Information delivery systems
- Public awareness regarding EM
- Interagency coordination
- Sources of information and data
- Partnerships
- Volunteers
- Public involvement in decisionmaking
- Conflict management (prevention and resolution)
- Agency's structure, mores and organizational philosophies (includes concern about efficiency of operation, empowerment, etc)
- Agency's land ethic (stewardship)
- Laws, regulations, and policies
- Budgets
- Agency's diversity of cultures and thoughts
- Public attitudes/perceptions about managing agency
- Management models, tools, methods and other management technologies, including monitoring
- Understanding political environments and pressures
- Approved plans
- Obligatory documents (permits, mining claims, rights-of-way)

"Other" physical factors of the environment

Note that many of these "other" factors will be identified by the other ecosystem planning members representing either the physical or biological components.

To operationalize the human dimensions, it may be useful to form a matrix with the dimensions discussed above on one side and characteristics of useful data on the other. This matrix would then constitute both general guidance and a check list for human dimension information that would be useful in ecosystem planning. Each plan will be different and will undoubtedly result in a different set of relevant data and identify data needs that are not contained in this initial matrix.

Some data characteristics that are helpful to examine for each human dimension include the following:

- **Spatial Scale** — The geographic scale at which a human dimension factor should be evaluated for sustainable ecosystem management.
- **Data Format** — The manner in which data for the human dimensions factor should be displayed.
- **Data Availability** — The status of the information needed.
- **Data Accuracy** — The quality, consistency, and accuracy of the data.
- **Planning Level** — The appropriate planning scale for gathering and evaluating data for each factor.
- **Administrative Responsibility** — The Forest Service unit with primary responsibility for data management.

INTEGRATING SOCIAL DATA AND ANALYSIS INTO ECOSYSTEM APPRAISAL AND MANAGEMENT

Successful integration of human dimensions into ecosystem management and planning is signified when the "line" between biophysical and social data blurs to the point of becoming indistinguishable. The relevant question for appraisal of the condition and potential of ecosystems then becomes which data are relevant to the questions, issues, management options, or any other objectives that are being considered.

The first phase of ecosystem appraisal is to identify the question (issue, concern or need) which is initially mandating the appraisal. The question can range from a general agency policy to do an appraisal of the health, condition and situation of all ecosystems on national forests to a more specific question about the production possibilities for a specific output at a specific site, e.g., dispersed recreation. Careful definition of the question will easily lead to the identification of which biophysical and human dimension characteristics of an ecosystem and its setting need to be described and analyzed.

In such analyses for sustainable ecosystem management, some data elements within all of the human dimensions identified earlier are likely to be relevant. The scale(s) at which these data elements are relevant will depend on the scope of the question or mandate being addressed (from local, site specific questions to questions of international significance). The scales at which human dimensions are analyzed should, then, coincide with the scope or level of significance of the ecosystem question or issue being addressed.

Analysis and presentation for the ecosystem appraisal can be organized as follows:

1. Identify the question and the scales relevant to the question, and area or site.
2. Inventory the relevant biophysical and human dimension attributes to include internal (National Forest level) scale attributes, but also attributes or characteristics at influence zone, market area and other relevant external scales.
3. Describe the conditions, status, trends and values important to deciding the ecosystem question with human dimension data integrated into the analysis.
4. Define desired future conditions based on both biophysical and social analyses.
5. Identify and define appropriate management strategies and predict likely direct and indirect outcomes and tradeoffs from implementation of these strategies.
6. Establish monitoring and evaluation processes, and adaptive management systems to consider all aspects of the biophysical and human dimensions relevant to the situation.

Key to widening the acceptance and integrated use of human dimension data in ecosystem management is convenience, cost, simplicity and the ability to see relationships of social characteristics to biophysical characteristics. Geographic Information System (GIS) technology provides this simple, yet very powerful, capability.

While biophysical data may be captured and displayed at scales and levels of resolution different from those relevant to human dimensional data, differential levels of amount, intensity, quality, recency, etc., of both sets of data may be spatially described, displayed and superimposed. This superimposition provides the manager, scientist and public a visual representation through mapping for seeing system relationships, potential conflicts, and complementary relationships that exist. At some scale, all human dimension data are "mappable". For broad public attitudinal data, influence zone or even market area may be the lowest level of resolution possible. But across population areas within influence zones and market areas, differences in gradient may be distinguished and displayed relative to relevant atmospheric, climatic, soil, water, and biological dimensions.

CURRENTLY AVAILABLE DATA AND METHODS

Following is a brief listing and description of some of the available human dimension data — these examples relate to many of the human dimension data elements in the classification lists.

Recreation Visitor Data

In cooperation with Forest Service Research, National Forest System routinely collects and analyzes data describing the demographics, preferences, attitudes, values and economic impacts of forest recreational visitors. OMB-approved surveys of wilderness users through the Intermountain Station and of CUSTOMER Report Card, and CUSTOMER Comment Card studies led by the Southeastern Station are examples.

Recreation Use Data

The Forest Service and other agencies routinely collect information describing the incidence, amount, type and location of recreational use of forest areas (Recreation Resource Information System — RRIS). These data can be associated with ecosystem delineations.

General Public Information

The National Survey on Recreation and the Environment (NSRE) is underway to collect information on the U.S. public's participation in outdoor recreation, attitudes toward management issues, accessibility issues, wilderness and wildlife. The USDI Survey on Hunting, Fishing and Nonconsumptive Wildlife

Activities is another example. These and similar surveys are scientifically valid data sources and are OMB approved to provide subregional resolution.

Anecdotal Data

Newspapers, demographic magazines, other media and generally published materials provide a wealth of information and data on public attitudes, fads, concerns, and new issues in resource management.

Census and other Secondary Sources

Census of Population, Census of Agriculture, business indicators, labor statistics, opinion polls, and a host of secondary source data help define the social climate within which ecosystem management and appraisal reside.

All of the above data sources are available at low or reasonable costs for developing a human dimension baseline and for monitoring change. Used in conjunction with GIS, these data form a powerful basis for increased understanding of the human dimensions of ecosystem policy, management and use.

THE CURRENT SITUATION

The broad implications of adding the Human Dimension as a full partner in sustainable ecosystem management effort may seem daunting to many people who have been focused on the physical and biological dimensions. Please be reassured; the Agency is not starting from zero. A small but active group of Forest Service research social scientists is exploring the vast amount of social science research information, while also doing original Forest Service related research. In addition, each Regional office has a Social Science Coordinator. The Washington Office Environmental Coordination office has a social scientist on staff, as does the RPA staff. A scattering of social scientists do exist at the Forest and District levels (archaeologists, anthropologists and sociologists). Forest Service social scientists also include more than 75 economists. In addition, many employees in the Recreation, Heritage Resource, Wilderness, Wildlife and Public Affairs staffs are very much involved in the human side of ecosystem management.

FOREST SERVICE RESEARCH ACTIVITIES RELATED TO HUMAN DIMENSIONS

The significance of the relationship between people and the environment to sustainable resource stewardship is recognized in a wide range of recent research related management and policy documents (c.f. National Research Council, 1990; Consortium for International Earth Science Information Network, 1992; Salwasser, MacCleery, and Snellgrove, 1992; Sample, 1991). Having recognized the significance of the human, physical and biological dimensions of ecosystem management the next issues to be addressed are: (1) what does the relationship between people and the environment entail and (2) how can it be incorporated into sustainable resource stewardship. As the agency approaches these issues, the question becomes "what is the human dimension of Ecosystem Management and how can it be integrated into forest management?"

Many Forest Service research projects and scientists are exploring these questions. The following is a review of issues addressed by Forest Service Research relevant to the human dimension of Ecosystem Management.

Recreation and Wilderness

The longest standing areas of research related to the human dimension of resource management are those related to recreation and wilderness uses. Research topics include the benefits of leisure and recreation, visitor profiles, preferences, and satisfaction, impacts on the resource due to visitor use, and trends in visitor use and activities.

Ethnic Diversity and Urban Forestry

Closely related to recreation research, this area focuses on the increasingly urban and diverse publics using and concerned about forest lands. Issues being addressed include the preferences and needs of ethnically and racially diverse recreation customers and values held toward natural resources by urban and/or rural residents.

Rural Development

The 1980's brought attention to the plight of rural America. In an attempt to address issues related to the relationship between the agency and rural communities, Forest Service Research developed a research strategy, "Enhancing Rural America." Research focuses on understanding the impacts of land management decisions on communities, the economic and social impacts of tourism on rural communities, and diversifying rural economies.

Relationships Between People and Natural Resources

This is a focus area for Forest Service Research and is of increasing importance. Since 1973 research at the Rocky Mountain Station has focused on the values and benefits of amenity goods and services. This work has covered both economic and economic measures of value and benefit. That research has contributed significantly to development of several on-line social-science-based management technologies including the Recreation Opportunity Spectrum (ROS), Wilderness Opportunity mapping, and benefits-based management. It has also advanced the state of the art of practice of estimating the economic value of amenity goods and services, including their "existence values." Three recently established efforts that have direct implications for the human dimension of ecosystem management include the creation of two new research work units: the Social and Economic Dimensions of Ecosystem Management unit and the Integrating the Ecological and Social Dimensions of Forest Ecosystem Management unit. Additionally, social research done at the Pacific Northwest Research Station is part of the Consortium for the Social Values of Resource Management. In addition to these efforts, many existing research projects are turning their attention to human dimensions issues.

LEGISLATION REVIEW — HUMAN DIMENSION FOCUS

A summary of selected references to authorities or enabling legislation that currently provides for, requires, and/or recognizes the important role of the Human Dimension in Agency management activities and decisions. Note that explicit references to elements of the human dimension have been in Agency related legislation for a long time.

While the term "human dimension" does not appear as such, it is implicit in the specific references to cultural, social and/or economic well-being; community welfare, people, publics, the public good, etc.

- The 1990 Farm Bill (Title 23, Subtitle G—Rural Revitalization through Forestry) and other laws, regulations, and policy give the agency direction to participate in community-based rural development activities.
- The National Forest-Dependent Rural Communities Economic Diversification Act of 1990 (provided for in the above Farm Bill subtitle) provides the Agency with a special opportunity to help eligible rural communities located in or near National Forests to organize, plan and implement rural development efforts.
- Weeks Law (3/1/11) authorized the Secretary of Agriculture to provide fire protection on State and private lands tiered to watersheds. Also authorized the Secretary to sell or exchange Forest Service lands to States where it was in the benefit of the public interest to do so.
- Twenty-five Percent Fund Act (5/23/08) authorized the Secretary of Agriculture to extract 25% of the timber and other forest products receipts from that area in which the timber came to provide for public schools and roads for those counties in which the Forests are situated.
- Townsite Act (7/31/58) authorized the Secretary to set aside and designate as a townsite up to 640 acres for indigenous communities adjacent to public lands, where community objectives outweigh public objectives and values...
- National Historic Preservation Act (1966)
- National Environmental Policy Act (1969) is the nation's basic charter for protection of the environment. NEPA's main thrust is evident in its preamble: (a) to encourage productive harmony between people and their environment and to (b) prevent or eliminate damage to the environment...and stimulate the health and welfare of people. NEPA requires the interdisciplinary use of the natural and social sciences in Federal planning and decisionmaking which may affect the human environment (Sec. 102(2)(A)). Section 101 authorizes all practical means to foster and promote general welfare...and create conditions...where man and nature can exist in productive harmony...to preserve important historic, cultural, and natural aspects of our national heritage...and maintain an environment that supports diversity and variety of individual choice...An important aspect of NEPA is that it can serve to coordinate consideration of...other environmental statutes...
- Forest and Rangeland Renewable Resources Planning Act of 1974 declared the public interest to be served by the Forest Service. As such, the renewable resource program must be based on a comprehensive assessment of present and anticipated uses, demand for, and supply of the renewable resources from the nation's public and private forests and rangelands...It declared both a responsibility and opportunity to ensure and maintain a conservation posture that will meet requirements of our people in perpetuity.
- The Cooperative Funds and Deposits Act (12/12/75): Gave authority to the Secretary of Agriculture to negotiate and enter into cooperative agreements with...other agencies, organizations, institutions...when [he] determines the public interest will be benefitted and there are mutual interests.
- Federal Land Policy and Management Act of 1976 gave authority to the Secretary of Agriculture to dispose of public lands, through exchange, when determining the public interest would be well served in so doing. Describes multiple use as means of managing lands and resource values so ...as to best meet present and future needs of the American people.

- National Forest Management Act of 1976 amended the 1974 RPA. The NFMA states: "...to serve the national interest, the renewable resource program must be based on comprehensive assessment...through analysis of environmental impacts...the Agency has the responsibility and opportunity to be a leader ...to maintain natural resource conservation posture to meet the requirements of a people in perpetuity.
- Cooperative Forestry Assistance Act of 1978 gave authority to the Secretary for financial, technical, and related assistance to State Foresters or equivalent State officials to further provide technical information, advice and related assistance to private landowners, etc., in management assistance, insect and disease control, rural fire protection...and rural forestry.

CONCLUSIONS - WHERE DO WE GO FROM HERE?

Much needs to be done before the human dimension is a full partner in ecosystem management. A commitment to the importance of the human dimension has been expressed by a number of the Agency's top leadership — we must follow through and meet the commitment. Management questions need to be formulated to assist in identifying what types of information are needed. Appropriate levels of professional staffing and funds will have to be provided. Analysis tools and integrated information inventory processes, including map layers, scale analysis, and GIS, will need to be developed. Finally, research needed in order to develop an understanding of how the human dimension layers relate to the other dimensions of ecosystems. The interdisciplinary team process will focus on identification of tradeoffs and other implications involved in managing ecosystems. When coupled with full public participation, this information will set the stage for the Forest Service to better manage the invaluable resources it is responsible for.

Human dimensions must be an integral part of ecosystem management, - not a footnote.

"People are part of ecosystems and as such humans shape ecosystems and are shaped by them. People value and desire a broad spectrum of benefits (including survival) from ecosystems. To make effective ecosystem management decisions, the Forest Service must have a scientifically sound and integrated understanding of the physical, biological and human dimensions of ecosystems. The human dimension of ecosystem management must include information about people's traditional and changing perceptions, beliefs, attitudes, behaviors, needs, and values, and the past, present and possible future influences of humans on ecosystems."

CREDITS

This paper is based on three creative and interrelated projects to date to define the human dimensions of ecosystem management. These three projects are the Human Dimensions Task Group for the Forest Service's Ecosystem Management Staff, a workshop of all Regional Director's of Recreation Management with the Washington Office Recreation Staff, April 26-29, Washington D.C. and finally a working group in Region 3 of the Forest Service that has recently further clarified the human dimension. The members of the Human Dimensions Task Group are Greg Super, Geraldine Bower, Valerie Chambers, Deborah Carr, Jill Osborn, Bev Driver, Ken Cordell, Gloria Flora, Herb Mittmann, Steve Galliano, Warren Bacon, Cynthia Manning, and Susan Yonts-Shepard. The members of the Region 3 subgroup are Jon Bumstead, Carolyn Holbrook, Bev Driver, and Maria Garcia.

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The Aesthetic Experience of Sustainable Forest Ecosystems

Paul H. Gobster[1]

Abstract — The social acceptability of "Ecosystem Management" and other sustainable forest ecosystem approaches rests in large part on people's aesthetic response to management change. For many, this response is based on a "scenic aesthetic" that is narrowly defined and largely visual in nature. The process of change is often perceived negatively, especially when it involves the death of trees by natural or human-induced causes. The scenic aesthetic remains the culturally dominant mode of appreciation; it is reinforced by research models of landscape perception and by landscape management practices, hindering progress towards other social goals such as biological diversity and ecosystem health. In contrast, an "ecological aesthetic" as espoused by Aldo Leopold and others requires a learned experience of the multimodal, dynamic qualities of forest environments, and appreciates both subtle and dramatic changes exhibited in the cycles of life and death. As such, adoption of an ecological aesthetic could help resolve perceived conflicts among social goals. Suggestions for planning, management, research and theory gleaned from an ecological aesthetic show how we might achieve sustainable forest ecosystems that are understood and appreciated by our public.

Our first and most immediate response to the environment is often an aesthetic one (Kaplan 1987). Although the adage that you "can't judge a book by looking at its cover" bears a great deal of truth, our evaluation of a place frequently depends on what we see from an aesthetic point of view. In the forest landscape, this implies that the appearance of the environment reflects the quality and care that goes into its management, and treatments that conflict with our aesthetic preferences may be construed as signs of poor management (Hull 1988, Nassauer, 1988).

In this paper I argue that current approaches to forest visual management practice and research are inadequate for dealing with aesthetic issues in the context of sustainable ecosystem management. Since their inception some thirty years ago, these approaches have tended to emphasize the formal, visual, and static characteristics of landscape scenery, a response to the dominant cultural mode of landscape perception and experience. However, forest ecosystems managed for sustainable values may exhibit few characteristics one typically thinks of as scenic, and thus under the current paradigm of visual management practice and research we have few guidelines for resolving aesthetic and biological/ecological goal conflicts. The primacy that aesthetics plays in people's evaluation of environments suggests that in order for sustainable ecosystem management to be fully accepted, a better understanding is needed of how aesthetic and biological/ecological outcomes are perceived and interact. In this paper I attempt to show how aesthetic appreciation of sustainable forest ecosystems requires an expansion of our understanding of what beauty in the landscape can mean and provide to people, and how a redefined program of landscape practice and research can be instrumental in discovering this beauty and communicating it to the public. I conclude with some practical ideas for bringing this sustainable "ecological aesthetic" into the vocabulary of landscape practitioners and researchers, to provide a better integration of aesthetic and biological/ecological goals in the management of forest ecosystems.

"Sustainability" has a variety of meanings when applied in the context of forest ecosystems (Gale and Cordray 1991). My use of the term sustainability focuses on the ecological aspects of sustainability, specifically on management approaches and practices that aim to restore and maintain the ecological structure and function of ecosystems, and preserve and enhance the health

[1] *Research Social Scientist, USDA Forest Service, North Central Forest Experiment Station, Chicago, IL.*

and diversity of species and ecological communities. Examples of management approaches include "New Forestry" and "Ecosystem Management" (e.g., Franklin 1989, Robertson 1992), where forest and related wildland ecosystems are managed for multiple resource values, including commodity values such as timber; and "ecological restoration" (Jordan et al. 1987), where commodity values usually do not enter the picture. Examples of management practices implemented to achieve desired future conditions include direct manipulation through cutting, burning, and other intentional activities, and indirect management that permits or encourages natural processes and disturbances like fire, timber falls, and diseases to accomplish sustainable management goals.

FOREST AESTHETICS IN CULTURE, MANAGEMENT, AND RESEARCH

People's aesthetic preferences arise from a number of different sources, but of these, our dominant culture has played the major role in shaping our aesthetic preferences for landscapes (Rees 1975). I feel three cultural legacies are particularly important in understanding our current preferences for forest landscapes and forest landscape management: these include an attraction to an idealized nature; an orientation to a static, visual mode of landscape experience; and an aversion to disruption and change. Together, these legacies are responsible for what I will call the "scenic aesthetic" mode of landscape appreciation. The preponderance of the scenic aesthetic in our society makes it difficult for many people to appreciate the more subtle, experiential, and dynamic qualities that often characterize sustainable forest ecosystems, qualities that relate to a much different, "ecological aesthetic." In the following sections I will discuss how these two aesthetics differ, and describe how the emphasis of contemporary landscape research and practice on the scenic aesthetic could affect our ability to implement sustainable ecosystem practices that are aesthetically acceptable.

Idealized Nature

Empirical studies of people's landscape perceptions have identified important attributes of wildland forests that contribute to visual quality. Research findings indicate a high visual preference for "near-view" forest stands with large trees (Arthur 1977, Buhyoff et al. 1986), an herbaceous ground cover (Patey and Evans 1979, Brown and Daniel 1984), and an open mid-story with high visual penetration that affords a park-like character (Brown and Daniel 1984, Ruddell et al. 1989). Additionally, many "vista" forest landscapes of aesthetic appeal exhibit distant views and high topographic relief (Propst and Buhyoff 1980, Buhyoff et al. 1982, Gobster and Chenoweth 1989).

The kinds of forests characterized by these attributes conform very closely to those popularized by landscape painters of the 17th and 18th century, who together with writers, landscape designers, and gentlemen travelers brought about an aesthetic appreciation for landscapes that were natural in character (Huth 1972, Nash 1973, Cox 1985). In the U.S., painters of forest landscapes emphasized the dramatic panoramas of the mountainous west and the lush, tidy, pastoral views of the east. Their idea of natural beauty was a highly selective one, defined by a rigid set of criteria. Landscape painters often stylized nature, and composed a scene by adapting formal design principles to enhance the beauty of the nature they saw (Clark 1949).

This natural, scenic ideal is evident in current landscape management and research programs whose objectives are to assess aesthetic quality. "Visual resource management" programs such as the Forest Service's "Visual Management System" (1974) were developed to deal comprehensively with aesthetic issues on our public forest lands. Like the methods of the landscape painters and designers, aesthetic quality is defined in part on the basis of formal principles of variety in line, form, color, and texture. Landscapes that exhibit these features are evaluated as "distinctive" and given high levels of protection, while landscapes that are "common" or "minimal" in their variety are allowed to be more intensively used for timber harvesting or other purposes. While relatively few empirical studies have used formal or "artistic" design attributes to identify landscape preferences (Gobster and Chenoweth 1989), the focus on evaluation and comparison is a trait common to most studies, and methods are often geared to finding the "most scenic" landscapes for protecting.

Our current orientation to an idealized nature has made it difficult to merge aesthetic objectives with those relating to the management of sustainable forest ecosystems. Many sustainable forest ecosystems do not display the formal, compositional properties of an idealized nature. In my own region of the Lake States, many of the areas identified as valuable for the protection of endangered species or ecosystems are unspectacular areas of flat, interior forest, prairie, marsh, bog, and barren land. Such areas frequently merit visual variety ratings of "common" or "minimal," and are thus prone to be discounted as unworthy of aesthetic consideration. By the same token, management practices enacted to maintain or enhance ecological function such as prescribed burning or the retention of downed wood often detract from the tidy, stylized naturalism of the scenic ideal. By maintaining a limited standard of aesthetic value such as "visual" or "scenic" quality, we are negating many of the attributes of biologically diverse ecosystems that, in product or process, can contribute to a richer, multidimensional understanding of the aesthetics of nature.

Aldo Leopold was one of the first to point out a different type of aesthetic in the natural landscape, one that did not conform to the canons of idealized beauty expressed by popular scenery. The elements of this "ecological aesthetic" were referred to implicitly in many of Leopold's essays, and have more recently been brought to light in the writings of Callicott and his colleagues (e.g., Callicott 1992). With respect to the formal qualities of an idealized nature, Flader and Callicott (1991) conclude: "For [Leopold], the esthetic appeal of the country... [had] little to do with its adventitious colors and shapes— and nothing at all to do with its scenic and picturesque colors— but everything to do with the integrity of its evolutionary and ecological processes" (p. 9-10). Leopold thus expands our concept of natural beauty to encompass a wide range of sustainable forest ecosystems. By appreciating forest ecosystems for their biological diversity and health, we redefine the goal of much which is done in landscape research and management from identifying and protecting the *most* beautiful forest sites, to discovering and interpreting the varieties of ecological beauty present in *each* forest site (Evernden 1985).

Landscape Experience

The scenic aesthetic is driven by a second factor that is highly related to our culture, that is the way in which we experience the beauty of nature. Terms and activities from the romantic period reveal our cultural biases of landscape appreciation. "Scenic" and "picturesque" beauty were coined to refer to landscapes that exhibited the desired formal aesthetic qualities found in landscape paintings, and use of the term "landscape" took on an artistic meaning, as a view seen from a specific perspective (Rees 1975). The aesthetic experience of landscapes thus came to be associated with the view of a static composition, and such an activity was referred to as "sightseeing" or "picturesque touring" (Adler 1989). Carlson (1979) describes the birth of the "landscape cult" in the 18th and 19th centuries, who developed sightseeing as a leisure pastime, and even went so far as to use a device called the Claude Glass, through which a landscape could be viewed in "proper" framing, color, and perspective.

Scenic touring continues to be the way most people experience the forest landscape. "Driving for pleasure" and "sightseeing" are among the top activities of visitors to public park and forest areas (Cordell et al. 1990), and while the Claude Glass is no longer in use, the forest landscape is similarly framed and experienced through the windshield of the automobile, the viewfinder of the camera, and the designated scenic overlook. Like a painting, our experience of a landscape is often limited to a view at one point in time, and with most of us living in urban areas, we rarely get to experience how forest landscapes change on a daily, seasonal, or yearly basis.

Because of the high popularity of sightseeing by car, forest landscape management often focuses on visual enhancement and mitigation activities along road corridors. This is evidenced in the National Forest's "Scenic Byways" program (Robertson 1988), and in visual management criteria for identifying areas of high "visual sensitivity" (USDA Forest Service 1974). These areas receive priority treatment as visual retention zones, and are managed to provide a naturally appearing forest condition with few signs of disruption. Viewshed management strategies vary by region, but in areas with flat topography this might only mean a narrow "peek-a-boo strip" of uncut trees to mask forest harvesting activity to the unaware motorist zooming by (Wood 1988).

Most research in landscape aesthetics has done little to explore varieties of aesthetic experience beyond the scenic. Methods to assess aesthetic perceptions often consider only "visual quality" or "scenic beauty" as dependent variables, which are operationalized by a simple check mark on a rating scale (Daniel and Boster 1976). A photograph or slide substitutes for the actual landscape in most studies, which, like the landscape painting or Claude Glass, is a flat, framed snapshot of a single point in time. The entire assessment process usually requires only a few seconds for a person to view and rate each landscape scene, further reducing the "landscape experience" to a momentary judgment.

Because many sustainable forest ecosystems lack the formal qualities of an idealized nature, it is difficult to experience them as one would a landscape painting. There may be no prospect, no "view" in the traditional sense, and thus nothing "scenic" to behold (Evernden 1983). In Leopold's ecological aesthetic, however, there is much to experience in these non-picturesque landscapes. Aesthetic qualities engage all of the senses, not just sight. Ecosystems are appreciated for their detail and for the "big picture," but ecological beauty is most highly manifested in the interplay across scales, such as in the existence of wild species sustained by a community that retains a high degree of ecological integrity. Aesthetic experience for Leopold is a cumulative process that begins in the mind of the individual and is refined and nurtured through an intimate relationship with the environment over time. Most importantly, Leopold's landscape experience ties aesthetics together with ethics, giving moral justification to the pursuit and protection of beauty in the environment: "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise (Leopold 1981: 224-225). This changes our idea of the aesthetics of nature from one of viewing scenery as a passive form of entertainment to one that engages our hearts and minds towards participatory action.

Disruption and Change

A third factor that comprises our scenic aesthetic has not received much direct attention by landscape writers or researchers, yet is pervasive in our culture. This is our aversion to change in the forest landscape, especially change that signals a disruption in our static ideal of beauty. This aversion is evident in our empirical studies which show radical changes in scenic beauty "flows" over the course of a growing cycle (Hull and Buhyoff 1986, Palmer 1990, Ribe 1991), and negative responses to standing dead trees (Benson and Ullrich 1981, Ribe 1990), downed wood (Vodak et al. 1985, Brown and Daniel 1986; Ribe 1991), insect and disease outbreaks (Buhyoff and Lueschner 1978, Hollenhorst et al. 1993), and fire (Anderson et al. 1982, Taylor and Daniel 1984). These findings in many ways conflict with what ecologists are telling us about the beneficial effects that some kinds of disruptions— both natural and human induced —can bring to forest ecosystems (e.g., Ostry and Nicholls 1992, Maser et al. 1979, Hunter 1990, Baker 1992).

In one sense these research findings are simple expressions of our preference for tidiness, mature trees, and a lush forest understory, but in another sense they may reflect deeper cultural aversions to change. One of the most deep seated of these is change that brings on disease and death. Human diseases have long been associated with evil, and the metaphors we use to describe diseases, those afflicted, and actions towards treatment are often equated with warfare against evil (Sontag 1978). Fear of death is universal to most societies, and even though many religions believe in an afterlife, our concept of death has a strong sense of permanency (Becker 1984). Whether or not these fundamental attitudes towards human disease and death are linked to how we think about forest diseases and the death of trees, it is clear that they exhibit many of the same characteristics. For example, our culture has tended to view forest insect outbreaks in a mostly negative way, and has used warfare-like tactics (including propaganda campaigns) to try to wipe them out, however unsuccessfully (Carson 1962). We have long viewed healthy trees as a sign of a healthy forest (Ostry and Nicholls 1992), and as with human diseases, have tended to focus on treatment of the symptoms of forest diseases rather that trying to understand the causes. And because our ideal image is of a mature forest frozen in time, the death of trees by natural forces or human intention may convey a permanent end rather than a point in a cycle (American Forest Council 1991).

Following the popular scenic aesthetic, many of our current landscape management guidelines attempt to reduce the noticeability of change. Regeneration sites are located away from areas of human use, are screened, or are designed to blend in with the forms and lines found in the natural landscape (Bacon and Twombly 1980). While the presence of wildlife adds considerable scenic interest to the landscape (Hull and McCarthy 1988), dead "snag" trees and slash piles created for wildlife food and cover are also located so that they minimize disruption to the visual scene (USDA Forest Service no date). Prescribed fire can also improve the beauty and diversity of the understory, but the immediate negative visual effects are mitigated by leaving unburned islands, limiting the amount of road frontage that is treated, and restricting burns to periods of low visitor use (Bacon and Dell 1985). The "illusions" created by these techniques further the idea that a natural forest is one that is mature, tidy, and unchanging (Wood 1988).

Landscape research has tended to focus on people's aesthetic response to discrete changes in the landscape rather than to understand how the dynamics of change are perceived or experienced. For example, the visual effects of a prescribed burn or the slash left from a timber harvest can be highly negative, but in many environments their duration is quite brief (Bacon and Dell 1985). Although such studies are consistent with the fact that most people see the forest at only snapshots in time, it is possible that methods that explore the experience of landscapes might arrive at a much different understanding of the aesthetic values of change.

As a keen observer of nature, Leopold's ecological appreciation was closely tied to the dynamics of change. In his essays in *Sand County Almanac*, Leopold (1981) gives many examples of how ordinary places take on aesthetic significance through the experience of change. In "Prairie Birthday," Leopold's daily observations of prairie flora growing around a "backward farm" make clear how the beauty of a diverse environment unveils itself through the course of a growing season. In "A Mighty Fortress," he conveys how tree insects and disease transformed his "ailing" woodlot into a wildlife haven. And in what is perhaps his strongest appeal to an ecological aesthetic, in a "Conservation Esthetic," Leopold states that the perception of beauty and quality comes with an understanding of the natural processes through which ecosystems evolve and are maintained.

ADOPTING AN ECOLOGICAL AESTHETIC IN THE MANAGEMENT OF SUSTAINABLE FOREST ECOSYSTEMS

This review has shown that our orientation to the scenic aesthetic is strongly grounded in our culture. In forest ecosystems this aesthetic is reinforced by the places we designate for recreation, and by the methods through which we manage forests for aesthetic enjoyment. Research knowledge of people's aesthetic preferences for forests accumulated over the past three decades is formidable, but it, too, is limited in its methods and scope of inquiry, and tends to "mop up" questions relating to our understanding and application of the scenic aesthetic rather than expand aesthetic theory through discovery and expression of alternative paradigms.

Leopold's ecological aesthetic provides such an alternative, one that offers promise in merging the goals of aesthetic preferences and ecological sustainability. In a recent paper that discussed some related aspects of scenic and ecological aesthetics (Gobster in press), I argued that because the scenic aesthetic was so deeply entrenched in our culture it would be difficult to get people to "see" beauty in an ecological sense. To sidestep the wait for such a cultural evolution to occur, I outlined a synthetic approach called "appropriateness analysis" that might help to resolve immediate conflicts between managing forests for aesthetic and biodiversity goals. At the same time, I suggested that trade-off analyses such as appropriateness analysis were at best a short-term fix, and that in order to move towards a more sustainable forest landscape that is also socially acceptable, landscape practitioners and researchers needed to develop the ideas and tools to begin to understand and implement an ecological aesthetic here and now.

In the second part of this paper I wish to flesh out some ideas of how an ecological aesthetic might be realized. Many of these ideas have been stated previously by others, but by bringing them together under the focus of sustainable ecosystems I hope they might provide the reader with some practical insights on how we might incorporate ecological thinking into planning and program development, on-the-ground management, and research and theory development in landscape aesthetics.

Some Ideas for Planning and Program Development

Continue to move "visual management" towards an ecological approach

The Forest Service's Visual Management System (USDA Forest Service 1974) and related policies and programs have gone far to bring visual quality issues into the forest planning process. In many cases, however, landscape management for visual quality has been reduced to one of mitigating the visual impacts of timber harvesting and other resource development activities that do not conform to public expectations of a "naturally appearing forest." Sustainable forest ecosystem management offers new opportunities to help redefine public expectations of naturalness, and landscape management programs should recognize the need for an "expanded" aesthetic that incorporates principles of ecological sustainability explicitly into methods and practices. Revisions of the Visual Management System handbook are currently underway, and show a sensitivity to ecological management concerns (Galliano et al. 1992). As resource-specific handbooks and training programs are updated, these, too, should reflect a broader, ecological aesthetic in landscape management principles and practices. One recent example that moves in this direction is a regionally-produced publication that uses landscape ecology as a basis for forest landscape analysis and design (Diaz and Apostol 1992).

Expand the concept of "scenic byways" programs

Scenic roads programs have been developed by the Forest Service and other agencies in recent years to showcase outstanding natural and cultural scenery available to the automobile traveler (USDOT Federal Highway Administration 1988). While the emphasis is currently on the scenic aesthetic as described in this paper, programs like the Forest Service's Scenic Byways program (Robertson 1988) offer significant opportunities to interpret the ecological aesthetic of sustainable forest ecosystem management to mass audiences. Interpretive signs, roadside pullouts, short-loop trails, and other suggested byway developments could be used to bring people out from behind the windshields of their cars and towards a deeper understanding and appreciation of sustainable forest ecosystems.

Incorporate contextual considerations into ecological management

Perceptions of the appropriateness of management activities are dependent on the context or setting in which management change is to occur. Tlusty (1992) provides a conceptual model for understanding how different aesthetic management criteria might be applied to different types of forest settings. He identifies three major setting types: "Functional" settings like wilderness areas or plantations, where management emphasis is not on aesthetics, and aesthetic management is thus "hands off" or mitigative in nature; "recreational" settings where emphasis is on the visual, scenic landscape and aesthetic management activities conform to people's notions of idealized nature; and "ecosystem" settings, where emphasis is on sustaining the structure and function of the ecosystem and aesthetic management aims toward principles conveyed by Leopold's ecological aesthetic. The "Recreation Opportunity Spectrum" (USDA Forest Service 1986) offers a somewhat different concept of settings, but could also be used to understand and plan for ecologically sustainable forest management activities that are appropriate in scale, duration, and other considerations as one moves across the wilderness-to-urban spectrum (Gobster in press).

Some Ideas for On-the-Ground Management

Show a "conspicuous experiential quality"

Visual mitigation practices such as screening, edge shaping, or location are commonly used to reduce the impacts of resource activities that might not meet people's expectations for a naturally appearing forest environment. Should sustainable ecosystem management practices that violate this same scenic ideal be similarly mitigated? Thayer (1989) argues not, maintaining that the "visibility and imageability of the sustainable landscape is critical to its experiential impact and the rate at which it will be adopted and emulated in common use" (p. 108). This implies that in order for an ecological aesthetic to become understood and appreciated by the public, it must be seen and experienced. This "conspicuous experiential quality" will help speed the diffusion of change in aesthetic expectations (Thayer 1989).

Use design cues to "reveal" ecological beauty

Nassauer's research on landscape restoration in agricultural (1992) and suburban (in press) settings suggests that design cues can convey powerful messages that "messy" ecological practices show human care and stewardship rather than neglect or mistreatment. In other words, "conspicuous experiential quality" need not be of the "in your face" variety, and design cues can help reveal and express the intentions behind sustainable management practices. In settings where recreational use dominates, these cues might include some picturesque conventions like framing or the use of texture, height, and color contrast to call attention to sustainable land use practices. These practices might be small in scale and of limited duration, but would be visible to the recreationist, perhaps along a winding, well-maintained nature trail. Selective cutting, and even some planting of immediate foreground views with native but showy under-, mid-, and overstory species might be done to enhance the visual, scenic effects. In forest areas away from concentrated recreational use, picturesque conventions might by replaced by less stylistic cues like mowing or low-key fencing that still convey human intent and land stewardship. In backcountry areas, cues might be subtle or missing altogether— perhaps unobtrusive marker posts in representative areas, keyed to a brochure available off-site. For these sites, care is exhibited by ecological integrity and largely up to forest users to discover it.

Use information to interpret sustainable forest ecosystem management practices

For Leopold, knowledge was an important precursor to the comprehension of ecological beauty. Information can be an important tool in conveying knowledge about the intent and purpose behind sustainable management practices, especially for some management activities like burning, where it is difficult to employ design cues to improve public acceptability. On-site information such as signage, interpretive nature trails, stewardship programs, and the like can aid in communicating messages to the public. Newsletters or brochures put out by many forests and restoration groups are useful off-site ways to spread the word, as are local newspapers. It is critical, however, that this information be expressed with sincerity and in an objective manner to avoid suspicion that managers are trying to "fool the public" (Wood 1988).

Involve the public to gain a deeper understanding and experience of "ecological" beauty

Experience is the essential counterpart to information for attaining knowledge and appreciation of sustainable ecosystems. Experience can be facilitated through the design of self-guided nature tours; by the encouragement of nature-oriented recreation like birding, plant identification, hunting, and nature photography; and by providing other forms of unassisted nature experience opportunities. Directed activities are particularly valuable ways through which forest users can gain experience and appreciation of natural systems and processes maintained through sustainable management practices. Guided tours are one important way to reach large audiences, and have shown potential in communicating the benefits of "new forestry" practices (Brunson 1992). Public participation in ecosystem management activities is less easy to accomplish on a large scale, but can be extremely effective on a project basis. Student internships and summer youth camps sponsored by national forests or scouting groups are two types of existing programs that could be geared towards ecological management. Ecological restoration programs have made effective use of volunteers, who often dedicate much of their free time in activities like cutting, burning, seed collecting, and planting. In the Chicago area where I live, the Nature Conservancy's Volunteer Stewards Network has grown to over 5,000 members in the past 15 years, and for many in the network, restoration has become a leisure activity that has deep aesthetic, symbolic, and spiritual implications (Lonsdorf 1993).

Some Ideas for Research and Theory Development

Investigate the attributes of sustainable forest ecosystems that relate to aesthetic quality

Much of the past research on people's perception of landscapes has been directed towards identifying "universal" predictors of landscape quality (e.g., Wohlwill 1976), and looking at the visual impacts of different resource-oriented management practices (e.g., Ribe 1988). With new forestry, ecosystem management, ecological restoration, and other sustainable ecosystem approaches attaining wider application, forest landscape perception research has the opportunity to expand this orientation in some significant ways. One of these is in the types of management practices that are studied— we need more in-depth studies that look specifically at sustainable ecosystem management practices such as prescribed burning, snag trees left for wildlife, and cutting patterns that minimize forest fragmentation. Along with basic management practices, we also need more information on people's aesthetic responses to the structure and function of forest ecosystems. In this respect, a key need is for information about the perception of change, especially on how the dynamics of change are perceived and experienced. Finally and most importantly for developing ecological aesthetics, we need information about the unique qualities that make each forest type and ecosystem aesthetically significant. Evernden's (1983) criticism of visual management's emphasis on "thingness" in relation to the aesthetic qualities of prairie ecosystems vividly illustrates the importance of studying natural landscapes on their own terms.

Investigate the people's aesthetic experiences of sustainable forest ecosystems

The search for landscape attributes, whether general or unique to a given ecosystem, provides only part of the knowledge needed to expand our understanding of an ecological aesthetic. The other side of the equation where more information is needed is on the nature of the aesthetic response itself. Because most studies have operationalized people's aesthetic responses to landscapes as "visual quality" or "scenic beauty" ratings of photographic environmental surrogates, we know very little about how real places are experienced (Hull and Stewart, 1992), or about the wider nature of aesthetic responses. Zube et al's. (1982) framework for landscape perception research laid out a rich source of ideas for understanding the aesthetic experience of landscapes, a set of ideas which take on a heightened sense of significance in the context of an ecological aesthetic. Using the terms of their framework, the aesthetic of the surrounding, multimodal, information-rich environment of sustainable forest ecosystems is one that is appreciated by movement and exploration rather than by gazing at a view, is perceived and experienced through multiple senses simultaneously, is interpreted through affective, cognitive, and symbolic meanings, and invites participation and meaningful interaction. Each aspect of the Zube et al. (1982) framework deserves renewed attention as aesthetic perception studies of sustainable ecosystems increase in the years ahead.

Colleague Rick Chenoweth and I have begun to look at aesthetic experiences in the landscape in a somewhat different way, not only examining the attributes of the landscape and the characteristics of the experience, but also trying to understand the "ecology" of experiences in time and space and in relation to social and situational conditions, and the value aesthetic experiences play in people's lives (Chenoweth and Gobster 1990, Gobster and Chenoweth 1990). While not addressing sustainable ecosystems in particular, we think our work holds significant potential for understanding how aesthetic experiences in non-spectacular ecosystems like prairies might differ or are valued in comparison with experiences in more "traditionally" scenic areas. For example, many people's aesthetic experiences from our work showed the importance of environmental knowledge and of the dynamic, ephemeral characteristics of the environment—some of the same qualities exhibited in Leopold's ecological aesthetic.

Expand the repertoire of methods

Investigations of some of the ecosystem- and experience-related phenomenon mentioned above will require new and innovative methods. "Experiential approaches" to landscape assessment include a wide range of qualitative and quantitative methods (Gobster 1990), and hold significant promise for understanding how sustainable ecosystems are perceived and experienced. For example, our study of aesthetic experiences had study participants carry small diaries with them to write down quantitative and qualitative information about their aesthetic experiences (Chenoweth and Gobster 1990). Hull et al. (1992) similarly used diaries and beepers to obtain quantitative ratings of people's moods, satisfaction, and perceptions of scenic beauty in order to understand the dynamics of recreation experiences. Focusing, a technique from experiential psychotherapy, is a qualitative method that has been suggested as a means to understand the non-verbal "felt senses" that result from aesthetic experience of the environment (Schroeder 1990). Other qualitative approaches such as first-hand aesthetic description (Berleant 1992), literary analysis (Porteous 1986), and observation (Seamon and Nordin 1980) have been described as phenomenological alternatives to positivist methods of analysis (Seamon 1982), and hold particular promise in exploring the aesthetic experience of sustainable ecosystems.

Build ecological aesthetics into landscape perception theory

Finally, research on the perception of sustainable forest ecosystems has the potential to advance our theoretical understanding of environmental aesthetics. Carlson's (1993) review of landscape assessment concludes that most of the theoretical development has focused on models and theories that attempt to identify and put into an organizational framework the basic person- and landscape- related variables that help to explain landscape preferences. While this work has been valuable in explaining which landscapes are preferred, current theory in landscape assessment does little to *justify* why they are preferable. For example, just because the complete removal of slash and downed wood may make a forest stand more scenically preferred, it does not justify this preference. As Carlson (1993) concludes, justificatory theory is needed in this field because: "We need not only to be able to explain what is preferred and desired by way of landscapes, but also to establish what is preferable and desirable. Only by references to the preferable and the desirable do we have the ultimate grounding for landscape evaluation and for the more practical matters of landscape planning and design" (p. 55).

In this sense, Leopold's land ethic provides an important foundation to a justificatory theory of landscape aesthetics. By uniting beauty with ecological stability and integrity, Leopold's land ethic provides a normative justification for preferable and desirable landscape management that enhances the sustainability of forest ecosystems for human, biological, and ecological values.

CONCLUSION

Evidence of evolving land management approaches in urban, agricultural, and wildland environments shows that the concept of ecosystem sustainability is becoming accepted at least on some basic level by many professional and lay persons. But for most, this acceptance has been largely because of an intellectual understanding, and not because the products or processes of sustainable landscape management are inherently preferred. Our cultural ties to the scenic aesthetic run deep, and because of the primacy of aesthetics in environmental perception, a greater commitment toward the adoption of innovative methods for ecological sustainability has not been forthcoming.

Leopold's ecological aesthetic offers guidance for merging biological and ecological concepts of sustainability with aesthetic appreciation. Experience is a key component of this aesthetic, where both intellectual and affective capacities engage an individual to understand, appreciate, and ultimately act upon the environment in a purposeful way. This last point is a crucial one for greater adoption and acceptance of sustainable forest ecosystem management, and suggests that approaches that foster experiential contact with natural systems and processes can lead to positive behaviors to protect them. The ideas discussed in this paper provide some first steps for how we can help to advance this evolution of change, not only among our public groups but also in our own institutional cultures of landscape management and research.

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Declining Southwestern Aquatic Habitats and Fishes: Are They Sustainable?

John N. Rinne[1]

Abstract — Aquatic habitats and native fishes in the arid American Southwest have declined drastically in the past half century. Physically, changes are attributable to dams and diversions, channelization, and drying of streams. Biologically, the extensive introduction of non-native species of fishes has negatively impacted the native fish fauna through hybridization, competition, and predation. Sustaining both the aquatic and native fish resources will require a combination of securing habitats, changing both species and land management strategies, and continued research on the biology and habitat requirements of native species.

INTRODUCTION

Historically, riparian-stream areas of the southwestern United States supported aquatic habitats critical for survival of native fishes. In the past half to three quarters of a century, these vital resource areas and the native fishes they supported have declined markedly. Extensive aquatic habitats have been modified or lost through construction of major and minor dams, water diversion, channelization, and groundwater mining. Superimposed upon habitat modification have been biological alterations, principally the introduction of non-native fishes. Although the extent of these biological and hydrological alterations has slowed dramatically, in part because of absence of opportunity, their legacy persists.

Extensive research and management effort has been expended over the past quarter century to curb this dramatic loss of habitat and fishes and to reverse its trend. The objective of this paper is to present an overview of both past and ongoing conservation efforts designed to sustain aquatic habitats and native fishes in the Southwest. I will discuss and summarize, through both the literature and case history examples for native fishes principally in Arizona: 1) the nature of the decline of aquatic resources in the region, 2) suggest necessary actions to halt and perhaps reverse this decline, and finally, 3) attempt to answer the question: Are aquatic habitats and fishes sustainable in the American Southwest?

[1] *John N. Rinne is a fisheries research biologist and National Fisheries Research Coordinator with the Rocky Mountain Forest and Range Experiment Station, USDA Forest Service, Flagstaff, Arizona.*

THE FISHES

The native fish fauna in streams in Arizona (Minckley 1973) and the arid American Southwest (Miller 1961, Rinne and Minckley 1991) is depauperate compared to that of drainages further east. Since the late 1800s, only about 30 taxa of fishes have been recorded in the waters of Arizona; several are now extinct (Minckley 1973, Miller et al. 1989, Rinne 1990a). By comparison, that many species may inhabit a single creek in eastern streams. Endemism is high in fishes in the Southwest, and specialization of forms is the rule. In the Gila River, which drains the major portion of Arizona, almost a third (5 of 17 species) of the native species belong to monotypic genera (Miller 1961, Minckley 1973).

The progressive depletion of native fishes resulting from introductions of non-native fishes has endangered many native fish over the entire United States (Kirch 1977, Deacon 1979, Deacon et al. 1979, Williams et al. 1989). Many species of fishes new to the West and Southwest were introduced largely for recreational or sport fishing (table 1). Since the turn of the century, the fish fauna of Arizona has almost tripled through widespread intentional and accidental introduction of non-native fishes (Rinne 1990b). non-native fish introductions were primarily for sport, bait, or as biological control agents (Miller and Alcorn 1946, Miller 1952, Deacon et al. 1964).

Specific cases abound in the literature that report replacement of native fish species by those introduced, apparently through biological competition (Deacon et al. 1964, Minckley 1973, Deacon and Minckley 1974, Rinne et al. 1981, Courtenay and Meffe 1989), hybridization (Rinne 1988, Rinne and Minckley 1985, Rinne et al. 1985) or direct predation (Schoenherr 1981, Meffe et al. 1983, Minckley 1983, Meffe 1985, Blinn et al 1993).

As a result, 60% of the native fishes in the Southwest are listed by federal and state agencies as threatened, endangered, or of special concern (Johnson and Rinne 1982, Williams et al. 1989, Rinne, 1990b).

THE HABITATS

In the early 1900s, humans moved westward and began settling in the arid American Southwest. Major (fig. 1) and minor dams, diversions of water for agriculture, and pumping of groundwater were necessary for man to survive. Water development projects vastly altered the hydrologic regimes of the area, and imposed drastic additional demands upon the naturally changeable rivers and streams and their fishes (fig. 2). These added impacts when combined with the naturally occurring changes of flood and drought, often exceeded the limits of adaptation of many species. The combination of rapid quantitative and qualitative changes of hydrologic conditions, man's continued alteration of aquatic habitats and natural hydrologic cycles, and the introduction of fishes foreign to these waters caused a dramatic decline in the native fish fauna; some species became extinct (Miller et al 1989, Minckley 1973). At present, the remaining fish fauna of the desert Southwest is endangered. Conservation of this valuable natural resource is and will continue to be a strong challenge to fish managers and biologists of the region.

Table 1. — Introduced recreational or sport fishes of Arizona and the lower Colorado River, 1900-1971. Those species currently established are denoted by an asterisk.

COMMON NAME	SCIENTIFIC NAME	YEAR INTRODUCED
White sturgeon	Acipenser transmotanus	1967
Coho salmon	Oncorhynchus kisutch	1967
Sockeye salmon	Oncorhynchus nerka	1957
Cutthroat trout*	Oncorhynchus clarki	ca. 1900
Rainbow trout*	Oncorhynchus mykiss	ca. 1900
Golden trout	Oncorhynchus aguabonita	1971
Brown trout*	Salmo trutta	1924
Brook trout*	Salvelinus fontinalis	1920
Grayling*	Thymallus articus	1943
Northern pike*	Esox lucius	1967
Muskellunge	Esox masquinongy	1970s
Striped bass*	Morone saxatilis	1969
White bass*	Morone chrysops	1960
Yellow bass*	Morone mississippiensis	1929-32
Smallmouth bass*	Micropterus dolomieui	1942
Largemouth bass*	Micropterus salmoides	1935
Rock bass	Ambloplites ruprestris	1960
Warmouth*	Lepomis gulosus	1950s
Redear*	Lepomis microlophus	1947
Green sunfish*	Lepomis cynellus	before 1926
Pumpkinseed*	Lepomis gibbosus	1950
White crappie*	Pomoxis annularis	before 1924
Black crappie*	Pomoxis nigromaculatus	1930s
Yellow Perch*	Perca flavescens	1930s
Walleye*	Stizostedian vitreum	1960s
Sargo	Anisotremus davidsoni	1960s
Bairdella	Bairdella icistia	1960s
Orangemouth corvina	Cynoscion xanthulus	1960s
Mozambique tilapia*	Tilapia mossambica	1960s

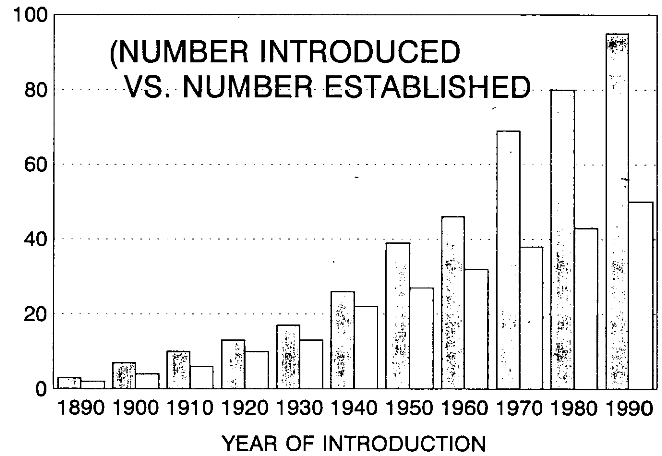

Figure 1. — Chronological history of non-native fishes introduced and ultimately established in the waters of Arizona, 1890-1990.

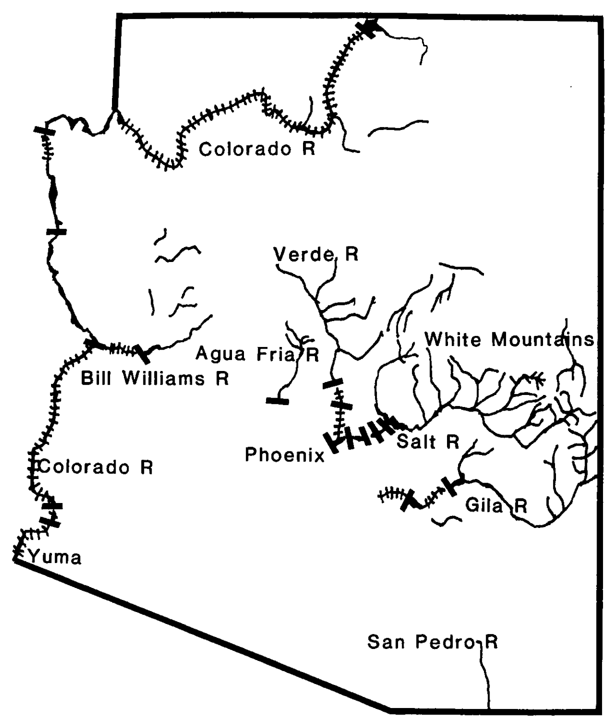

Figure 2. — State of Arizona, indicating major dams (solid bars) and stream habitat modified (cross hatching) by these dams.

HYDROLOGIC IMPACTS

Salt River

The hydrology of the Southwest has been altered dramatically. In Arizona, 80% of mainstream river habitats have either been physically and chemically altered or completely lost through drying (fig. 3). The first U. S. Bureau of Reclamation dam, Roosevelt, was completed on the Salt River in 1911. This dam and a series of 3 more (Stuart Mountain, Morman Flat, and Horse Mesa) within the next 25 years effectively dried the Salt River at Tempe. Horseshoe and Bartlett reservoirs soon followed on the Verde River, a tributary of the Salt. In the late 1800s more than a dozen native fishes swam in the waters of the Salt River at the Tempe Bridge (Deacon and Minckley 1968). Presently, only an assemblage of introduced species such as carp, catfish, sunfishes, and cyprinids from the bait industry can be collected in the few remaining artificial aquatic habitats created by gravel mining, developments, and sewage effluent in the reach of the Salt in the greater Phoenix metropolitan area.

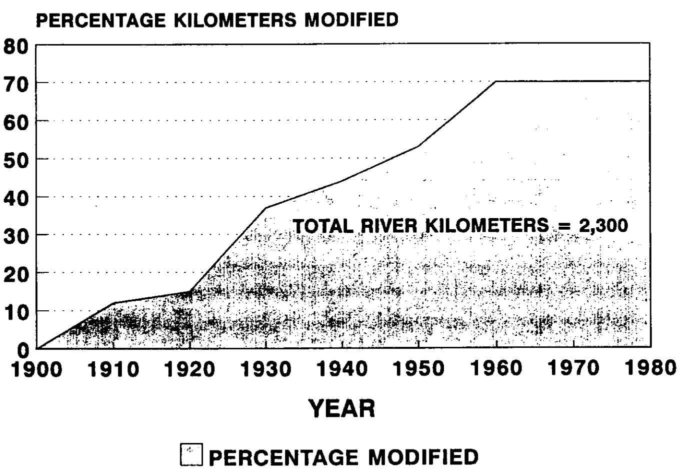

Figure 3. — Percentage of mainstream river habitat modified by major dams in Arizona, 1900-1980.

Gila and Colorado Rivers

Similar to the Salt, the Gila River was dried following completion of Coolidge Dam. Historically, the Gila River was navigable and was comprised of large marshy areas and oxbow lakes that would become several kilometers wide in flood (Rea 1983). Gila topminnow (*Poecilliopsis occidentalis*) and desert pupfish (*Cyprinodon macularius*) (see below) once abounded in these waters. The large Colorado salmon or squawfish (*Ptychocheilus lucius*) once ran in spawning schools up the Gila River, and in the late 1800s penetrated upstream at least as far as Ft. Thomas, over 200 km southeast of Phoenix (Kirsch 1889, Minckley 1973). Except for local convectional monsoon storm runoff and periodic flooding from winter storms and spring runoff, these once extensive and diverse aquatic habitats are now rivers of sand characterized by intermittent or subsurface flow only.

The taming of the Colorado River began in 1935 with closure of Boulder Dam impounding Lake Mead. Again, successive dams below (Parker and Davis) and above (Glen Canyon) this initial structure completely controlled and altered this highly diverse and hydrologically variable aquatic habitat, which was home for four large river cypriniform fishes: razorback sucker (*Xyrauchen texanus*), Colorado squawfish, and bonytail (*Gila elegans*) and humpback (*G. cypha*) chubs (see below) (Rinne and Minckley 1991). The Colorado is perhaps the most controlled and modified river in the world, presently consisting of dams, diversions, canals, and channelized waterways.

By comparison, very few streams such as Aravaipa Creek in southeast Arizona support an intact assemblage of native fishes (Barber and Minckley 1966, Rinne 1992). Streams in the Southwest such as Aravaipa that are generally unmodified by dams or diversions will effectively sustain a native fish fauna. Although non-natives are introduced accidentally from stock tanks on the watersheds during storms, subsequent floods appear to remove introduced fishes while sustaining native species.

WATERSHED IMPACTS

The upper elevation, more mesic forested areas of central Arizona and the White Mountains have sustained extensive timber harvest and grazing (Rinne and Medina, in press). The post war (1950s-60s) westward movement of the human population increased use of the national forests through hunting, fishing, and general recreational activity. The 1960 Multiple Use and Sustained Yield Act, along with other environmental legislation in general, resulted in more emphasis for all uses of public lands (Rinne and Medina, in press). Accompanying the environmental movement was the paradigm that all uses across the landscape are interrelated, and that one use affects the others.

The concepts of the intimacy of the watershed and its use and the functioning and health of riparian-stream systems (Debano and Schmidt 1989) and the disciplines of hydrology and fisheries (Heede and Rinne 1990) are paramount to sustaining native fishes. The idea of approaching land management on a watershed/ecosystem basis developed in the late 1980s (Szaro and Rinne 1988). Extensive literature was generated in the 1980s on the effects of grazing on stream habitats and fishes (Platts 1979, 1981, 1982, Kauffman and Krueger 1984, Rinne 1985, 1988). The Clean Water Act will place increased scrutiny of the quality of water issuing from watersheds into streams supporting native fishes.

Biological Impacts

Hybridization

Hybridization between native and introduced fishes in Arizona is best illustrated with salmonids. Rainbow (*Oncorhynchus mykiss*) and cutthroat (*clarki*) trout were first introduced from around the turn of the century (table 1). The rainbow trout has probably been more widely stocked in Arizona, and because of its spring spawning habits, freely hybridizes with the native Apache trout (*apache*). As a result, the native trout has been reduced drastically in range and numbers (U. S. Fish and Wildlife Service 1979, Rinne 1988), and its present distribution in pure form is negatively correlated with past stocking records of rainbow trout (Rinne and Minckley 1985). The native Apache trout had become restricted to less than 5% of its former range in Arizona (Harper 1978). In the late 1800s anglers could easily catch 100 per hour (U. S. Fish and Wildlife Service 1979). Presently, fishing is carefully regulated in some of the streams containing this "featured species" of sportfish (Arizona Game and Fish 1985).

In part, the massive range reduction of the Apache trout can be attributed to habitat alteration and competition with brown and brook trouts. Nevertheless, because less than a dozen of over 40 streams examined in the White Mountains of Arizona between 1977 and 1982 contained Apache trout populations that had not hybridized with other salmonid species (Rinne 1985), hybridization can be offered as a valid and major factor responsible for the marked decline of this once abundant native species.

Based on hatchery produced F_1 hybrids, Rinne et al. (1985) suggested that either pre- or post-mating isolating mechanisms may be present between the two species. Hybrids, however, have been readily produced by the U. S. Fish and Wildlife Service, Williams Creek National Fish Hatchery (Bob David, USFWS, pers. comm.). Further, much of the hybridization that occurred between Apache and rainbow trouts historically has diminished with the cessation of stocking of fingerling rainbow trout. Trout planted as fingerlings grew to adulthood in the wild and presumably more readily interbred with the native trout (Rinne 1985). Recently, utilizing state of the art genetic techniques, Dowling and Childs (1992) reported hybridzation between rainbow and the native trout to be regulated and dampened maternally by the latter. Based on a genetic analysis of introgression, Carmichael et al (1993) paint a bleak picture for the sustainability of the rare native trout in the White Mountains. Nevertheless, the combination of either using "catchables" or prohibition of stocking of rainbow trout in streams containing native trout and the apparent innate genetic trait suggested by Dowling and Childs (1992) have dampened the historic, more extensive hybridization effect.

Competition

Competition is another major mechanism of interaction that results from the extensive non-native salmonid introductions. Both brown and brook trouts have replaced the native trout in Ord Creek, Fort Apache Indian Reservation. Despite attempted stream restorations with fish toxin (1977 and 1981), and reintroduction of Apache trout to Ord Creek, brook trout continue to dominate the fish population in this stream. In 1977 they comprised 85% of the total number and 78% of the total biomass of adult trout (Rinne et al. 1981). Relative abundance of non-native to native trout could result in competition for food and space. Preliminary laboratory experiments have demonstrated that adult brook trout are more aggressive than Apache trout, and could interfere with feeding and spawning success of the native trout.

Competition as a negative impact on native fish populations is also demonstrated in the cyprinid or minnow family. The red shiner (*Cyprinella lutrensis*) has been both accidentally introduced by the bait industry and intentionally as a forage species. In Midwestern streams where it is native, the red shiner occupies intermittent and constantly flowing streams with an assemblage of other minnows. However, it survives admirably under intermittent flow conditions characterized by stressful levels of pH and dissolved oxygen, and high turbidities and temperatures (Metcalf 1966, Cross 1967, Matthews and Hill 1977). The red shiner, a habitat generalist, typically becomes abundant and dominates fish assemblages in Midwestern streams that have sustained habitat degradation (e.g., increased turbidity and temperatures). This dominance usually occurs to the detriment of more specialized fishes (Matthews and Maness 1979). The low faunal diversity of streams in the arid Southwest, combined with natural variations of aquatic habitat conditions and those induced through extensive habitat alteration, provide more suitable conditions for establishment and proliferation of the red shiner.

The red shiner is presently widespread in aquatic habitats of the Southwest. It has been suggested to have contributed significantly to the decline of native fish populations in Arizona (Minckley and Carufel 1967, Minckley 1973) and the Southwest (Minckley and Deacon 1968). For example, this cyprinid has an inverse distribution pattern in Arizona to two native, federally threatened cyprinid species, spikedace (*Meda fulgida*) and loach minnow (*Tiaroga cobitis*) (Minckley 1973). The mechanism of displacement or competition is not known, but, Rinne (1991) suggested utilization of the same physical habitat in the Verde River, Arizona, by adult red shiner and juvenile spikedace may be one mode of competition. Similarly, the red shiner has increased in lowermost reaches of the Virgin River, Arizona, while the native, endangered woundfin (*Plagopterus argentissimus*) has decreased (Cross 1978, Deacon 1988).

Predation

Evidence of the importance of predation on natives by introduced fishes is increasing. Recently, Blinn et al. (1993) demonstrated that rainbow trout have a significant impact on the native Little Colorado spinedace, *Lepidomeda vittata vittata.* Laboratory and field experiments have shown effective predation upon small (50 mm) spinedace. Inverse, linear and site specific distributional patterns in the field also indicate the negative impact of this introduced trout on the native cyprinid.

Centrarchids have been widely introduced for sport fishing in Arizona. In the lower Colorado River, species of this family have almost displaced the native fish fauna, presumably through predation on the eggs and young of native species (Minckley 1979). The green sunfish (*Lepomis* cyanellus) appears to be a strong contributor to replacement of the native Gila chub (*Gila intermedia*), presumably via predation. The sunfish is abundant in downstream reaches of Sycamore Creek, Prescott National Forest. Although the chub is abundant in the headwaters where the sunfish is absent, it is totally absent in the lower reaches where sunfish abound. A similar pattern of replacement has been documented in Sabino Canyon, Coronado National Forest, by J. A. Stefferud (Tonto National Forest, pers. comm.).

The drastic decline of the razorback sucker in the lower Colorado River has been attributed in part to predation on razorback ova, larvae, and fry by catfish (Minckley 1983). Marsh and Brooks (1989) reported predation effects by flathead and channel catfish when an attempt was made to re-establish the now- threatened razorback sucker in its native range. In one 2-day period it was estimated that up to 900 juvenile razorback suckers (45-168 mm SL) were eaten by these two introduced ictalurids in the Gila River, Arizona. Other formerly abundant, but now rare, native species (spikedace, loach minnow, Colorado squawfish of the Gila River, the lower Salt River, the Verde River, and the lower Colorado River have been replaced by introduced fishes. Channel and flathead catfish are abundant in all these waters. These inverse patterns of distribution of native species and the two catfishes combined with the data of Marsh and Brooks (1989) is strong evidence that predation is occurring.

The mosquitofish (*Gambusia affinis*), which Myers (1965) labeled the "fish destroyer," has been introduced worldwide as a biological control agent for mosquito larvae (Shoenherr 1981). Such introductions often are conducted despite the presence of native fish species that consume mosquito larvae. In Arizona, the mosquitofish has been implicated as a competitor responsible in large part for the drastic reduction of the range of the endangered native Gila topminnow (Minckley 1973, Minckley et al. 1977, Meffe, et al. 1983). The topminnow and the native desert pupfish were formerly widespread, and both were undoubtedly quite effective in mosquito control. Topminnow began to decline in the early 1900s with alteration of habitat by humans (Miller 1961). Nevertheless, topminnow was considered to be yet one of the most common native fish species in the lower Colorado River Basin in the 1930s (Hubbs and Miller 1941). Mosquitofish were introduced in California in 1922 and were collected from the Salt River at Tempe, Arizona, in 1926. Replacement of the topminnow by mosquitofish is usually rapid (Minckley and Deacon 1968, Minckley 1969); however, in the 1970s, the two species were reported to co-occur in southern Arizona in the Santa Cruz River system and Sonoita Creek (Minckley et al. 1977). In both instances, it appeared that topminnows were being sustained as a result of high carbonate waters in upstream, springhead refugia.

Because of the similarity of life history characteristics of topminnow and mosquitofish, competition for resources appeared to be the mechanism of replacement of the former by the latter (Shoenherr 1981). However, extensive studies both in the laboratory and in the wild by Shoenherr (1974) suggest mosquitofish effectively eliminate topminnow by predation on the fry and, secondarily, by reducing survivorship of adult females. Further, Meffe (1984) demonstrated that coexistence of topminnow with mosquitofish may be dependent on habitat complexity. In this same paper, he also discussed flood disturbance as a factor permitting persistence of topminnow in presence of mosquitofish. Meffe's (1985) field and laboratory studies corroborated Shoenherr's earlier (1974) results indicating mosquitofish replaces the native topminnow largely through predation. Based on the above case histories, predation, as a mechanism of interaction between native and introduced fishes, may be the primary reason for decline of native fish diversity in Arizona and the Southwest.

CONSERVATION EFFORTS TO SUSTAIN NATIVE FISHES

Conservation efforts for native fishes in desert environments of North America had their inception in the late 1960's and early 1970's. The National Environmental Policy Act (NEPA) of 1969, the Endangered Species Preservation and Conservation Acts of 1966 and 1969, and ultimately the 1973 Endangered Species Act laid the legislative groundwork for conservation and sustaining of all threatened and endangered species and their habitats. Although legislation laid the groundwork, it remained for agencies and individuals to implement of these Acts.

Individuals interested in desert fishes as a resource immediately took a stand to preserve species being rapidly and markedly impacted by the demand for water for agricultural and housing developments. In April 1969, a group of a less than a dozen individuals from several agencies gathered in Death Valley to prevent the potential loss of a group of rare species of desert fishes as a result of pumping of aquifers and drying desert springs in the Ash Meadows system. Out of these efforts, the Desert Fishes Council was founded in November 1969 and now numbers over 300 members (Pister 1981, 1990, 1991). This group presently addresses the conservation of desert fishes and their habitats throughout western United States, and northern Mexico, and has recently become more international in scope. A little over two decades after founding of the Council, a book ("Battle Against Extinction, Native Fish Management in the

American West," Minckley and Deacon 1991) was published documenting the struggle to conserve (and often preserve) native fishes, mostly in arid environments of the American West.

In the past, conservation efforts in the southwestern United States for the recovery and sustaining of native fishes in their native habitats have taken one or a combination of approaches. Conservation activities are normally outlined in "Recovery Plans" for species, which are documents for federally listed species drafted by a recovery team composed of multi-agency personnel. To address the extensive loss of aquatic habitat through hydrologic alteration, most recovery plans for listed fish species include objectives and activities that will secure, maintain, and enhance habitats (Rinne and Turner, 1991).

Removal of non-native fishes, a common practice in efforts to restore and sustain native fishes (especially salmonids), usually involves treatment with piscicide (Rinne and Turner 1991). The success of habitat renovation attempts is often reduced because of size and/or complexity of habitat, lack of security of habitat through ownership or special management, variable conditions of habitats from year to year, and the ever-present threat of reestablishment of introduced species.

Johnson and Rinne (1982) first expressed the need to move from protection and listing of native Southwestern species, as dictated by the 1973 Act, to active recovery. Large-scale rearing of fishes in hatchery environments (Rinne et al. 1986), coupled with intensive, long-term re-introduction into what is deemed "favorable habitat" in the wild, also has been used extensively in conservation efforts (Johnson 1985, Simons 1987, Minckley and Deacon 1991). Several extensive re-introductions of razorback sucker into un-renovated streams and rivers in Arizona (Johnson 1985) have failed largely because of predation by non-native species as discussed above. Over 10 million razorback suckers were stocked in waters of the Gila River Basin, Arizona in the 1980s; fewer than 120 have been recaptured, mostly within two weeks of stocking.

The apparent "successful recovery" of the Sonoran topminnow through extensive and widespread introductions similar to efforts for razorback sucker is questionable because of unprotected habitats (Simons et al. 1989). Purchase and management of riparian/stream habitats by private agencies such as The Nature Conservancy have been a boon to desert fishes.

Improving aquatic habitat through instream improvements has become a common practice for recovery of salmonids (Rinne 1981). However, because such instream, site specific improvement is often conducted without regard to surrounding land use and resultant condition of the supplying watersheds (Szaro and Rinne 1988, Lafayette and Rinne 1991), the probability of failure increases.

THE SUSTAINABILITY QUESTION

The above discussed literature and case histories of individual species of fishes and the overall activity to sustain them bring us to the bottom line, "Can native Southwestern fishes and their habitats be sustained?" In part, the answer to this question lies in 1) securing habitats, 2) species management strategies, and 3) ecosystem or landscape versus project or target management of natural resources.

First, we must undertake conservation activities where the habitat is secure (e.g., federal, state, and private conservation lands and special management areas), or relatively so, and where we can reasonably be most effective. Acquisition of large, important habitats such as the San Bernadino Ranch (U.S. Fish and Wildlife Service), White Mountain Hereford Ranch (Arizona Game and Fish Department), and numerous riparian-stream areas (The Nature Conservancy) has been and will continue to be critical to sustaining native fish species. Recent land exchanges by the U. S. Forest Service on Nutrioso Creek and purchase of private lands along the Little Colorado River, both important habitats for the threatened Little Colorado spinedace, are prerequisite to sustaining native fishes in the wild. Special designation and management of riparian-stream areas by federal agencies, such the San Pedro National Conservation Area and Bonita Creek Riparian Conservation Area (U. S. Bureau of Land Management) are both timely and critical. Species on federal lands are afforded protection from negative impacts by the Endangered Species Act. Finally, cooperative interagency management that is synergistic in recovering species and sustaining biodiversity in our streams and rivers should be adopted (Williams and Rinne 1992). Multi-disciplinary approaches incorporating hydrologists, biologists, geneticists, geologists, and botanists that address the overall ecosystem, or biodiversity philosophy of conservation, must be formulated.

Secondly, too often, past management and conservation efforts for native desert fishes were, of necessity, undertaken under crisis situations and were narrowly focused and monospecific in nature. In some cases, isolated desert springs leave no alternative to a species-by-species approach to recovery. On the other hand, conservation of desert fishes in the future must consider the total diversity (i.e., genetic, species, ecosystem, and landscape) of the fauna relative to alteration of aquatic systems (Rinne 1990). Where possible, conservation activities must begin to move away from single-species management toward ecosystems (i.e., river systems) or groups of species (Johnson and Rinne 1982, Rinne 1984, Williams et al. 1985, Rinne 1990b). Further, species not currently on official federal lists (i.e., state lists, U. S. Fish and Wildlife Service candidate species list, American Fisheries Society special concern list, U. S. Forest Service sensitive species lists) should be the object of our immediate conservation.Successful strategies for sustaining native fishes must include the multitude of fish species that are becoming rare but are not yet federally listed.

Management and research activities that consider such candidate, vulnerable, restricted, sensitive, or special species lists will become more proactive. For example, in the Southwest Region of the Forest Service (Arizona and New Mexico), there are more than 230 sensitive vertebrate species, 15 of which are fishes. Unfortunately, much endangered species management and research has adopted a "flagship" philosophy, wherein most budgeting and effort are directed toward a few high-profile species (e.g., bald eagle, grizzly bear, whooping crane, and currently in the Southwest, Mexican spotted owl). Although flagship efforts may indirectly spill over and affect other species needing immediate conservation attention, this is a serendipitous but less desirable way to manage ecosystems (Rinne and Medina, in press). Granted, we cannot direct intensive conservation efforts at 230 species, but through timely conservation we can increase the probability that many of these species will never be listed as threatened or endangered. Further, if we design our conservation efforts on an ecosystem basis (see below), a greater number of species in need of help will automatically be addressed.

Thirdly, because of occurrences such as the Virgin River, Ord Creek, and numerous past renovations with specific agency goals (Binns 1967, Holden 1991, Rinne and Turner 1991), a philosophy of management should be adopted that considers the ecosystem and its total diversity (Williams et al. 1985, Rinne 1990). To be successful, such a philosophy necessitates a multi-agency and multi-disciplinary approach (Pister 1990, Rinne 1990b, Williams and Rinne 1992). That is, the broader the landscape a recovery or conservation plan addresses, the greater the probability that more than one agency will be involved in management. Equally important, the probability is greater that a larger component of total fish diversity rather than single, indicator, or threatened and endangered species will be addressed (Rinne 1984). Recovery teams and plans such as those of the Colorado Fishes and the Desert Fishes Recovery teams nominally address this philosophical approach to conservation of desert fishes. However, because of the extent of the Colorado River Basin, for example, (and other major desert rivers), its drastic physical alteration, and agency jurisdiction that divides conservation efforts and philosophies into upper and lower basin entities, a holistic or ecosystem approach has not been realized.

Finally, agency targets and recovery plan objectives, although well-intentioned, often can be detrimental to the recovery of a rare fish over the long term. Although goals for delisting the Apache trout and Sonoran topminnow may be quantitatively obtainable, they may be qualitatively unsound (Rinne 1990). A recent chemical renovation of the Virgin River with piscicide to remove the introduced red shiner and restore the endangered woundfin minnow further illustrates this point. Placing a higher priority on completion of the operation on schedule rather than insuring that all components of this large-scale, multi-agency activity were in place, resulted in treatment of excessive reaches of river and negative impacts on other native species. The lack of success of chemical renovation of Ord Creek to recover the native Apache trout strongly suggests that no matter how well-intentioned or planned the project, success is not easily achieved (Rinne et al. 1981), and indeed as occurred in the Virgin River and in Ord Creek (Minckley and Mihalik, 1981), non-piscine aquatic biodiversity also is impacted.

SUMMARY AND CONCLUSIONS

Great strides have been made in understanding the issues of conservation and perpetuation of biodiversity of fishes in the American Southwest since enactment of the Endangered Species Act. Case histories discussed demonstrate the successes, but at great cost in time and money. Although piscine diversity of this xeric region is low, the uniqueness of the taxa dictates continued efforts to sustain fishes in their natural environments. Minimal success in conserving native fishes has been largely attained at a species level; genetic and ecosystem levels only now are emerging as important components of the effort to conserve this unique native fish fauna. A landscape biodiversity approach to conserving desert fishes has the least probability of being realized because of the often disjunct and isolated nature of aquatic ecosystems interposed with the continued development of the arid regions of the Southwest.

It behooves us to adopt innovative approaches in future conservation efforts for this regional fish fauna. To do less would discredit extensive efforts by agencies and individuals in the past that frequently were necessarily crisis management endeavors to secure the very existence of rare fishes. Indeed, it is time to move from "saving" (Johnson and Rinne 1982) native fishes in the arid Southwest to "recovering" and sustaining them in their native habitats. Many of these fishes occur in isolated habitats in a region characterized by the lack of the basic life medium of fishes—water. We have come far, yet have far to go. Only if the overall aquatic and influencing resources are given priority in our conservation efforts will aquatic habitats and native fish biodiversity be maintained for perpetuity.

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Landscape Aesthetics, Ecology, and Human Health: In Defense of Instrumental Values

Russ Parsons[1], Terry C. Daniel[2], and Louis G. Tassinary[1]

Abstract — For the past forty years, following Aldo Leopold's (1949) lead, ecologists and environmental ethicists have lobbied for the inherent or intrinsic value of nature as a basis for environmental conservation. Though they start from different perspectives, ecologists and ethicists often borrow heavily from each others' literature, employing similar arguments to reach the same conclusion: Nature is valuable in and of itself. Writers in these fields generally concede that the concept of value makes no sense in the absence of a valuing perceiver (i.e., one cannot properly speak of the *intrinsic* value of nature); but, but they do maintain that natural objects, such as a tree, a meadow or a countryside, are *inherently* valuable. That is, they are valuable simply by their existence, completely independent of the utility or benefits they may confer upon a perceiver. Utilitarian or beneficial values are referred to as *instrumental* values, and they are generally considered to be an anemic basis for the new conservation ethic towards which environmentalists have lately been moving (Callicott, 1985).

Without attempting to dispute the claim that nature may have inherent value, we argue that instrumental values have an important part to play in land management, and that they can be used as effectively as the concept of inherent values to inform conservation ethics. Most importantly, the instrumental values of natural environments have yet to be fully articulated by environmental researchers. In particular, the concept of environmental value cannot be understood without reference to the evolutionary utility of emotional responses. Research and theory from the fields of landscape aesthetics and environmental psychophysiology suggest that human responses to environments are, in part, genetically determined, and that views of natural landscapes may have the potential to positively influence physical and psychological health. An evolutionary perspective towards values in general, and towards human responses to environments in particular, is wholly appropriate for the ecologically-based approach to land management recently promulgated by the U.S. Forest Service. The emphasis on instrumental values that derives from an evolutionary perspective can be an important complement to the development of a conservation ethic based on the concept of the inherent value of nature.

The relevance of ecology to human value is not so much that it provides any basis for judgment, but that it shows the kind of thing the judgment has to be about, namely, a system of interacting activity - different aspects of human personality, interacting with one another and with natural and artificial surroundings - rather than single isolated traits. The whole richness of the interacting system in which man [sic] is involved is not fully expressed in his embryonic development, even if we include all his activities as a youth and adult. Human values inhere in what one might call "human

[1] *Environmental Psychophysiology Laboratory, College of Architecture, Texas A&M University, College Station, TX, USA.*

[2] *Psychology Department and School of Renewable Natural Resources, University of Arizona, Tucson, AZ, USA.*

evolutionary ecology". Unfortunately the science we have developed so far can hardly sketch even the crudest outline of this....the most basic science of the future. — C.H Waddington, 1978

INTRODUCTION

This paper is an attempt to apply concepts from evolutionary biology to environmental aesthetics, human health and environmental ethics, and to draw out of that exercise potential implications for environmental management. A review of recent biological and ecological writings reveals a preoccupation with environmental values - not *biological* values, such as the value of a given soil composition for plant growth - but, *human* values or ethics as they relate to natural environments, ecosystem sustainability, biodiversity, etc (e.g., Leopold, 1933; Murdy, 1975; Soulé, 1985; Waddington, 1978). At the same time, authors in the field of environmental ethics have been concerned with biological issues, such as ecosystem sustainability (e.g., Callicott, 1985; Rolston, 1984), and both the ethicists and the ecologists have repeatedly called for a new, nonanthropocentric environmental ethic that firmly establishes the *intrinsic* value of nature, apart from any reference to human needs or welfare (i.e., *instrumental* values). Intricate logical arabesques and extensive cognitive rationalizations are used by these authors to establish nonhuman notions of right and wrong with respect to natural environments.

One assumption often made by writers in these fields is that any system of ethics that acknowledges the centrality of humans necessarily subverts goals associated with ecological sustainability, such as biodiversity. As Kagan (1984) has suggested for western philosophers in general, one reason environmental ethicists might have for basing environmental ethics on nonanthropocentric logic is a basic distrust of human nature, considering it to be essentially selfish, cruel and deceitful. Frequent disparaging references to narrow or "economic" human self-interests in various environmental literatures supports this idea.

By contrast, in the research conducted in our respective labs, and more generally in trying to understand the research on landscape aesthetics and the potential health effects of natural environments, we have been unabashedly anthropocentric, focusing on some of the *instrumental* values of nature. We have found it very useful in this regard to take an evolutionary perspective on human interactions with their environments, a view that has been supported by findings that reveal strikingly similar aesthetic preferences across vastly different populations, as well as significant overlaps between preferred environments and natural environments that have been reported to be stress-reducing. This visually preferred, stress-reducing environment type has been called an evolutionary or savanna environment, because its forms and contents resemble those of *Homo sapiens* speciation.

While we agree that ethics is relevant to any discussion of human relations to natural environments, we construe environmental ethics anthropocentrically as well, rather than anthropomorphically. That is, we understand environmental ethics to be a subdiscipline of human ethics that examines the morality of human behavior as it relates to natural environments; thus, our view is anthropocentric. As we argue below, however, and as others have argued as well (e.g., Harlow, 1992; Murdy, 1975; Seligman, 1989; Weston, 1985), this implies only that the values humans hold are human-centered, and not that *only humans* have value. Similarly, though our understanding of environmental ethics is not anthropomorphic, and thus we do not impute human characteristics such as innate moral rights to nature, we recognize that nature can be and often is very highly valued.

As scientists, we suggest that an appropriate stance with respect to environmental ethics is one of description and explication *in the service of* environmental management, and not prescription *as a guide to* environmental management (see Campbell, 1975). We suggest further that calls for a new environmental ethic may be premature; that there is still a great deal to be learned both about how people value environments and about the nature and scope of human needs. In the second section of the paper we present several brief historical illustrations of relationships humans have had with the environments they have occupied. These examples suggest the breadth of human relationships to environments, indicating that humans have been both destructive and constructive in their environmental manipulations; that humans are quite capable of simultaneously exploiting and sustaining ecosystems[1]; and that one tendency of human manipulations of the land has been to create environments reflective of our evolutionary past. The forms and contents of evolutionary environments are also relevant for understanding current research in landscape aesthetics, and towards the end of this section we review a recent study that is emblematic of findings in the area.

In the third section of the paper, to suggest the complexity of even the most basic human needs, we examine Geist's (1978) evolutionary model of human health. This model challenges narrow, economic interpretations of human self-interests and emphasizes the potential importance of ancient humans' interactions with their environments for present day approaches to human health. In the final section of the paper, we conclude by describing an evolutionary approach to ethics, highlighting the importance of emotional

[1] *The sense in which "exploit" (and its forms) is used here and below is synonymous with "use," as is commonly understood in the evolutionary literature. Thus, "environmental exploitation" is not coextensive with destruction of an environment or ecosystem. Rather, it is an ecologically neutral phrase referring to an organism's use of its environment, and that use may or may not be ecologically destructive.*

responses. One conclusion we draw from this analysis, and those of the preceding sections on aesthetics and human health, is that to understand human interactions with their environments, it is crucial to understand human emotional responses to environments and the evolutionary history of those responses. This appears to be true regardless of the nature of the interaction, whether it involves manipulations of the landscape, aesthetic responses, health effects or ethical valuations of the environment.

A second conclusion we reach is that "instrumental values" is a much broader category than is typically supposed, one that cannot simply be equated with narrow economic self interests. By implication, environmental management policies that incorporate evolutionary approaches to environmental aesthetics, human health and environmental ethics as laid out here will not necessarily be ecologically destructive. Rather, the incorporation of an evolutionary approach leads to the conclusion that biodiversity of ecosystems, whether naturally occurring or humanly maintained, is an *instrumentally* valuable goal.

LANDSCAPE AESTHETICS AND HUMANITY'S RELATION TO THE LAND

...[the] ideological opposition of culture and nature - with no mediating term - has had real consequences. More often than need be, Americans confronted with a natural landscape have either exploited it or designated it a wilderness area. The polluter and the ecology freak are two faces of the same coin; they both perpetuate a theory about nature that allows no alternative to raping it or tying it up in a plastic bag to protect it from contamination. — Frederick Turner, 1991

In this section we will examine how people have manipulated the environments they have occupied by citing several brief examples of human interventions in the land. These examples are drawn from more extensive historical analyses by Dubos (1972; 1980) and Glacken (1967). They are not meant to be comprehensive but are only presented as evidence suggesting that humans have been prone to environmental manipulation virtually since speciation. Though humans have been ecologically destructive, they have also been ecologically responsible, both in western and nonwestern cultures. In particular, one tendency of human land manipulation has been to create landscapes reflective of human evolutionary environments. Evidence from research on landscape aesthetics indicates that present day responses to savanna-like environments are almost uniformly positive; and exposure to savanna-like environments is also believed to be healthful or stress-reducing, a point we will examine further in the following section. We begin this section with definitions of several terms often used to describe environments.

Definitions

Evolutionary environment is a term Geist (1978) has used to assign to the environment of speciation, that is, the environment in which a species evolved from its previous form to its present form. Given that there is some theoretical ambiguity about the specific means of human speciation (see below), whether through dispersal and expansion or by gradual evolution *in situ*, pinpointing the specific environment type where *Homo sapiens* first emerged is no trivial matter. Geist (1978) argues for a dispersal or expansionist model, and locates the human evolutionary environment in the cold, open woodlands and grasslands of interglacial Europe. Jolly and Plog (1979), on the other hand, favor an evolutionist model of gradual and parallel speciation over the huge geographic range of *Homo erectus*, which extended from Africa north to Europe and the Middle East and east to Asia (Brace & Montague, 1977; Campbell, 1985; Geist, 1978; Jolly & Plog, 1979).

For our purposes here, choosing between these alternatives is less important than highlighting some common features among these candidate environments for human speciation. Whether in Europe, the Middle East, Asia or East Africa, the environments in question are typified by broad open expanses of grasslands supporting a variety of grazing fauna, punctuated by occasional groupings of trees and smaller vegetation types, including tubers and other plants with edible roots and stems. (Campbell, 1985; Dubos, 1980; Geist, 1978; Jolly & Plog, 1979). The selective pressures in such an environment would favor cooperative representatives from the genus *Homo* with good communication skills, large body size, excellent manual dexterity, good eyesight - in short, those features that support a hunting and gathering lifestyle.

A *savanna environment* is any region where the mean distance between trees exceeds canopy diameter (Dubos, 1980). This broad definition allows the term savanna to accommodate not only the candidate environments for human speciation just mentioned, but it also includes the extensive geographic range of the genus *Homo*. This range includes areas as far flung as the grassy plateaus of East Africa, the broad valleys of Mesopotamia, the open woodlands and colder grasslands of southern Europe, and the islands of Indonesia (Cambell, 1985; Dubos, 1972; Jolly & Plog, 1979). More importantly, this definition suggests that savannas were likely the environments of our immediate evolutionary precursors, *Homo erectus* and *Homo habilis*, as well as those of our own speciation. Thus, savannas helped to shape our environmental responses for several million years before human civilization.

In what follows we will also have cause to refer to natural environments and wilderness environments. For *natural environments*, we follow Ulrich (1986) and regard as natural any environment where vegetation predominates, in any of its forms. Thus, human artifacts, such as buildings, roads, cars, etc. may be present, but only as minor, insignificant features of the environment. It should be noted that, by this definition, rural agricultural land is considered natural, whereas a rigid adherence to the focus on vegetation per se would exclude environments, such as arid deserts or barren polar regions, with little or no vegetation. Here we make an exception and construe these environments as natural. The aim here is not to be scrupulously logically consistent, but to reflect a "common parlance" conception of natural environments revealed by literature that has been reviewed by Ulrich (1986) and Wohlwill (1976) among others.

By *wilderness environments*, we refer to natural environments that have no human artifacts (buildings, cars, etc.), and only minimal evidence of human influence (e.g., hiking trails). This definition differs from those that might be offered by strict wilderness preservationists, who would balk at the notion of *any* human influence in a wilderness area. However, given the extensive reach of air pollution, water pollution, acid rain, overflight noise, etc., such a strict definition of wilderness, would exclude too many lands commonly thought to be wild (Dubos, 1980; Oelschlaeger, 1991).

Historical Examples

There is a tendency among western environmental ethicists and, to some extent, ecologists as well, to view past human relations to their environments as being more ecologically sensitive than those of contemporary societies (e.g., Oelschlaeger, 1991). This view is based, in part, on the belief that early humans lived more intimately and harmoniously with nature than do modern humans, took from the land only what they needed, and consequently were less destructive of the environment. Common, also, is a belief that nonwestern cultures, especially Oriental cultures and aboriginal peoples outside of Europe, have behaved more responsibly towards the land than have their western counterparts (e.g., Roszak, 1969; Udall, 1964). An evolutionary perspective, however, casts doubt on the notion that the modern tendency to alter the land is either era or culture-specific. Historical analyses (e.g., Dubos, 1972; 1980; Glacken, 1967) indicate that since the time of Paleolithic hunter-gatherers, but especially with the advent of Neolithic agriculturalists, humans have altered their environments in substantial ways, often having destructive environmental effects.

It is almost certainly true, as many writers have suggested (e.g., Dubos, 1972; 1980; Oelschlaeger, 1991), that Paleolithic humans were more aware of the biological rhythms of nature than are modern urban humans. But, given the vast differences between the typical surroundings of the two groups, and the respective skills needed to "earn a living" from the environment, the point seems both trivial and obvious. Similarly true, trivial and obvious is the contention that Paleolithic humans were less destructive of their environments than are moderns. What is not so obvious, however, but what is often asserted (or strongly implied) as well, is that the former point necessarily leads to the latter: That is, *because* early humans were more attuned to nature they were perforce less destructive of their surroundings. There are several reasons to doubt this explanation of the differences in ecological destructiveness between modern and pre-urban humans

From an evolutionary perspective, one might view both the ability to discern the biological relationships of one's environment and the tendency to use that knowledge to exploit the environment as a pair of coordinated means to improving reproductive fitness, means that are common to Paleolithic *and* modern humans. Exploitation of available nearby environments is thought to have been a common response of early humans and their precursors to population pressures that taxed the carrying capacity of a given environment (Campbell, 1985); and, more generally, ordinary pressures of organic competition are believed to favor exploitation of resource rich environments (Campbell, 1985; Geist, 1978). Thus, manipulation and exploitation of the environment may well be genetically influenced tendencies. For this reasoning to be plausible, there should be evidence that ancient humans were exploitative.

There is good empirical evidence that Paleolithic humans were quite willing to manipulate their environment, occasionally being exploitative beyond their needs (though certainly not on the scale of modern human societies), as were other large predators such as lions, wolves, and tigers (Dubos, 1980). There is fossil evidence that Paleolithic humans engaged in several forms of cooperative hunting, one of which involved luring herds of various types of social ungulates to their death by stampeding the animals over cliffs. This practice undoubtedly left more meat than could be consumed by the group or defended from other predators (Dubos, 1980; Geist, 1978). Spots where such hunting occurred are known as "jumps" or "deadfalls", and are remarkable not only for the large number of animals killed, but also for the tendency of Neanderthal habitation sites to be clustered nearby (Clark, 1970), suggesting that this type of hunting was relatively important for early humans. There is also evidence that Paleolithic hunters regularly burned large tracts of forests and grasslands in pursuit of their prey (Dubos, 1980; Oelschlaeger, 1991), again suggesting a willingness to manipulate the environment to their advantage.

These examples attest to the exploitative tendencies of Paleolithic hunter-gatherers in ancient Europe and Africa; and others could be cited as well, indicating that preagricultural hunter-gatherers of other times and places, such as preagricultural Native Americans, engaged in similar hunting practices (Guthrie, 1971; Wheat, 1967). It is also the case that cultures often cited as having ecologically benign environmental philosophies and/or lifestyles, such as the Japanese, Chinese, Australian aboriginals and South and North American aboriginals, have known periods of great environmental degradation. Whether it be through deforestation, burning of grasslands, overgrazing by domesticated animals, or hunting beyond their needs, all of these reputedly "ecological" cultures have been environmentally destructive (Dubos, 1972; 1980; Glacken, 1967). Indeed, they may have developed their benign environmental philosophies in recognition of past environmental indiscretions (Dubos, 1980).

Examples such as these serve to contradict the notion that acute awareness of and intimacy with natural surroundings *necessarily* produces restraint in human actions toward their environment. More likely explanations of recent large-scale environmental degradations lie in ever increasing population sizes and advances in technology that facilitate exploitation of the environment. Thus, considered in light of the long history of human environmental manipulation after the development of agriculture, which shows ever increasing and more facile manipulations (see Glacken, 1967), these examples of ancient environmental manipulations lend support to the idea that the tendency to manipulate one's environment is a genetically influenced trait.

Having reached this conclusion, we hasten to add two important caveats. First, "genetically influenced trait" is a carefully qualified phrase intended to recognize the importance of sociocultural and other environmental influences on one's tendency to manipulate the environment, yielding the expectation of considerable variability in the manner and extent to which people manipulate their environments. Indeed, genetic heritability alone implies phenotypic variability at some point during the history of a trait, or there would be nothing for natural selection to work with (Lewontin, 1984).

Second, we have also carefully worded this conclusion to suggest the ecological neutrality of the human predilection towards environmental manipulation. Human interventions in environments are not necessarily ecologically destructive, though they often change the specific form of ecosystems, displacing some species and introducing others. Dubos (1980) contends, for instance, that there are examples throughout the world, such as the Ile de France, where humans have raised crops almost continuously from the time forests were first cleared from the land by Neolithic agriculturalists, and these areas are still fertile, productive, ecologically rich areas. Landuse patterns and farming techniques established by the Benedictine monks during the middle ages are a prime example of this ecologically sensitive, yet humanly sustaining land intervention capability, a capability that has been found in different cultures throughout the world (Dubos, 1972). That humans can benefit nature through their interactions with the environment has been clear, as well, to some of this country's more revered environmental writers (e.g., Leopold, 1933; Thoreau, 1993). Thus, what this brief historical sketch suggests is that humans have always manipulated their environments, sometimes enhancing ecological sustainability, other times reducing it.

Although there is variability in how humans have manipulated their environments, history is strewn with examples of interventions of a particular type. It is only relatively recently (within the past several hundred years) that humans have come to regard wilderness areas positively. Throughout most of human history, densely forested lands, forbidding mountains, swamp or marsh lands and desert areas have been considered inhospitable lands, fit only for wild beasts (Glacken, 1967; Nicolson, 1959; Oelschlaeger, 1991). As early humans spread beyond savanna-like environments of their speciation, they often settled near water, and they either cleared forests or irrigated arid land to accommodate their settlements and agriculture. Dubos (1972; 1980) has argued that human settlement patterns and their subsequent land manipulations reflect our biological past, mainly through optimizing prospect and refuge opportunities (Appleton, 1975) and settling near water when it was possible, and manipulating the environment to approximate these and other savanna-like characteristics when it was not.

Dubos suggests that part of the human inheritance is a set of environmental needs shaped by our evolutionary environments, and that these environmental needs can help to explain present behavioral patterns such as an:

> *"...almost universal and subconscious fear of forested wilderness; certain features of design that are common to all schools of landscape architecture; the preference of all human beings for the same narrow range of environmental temperature; the biochemical similarity of nutritional requirements in all human groups; the fact that all the plants we cultivate belong to sun-loving species (as do the plants growing in savanna kinds of country) and cannot therefore grow in the shade of a dense forest" (Dubos, 1980).*

Landscape Aesthetics

We do not intend to argue the specific merits of Dubos's thesis, but we do wish to point out that it is consistent with the literature on landscape aesthetics and scenic beauty preferences, as well as with theory and initial research on the stress-reducing qualities of nature. This research, which has been reviewed elsewhere (Kaplan & Kaplan, 1989; Parsons, 1991; Ulrich, 1983; 1986), suggests that certain abstract visual characteristics, as well as more concrete content elements, contribute significantly to explanations of perceived scenic beauty. These visual characteristics include relatively open fore and mid grounds; complexity tempered by comprehensibility; relatively smooth textures that indicate ease of travel; and, most importantly, a quality that is commonly referred to as mystery, in which a bending path, a stand of trees, a rise in topography, or any other visually interposed element partially blocks one's view while suggesting the availability of more environmental information beyond.

Though these abstract characteristics could be used to describe any environment, including a largely artifactual urban environment, research findings indicate that predominately natural environments are generally preferred over predominately urban environments (see Parsons, 1991; Wohlwill, 1983, for reviews). Important vegetational types include trees, grasses, forbs and shrubs (Daniel & Boster, 1976; Ulrich, 1986), and when these contents are combined to produce the abstract visual characteristics just mentioned, the picture that emerges is remarkably similar to the savanna environments described above: fairly open grassy areas, allowing relatively deep visual penetration and unobstructed travel; and, these grassy areas are punctuated by occasional clumps of trees and shrubs that partially block the view suggesting more environmental information beyond, with the whole presenting a somewhat complex yet comprehensible scene.

As an illustration, we will describe a typical study from this body of research, which demonstrates the potential emotional responses of humans to landscape scenes. We suggest that these emotional responses to landscapes can serve as a link among aesthetic responses, potential health effects of natural environments and environmental valuations (ethics). In addition, we suggest that emotional responses in each of these domains are, in part, a function of human habitat selection systems developed in the savanna-like environments of our evolutionary past[2].

Yi (1992) has recently completed a study in which she compared the effects of culture, occupation, symbolic significance of the landscape, and beauty of the landscape on subjects' assessments of scenic beauty, picnic preferences and residential preferences. She compared: Koreans and Texans; farmers, landscape architecture students and others; and, landscapes with positive semantic associations for Koreans (e.g., location of a Buddhist temple), landscapes with positive semantic associations for Texans (e.g., campus of a highly regarded university), and semantically neutral landscapes. Photographs representing high- and low-beauty exemplars in each of the landscape categories were presented to subjects for the three preference judgments[3].

Though this was a fairly complex study, testing an ambitious model of cognitive and affective responses to landscapes and involving multiple independent and dependent variables, the results were strikingly uniform. As Figure 1 shows, Koreans and Texans reported very similar scenic beauty, picnic and living preferences (rows in Figure 1), regardless of the semantic associations of the landscapes being judged (columns in Figure 1). Indeed, though there are some statistically significant differences for these factors across the three judgments, the theoretically assigned *aesthetic value* of the scenes (landscape scenic beauty) is clearly the most important factor. The results were nearly the same when preference ratings were compared among farmers, landscape architects and others. And, both of these findings are underscored by an examination of the effect sizes of the independent variables (Figure 2), showing that scenic beauty accounted for 27% - 40% of the variance in the three preference judgments, while the other factors (i.e., cultural, semantic, occupational) combined accounted for less than 10% of the variance in each of the judgments.

As mentioned above, these findings are consistent with other work in this area. For instance, aesthetic quality significantly predicts both the choice and the willingness to pay for the use of campsites (Daniel, Brown, King, Richards & Stewart, 1989); and, when asked to rate different examples of a savanna tree species (*Acacia tortilis*), respondents from the U.S., Argentina and Australia preferred those examples that indicated a high quality savanna environment (Orians & Heerwagen, 1992). Thus, this study by Yi (1992), when placed in the context of related research, highlights the present day importance of emotional responses to savanna-like environments (as seen through the lens of landscape aesthetics or scenic beauty), indicating a positive effect of such environments for both short-term (place to picnic, camp) and long-term (place to live) interactions with the environment, as well as for aesthetic responses.

In the next section we describe Geist's (1978) evolutionary model of human health, concentrating on the aspects of the environments of human speciation that are relevant both to health and aesthetics. This examination is prompted in part by the desire to understand potential health effects associated with time spent in natural environments, beliefs for which are longstanding, and evidence for which is only beginning to unfold. But, this examination is also presented as an example of the complexity of human needs and their relations to the environment, suggesting that human interests cannot be narrowly defined.

HUMAN HEALTH

Negatively phrased, parochial definitions of health, focusing on the absence of disease or injury, have rightly been criticized for some time as being inadequate. Unfortunately, more positively phrased definitions are often vague. The World Health Organization, for instance, has defined health as "... a state of complete physical, mental and social well-being, and not merely the absence of disease or infirmity" (as quoted in Geist, 1978). Because it is not clear how one measures the complete physical, mental and social well-being of an individual, the utility of this definition (and others like it) is limited. Geist (1978), an evolutionary biologist, has developed a theory of health that includes a specific set of criteria against which human health can be evaluated. Though the theory is by no means accepted wisdom in the medical community, it has several advantages for our purposes here. First, it provides a working definition of health with enough precision to generate testable hypotheses about how the management and design of environments can influence human health. Second, Geist's theory constitutes an evolutionary approach to health, making it a felicitous choice for the evolutionary framework we use to discuss relationships among health, environmental ethics and landscape aesthetics.

According to Geist, the human genome provides the potential for any individual to develop into an exceptional phenotype that is especially capable of dealing with environmental contingencies when exposed to unexploited environments. Such an individual is "...characterized by exceptional development of tissues and behaviors of low priority", that is, one who is

[2] See Jaenike & Holt (1991) for a review of evidence for genetic variation in habitat preferences.

[3] High- and low-beauty scenes were chosen by the investigator to maximize and minimize respectively the above mentioned abstract and content variables associated with visual quality in landscapes.

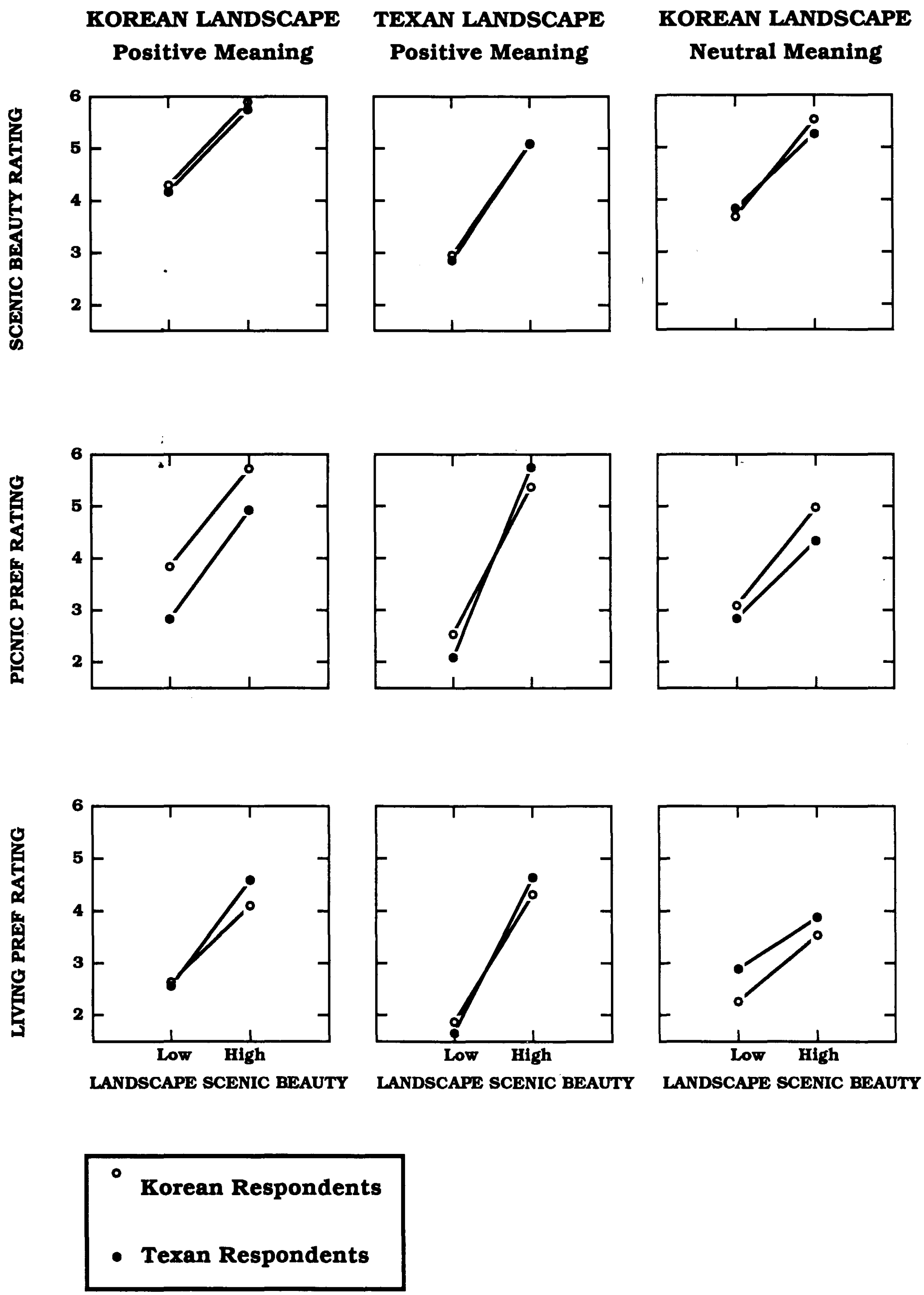

Figure 1. — Scenic beauty, picnic preference and living preference judgments of Korean and Texan respondents (7-point scales, 7 high). Subjects in both groups rated Korean landscapes with positive meanings for Koreans, Texan landscapes with positive meaning for Texans, and semantically neutral Korean landscapes. Landscape scenic beauty was manipulated within these landscape types according to theory in landscape aesthetics (see text). All subjects rated three examples each of low landscape scenic beauty and high landscape scenic beauty within each of the semantic landscape categories. (Adapted from Yi, 1992)

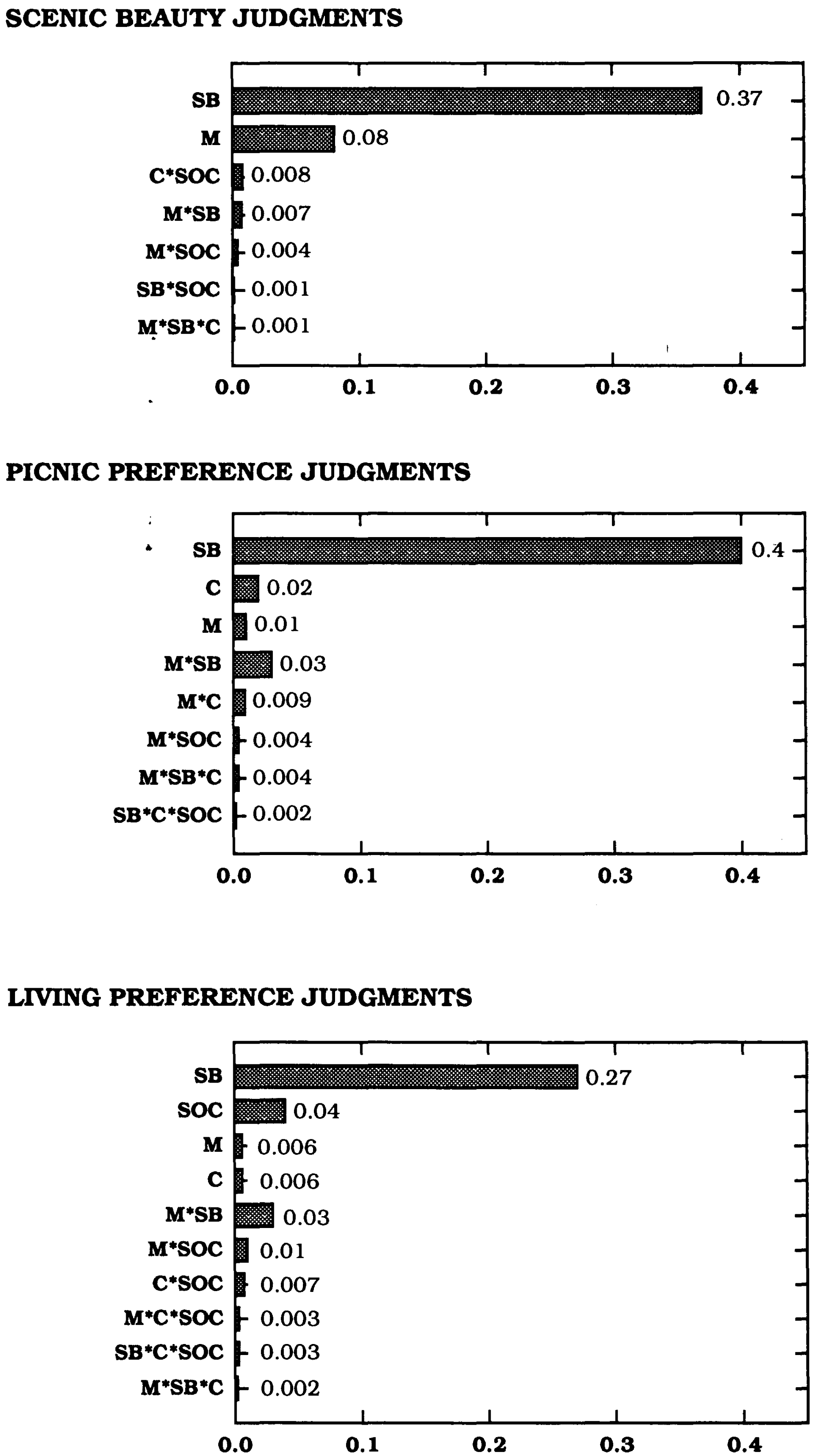

Figure 2. — Effect sizes of the significant main effects and interactions for scenic beauty, picnic preference and living preference judgments ANOVAs. The abbreviations used for the independent variables are: SB = scenic beauty of the landscape (Low/High); C = culture (Korean/Texan); M = meaning of the landscape (Korean Positive/Texan Positive/Neutral); SOC = social/occupational class (farmers/landscape architecture students/others). (Adapted from Yi, 1992)

required to spend relatively little time in maintenance activities (hunting, gathering, finding shelter, etc.), and can spend relatively more time developing behavior and structures that are "...adaptive in dealing with conspecifics and new environments during dispersal." Following Waddington (1957;1960), Geist argues that evolution only occurs during dispersal, when members of a population living on the geographic edge of a habitat exploit an unoccupied environment that is rich in resources. Traits that are adaptive under conditions of resource abundance gain full phenotypic expression, and natural selection acts to remove those genotypes that do not maximize the adaptive traits. Thus, according to this scheme, health is defined in terms of the fullest phenotypic expression of those traits that were adaptive when prehumans evolved to *Homo sapiens*, and those traits are referred to as the diagnostic features of the species (Campbell, 1985; Geist, 1978).

It is important to recognize that this model of human speciation, evolution by small isolated populations that then expand to displace their predecessors (the expansionist model), can be contrasted with an extreme evolutionist model (Jolly & Plog, 1979). According to this latter model, *Homo sapiens* evolved from *Homo erectus* gradually in many places and in parallel, without isolated exploitation of an uninhabited resource rich environment, and without the need for displacement. The predecessors simply gradually changed into the successors *in situ*. The fossil record offers no clear choice between these models (Jolly & Plog, 1979), nor is it clear that they are necessarily mutually exclusive. Thus, our emphasis here of Geist's theory of health does not reflect a strong endorsement of the expansionist evolutionary model. Rather, by highlighting this theory of health we hope to introduce a level of precision into discussions of the potential health implications of human evolutionary environments that has been lacking in the past (e.g., Kaplan & Kaplan, 1989; S. Kaplan, 1987), including in some of our own writings (Parsons, 1991; Ulrich & Parsons, 1992).

Optimum environments. We have suggested that health could be defined in terms of phenotypic expression, and we have alluded to the idea that the phenotypic expression of the human genotype depends upon the environment of ontogenetic development. Geist (1978) has hypothesized that the phenotype to be expected under conditions of resource scarcity is considerably different than that expected under conditions of resource abundance. He has reviewed evidence in both animals and humans suggesting that conditions of resource scarcity produce relatively small, docile individuals of low competence, while those individuals raised in plentiful environments are larger, more likely to explore their environments and are more adept at handling a broad range of environmental contingencies. In each case, the phenotype produced is the one that maximizes reproductive fitness under the prevailing environmental conditions. Natural selection, then, defines health in terms of reproductive fitness, caring not whether the phenotype produced is large or small, inquisitive or timid, competent or incompetent across a variety of environmental settings.

As mentioned above, Geist (1978) suggests that humans emerged as a dispersal or expansion phenotype under conditions of resource abundance; and others who favor the gradual evolutionist model of speciation indicate that the environments occupied by *Homo sapiens'* precursors also were relatively plentiful. Thus, both models argue for large bodied, highly competent individuals as being the healthiest human phenotype, and by implication, the optimum environment needed to maximize the phenotype is one of resource abundance. Stated more formally by Geist, "The optimum environment for man [sic] is that which during ontogeny maximizes body size and still produces a disease-free organ system at the termination of body growth in early adulthood." Although stated in terms of physical development, Geist contends that this definition implies maximal development of psychological, social and intellectual capacities as well, because maximal body growth and organ health cannot be attained without the development of these other characteristics.

Geist (1978) presents a large, multifarious set of normative environmental criteria derived from the model that, if met, should lead to optimum health. Though we will not examine all of these criteria here, we will mention several that are directly related to outdoor environments and human emotional responses. Geist's model predicts that substantial interactions with outdoor natural environments during ontogeny are essential to good health. Such interactions help to confer knowledge about the passing of seasons, meteorological phenomena, the diversity of living things and natural rhythms of life and death, including information about how plants grow, blossom and bear fruit, as well as information about the movements and behavioral patterns of other animals. It is obvious, perhaps, how knowledge of this sort would have been useful to pre-urban humans, and it is interesting that Geist maintains that its acquisition still contributes to optimum health of modern humans, suggesting as it does that a cavalier attitude towards one's environment is unhealthful. In a similar vein, Hardin (1982) suggests that U. S. educational systems disproportionately concentrate on literacy at the expense of "numeracy" and "ecolacy", and that a well-developed, properly educated person would be conversant with knowledge, concepts and theories in each of these domains.

Geist also suggests that interactions with outdoor natural environments during development are essential because nature provides opportunities to *do* things, improving coordination, dexterity and motor skills through manipulation of the natural environment. These interactions help to hone other skills as well, including language, intellectual and social skills, through the use of nature-related metaphors and the exchange of environmental information with others (see Fernandez, 1973; 1974; and Geist, 1975).

Nature also provides opportunities to experience a wide range of emotions. Geist adopts an "optimum level of arousal" theory of emotion (e.g., Berlyne, 1966). In this framework arousal is modulated by matches between significant environmental stimuli and stored memories, and the most relevant stimulus patterns are those with some evolutionary significance. He maintains that one of the more crucial environmental criteria for good health is the provision of emotional security, which is largely acquired through the mastery of daily problems. The most important daily problems to be overcome are the acquisition of food and shelter, and savanna environments offer significant advantages in this regard. Immediate threats from predators are more easily handled on a relatively open savanna plain than in a heavily forested environment because the greater visibility of open savanna environments allows early identification of predators, the occasional clumps of trees provide shelter both from predators and from the elements; and, the grassy expanses allow ease of movement, aiding escape. The presence of grasses, trees and flowering plants also imply the presence of water, suggesting the potential longterm habitability of the environment. The ready fulfillment of food, shelter and safety needs (and the concomitant reduction of anxieties about them) in such environments could well lead to a conscious association of emotional security with savannas, which could in turn have salubrious effects, further enhancing the reproductive fitness of an initial positive response that takes advantage of the immediate survival benifits (see Parsons, 1991, for a theoretical account of potential health effects, and Ulrich *et al.* 1991b, for an empirical demonstration of the stress-reducing effects of natural environments).

We want to be clear, however, that we view emotional security in this example as one instance of a range of positive emotions one might consciously associate with savanna-like environments (see below). By listing the food/shelter/ safety-related advantages of savannas we do not mean to imply that a conscious assessment of these advantages is what produces the initial positive emotional response. Rather, an evolutionary perspective suggests that those of our precursors who by chance responded positively to savanna environments likely spent more time in them, reaped the benefits of them and consequently left more descendants than those who were indifferent or responded negatively. Thus, a present-day legacy predisposing humans to respond positively to savanna-like environments is thought to be one among a number of influences on human emotional responses to environments. The specific nature of those emotional responses (including possible negative responding) is also a function of one's experience with environments as well as unique situational constraints.

The broader point that deserves to be reiterated is that an evolutionary analysis of factors contributing to human health and well-being includes significant interactions with nature during ontogeny and beyond, interactions that center on the emotional value of the survival capacities (e.g., food, shelter, safety concerns) of one's environment. In the next section, we examine an evolutionary model of ethics that also focuses on emotional responding.

ENVIRONMENTAL ETHICS

The historical origin of the present ecological crisis is...not in Genesis 1:28 but in the failure of people to anticipate the long-range consequences of their activities - consequences that have recently been aggravated by the power and misuse of modern technology.
— René Dubos, 1980

As the foregoing suggests, we largely agree with this assessment by Dubos, though we would add that the ability to foresee long-range consequences[4] must be accompanied by a motivation to change and the power to effect that change if ecological crises are to be averted. The power to effect change is primarily a political issue, but, understanding the motivation to change environmental behaviors requires an analysis of environmental values. In this section we will briefly summarize an evolutionary model of human values proposed by Kagan (1984) that, both in its general form and its emphasis on the importance of emotions in human value systems, is representative of many recent biological approaches to ethics, morality and values (e.g., Alexander, 1987; Pugh, 1977; Richards, 1987). We will also illustrate how this general model of ethics might be applied to environmental issues. Finally, to conclude this paper we suggest that the grounding of environmental ethics in emotional responses to environments echoes the importance of emotional responses for human manipulations of the environment, aesthetic responses and potential health effects as outlined in the preceding sections, and that an understanding of emotional responses to environments in each of these domains is facilitated by supposing an evolutionary predisposition to respond positively to savanna environments.

In *The nature of the child* (1984), Jerome Kagan has sketched an evolutionary model of ethics that highlights the importance of emotions in the development of value systems. Kagan suggests that the possibility of universal moral standards is typically rejected in favor of moral relativism because of the large variety of moral standards proposed at different times and in different places. If there is some universal set of moral standards, why has there been such a variety in articulated values? Kagan believes that though surface behaviors and specific stated standards may change, they are nevertheless grounded in "...a set of emotional states that form the bases for a limited number of universal moral categories that transcend time and locality" (1984). This belief is supported by research indicating that there is a relatively small number of human emotions present at birth (Izard, 1971; Izard & Beuchler, 1980), and that people from different cultures both experience and express emotions in similar ways (Izard, 1971; Ekman, Friesen & Ellsworth, (1972), suggesting that there is some stable set of

[4] *This essentially is Hardin's (1982) definition of ecolacy.*

emotional capacities. Kagan's (1984) research with small children indicates, among other things, that children almost invariably begin to grasp distinctions between good/bad and right/wrong between the first and second years of life. Despite this evidence suggesting that there are genetic influences on emotional capacities and the ontogenetic development of ethical standards, there is no tendency in Kagan's writings, nor is it our intention here, to propound a *normative* set of universal moral categories. Rather, we are interested in the descriptive and explanatory utility of universal emotional capacities that may underlie human value systems.

Kagan's model focuses on unpleasant emotions that accompany temptations to violate a standard. He proposes five classes of unpleasant emotions that accompany such violations: different types of anxiety in response to threatened or actual physical harm, social disapproval or task failure; feelings of responsibility when one causes harm or distress to another; feelings of fatigue or boredom after repeated gratifications of a desire; feelings of uncertainty in the face of discrepant events that are hard to understand; and, the recognition of inconsistencies among one's beliefs or between one's beliefs and behaviors. According to Kagan:

> *Because people do not like to feel afraid, feel sorry for someone less privileged, or to feel guilty, bored, fatigued or confused, these unpleasant states will be classified as bad; and people will want to replace, suppress or avoid them. The acts, motives, and qualities that accomplish these goals will be good and, therefore, virtuous. But the specific concrete conditions that provoke these unpleasant emotions will differ with time and location; and so, too, will the specific acts and qualities that suppress them (Kagan, 1984).*

Thus, a small set of fundamental human emotional states, which people are innately prone to avoid, interacts with economic, political and social conditions to produce the standards for behavior in a given society. For example, if the conditions of a given society produce anxieties about the survival of the society (e.g., in the face of an invading army) that are greater than those associated with personal safety, then physical courage will be highly valued, because it helps to assuage the former anxiety. Extending this analysis to a personal system of morals, it is reasonable to suppose as well that the vagaries of one's particular upbringing influence the specific conditions that elicit the proposed fundamental emotions and, in turn, the behaviors one considers virtuous. So, in the physical courage example, though society may highly value physical courage and personal sacrifice for the good of the community, an incompletely socialized individual, or one disaffected by limited access to society's benefits, may regard such sacrifices with significantly less élan.

Though it may not be immediately obvious, application of this model to human-environment interactions can help to explain the history of attitudes towards and manipulations of wilderness environments in the United States. Early European settlers in North America considered this continent (at least what they knew of it) to be a vast wilderness, and in their minds 'wilderness' was unambiguously pejorative (Nash, 1967).[5] William Bradford, a passenger on the *Mayflower*, described the New World as a "hideous and desolate wilderness" (Morison, 1952); and, with few exceptions, those who came after him were similarly disposed until well into the middle of the 19th century (Nash, 1967). For the past 150 years or so, however, attitudes towards nature in general, and wilderness in particular, have been changing in this country, reflecting a positive regard for wild lands that had been part of the European landscape aesthetic for the previous 150 years. These changes in attitude eventually inspired the early conservation and preservation movements in *fin de siècle* North America, as well as the more recent concerns for the environment in the latter half of this century.

Nash (1967) suggests that the primary reason for the antipathy towards nature expressed by the early European settlers was the harsh life that survival in the wilderness entailed. The procurement of even basic necessities, such as food and shelter, constituted a constant struggle with the environment, and doubtless was a major source of anxiety. Under such circumstances, Kagan's model would predict that any human qualities and behaviors that could prevent or eliminate those anxieties would be considered good and virtuous. Thus, we would expect that the clearing of land (felling trees, draining swamps), planting of crops and establishment of towns would be considered good and proper behavior towards the environment, as was indeed the case (Nash, 1967). Presently, there are different conditions that provoke food/shelter/safety-related anxieties with respect to natural environments, including wilderness environments. These conditions (environmental degradations caused by various pollutants, the depletion of resources, extinctions of species, etc.) have been well-documented over the past 60 years, and can be broadly construed to evoke anxieties about the integrity of our ecological support systems. Under these new conditions, Kagan's model would predict a corresponding shift in the motivations, qualities and behaviors regarding wilderness and other natural environments that would be needed to allay these anxieties, such as a shift towards motivations and behaviors to preserve ecological integrity, and those new motivations and behaviors would now be considered ethically proper.

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[5] *We concentrate here on European settlers and their descendants' attitudes towards wilderness partly because their attitudes have changed most dramatically in the past several hundred years (see Nash, 1967; Oelschlaeger, 1991), but also because there is good evidence that many of the aboriginal inhabitants of North America did not greatly differentiate their culture from the environments they lived in, and thus the notion of wilderness as something distinct from society had little meaning for them (Callicott, 1983; Oelschlaeger, 1991). For example, Sigurd Olson discerned no attitude among the Athabascan Cree that resembled the semantic antipodes that civilization and wilderness represented for him when he heard a distant train whistle while on a solitary camping trip (Olson, 1958). See Hardin (1982) for a cogent argument why promiscuous altruism cannot persist.*

Of course, this one example provides a limited illustration of how this evolutionary model of ethics might be applied to human-environment interactions. One limitation involves the manner in which anxiety is resolved. Because the model proposes that *anything* that mitigates the relevant fears and anxieties would be considered good, a shift towards preservation of ecological integrity is only one possible solution in this example. To the extent that "technological fixes", for instance, alleviate environmental concerns, they too will be considered good (although not necessarily by the same person). Thus, someone who takes great solace in the knowledge that "floating corrals" and genetically engineered microbes can be used to contain oil spills; that food can be grown in space (or in *Biosphere II*); or that cheap, clean, limitless fuel can be had through nuclear fusion is someone who can be expected to value technology and scientific innovation over conservation and reduced consumption. The important point here is that, though the social, economic and political conditions of a given society may elicit the negative emotional states to be avoided (and thereby help to determine the behaviors, motivations and character traits that are considered virtuous), the specific conditions of one's development and current place in life are also relevant, helping to account for the manner in which anxieties and other negative emotions are avoided.

A second limitation of this illustration concerns its focus on immediate survival-related anxieties, such as access to food, shelter and safety. Though these are the most important evolutionary concerns, both in terms of immediate physical survival as well as for general health and emotional well-being (see section on Geist above), values that have some relevance for natural environments and ecological sustainability could also be grounded in any number of the classes of emotion proposed by Kagan to underlie human ethics. Several other possibilities are listed in Table I. For the most part, the concerns and anxieties listed are self-evident, as are the corresponding motivations and emotional qualities, which are suggested as a subset of possible "valued palliatives". This table is not meant to be comprehensive, but is presented only to suggest how various concerns in contemporary societies might give rise to environmental values. Those familiar with the literature on the recreational and health-related benefits of nature (e.g., Driver & Knopf, 1976; Driver & Rosenthal, 1978; Hayward & Weitzer, 1984; Kaplan & Kaplan, 1989; Ulrich & Parsons, 1992; Ulrich, Dimberg & Driver, 1991a) will recognize the suggested palliative qualities as oft-reported goals of outdoor nature experiences that range from simply sitting in an urban garden or park to camping in backcountry wilderness. Thus, many of the values people typically ascribe to natural environments can be potentially accounted for, at least in part, by an evolutionary approach to ethics.

We do not wish to belabor discussion of this example or the particular model from which it is derived, as this model represents only one among several potentially useful evolutionary approaches. Nevertheless, the last anxiety/palliative value pair listed in Table I merits closer examination. Further

Table 1. — An illustrative list of concerns, negative emotions and anxieties commonly produced by the conditions of contemporary societies, accompanied by potential palliatives to be had through interactions with nature.

CONCERNS/ANXIETIES	VALUED PALLIATIVES
FOOD/SHELTER/SAFETY	PHYSICAL/EMOTIONAL SECURITY
RUSH/DIN OF URBAN LIFE	QUIET/TRANQUILLITY/ESCAPE
ROOTLESSNESS/ALIENATION SOCIAL ANOMIE	SPIRITUAL/RELIGIOUS FEELINGS ONENESS WITH NATURE
MILD FREE-FLOATING OR AMBIGUOUS ANXIETIES	PLEASANT AESTHETIC/CALMING RESTORATIVE RESPONSES
LONGTERM SURVIVAL CONCERNS	PROMISCUOUS ALTRUISM INTRINSIC VALUE OF NATURE

discussion will help to emphasize two important points about values clearly articulated by evolutionary approaches to ethics in general. First, human value systems enhance reproductive fitness. The human ability to discriminate between good and bad, right and wrong, friend or foe, habitable or uninhabitable environments, prey and predator, etc., and to assign value to varying gradations of these (and other) qualities bears directly on one's reproductive success. This is true of other animals as well, though they may not make all of the same types of discriminations that humans do. From an evolutionary perspective, nonhuman animals can be presumed to have value systems (if not ethical standards) based on their own reproductive fitness requirements. Thus, just as human value systems are predominantly anthropocentric, a spider's value system, for example, is presumably arachnocentric (Murdy, 1975). This latter proposition implies that value discriminations need not be conscious, nor do they require extensive cortical development, which in turn leads to the second point of emphasis about values articulated by evolutionary models of ethics: The centrality of emotions.

That environmental values may be based on human emotions is not a new proposal (e.g., Daniel, 1989; Parsons, 1991; Ulrich, 1983), but it has been largely ignored by environmental ethicists. As mentioned at the beginning of this paper, rational, logically derived nonanthropocentric support for the intrinsic value of nature has been the holy grail of environmental ethics for the past 20 years. At the bottom of Table I, we imply that the tendency to believe in the intrinsic value of nature can potentially be accounted for by Kagan's (1984) evolutionary model of ethics, which, as we have seen, is based on emotional responding. We have listed a belief in the intrinsic value of nature as one possible salve for anxieties about longterm survival, which are essentially those concerns listed in the first cell of Table I, but in some indeterminate future. Just as expanding socio-spatial circles of concern/altruism can grow to include not only one's immediate family and friends, but those of one's town, state, nation and, ultimately, all humanity, such concerns may also be extended temporally for oneself, one's descendants, and one's altruistic reciprocators and their descendants.

We have listed two common palliatives for these kinds of expansive concerns, promiscuous altruism and a belief in the intrinsic value of nature. The former term was coined by Hardin (1982) to describe the global "brotherhood of man" approach to human relations, a virtue which can be supposed to be inspired by similarly global threats to existence, such as nuclear war.[6] In a like fashion, threats to existence focused less on internecine battles and more on the integrity of ecological support systems, such as global warming, can be thought to inspire virtues such as the intrinsic value of nature. An evolutionary perspective suggests that, *ceteris paribus*, the strength of one's concerns is inversely proportional to the spatiotemporal propinquity of the objects of those concerns. In both of these cases, however, uncertainty about the spatial or temporal distance of the threats probably lends ambiguity to their perceived immediacy and thus considerable variability in the corresponding palliative virtues. Despite this variability, even the most expansive of concerns and virtuous responses can nevertheless be seen to be ultimately self-serving to the extent that they enhance reproductive fitness (or are likely to). Thus, there is no small irony in suggesting that those who lobby hardest for the intrinsic value of nature may be motivated, in part (and perhaps unconsciously), by concerns for their own reproductive fitness, an eminently instrumental goal.

CONCLUDING REMARKS

We recognize that the brief motivational assessment just presented is somewhat sketchy. We recognize too, that the particular evolutionary model we have chosen to illustrate has its limitations, such as Kagan's focus solely on negative emotions as the mechanism by which ethical standards develop.[7] Unfortunately, applications of evolutionary approaches to ethics have not been forthcoming in the environmental ethics literature. Given what is known about other human interactions with natural environments mentioned in this paper, human ethical valuation of environments is bound to be a complex phenomenon. Evolutionary models of ethics suggest that, as in those other areas, emotional responding to the environment is likely an important component of environmental valuation. One significant implication of this is that prescriptive, wholly cognitive ethical arguments for the intrinsic value of nature may be less convincing than they could be because they are addressing a limited component of human valuation systems. For instance, a spontaneously *felt* experience of oneness with nature during a wilderness outing, as is occasionally reported (e.g., Scott, 1974), may be far more effective at inculcating ecologically responsible behaviors than is an argument for the intrinsic value of nature that requires (at least a rudimentary) understanding of quantum mechanics to logically establish the subatomic oneness of matter and energy (Callicott, 1985). At the very least, it is reasonable to suggest that the persuasiveness of cognitive arguments in the development of environmental values can be enhanced by efforts to establish early emotional attachments to the land. However, given the evidence cited regarding environmental manipulations, aesthetic responding and potential health effects of natural environments, it may be the case that humans are predisposed to develop positive emotional attachments to certain types of landscapes; attachments that, once made, play a large part in determining environmental values.

As with the other areas mentioned in the earlier sections of this paper, evolutionary approaches to ethics also suggest a far richer description of instrumental values than is typically the case in the environmental ethics literature. According to Kagan's model, any anxieties that can be assuaged by interactions with natural environments, as suggested in Table I, could potentially generate ecologically responsible environmental values. Indeed, even the tendency to attribute intrinsic value to nature may itself be instrumentally valuable. However, suggesting that instrumental values may cover a broad range of human responses to environments does not imply that all instrumental values are equal, a point which has been made for intrinsic values as well (Thompson, 1983), and which is acknowledged by even the most ardent *deep* ecologists (Reed & Rothenberg, 1993). This implies, in turn, that not all environmental qualities will be equally valued. From the evolutionary perspective reviewed here, for instance, the spatiotemporal immediacy of the concerns that generate instrumental values, including environmental values, is crucially important. Unfortunately, given the complexity of ecosystems, the immediacy of many environmental concerns is neither obvious nor always easily communicated. Thus, for those concerned with encouraging ecologically responsible behaviors, a second important implication of evolutionary approaches to environmental ethics is the need for an emphasis on what Hardin (1982) has termed "ecolacy," the skills necessary to understand the immediacy of complex environmental concerns. This theoretical emphasis on ecolacy is supported by considerable empirical research indicating the importance of education as a contributing factor both to the development of environmental attitudes and the display of environmental behaviors (see Stern, 1992, for a review).

To reiterate our main conclusions: First, in each area of human-environment interactions discussed, human manipulations of environments, potential health effects of environments, aesthetic and ethical valuations, emotional responding is an important component of the interaction.

[6] *See Hardin (1982) for a cogent argument why promiscuous altruism cannot persist.*

[7] *Though this does not imply that an emphasis on negative emotions is wrong. Numerous conditioning studies suggest that "...biologically fear-relevant stimuli are prioritized for very fast processing..", which can occur in the absence of awareness, while similar effects have not been found for neutral or positive stimuli (Öhman, 1993).*

Second, the beneficial effects of human interactions with natural environments are considerably broader than the narrow economic self-interests ordinarily cited, a point which is most cogently shown in the examination of Geist's evolutionary model of health. And finally, these first two conclusions have important implications for those interested in environmental ethics, suggesting as they do that calls for a new, nonanthropocentric basis for environmental ethics are likely premature; that there is much work to be done to simply understand human value systems; and that evolutionary perspectives on human-environment interactions can contribute significantly to that understanding.

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A Planning and Analysis Process for Including Social and Biophysical Considerations in Sustainable Ecosystem Management

B. L. Driver, Brian Kent, and John G. Hof[1]

Abstract — This paper describes a land management planning and analysis process that can guide implementation of the policy of "sustainable ecosystem management" recently adopted by several federal agencies including the USDA Forest Service. This process builds on and expands considerably the planning process used recently by the USDA Forest Service to develop plans for the National Forests and Grasslands. It identifies and describes the types of analyses and social and bio-physical information for sustainable ecosystem management that is responsive to human needs for the variety of goods and services that are produced on natural ecosystems.

BACKGROUND

In a June 4, 1992 letter to the Regional Foresters and Experiment Station Directors of the USDA Forest Service, Dale Robertson, Chief of that agency, announced "...that the Forest Service is committed to using an ecological approach in the future management of the National Forests and Grasslands. By ecosystems management, we mean that an ecological approach will be used to achieve the multiple-use management of the National Forests and Grasslands. It means that we must blend the needs of people and environmental values in such a way that the National Forests and Grasslands represent diverse, productive, and sustainable ecosystems." Robertson went on to list the four "basic principles" that "will apply to the future management of the National Forests and Grasslands:

1. "Take Care of the Land"....
2. "Take Care of the People and Their Cultural Diversity" by meeting the basic needs of people and communities who depend on the land for food, fuel, shelter, livelihood, recreation, and spiritual renewal.
3. "Use Resources Wisely and Efficiently to Improve Economic Prosperity"...
4. "Strive for Balance, Equity, and Harmony Between People and the Land"... (Robertson, 1992).

Since the announcement of this new philosophy of management by the Forest Service, other federal agencies (e.g., the Bureau of Land Management) have adopted and announced an ecosystems approach to management. These new directions, especially the goal of *sustaining* natural ecosystems, have created much dialogue and study within these agencies, as evidenced by the creation of many national and regional task forces to explore various facets of what this new approach to management really means and how it can best be implemented.

The above quotes from Robertson's June 4th letter make explicit that sustainable ecosystem management (SEM) means continuation of multiple-use management to meet the land-dependent needs of people. Nevertheless, this new approach to management has stimulated much debate about how these human needs can be met and still sustain the natural ecosystems which must provide the goods and services required to meet those needs? This is a pertinent question to which this paper is addressed. The paper develops and explains a resource planning and analysis process that incorporates the types of social and bio-physical information needed for SEM.

A planning and analysis framework of the type presented in this paper is particularly timely given the need to revise existing plans that were brought on line before the policy of SEM was adopted. Our information is that about 70 of the National Forest plans would have been scheduled for revision within the next

[1] *The authors are Research Forester, Project Leader, and Chief Economist, respectively, with the USDA Forest Service, Rocky Mountain Forest And Range Experiment Station in Fort Collins, Colorado*

five years in order to comply with the NFMA planning regulations (Federal Register 1982) even if the new policy of SEM had not been adopted. It is important, therefore, that any new process complement and supplement the existing framework used in the USDA Forest Service rather than take a sharp departure. For that reason the planning and analysis process presented here builds on planning approaches currently being used by the Forest Service, including the NFMA regulations, while incorporating the new requirements for SEM.

To identify and define social information/variables that needed to be considered in SEM, we relied heavily on the USDA Forest Service's National and Southwestern Region (Region 3 headquartered in Albuquerque, New Mexico) task forces on Integrating Human Dimensions into Ecosystems Management, a Special Scientific Advisory Committee on Ecosystems Management created by Region 3, and other sources.

THE PROCESS

The sequential steps of the SEM planning process we developed will first be listed as actions that must be taken to effectively, responsively, and efficiently, integrate social values into SEM. We will then elaborate the major activities that need to be undertaken at each step. These usually analytical or evaluative activities will define most of the types of information from the social and bio-physical sciences that are needed to complete each step.

The process we describe is based on the idea that specific analytical/evaluative requirements should determine which types, what amounts, and the quality of the data that should be collected and analyzed during the SEM planning process. It is inefficient to collect data if they contribute only marginally to the accuracy and reliability of the result of any quantitative analyses or the overall quality of the plan.

Steps of the Process

To present a comprehensive overview of our SEM planning and analysis process, we first list each step without elaboration. Most of those steps are self-explanatory. Where they are not, the purposes of each step should become clear in the next section where the activities needed to be undertaken at each step are elaborated.

1. Identify Need for Plan Revision [Purpose and Need]
2. Identify Constraints on Planning Process [Define Planning Criteria]
3. Identify and Organize Groups That Will Be Involved in The Planning Process
4. State General Planning Goals Within SEM Framework
5. Evaluate the Planning Agency's Institutional Setting and *Modus Operandi*
6. Evaluate the Local Social Context
7. Define and Evaluate Goods and Services
8. Evaluate Biophysical Conditions and Requirements of the Ecological Units
9. Evaluate Capabilities and Suitabilities of the Ecological Units to Supply Goods and Services. [Analysis of Management Situation]
10. Define Range of Desired Future Conditions of the Ecological Units
11. Develop Alternative Plans. [Formulation of Alternatives]
12. Visualization of Consequences of Alternative Plans with Public Education About Pros and Cons of Each Plan [Evaluation of Alternatives]
13. Prevent, Mediate, and/or Resolve Conflicts
14. Select the Plan to be Recommended for Implementation [Draft Forest Plan and EIS]
15. Public Review of the Recommended Plan
16. Develop and Propose The Plan That Will be Implemented [Final Forest Plan and EIS]
17. Implement Plan
18. Monitor and Evaluate Plan Implementation Results

The reader familiar with the steps of national forest land management planning that are described in "the Regs" (Federal Register, 1982) can see that we have kept all of those steps, as indicated by the selected titles of some of them in brackets above. The following section shows that we also expanded and/or emphasized the activities required in those "old reg" steps, especially the ones concerned with public involvement and participation. In addition, we have added new Steps Nos. 3, 5, 6, 8, 10, and 13 with the purposes of those additions being to: assure better involvement of all "stakeholders" and prevention and early resolution of conflicts; better public understanding of the planning process and the feasible alternative plans; incorporation of the goal of achieving SEM; and better integration of the human dimensions of SEM into the process. These modifications and additions will be clarified in the following section.

Elaboration of Activities Required at Each Step

In this section activities that comprise each step of the process are elaborated.

1. Identify Needs for Plan Revision

This step is identical to Step(b) of the National Forest planning process that is described on page 43044 of the Federal Register (1982). The major tasks are to identify and define the needs (i.e., issues, concerns and opportunities) that drive the planning process. Those needs include those that are identified by the public as well as those identified by professional land managers. Effective identification and definition of public needs depends on efforts by the planning agency to involve all "stakeholders" in the planning and management of the public lands through the creation and nurturing of what has been called "collaborative partnerships" (Bruner, 1991), where trust and mutual respect must be established. This should be an on-going process of "public involvement" rather than attempting to get "public input" at the time of plan revision. These commitments and efforts are necessary not only to assure comprehensive and explicit identification of the needs for planning analyses but to help prevent and resolve conflicts—and costly litigation—early in that process. Along this line of reasoning, Wondolleck (1988) suggests these five ways to help prevent conflicts: build trust, build understanding, incorporate conflicting values, provide opportunities for joint fact finding, and encourage cooperation and collaboration. On-going collaboration with all stakeholders will help accomplish these five ways of preventing conflicts.

2. Identity Constraints on Planning Process

This step requires an evaluation of the constraints that will establish bounds for the planning activity and analyses. Included are legislative and administrative directives such as the Forest Service's legal and administrative directives to help maintain and improve the stability of rural resource-dependent communities or protect sacred sites of American Indians. Particular concern should be given to the planning units' current and future likely fiscal resources; the plans should be realistic and not recommend actions for which adequate funding is unlikely. This step should also address the institutional and other constraints on practicing SEM; for example, does the planning agency administer the land areas that comprise particular ecological units or only portions of them? If so, can cooperative arrangements with the other owners facilitate SEM or not? Any constraining effects of agreements with other parties must also be considered. Criteria for identifying data requirements and analysis methodologies are also developed in this step.

3. ID and Organize Groups That Will Be Involved in the Planning Process

This step is the first opportunity to bring outside interest groups into the process in a meaningful way. It involves setting up the planning team, with clear assignment of responsibilities, and creation of one or more vehicles to facilitate public involvement such as a citizens' advisory group. The planning team should include representatives of different disciplines/land uses. An advisory group should include members that represent different interests and perspectives, including people/groups who do not always agree with the planning agency's policies and practices. It is essential to use this step to bring all interested parties into the process early and begin work to develop their "ownership" in the planning effort and its results.

4. State General Planning Goals within SEM Framework

This step calls for cooperative work on the part of all interested parties to arrive at a general set of planning goals. Following terminology adopted for Forest Service land management planning "a goal is a *concise* statement that describes a *desired condition* to be achieved in the *future*. It is normally expressed in broad, general and nonquantitative terms that are *timeless* in that there is no specific date by which the goal is to be achieved. They may be specified on a forest-wide basis, for sub-forest areas or on a local planning area basis." As an example, one important goal is to achieve SEM. Another might be to preserve the land-dependent lifestyles of local indigenous sub-cultures—to the extent possible. These goals should be limited in number and relate directly to issues, concerns and opportunities identified previously.

5. Evaluate the Planning Agency's Institutional Setting and *Modus Operandi*

This clearly new step was added to assure that both the agency personnel and members of the public involved in the planning effort clearly understand the institutional context within which the planning and analysis will be conducted, especially the resources available to do the planning. This process should examine questions such as the following:

- Does the planning unit use the latest technology and modern management science including planning and optimization techniques?
- Has the planning unit been active, innovative, and successful in public involvement activities?
- What are the agency's polices and practices regarding maintaining and promoting rural community stability and development, eco- and heritage tourism, and maintenance of resource-dependent lifestyles of local residents such as subsistence users of the public lands?
- Does the planning unit actively promote partnerships and the use of volunteers?
- Is the planning unit active and successful in negotiating interagency agreements?
- Has past management and use emphasized commodity or amenity uses or both?
- Have adequate fiscal resources been made available?
- Has the planning unit been the subject of much litigation?
- Are the local people and interest groups generally supportive or adversial?
- Does the agency have a good reputation, and is it respected locally?
- Has the planning unit had "good press"?

- Does the planning unit emphasize efficiency of its operation?
- Does the agency and/or the planning unit endorse an explicitly stated land management ethic?
- What is the planning unit's commitment to, and record, on civil rights and equal employment opportunities?

Answers to these questions should reflect actual conditions and not good intentions.

6. Evaluate the Local Social Context

Similar to Step 5, the purpose of this new step here is to gather *contextual* information necessary for the planning unit to understand and work effectively with its local constituents. The social context posses questions about both the historic and present social conditions of the local area that will be impacted by the plan being developed. Historic information includes identification of any Traditional Cultural Properties of the American Indians and of national and local heritage sites, understanding of historic cultural uses by all sub-cultures, and being familiar with the past history of the planning agency's management of the area and the land use history of that area including the history of settlement and significant changes in land use.

Questions directed toward understanding the existing social context include

- What are the important local institutions and the nature of the local social infrastructure?
- Are there highly resource-dependent local communities?
- What is the industrial mix or economic diversity of the region?
- What are the market areas for the goods and services demanded from the planning area?
- What are the spatial locations of local major sub-cultures?
- Are there local tourist destination areas, and how significant are they to the local economy?
- What are the "demographics" of the local area regarding age structures and trends in population growth?
- What are the key local economic indicators regarding the economic structure/industrial diversity, employment and unemployment levels, income levels and distributions and economic trends?
- Is there local community cohesion (unity and cooperation) and stability (ability to absorb and manage change)?

7. Define and Evaluate Goods and Services

The "needs" assessment called for here is one of the most important, albeit difficult, activities of the process. As mentioned above, the most complex part of SEM is delivery of as wide a spectrum of goods and services from the natural ecosystems as is feasibly and economically efficiently possible while assuring that the ecosystems maintain their health, evolutionary physical integrity and capacity to deliver goods and services to meet human needs over the long run. This complexity cannot be understood and addressed unless an accurate and reliable needs assessment is made for all of the planning unit-dependent goods and services. The list of those needs includes those for the so-called commodities and amenities. Examples include traditional subsistence uses as well as public desires and expectations for: recreation opportunities including the setting attributes, facilities, and managerial programs, that facilitate particular types of recreation activity; both eco- and heritage-tourism accommodations and sites; scenic resources and scenic by-ways; interpretative, educational services, and health and safety related services; and opportunities to hunt and fish and to learn and do scientific research. Included too are requests from local communities for help in maintaining their stability and economic vitality and for a wide variety of special uses. Some of these are difficult to define much less quantity. They include uses related to spiritual, religious, and other hard-to-define human values such as maintenance of a particular lifestyle or sub-cultural tradition. At times, limited information will not permit accurate assessment of some types of uses that must be considered. In those cases, recourse must be made to professional judgment. While we encourage use of accurate and reliable data, we caution against very expensive surveys that only *marginally* increase the accuracy and validity of the data.

8. Evaluate Biophysical Conditions and Requirements of the Ecological Units

The purpose of this step is to gather the data necessary to define the biophysical states, requirements, and conditions of the natural ecosystems that will be impacted by the plan. We offer the following recommendations which are derived largely from personal conversations with ecologists in the Rocky Mountain Forest and Range Experiment Station and in the Regional Office of the Southwestern Region (Region 3) of the Forest Service.

- Referent bio-physical conditions must be established that will establish clear criteria of the desired structure and functioning of the ecological units. These criteria must establish at least broad ecological objectives for specific ecosystems or related sets of ecosystems. These criteria and ecological objectives will be used as tests of ecosystems sustainability.

- Once the biophysical-based referent criteria and objectives have been established, inventories and evaluations must be made to determine which ecosystems and subsystems meet those criteria, which do not—or likely will not in the near future—and what actions are needed to meet the criteria if and when they are not met.
- Only biophysical data that is clearly needed should be collected. In line with this reasoning, we endorse the recommendation in the preliminary report of the Forest Service's Region 3 Committee of Scientists that two types of biophysical "ecosystems needs assessment" be made in sequence[1]. The first, a "course filter analysis" will be made to identify areas needing more in depth analyses. Then a "fine filter analysis" will be made for those critical or sensitive areas where problems (e.g., threatened or endangered species) exist or are likely to exist regarding the sustainability of the ecosystem because of those problems. The report of the R-3 Committee of Scientists states that the Nature Conservatory has estimated that 85-90 percent of all species can be protected by the course filter analyses (Hunter 1991), which is much less costly to conduct than the fine filter analyses.

9. Evaluate Capabilities and Suitabilities of the Ecological Units to Supply Goods and Services

This step is similar to Step(e) ("Analyses of the Management Situation") of the National Forest planning process as described on page 43044 of the "Federal Register" (1982). As stated there, the purpose "...is to determine the ability of the planning area...to supply goods and services in response to society's demands...to provide a bases for formulating a broad range of reasonable alternatives." Optimization modeling will play a key role in making these determinations. These evaluations will include documentation of the current level of goods and services provided and whether the desired and projected levels of goods and services can be provided in the future. While ecosystem sustainability must be kept in mind, the primary orientation in this step is to determine whether the resource base is physically capable of and suitable for such production. Thus, a full range of management alternatives for each ecological unit will be considered in this step, with the issue of sustainability being the focus of Steps 10 and 11. This analysis will include broad scale landscape assessments that will identify landscapes that are unusually sensitive to use, the visual qualities of the landscapes, landscapes with risk factors (e.g., flood plains, those highly susceptible to human-caused wildfires), the accessibility challenge levels under different amenity uses; the locations and scope of structures made by humans; Recreation Opportunity Spectrum and Visual Quality Objective classifications; special classifications (e.g., designated wilderness, scenic or recreation areas); and unique land forms and recreation sites. The new Forest Service Handbook on Scenery Management (now in draft but to be published soon) provides excellent guidance for these landscape assessments.

10. Define Range of Possible Desired Future Conditions of the Ecological Unit

This step is central to SEM because it uses public participation to integrate the results of Steps 7 (Needs Assessment), 8 (Biophysical Evaluation) and 9 (The Capability and Suitability Analyses) to develop general future scenarios for the ecological units. An important consequence of this will be the establishment of broad guidelines about what types of management actions can be taken within each ecological unit or related set of units. These guidelines have come to be called "desired future conditions" (DFCs). While this is a useful concept at some level of generalizability, we emphasize that DFCs cannot specify future ecological conditions that will in fact exist over considerable time; we do not have that knowledge about complex ecological processes. Therefore, DFCs can only target general conditions for the near future.

In general, the set of management actions that will be acceptable under these DFCs will be more restricted than the list considered in Step 9. Here, the needs assessed in Step 7 must be screened through the set of biophysical criteria established in Step 8 to assure sustainability of the natural ecosystems on which the targeted goods and services would be provided. The needs so screened will include those related to production of outputs such as timber, forage, etc, maintenance of desired lifestyles, maintenance of local community stability and economic vitality, and all other needs assessed in Step 7.

While achieving SEM is a clear underlying goal of the planning process being proposed, there might be instances where some desired future conditions will represent compromises between some measure of desired ecosystem sustainability and meeting of societal needs. For example, it is quite possible that not all ecosystems will be managed to preserve all species of flora or fauna so long as enough ecosystems are maintained to do so. Or some short-term, undesirable ecological impacts might be tolerated to sustain a particular societal need.

11. Develop Alternative Plans

In many ways, the preceding steps are preparatory for this step. Here (as in Step 9) optimization techniques from modern management science will be applied to help develop a set of planning area-wide alternative management plans which will

[1] Personal communication with Dr. Merrill Kaufmann, a member of that committee.

allocate each ecological unit within the planning area to one or more types of management. Information from previous steps is utilized to define sets of possible management actions for each ecological unit. While these sets of options may vary from alternative to alternative on any given unit, they always must be consistent with the objectives of SEM as represented by legal requirements and standards and guidelines. They also should be consistent with long-term goals as defined by the desired future condition scenarios developed in Step 10.

In some ways, this step functions much as it did in the first round of National Forest planning under NFMA, where linear programming models were developed using the FORPLAN system (Kent et al., 1991). However, major criticisms of these early efforts related to their inability to properly account for ecological concerns such as the spatial arrangement of management treatments on the ground or species biodiversity. Recent developments (Hof and Joyce, 1992, In Press; Hof et al., In Press) demonstrate the feasibility of incorporating these ecological considerations in optimization analyses. These improved capabilities are being incorporated in a new optimization system being developed at the Forest Service's Rocky Mountain Forest and Range Experiment Station. When development is complete, this system will permit the formulation of optimization models that incorporate wildlife species dispersal, habitat fragmentation, edge effects and ecologically based objective functions such as those measuring biodiversity. Linear programming, mixed integer programming and multi-objective function formulations also will be incorporated.

12. Visualization of Consequences of Alternative Plans

This is a public involvement and public education step oriented toward achieving a clear understanding of the consequences of each of the alternative plans developed in Step 11. Basically the purpose is to help the public, and the agency professionals, obtain a clear image of what the visual and other conditions of the ecological units will be at future dates if each alternative plan were implemented. This should enhance the rationality of the process of selecting a preferred plan and help prevent conflicts among individuals and groups that hold different values. While new techniques must be developed to facilitate better visualization of the consequences of possible alternative management actions, much progress has been made recently in that technology, which goes beyond the conventional use of maps, pictures, and narrative statements. These include use of digitized photographs and video tapes in which proposed developments have been dubbed in, use of game and other simulation techniques, of computer-based learning techniques, and of visual representations of the impacts of the proposed actions that have been implemented elsewhere.

An advantage of the use of optimization modeling as described in Step 11, is that it provides much information characterizing the effects of each alternative generated. Recently, optimization analysis software has been ported to microcomputers (Kent et al., 1992), and much supplementary software exists on these computers to further facilitate the process of characterizing alternatives. See Ager et al. (1991) for examples of how this was done in Region 6 of the USDA Forest Service during the first round of forest planning. Much of this information can be used to further supplement the visualization process.

An important part of this step is public education, not only about the consequences of each plan but the technical and other (e.g., political) reasons why each management action was proposed. This has become increasingly important in our society for which the latest national census of the population shows that at least 80% of the people reside in essentially urban areas. Although national and regional opinion polls show that the levels of environmental concerns of these citizens remain high and are increasing, research has also documented a negative correlation between population of place of residence and amount of objective knowledge about principles of resource management. For example, people in larger cities, on the average, have responded that hunting, not loss of habitat, is the major cause of decline in populations of wildlife. Put simply, although subjective concern is high, objective knowledge of the public is not always at a level that facilitates objective discourse about alternative natural resource management actions. All public land management agencies face this important educational challenge—one which will require more than public education about proposed management plans and include off-site education and possibly more work with the elementary and secondary school systems.

13. Prevent, Mediate, And/Or Resolve Conflict

Much of the tasks inherent in the title of this step should be accomplished during the previous steps, especially the prevention of conflicts and their mediation if and as they arise. Steps 1, 3, 4, 5, 6, 7, 10, 12, 13, 15 and 18 each provide opportunities for "conflict management" with Steps 3, 4, 10, 12, 13, 15 and 18 explicitly oriented toward it. When conflicts remain after all of these efforts, we recommend that professional mediators be hired rather than attempt to resolve the conflicts using mediators without adequate skills in mediation. For example, much progress has been made recently in social-psychology on persuasive communication and attitude change, but most professional resource managers do not have the training—or even the dispositions—to be effective in using this information. The costs of hiring consultants with this training generally is likely many orders of magnitude less than the dollar costs of litigation and the other social costs of unresolved conflicts. During this process of conflict management, it is quite likely that some compromised changes must be made in the desired future conditions established for each ecological unit in Step 10.

14. Select the Plan To Be Recommended For Implementation

This step follows logically from the previous analyses and consists of incorporating information obtained in Steps 11, 12 and 13 into development of a recommended plan. These efforts frequently require additional quantitative evaluation such as sensitivity analyses and the development of additional alternative plans. The final product of this step will be the draft plan and appropriate NEPA document.

15. Public Review of the Recommended Plan

This step seems self-explanatory, but opportunities should be taken here for conflict prevention and resolution. Also, if new issues arise after Step 13 was completed, they may be addressed during this step. The amount of effort needed here should be inversely proportional to the degree to which Wondolleck's (1988) five steps of conflict prevention were followed previously in the planning process.

16. Develop and Prepare The Plan That Will Be Implemented

This step involves making changes in the recommended plan that are needed because of the Step 15 public review. As with Step 14, additional alternative plans may need to be developed to respond to input received during Step 15. The final product of this step will be the final plan and appropriate NEPA document.

17. Implement Plan

Plan implementation is a complex process and the way it is carried out will determine the degree to which SEM principles are incorporated into actual management. We discuss this further in a later section of the paper.

An important part of plan implementation is project-level planning and implementation. While the Forest Service has a detailed system (i.e., Integrated Resource Management) for guiding project-level planning, that system was developed before the policy of SEM was adopted. Therefore, to help assure that project-level planning does result in SEM, the ecological filters referred to in Step 8 need to be applied in project-level planning to determine the impacts of the proposed project on the biophysical requirements and conditions of the ecosystems in which the project is located. Also, the impacts of each proposed project on the desired future conditions, established in Step 10, must be evaluated, and these impacts might cause these projects to be revised.

18. Monitor and Evaluate Plan Implementation Results

The concept of adaptive management, where the results of plan implementation are continuously monitored and evaluated, is a key component of SEM. These activities are necessary to ascertain the health of the ecosystems being managed and identify if management direction needs to be changed by amending or revising the forest plan. Because monitoring and evaluation for the first round of forest plans was not carried out with SEM in mind, much work needs to be accomplished to identify what really needs to be monitored to assess the compliance of management with SEM. Certainly, the referent bio-physical criteria (Step 8) and broad desired future conditions (Step 10) will provide useful guidelines for monitoring to achieve SEM.

Process Summary

While our expanded planning and analysis process seems complex and arduous, it really should help simplify that process in some ways by making explicit which types of analyses and data are needed for each step. The process shows which types of social and bio-physical data are needed and where they are needed.

INFORMATION/ANALYSIS STRUCTURE OF A DECISION SUPPORT SYSTEM FOR SEM

In the foregoing discussion, we have made numerous references to the need to utilize data, information and a variety of analysis tools to support SEM. In this section, we group key information and analysis components of a decision support system (DSS) designed to support SEM, briefly discuss those groups, and relationships between them. Our premise here is that while a key component of SEM is participatory decision making with each stakeholder having equal representation in the process, so much information needs to be obtained, managed, and analyzed to provide a context for decision making that the variety and scope of those data requirements becomes practically incomprehensible without some type of systematic organization. Our purpose here is to offer such an organization.

Figure 1 shows an organization for such a system with 6 major components identified in boxes and information flows or linkages identified by arrows. Three of the components, Resource Production Models, Information Systems and GIS relate to data and information. The Analytical Engine is the heart of the DSS and relies primarily on optimization techniques to conduct trade-off analyses that incorporate interactions between ecosystems, the resources they produce and the management of them. The Participatory Decision Making component utilizes

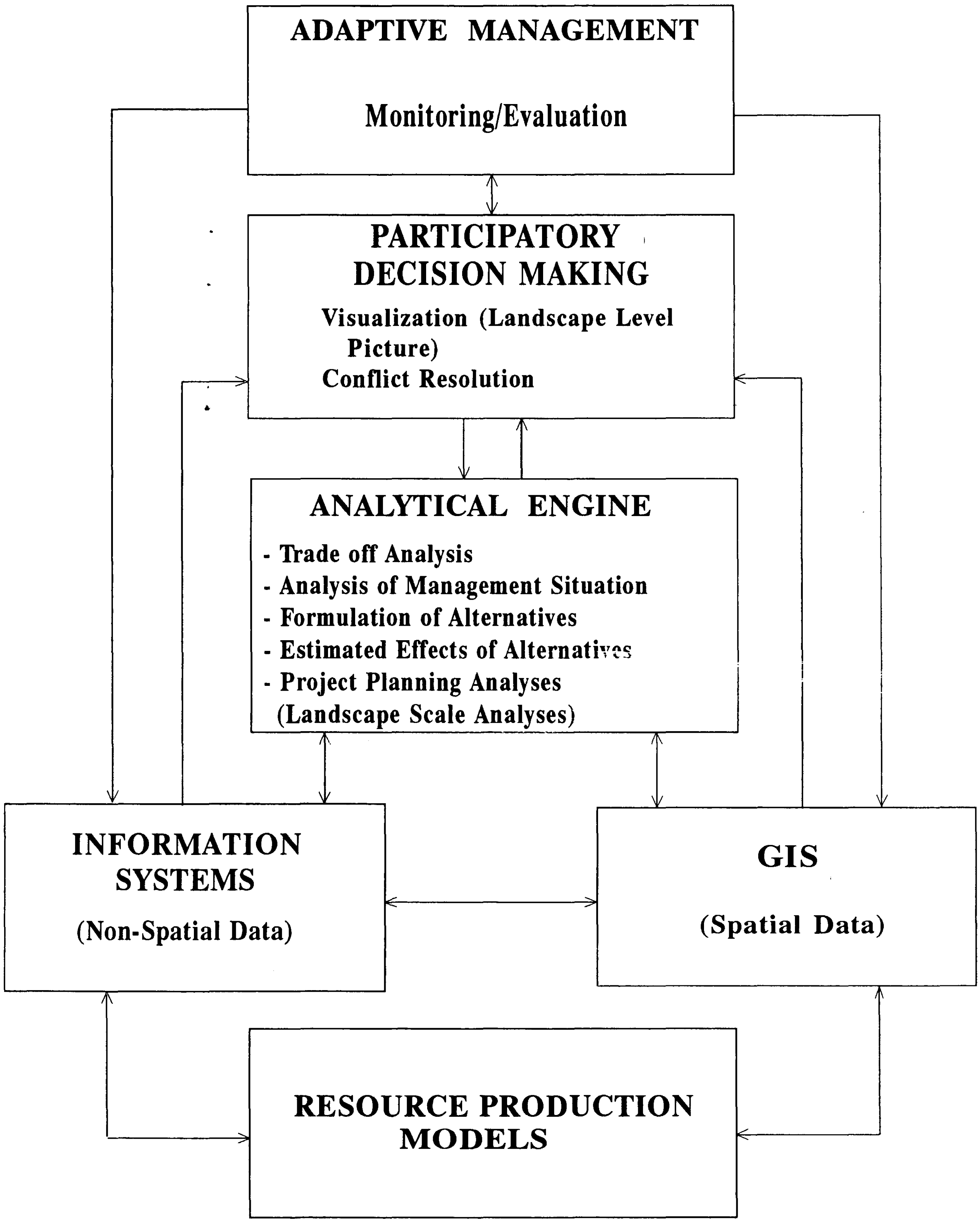

Figure 1. — Information/Analysis components for an ecosystem management based decision support system.

information from the analytical engine and elsewhere to provide input to all interested parties. The Adaptive Management component incorporates the monitoring and evaluation activities.

Data and Information

Reliable information of many kinds is essential to effective SEM. The collection and storage of accurate data about ecosystem conditions and trends are essential to sensible decision making (Steps 1, 2, 4, 7, 8, 9) and support all the other components of the DSS system. For our purposes, we categorize data as either non-spatial or spatial. Examples of non-spatial data include resource output levels under different forms of management and environmental and economic accounting information. Under ideal conditions, non-spatial data would reside on one or more relational data bases designed for consistent data storage formats across the agency and for easy access by other modules in the DSS. Unfortunately, in reality the situation in the Forest Service falls far short both in terms of data reliability issues and in terms of data base software issues.

Spatial data (sometimes referred to as mappable data) includes all information that has a spatial context. Examples include land area by vegetation type, slope, soils or other attributes; location of road networks, ownership boundaries or other linear features; and water bodies, mineral deposit sizes and locations. Geographic Information System (GIS) software systems are designed to link to a relational data base that is used to store spatial data in much the same way that non-spatial data are stored. GIS can also be used for certain types of data analyses, especially those that facilitate landscape scale and spatially oriented analyses. Examples include calculation of areas of suitable habitat for selected wildlife species, and amount of habitat edge (boundary between different habitat types), that would result from different management scenarios. Unfortunately many of the data quality problems that exist for non-spatial data exist for spatial data as well.

Interrelating these non-spatial and spatial data in meaningful ways will simplify the complex decisions that SEM implies. As noted above, past efforts at planning analysis within the Forest Service have been criticized for their inability to address the spatial implications of management; i.e. the implication for wildlife, water quality, sediment production, and many other issues of how various management actions such as timber harvesting and minerals extraction are located on the ground. Most of the earlier efforts did not utilize GIS, consequently the incorporation of this technology, along with improved formulation of optimization models as discussed below, offers considerable promise.

To conduct tradeoff analyses, information on resource output levels resulting from management practices is necessary. Ideally, the most reliable information of this type should come from Resource Production Models, the component identified at the bottom of figure 1. Because the USDA Forest Service manages portions of many large and complex ecosystems, this information is often unknown and must, where possible, be estimated. A variety of statistical techniques and computer simulation models have been developed to provide this information and much of the non-spatial data mentioned above is produced by these models. Unfortunately there are many implications of human management of ecosystems for which no derived or estimated production functions exist; these knowledge gaps increase the difficulty of conducting tradeoff analyses that are ecologically sensitive.

Three major data and information problems exist, the first being that many of the linkages between the Information Systems and GIS components shown in figure 1 still need to be developed. The second is that while the hardware and software technology needed for the information components exists, no standardized computing platforms and software have been selected and made available agency-wide within the USDA Forest Service. The third is that our knowledge of resource production models is incomplete, with much additional work being needed in their development and testing. While progress is being made on all of these fronts, it will be several years before a corporate information management system is fully in place.

Analytical Engine

Connecting the components in figure 1 into an overall DSS system should make it possible to trace and quantify indirect cause of change. However, the number and complexity of such indirect linkages easily overwhelms one's perceptual capacities, requiring the use of systematic analytical techniques to conduct the trade-off analyses that provide decision makers with the information they need. The analytical engine identified in figure 1 is the most important component of the DSS. Information from the three information components feed the system incorporated in the engine. Systems in the engine utilize this information in conjunction with information on management activities, constraints and ecological or other objectives to formulate alternative plans (Steps 9, 11, 14, 16).

Past plan alternative development efforts in land management planning have relied primarily on linear programming optimization modeling using the FORPLAN system (Kent et al., 1991). As noted above, criticisms of these efforts related to the inability to properly account for spatial interactions and the utilization of purely economic objectives. Recent work (Hof and Joyce 1992, In Press; Hof et al., In Press) suggests that mixed integer formulations offer considerable promise for incorporating spatial issues in optimization-based tradeoff analyses. Other work (Hof and Raphael, In Press) demonstrates the feasibility of incorporating ecological metrics such as measures of biodiversity into these analyses, thus increasing their utility as tools to support ecosystems management. Collectively, this work and other, ongoing efforts offer great promise for future optimization modeling to look at resource tradeoffs within the context of SEM.

Information obtained from optimization analysis results can be used to estimate effects of alternative plans. This effort can be further augmented through the use of other analysis techniques such as Input/Output modeling (Taylor et al., 1993).

The increased ability to incorporate spatial realities using mixed integer models will enable the analysis of tradeoffs resulting from different locations on the ground for resource production activities such as minerals extraction and timber harvesting. Past efforts to do this took place externally to the optimization analysis. However, there is a tradeoff due to the greater difficulty in solving mixed integer models as opposed to solving LPs, and due to the inability of humans to make sense of spatial analyses over parcels of land larger than, say, a watershed or viewscape. This suggest that spatial optimization analyses may be conducted for portions of national forests rather than for entire forests (which typically comprise 1-3 million acres) as was typically done with FORPLAN generated LP models. If so, these analyses will be used to support plan implementation (Step 17).

Participatory Decision Making

In the Participatory Decision Making component, data and information would be utilized in the earlier steps of our process (Steps 4, 7, 8) to assist in setting planning goals and in developing desired future conditions. The results of these efforts are transferred in the form of constraints and objectives to the optimization models developed using the Analytical Engine. Thus, they serve as inputs to the optimization analysis in order to help define the plan alternatives that are developed (Steps 9, 11, 14, 16). Visualization techniques can be utilized here to help in the evaluation of alternatives and in conflict resolution activities (Steps 3, 12, 13).

Adaptive Management

Adoptive management is widely recognized as a key component of SEM (Wondolleck 1988). Within the context of both our planning process and the NFMA regulations process, monitoring and evaluation (Step 18) is the adoptive management step. Annual monitoring and evaluation reports are required by agency planning directives, and are intended to provide a basis for assessing annually, the need to amend or revise the forest plan. To be consistent with SEM, increased attention will need to be devoted to monitoring ecosystem states (health) in addition to stocks and flows.

SUMMARY

The planning and analysis process developed to help assure sustainable ecosystem management expands the planning process used by the USDA Forest Service to develop plans for the National Forests and Grasslands. Those expansions focus on four areas.

1. Better public involvement, especially the on-going formation and nurturance of "collaborative partnerships" with all stakeholders to assure representation of their interests and ideas in the planning process, to make these stakeholders feel they have real ownership in the process, and to prevent conflicts.
2. Better incorporation of social and economic information into the SEM planning process.
3. Explicit consideration of variables and types of analyses needed to help implement the new "ecosystems management" policy of the USDA Forest Service. These include establishment of referent bio-physical criteria, ecological objectives, broad desired future conditions developed with public involvement, and application of course and fine filter ecological assessments.
4. Development and use of improved techniques to inform and educate the public about the nature, scope, and consequences of alternative management actions that can be taken. Particular attention here was given to needs to facilitate better visualization of the likely consequences of alternative plans both by the public and the planning agency's personnel.

We identified the various types of information and analyses needed for SEM, and we categorized the types of information and analyses needed for SEM and the relationships between them (figure 1).

We did not make specific recommendations about the types of professional skills needed for the analyses called for in our process. Our discussion of that process does make explicit the need for highly developed skills from a variety of disciplines especially the quantitative, managerial, and other social sciences including the social-psychology of conflict prevention and management. While schools of forestry are trying to change undergraduate and graduate curriculum to meet these needs, considerably more change is needed if SEM is to be practiced in a way that adequately integrates social and bio-physical information.

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Banquet Address:

The Biosocial Perspective

Thomas M. Bonnicksen[1]

Abstract — Resource managers act as mediators between a society and the physical environment from which people derive their resources. Managers urgently need a new perspective to perform this mediating role more effectively. The biosocial perspective satisfies this need by providing a theoretical foundation, an organizing framework (the biosocial model) and a process for managing relationships between people and their environment (The Impact Process).

THEORETICAL FOUNDATION

Conventional Perspectives

Environmental and Cultural Determinism

Two contradictory perspectives about society's relationship to its environment can be traced back to ancient Greece. The first perspective — environmental determinism — assumes that the physical environment exerts a controlling influence over society. The second perspective — cultural determinism — assumes that society controls its environment more than the environment controls society. Both ancient perspectives are at least partially correct as explanations for the relationship between people and their environment. The defect they share is their reliance on the assumption that this relationship operates in only one direction. Most scientists know that this assumption is flawed — people interact with their environment. Nevertheless, these ancient perspectives have persisted and developed new meanings in contemporary society.

Today some people advocate biocentrism as a perspective, which replaces environmental determinism. In this case, biocentrism does not try to explain relationships between people and their environment, it provides a normative mandate that dictates how people should act toward their environment. Biocentrism considers the earth or the environment as either the master of society or a deity that should be worshipped. Thus biocentrism means that satisfying human needs is less important than preserving the environment and protecting other species. Similarly, some people advocate anthropocentrism, which replaces cultural determinism. Anthropocentrism considers the earth or the environment as a servant or slave of society that should be exploited to serve human needs. Thus anthropocentrism means that satisfying human needs is more important than preserving the environment. Biocentrism and anthropocentrism share the flaws inherent in environmental and cultural determinism. It is foolish to think that people will sacrifice their own welfare on behalf of other species or that they will knowingly modify the environment in a way that jeopardizes human survival.

The Ecosystem Model

The ecosystem model represents another conventional perspective for organizing society-environment relationships. Its strength lies in an explicit recognition of interactions between a society and its environment. This model serves both scientific and normative purposes. Scientists use the ecosystem model to explain physical, chemical, and biological relationships between humans and their environment. Thus it is a scientific perspective that uses the machine as a model for society-environment relationships. Some people also advocate using the ecosystem model as a normative guide for governing society-environment relationships.

The ecosystem model has limited usefulness because it reduces humans to parts in a machine. It ignores the human capacity for foresight and abstract thinking. The ecosystem model also can only be applied to a specific geographical location. It is a four-dimensional model that includes three dimensions in space and one dimension in time. Therefore, an ecosystem is a quasi-mechanical system located in an arbitrarily defined volume of physical space at a particular time.

The ecosystem model cannot adequately organize society-environment relationships in industrial societies. The social boundaries of industrial societies, as defined by

[1]*Professor, Department of Forest Science, College of Agriculture and Life Sciences, Horticulture/Forest Science Bldg., Texas A&M University, College Station, Texas*

information networks, do not coincide with the boundaries of the ecosystems they manage. The ecosystem model also lacks explicit recognition of decision making processes. Industrial societies have communication and decision making networks that make them difficult to confine within anything less than a global ecosystem. The ecosystem model is inappropriate for representing society-environment relationships when decision making cannot be confined to the same geographical area as the environment being managed. The biosocial perspective provides an alternative framework that avoids the constraints of the ecosystem model.

The Biosocial Perspective

The biosocial perspective assumes that a culture does not have complete control over its physical environment anymore than the environment controls a culture. It is axiomatic that a society and its physical environment adapt to one another. The relationship between them is reciprocal. Each is produced and maintained by interacting with the other. In other words, the biosocial perspective assumes that a process of interdependency exists between a society and its physical environment. The biosocial perspective also assumes that humans are the dominant force in modifying the environment. Instead of considering the earth as master or deity, or as servant or slave, the biosocial perspective visualizes the earth as home and garden. In other words, home and garden are the same place. Thus, the power to cultivate and change carries with it the responsibility to exercise that power with wisdom and responsibility.

THE BIOSOCIAL MODEL

The biosocial model is a simplified representation of the biosocial perspective. It is a generic model that accommodates a variety of resource management issues. Unlike the ecosystem model, there are no geographical restrictions. The boundaries of the biosocial model encompass only the parts and relationships that are useful for addressing a particular management issue.

Subsystems

The biosocial model is composed of four parts; the management subsystem (a society), the ecological subsystem (its physical environment), and the inputs and outputs that tie them together (Figure 1). Because human society is self-aware it is separated from the physical environment. Humans in the management subsystem can consciously modify their environments and social relationships to adapt to changing conditions. The ecological subsystem is not self-aware, so biophysical laws and fixed relationships limit its ability to adapt to the management subsystem.

The management subsystem is composed of stakeholders, agents, and the larger society. Stakeholders are organizations with direct access to natural resources, such as timber companies and hikers. The larger society is composed of organizations with indirect access to natural resources, such as consumers of wood products, Congress and the courts.

The ecological subsystem is composed of primary resources and the larger biophysical system. Primary resources are the key parts of the ecological subsystem. In the biosocial model, a primary resource is a physical object, or a collection of objects, that is valued by a stakeholder, such as trees and wildlife. If a stakeholder does not value an object as a primary resource, it becomes a secondary resource. Secondary resources, such as soil, are part of the larger biophysical system because they are essential for producing primary resources.

Agents

Agents occupy the central position in the biosocial model because they act as mediators between stakeholders and other organizations, and between those organizations and their physical environment. Agents, such as the US Forest Service, manage the ecological subsystem directly to enhance the value of certain resources. In addition, agents manage resources indirectly through persuasion or regulations that control the resource use practices of stakeholders.

Agents play a pivotal and difficult role in resource management. Some agents accept full responsibility for difficult decisions and use their authority to make choices on behalf of stakeholders. This method of decision making, which is called authoritative control, becomes more hazardous as issues grow in complexity. Other agents avoid making difficult decisions by relying on such methods as technological control in which computer programs and other formulas prescribe courses of action. This method substitutes science for human values. Managers can also rely on market control and allow supply and demand to set the prices that influence stakeholder choices. Finally, managers can avoid making decisions by using ideological control and allowing the preferences of a dominant stakeholder to dictate choices to other stakeholders. The biosocial perspective, however, does not rely on authority for making decisions nor does it rely on methods designed to avoid making difficult decisions. The biosocial perspective assumes that many resource management issues are best resolved using cooperative control in which agents and stakeholders work together as partners to formulate and carry out decisions. The Impact Process formalizes and simplifies cooperative decision making.

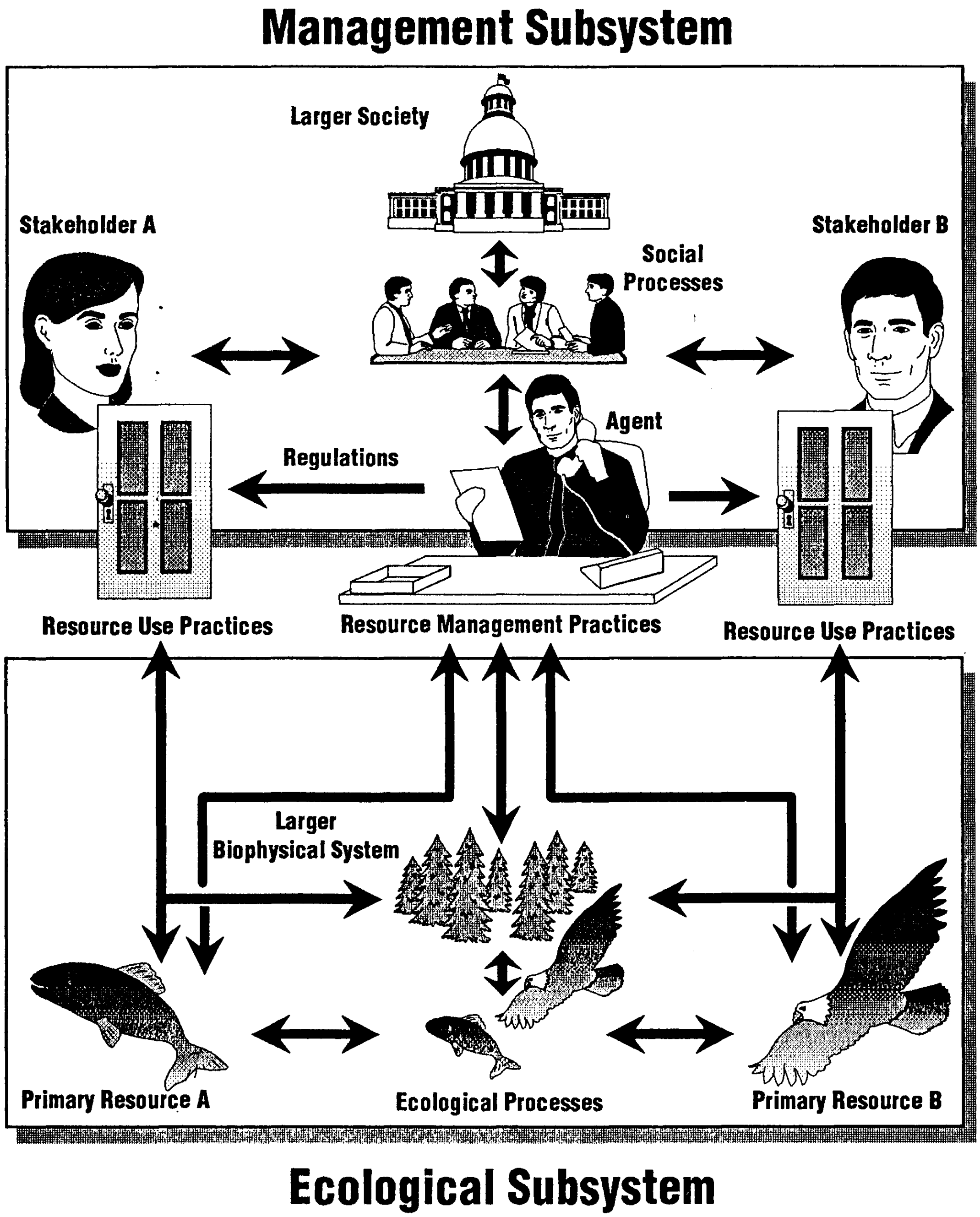

Figure 1. — The Biosocial Model.

THE IMPACT PROCESS

The Impact Process is a computer-aided group decision making procedure for using judgment to understand and resolve complex issues. The Impact Process is especially effective for resolving large-scale and contentious environmental and resource management issues that require cooperative decision making. Underlying The Impact Process is the belief that it is wiser to include affected groups in the formulation of decisions than to try to guess how they may react. It is also wasteful to ignore the knowledge possessed by people who spend their lives dealing with problems associated with an issue. The Impact Process facilitates the development of possible, cost-effective and acceptable decisions.

Over the past decade The Impact Process succeeded in producing a consensus on how to resolve a variety of complex issues. Such issues include protecting the northern spotted owl in California, wetland protection, shoreline erosion, beach and water access, forest management, river management, watershed management and strategic planning for the Texas space industry.

The Setting

The Impact Process brings people together in a workshop setting to explore alternatives for resolving an issue. A facilitation team conducts the workshop at a location that is convenient for the participants. Workshops follow a step-by-step procedure within a detailed schedule. This structured format ensures that participants use time efficiently and remain focused on the issue. A computer operator and assistant sit in the back of the room. Their equipment consists of a computer, a printer and a copy machine. This arrangement keeps the participants focused on the facilitator and each other, instead of the computer.

The process is fast. The time required to complete The Impact Process, and the number and type of workshops, depends on the issue. Most complex issues take a few weeks or months to resolve, and require two or three workshops. A simple well-defined issue might take one workshop to resolve. The time required for each workshop varies from one to three days.

Software

The software supporting The Impact Process gives participants an immediate response to the way they define an issue, and the potential consequences of their alternatives. The process uses two computer programs: EZ-IMPACTtm and EZ-RANKtm. EZ-RANK ranks issues and alternatives based on group preferences or specified criteria. EZ-IMPACT is a unique judgment-based simulation program. This software aids workshop participants in understanding an issue, and it converts that understanding into an operating mathematical model. As a result, sophisticated models can be built in as little as a few hours and updated in minutes. Both programs print customized forms for gathering information during a workshop.

The Process

The Impact Process follows a simple three-stage procedure of building up and narrowing down. First, a list of issues builds up and then narrows down to a set of critical issues. Second, a set of alternatives is examined and then narrowed down to those that are cost-effective and acceptable. Finally, participants select their preferred alternative to resolve each issue. These three stages consist of identifying issues, evaluating alternatives and ranking alternatives. Each stage usually requires a separate workshop (Figure 2).

The most important part of The Impact Process is deciding who should participate. The agent selects the participants. The facilitator maintains neutrality by advising the agent on the criteria for selecting participants. These criteria include relevant technical knowledge, a broad range of affected stakeholders, and ensuring that the participants are legitimate representatives of the stakeholders. The agent may also serve as a participant. The Impact Process structures discussions, but the participants provide relevant knowledge, the alternatives, the criteria for evaluating the alternatives, and they make the decisions. Therefore, the selection of participants is critically important because they determine the outcome of the process.

Identifying Issues

The identification of issues begins with stakeholders recommending a list of potentially important issues. Then they narrow the list down to the most important issues. A one hour session can generate over 100 candidate issues. Each stakeholder rates the candidate issues according to importance. Then each stakeholder identifies the single most important issue from their perspective. EZ-RANK uses the stakeholder ratings to produce a preliminary rank of the issues, and the software places the single most important issue identified by each stakeholder at the top of the list. An arbitrary cutoff in the ranked list provides the short list of critical issues.

Evaluating Alternatives

Stakeholders use variables to define their interests. A variable is the name of something that changes, such as timber production. To ensure that everyone discusses the same thing, a unit of measure, such as board feet, clarifies the meaning of timber production. In the ranking procedure, each stakeholder has the right to select one variable that best defines their interest. The stakeholder "owns" that variable. No other stakeholder can challenge its right to include that variable on the final list.

The next step involves projecting current trends in variables. Then stakeholders define how the variables interrelate with one another to produce these trends. The EZ-IMPACT software creates a computer model that reproduces these estimated trends using the relationships defined by the stakeholders. Next, stakeholders specify an objective for each variable in the model. All participants receive a table that shows the objectives of each stakeholder for each variable. Finally, the stakeholders use their model to design, simulate, evaluate, revise and select several alternatives that are possible, cost-effective and acceptable.

Ranking Alternatives

Stakeholders rank alternatives based on their acceptability and how well they meet implementation criteria. Typically, implementation criteria include feasibility, probability of success, cost, complexity of administration and flexibility. EZ-RANK generates the rankings. The stakeholders discuss and resolve differences between criteria and acceptability rankings to produce the final ranking. The selection of a preferred alternative ends the process.

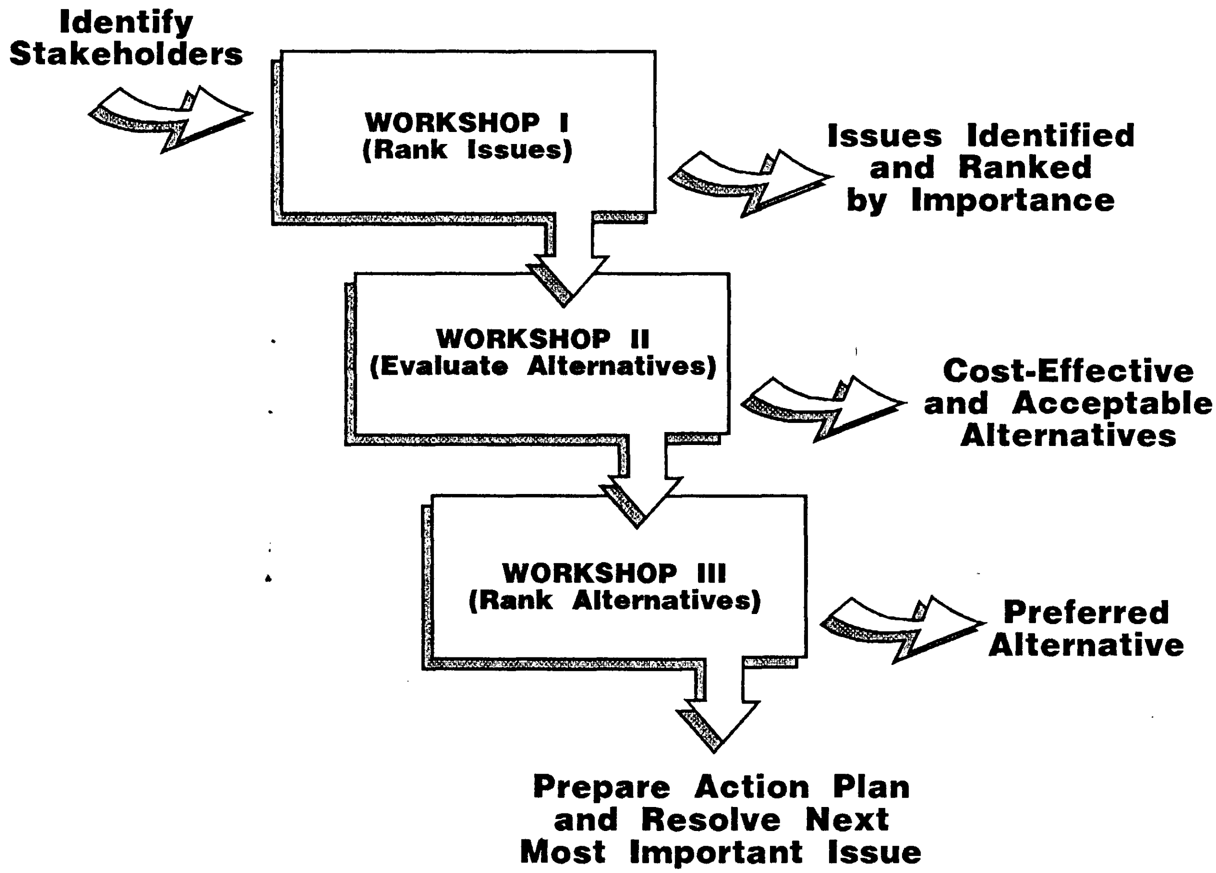

Figure 2. — The IMPACT PROCESS™ Workshops.

CONCLUSION

The Biosocial Perspective assumes that people and their environment form an interdependent system. It also assumes that people must intervene in that system responsibly because they play a dominant role in determining the condition of both society and the environment. The biosocial model organizes these interdependencies within a simple framework. The agent in the model acts as a mediator between society and its environment. The complexity of resource management issues requires agents to seek the help of stakeholders in making decisions through cooperative management. The Impact Process is a computer-aided group decision making procedure that facilitates cooperative management. The process identifies critical issues, evaluates alternatives and builds support for action. It is fast, portable and inexpensive. Thus, the biosocial perspective consists of a solid theoretical foundation for understanding complex resource issues, a model that organizes and simplifies issues, and a process that uses that model to facilitate cooperative management.

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Restoration Ecology of Coastal Riparian Areas: An Applied Approach

Adaptive Cope Team[1]

INTRODUCTION

Riparian areas of the Oregon Coast Range are largely dominated by red alder (Alnus rubra) following historic logging or fire disturbance. Conifers are often excluded from these sites due to the competitive nature of the alder.

Large woody debris from conifers is an important component in the stream ecosystem, affecting flows of water, nutrients and fisheries habitat. Downed conifers have a long life once they fall to the forest floor. Alder by contrast is short-lived both as an upright tree and as a downed log.

In order to provide a continuous source of conifer debris over time, conifers must be established within the present alder-dominated site.

OBJECTIVES

Determine which treatment or combination of treatments results in:

- Highest conifer survival by species
- Highest conifer seedling growth by species

METHODS

Six alder-dominated riparian sites throughout the Oregon Coast Range (Figure 1) were subjected to overstory and understory treatments, and seedlings of four conifer species were planted under the alder canopies. At each site, three overstory treatment plots (0.2 ha) were established: a control, partial overstory removal, and total overstory removal (Figure 2). Each overstory treatment contains an understory control and complete understory removal plots. Four conifer species (western redcedar, western hemlock, Douglas-fir, and grand fir) were underplanted at 2m X 2m spacing, with every other seedling tubed against browsers. Seedling height and diameter has been monitored at the end of each growing season

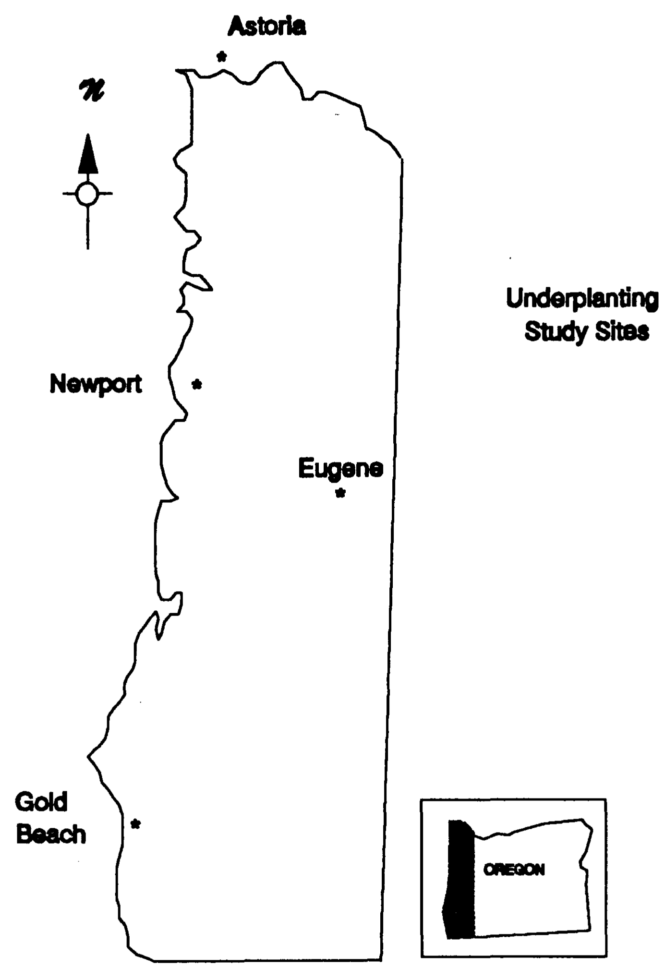

Figure 1. — Study sites for riparian underplanting project in coastal Oregon.

[1] *College of Forestry, Oregon State University, Newport, Oregon.*

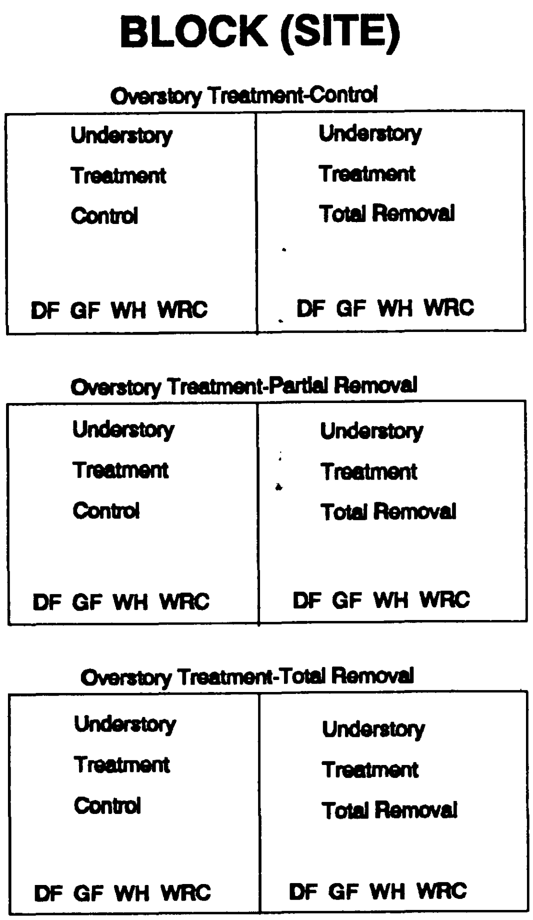

Figure 2. — Experimental design for underplanting study, showing the overstory, understory, and species treatments at each site (block). Underplanted conifers were DF=Douglas fir, GF=grand fir, WH=western hemlock, WRC=western redcedar.

RESULTS

Survival by species over all sites and treatments was quite variable, but some overall trends are apparent after two years (Figure 3). Western redcedar had high survival on most sites relative to the other species, however some decline in survival was seen by the end of the second year. With partial or no overstory removal, understory manipulation appears to be a key factor in survival of all species except western redcedar.

Height growth increment for seedlings was significantly affected by the overstory treatment ($P<.01$), species ($P<.0001$), and tubing effects ($P<.0001$). Mean height growth was significantly ($P<.05$) higher in both the total and partial overstory treatments than in the control, but height growth in the total and partial overstory treatments were not significantly different from each other. Mean height growth of western hemlock was significantly ($P<.05$) higher than all other species. Douglas-fir height growth was not different than grand fir, but was significantly ($P<.05$) greater than western redcedar.

SUMMARY

Though the results of this experiment are preliminary, some early trends are identified. With partial or no overstory removal, understory manipulation is important to the survival of all species with the exception of western redcedar. With total overstory removal, understory manipulation appears important to Douglas-fir survival in particular.

Total or partial overstory manipulation resulted in greater seedling height growth. Western hemlock outperformed all other species over all treatments for height growth. Tubed seedlings grew taller than non-tubed seedlings.

CONCLUSIONS

This study is beginning to answer questions concerning the establishment of conifer seedlings in riparian areas when light is the limiting resource. The results presented here represent initial response to treatments in the first two years. The study will be monitored for a total of six years. This applied research project compliments many fundamental riparian research projects administered by the COPE program, including studies of vegetation dynamics, hydrology, and fisheries and wildlife research.

This research is cooperatively funded by:

- Oregon State University
- Coastal Oregon Productivity Enhancement
- (COPE) Program
- Other State and Federal Agencies
- The Forest Industry
- County Governments
- Oregon Small Woodlands Association

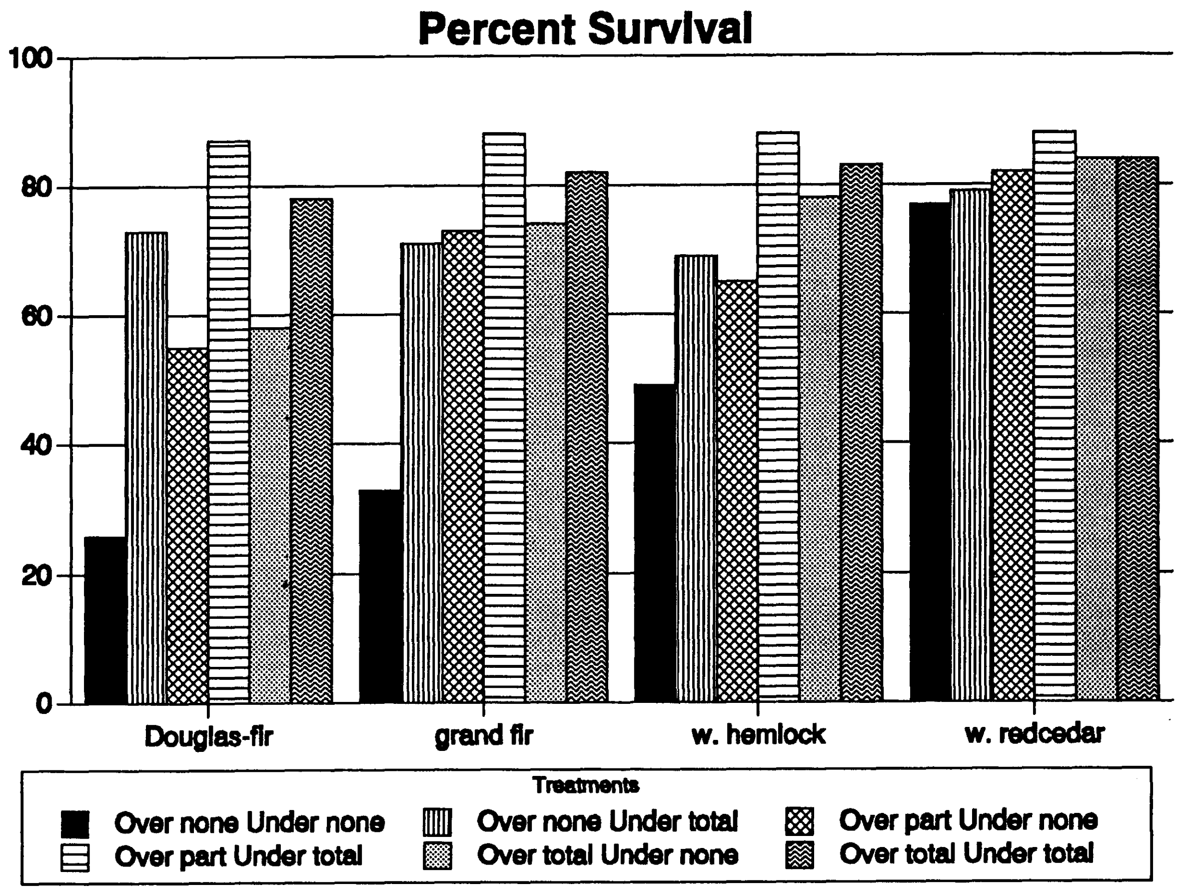

Figure 3. — Survival of seedlings at the end of two years in all combinations of treatments. Treatments legend describes the combination of treatments as Over none Under none = Overstory removal none, Understory removal none, etc.

Enhancing the Suitability of Habitats for the Endangered Stephens' Kangaroo Rat: A Long-Term Experimental Study

Mark C. Andersen[1] and Michael J. O'Farrell[2]

INTRODUCTION

When reserves are established for species that prefer a particular successional stage, it is important to know how to maintain the species' preferred habitat. We present here analyses of data from the first year of a projected five-year experiment designed to test the efficacy of treatments intended to enhance the suitability of habitat for the Stephens' Kangaroo Rat (*Dipodomys stephensii*, henceforth SKR), an endangered heteromyid rodent from Riverside and San Diego Counties in southern California.

Our methods of data analysis allow us to draw inferences about the effects of the treatments on both the density of SKR and on the community composition of the vegetation in the experimental plots. This allows us not only to determine which treatment gives the strongest positive effect on SKR densities, but also to determine the role of vegetation changes in those effects.

STUDY SITES

- Lake Mathews: 40 plots
- Shipley/Skinner Reserve: 18 plots
  (Both sites in Riverside County, CA)

Table 1. — Treatments

	LAKE MATHEWS	SHIPLEY/SKINNER
CONTROL	4 plots	8 plots
BURN	12 plots	4 plots
DISK & DRAG	12 plots	4 plots
BURN/DISK/DRAG	12 plots	2 plots

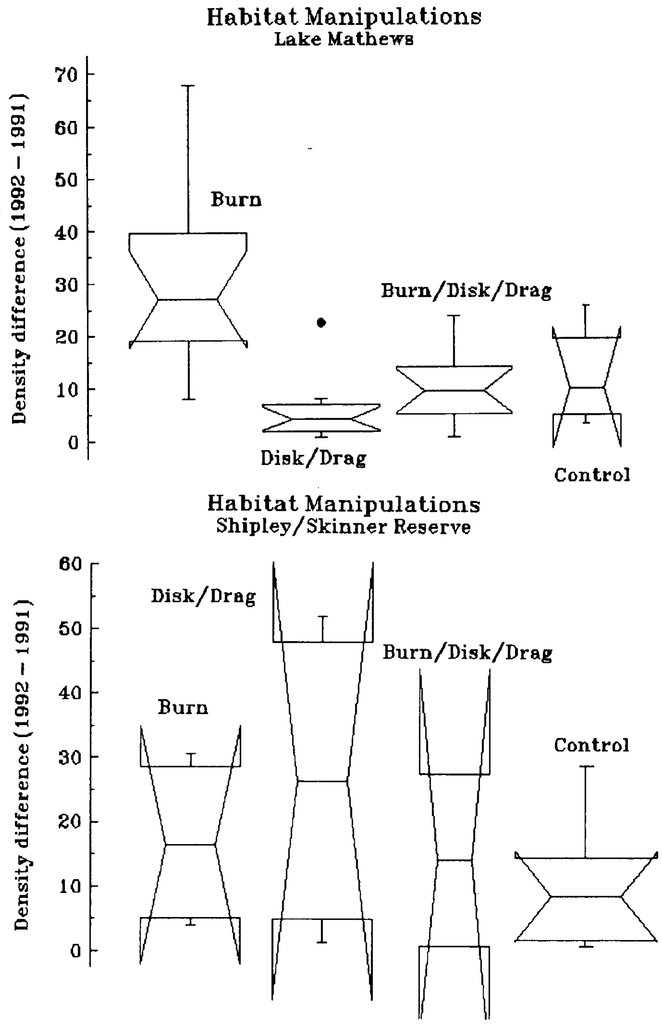

Figure 1. — RESULTS - SKR DENSITIES: 1

Explanation of Figure 1:

Upper & lower edges of boxes: Upper and lower quartiles of data.

Horizontal line: Median.

Notches: 95% confidence limits for the median.

"Whiskers": 1.5 times the interquartile range.

Isolated points: Possible outliers.

[1] *University of California Irvine, CA*

[2] *O'Farrell Biological Consulting, Las Vegas, NV*

ANOVA confirms no differences among treatments at Shipley/Skinner, "Burn" treatment the best at Lake Mathews

	LAKE MATHEWS	SHIPLEY/SKINNER
R-SQUARED	0.474	0.092
MSE	0.752	2.235
F	10.827	0.471
p	<0.001	0.707

Table 2. — Vegetation Effects

Enhanced by the "Burn" treatment:	
Artemisia californica	Lupinus polycarpus
Euphorbia albomarginata	Lasthenia chrysostoma
Hemizonia paniculata	Pectocarya linearis
Suppressed by the "Burn" treatment:	
Melica spp.	Eucrypta chrysanthimafolia
Apiastrum angustifolium	*Opuntia littoralis*
Emmenanthe penduliflora	*Heterotheca grandiflora*
Filago arizonica	Mentzelia affinis
Linanthus androsaceus	*Platystemon californicus*
Matricaria matricarioides	Cucurbita palmata
Lycium andersonii	Salvia mellifera
Stipa pulchra	

Lake Mathews Habitat Manipulation
Ordination Plot – Species Scores

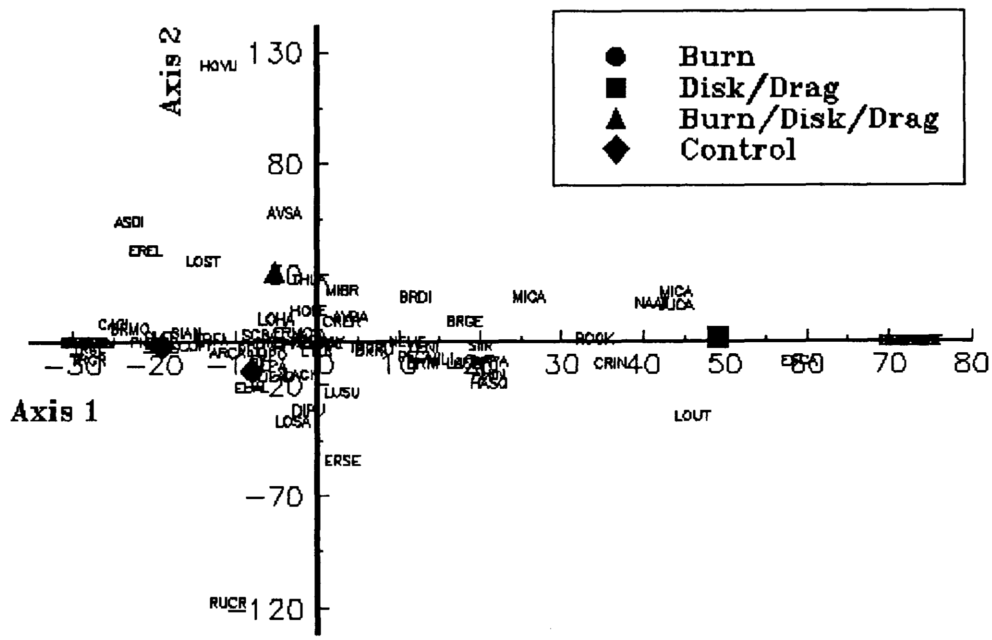

Figure 2. — Results of canonical ordination analysis. Four-letter abbreviations represent species, symbols are centroids for all plots receiving the corresponding treatment.

RECOMMENDATIONS

1) The "Burn" treatment produced the best results at the Lake Mathews site, which had lower SKR densities to begin with. Therefore, we recommend burning, at an interval yet to be determined, as a habitat management strategy for SKR.
2) We recommend additional long-term monitoring studies to determine an appropriate retreatment interval.
3) We recommend more widespread use of canonical ordination as a method for understanding the relationships among densities of target species, intrinsic habitat variables, and externally-imposed habitat management practices.

Regional Mitigation: A Means for Restoring Forested Ecosystems in Florida

B.F. Birkitt[1]

Florida environmental regulations require compensatory mitigation for the loss of wetlands or wetland functions associated with construction activities. After many years of requiring on-site mitigation, the state environmental agencies are beginning to recognize the ecological value of establishing larger, off-site regional mitigation areas, particularly to address impacts of linear projects such as roads, pipelines, and transmission lines.

A large natural gas pipeline company is proposing a 600 mile expansion of their existing line in Florida resulting in wetland impacts at crossings throughout the state. To minimize impacts, the pipeline has been sited along existing roads, railroads, and transmission line corridors. There will be no loss of wetlands.

Mitigation is required to compensate for the permanent loss of forested canopy in wetland areas and associated wildlife benefits resulting from maintenance of the pipeline right-of-way and temporary impacts to high quality forested systems.

Options for mitigation include:

- enhancement or restoration of wetlands
- preservation of high quality wetland systems
- creation of new wetlands

Enhancement or restoration of wetlands is the state's preferred option. Preservation is only acceptable under certain circumstances, and wetland creation is often discouraged except where site conditions are particularly favorable. For the pipeline project, wetlands on publicly-owned lands were identified which were in need of reforestation or hydrologic enhancement. A small creation site is also proposed at one location. Three regional mitigation sites located in the areas of predominant impact have currently been identified.

Documentation of the existing condition of the mitigation sites is being provided to determine the extent of environmental benefit and mitigation "credit" allotted for each site. Mitigation ratios of 2:1 to 4:1 are expected based on the limited impacts of the project and the benefits provided at the mitigation sites.

UPPER CHIPOLA RIVER REFORESTATION SITE

The Upper Chipola River site is a floodplain forest which has been clearcut in areas by forestry operations. Erosion and increased runoff are occurring and natural regrowth is sparse. To establish a more diverse community resembling the natural system and to ensure rapid reestablishment of a productive wetland, replanting of mixed hardwood species is proposed. Approximately 300 acres will be revegetated. The site is owned by the North Florida Water Management District and was selected because of their interest in revegetation of the site and its proximity to the pipeline route.

Vegetative transects were sampled at seven locations within the site including a natural area to determine appropriate native species for replanting. Two year bare root trees approximately 18-24 inches in height will be planted. Species include Florida maple, red maple, green ash, sweetgum, yellow poplar, black gum, various oaks and bald cypress.

HYDROLOGIC ENHANCEMENT SITES

The other two regional mitigation sites located at the Apalachicola/Florida River and Steinhatchee River involve hydrologic enhancement of existing wetland systems by breaching logging roads which block natural flow. The proposed sites are owned by the North Florida Water Management District and the Suwanneee River Water Management District, respectively.

APALACHICOLA/FLORIDA RIVER ENHANCEMENT SITE

The Apalachicola/Florida River site consists of floodplain swamp with numerous backwater sloughs. The most common dominant species are Ogeechee tupelo and water tupelo; other dominant tree species include bald cypress, water hickory, river birch, planer tree, black gum, green ash, and sweetgum.

The old logging roads currently obstruct normal drainage patterns. A head differential of as much a 3.83 feet from one side of the road to the other has been measured in some areas.

[1] *Dames & Moore, Tallahassee, Florida, USA.*

Water does not overtop the roads except following storms in excess of 10 and 25 year events and seasonal high water fluctuations of the river. Effects on the wetland system include: changes to the herbaceous strata, limited regeneration of tree species, and interference with the transport of detrital material and movement of aquatic organisms.

Hydrologic improvements proposed consist of installation of low water crossings or wooden bridges at 25 locations in the roadway network to reduce extended inundation periods and to more closely approximate natural sheetflow conditions. Approximately 930 acres of floodplain forest will enhanced by the activities proposed.

STEINHATCHEE SPRINGS ENHANCEMENT SITE

Hydrologic enhancement is also proposed at the Steinhatchee Springs mitigation site. This site is a hydric hammock which is managed by the Florida Game & Fresh Water Fish Commission as a Wildlife Management Area. Dominant tree species include bald cypress, red maple, sweetgum, water oak and laurel oak. At many of the sites, Carolina willow has encroached in the areas impounded by the roads. Runoff does not overtop the roads except for storms greater than 10 and 25 year events.

Six of the old logging roads on the site will be breached by culverts or low water crossings at 14 locations to restore or enhance historic drainage patterns. Willows will be removed and such areas will be planted with native tree species. Approximately 450 acres of forested wetlands will be enhanced at this site.

FUTURE OF REGIONAL MITIGATION AND BANKING

Regional mitigation is particularly suited for linear projects such as pipelines, roads, and transmission lines. It provides increased environmental benefits over small on-site mitigation efforts. The establishment of regional mitigation "banks" for future projects provides opportunities for continued development as well as significant environmental benefits.

In Florida, regional mitigation and mitigation "banking" are being encouraged as a means of restoring or protecting large ecosystems or wildlife habitat. Regional mitigation is becoming the norm for linear projects and the state is currently drafting a rule to regulate how mitigation banks are established and used. Several large wetland "banks" and wildlife "parks" have been established in the state. Many more are expected once regulations clearly authorizing the use of mitigation "banks" are in place.

Fire in Southwestern Riparian Habitats: Functional and Community Responses

D.E. Busch[1]

FIRE OCCURRENCE

Hydrogeological perturbations and exotic species introductions are among the factors indirectly contributing to shifts in riparian ecosystem processes. Among these altered processes is the incidence of fire, which appears to occur at greatly elevated frequencies in perturbed low-elevation riparian ecosystems and may be a novel form of disturbance in these systems.

Between 1981 and 1992, 166 fires burned 11,846 ha (27%) of the riparian vegetation in the lower Colorado and Bill Williams River floodplains (Fig. 1). The area burned annually and fire frequency were correlated between 1981 and 1988 ($r = 0.85$, $P < 0.01$).

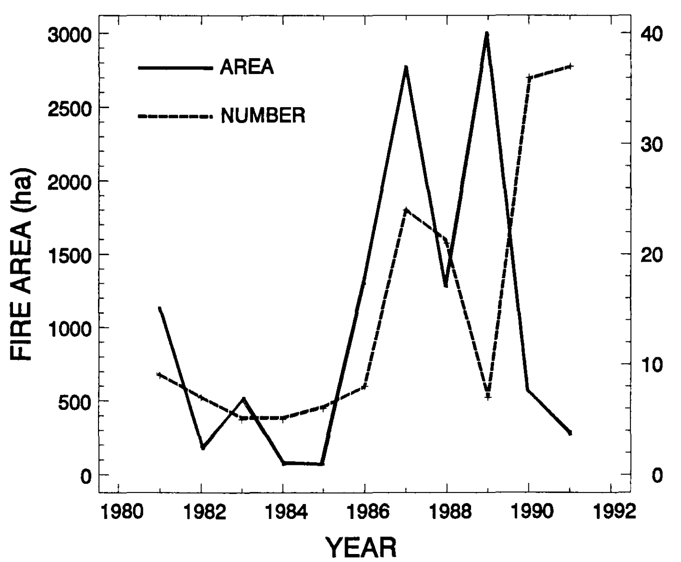

Figure 1. — Area and number of fires occurring between 1981 and 1991 in the lower Colorado and Bill Williams River riparian zones.

There was little difference between the structural classification of burned vegetation and the general population of riparian vegetation that had not burned. Comparisons of burned areas within each dominance type indicated that significantly less cottonwood/willow (*Populus fremontii/Salix gooddingii*) and mesquite (*Prosopis glandulosa* and *P. pubescens*) dominated habitat burned than would be expected based on the representation of these taxa in the riparian community. Fires in habitats dominated by saltcedar (*Tamarix ramosissima*) appeared to be disproportionately large, but this relationship was not statistically significant presumably due to the small subsample of stands that could be evaluated using GIS.

Assuming that the period evaluated is representative, fire could be expected to burn approximately 25% of the riparian vegetation, and 40% of the saltcedar-dominated habitat, each decade. This implies a disturbance interval shorter than that from fire in most forest associations, and one that is insufficient for full maturation of historically dominant cottonwood, willow, and mesquite. It appears that shrubby species that are less valuable from a wildlife habitat standpoint may benefit from riparian zone fires.

FUNCTIONAL RESPONSES

There is evidence for mechanisms which would facilitate increased community dominance of saltcedar and arrowweed (*Tessaria sericea*) following riparian zone fires. Rapid recovery in these taxa appears to be attributable to salinity tolerance and to mechanisms permitting efficient water uptake and use (Busch and Smith 1993).

Soil elemental analyses indicated that concentrations of nearly all soil constituents increased following fire. This contributed to a potential nutrient abundance, but also was a manifestation of elevated alluvium salinity. Boron was elevated in riparian soils following fire and lower leaf tissue concentrations suggest that saltcedar may derive tolerance to this element by an effective elimination mechanism. Based on leaf sodium concentrations, a dichotomy between halophytic (saltcedar and arrowweed) and glycophytic (cottonwood and willow) taxa was detectable, but no clear fire-related differences were apparent. There was little interspecific variation in leaf nitrogen concentration, but burned arrowweed had elevated concentrations of this element. This

[1]*Bureau of Reclamation, Division of Environment, Boulder City, Nevada, USA.*

may contribute to the higher water use efficiency following fire that was observed in recovering burned arrowweed relative to unburned controls.

Higher leaf stomatal conductance in all taxa and in both ecosystems was an indication of vigorous post-fire recovery. This may, in part, be attributable to increased radiation loads associated with the reduction of plant canopy cover following fire. However, decreases in water potential accompanying an increased transpiration load may signify water stress as a result of fire injury or inefficient recovery. Examination of transpiration-water potential regression parameters provided evidence for reduced post-fire hydraulic efficiency in burned Colorado River willows.

COMMUNITY RESPONSES

In agreement with functional analyses of fire recovery efficiency, the community importance of saltcedar and arrowweed was high following fire in all pre-fire dominance types. Fire also appears to contribute to the catastrophic declines in cottonwood populations that are in evidence in perturbed desert riparian ecosystems.

Relative cover and frequency of saltcedar was high in three burned riparian vegetation dominance types while arrowweed importance in burned riparian vegetation approached that of saltcedar. Marked increases in arrowweed abundance accompanied cottonwood declines in the cottonwood/willow dominance type. Willow maintained its importance in burned cottonwood/willow vegetation, but did not become established in other dominance types in the manner of arrowweed. Similarly, mesquite and other shrubs were present in burned riparian stands, but did not dominate even those areas that were classified as mesquite-dominated prior to fire.

As was expected of plants that recover from fire via resprouting, multiple linear regression analysis of burned riparian community ordination data revealed that the species that were dominant prior to burning were strongly associated with post-fire community composition. Subtle evidence of a successional trend was shown by the positive relationship of community structure with fire year, and the tendency of younger tree demographic classes and ruderal taxa to be found near the opposite extremes of Detrended Correspondence Analysis axis 3 from older demographic classes and slower growing species.

REFERENCE

Busch, D.E. and S.D. Smith. 1993. Effects of fire on water and salinity relations of riparian woody taxa. Oecologia 94:186-194.

Multi-Purpose Development of the Gila Drain Floodway

J. Dillon, J.H. Brock, and R.S. Gordon[1]

A major "greenbelt" reforestation effort appears feasible south of Phoenix as part of a multi-purpose floodway development project. This project, located within the northern "Borderlands" portion of the Gila River Indian Community, would involve the establishment of about 1 million trees on about 10 square miles of land by the 15th year, along with woody shrubs and herbaceous plants. The multiple uses proposed for the project area include the following:

1. Establishment of a reforested greenbelt to enhance the environment of Maricopa County in harmony with economic development planned by the Gila River Indian Community.
2. Provision of borrow material for new highway construction within the region being carried out or planned by ADOT.
3. Establishment of a regional floodway with naturalistic contours to convey, by gravity flow, excess storm waters from South Mountain and East Valley communities which are intercepted by the ADOT highway system.

Preparation of this assessment and preliminary plan was supported by funding provided by the Flood Control District of Maricopa County (FCD) and the Arizona Department of Transportation (ADOT). Participants in the assessment and planning process were the Gila River Indian Community (GRIC), representatives of the FCD and ADOT, HDR Engineering (a contractor to ADOT), and faculty members and graduate students from Arizona State University's School of Agribusiness and Environmental Resources (ASU-SABER), Department of Civil Engineering, and Office of Cultural Resource Management.

Phase I of this project involved two major tasks: the development of a (1) master plan/conceptual design and a (2) revegetation plan.

The *Gila Drain Master Plan* was a multi-disciplinary effort consisting of five sub-tasks: (i) a drainage report, (ii) a stormwater yield study, (iii) a vegetative inventory, (iv) a land use inventory, and (v) a cultural resources reconnaissance.

The revegetation plan was completed by ASU in May of 1993 and consisted of three sub-tasks: (i) a plant species recommendation, (ii) a reforestation cost/benefit analysis, and (iii) a planting and management plan.

The vegetation sample from which population estimates were drawn is classified as a desertscrub community of the Lower Colorado River Valley subdivision of the Sonoran Desert region. Species recorded conform to a saltbush series with a mesquite consociation, or "microphyllous desert", increasing westward along the site. The site itself is an alluvial plain between the South Mountain range to the north and the Sierra Estrella range to the south. The Gila River divides the plain south of the site. Elevations range from 1040.5 feet to 1115.4 feet, sloping westward, and average annual rainfall is about 200mm (7.87 inches). Soils are sodic and saline and fall into the alkali flat/saline loam/saline sandy loam category of the general range index. The soil association is Shontik/Casa Grande/Redun but closely borders the Gunsight/Carrizo/Cristobal association to the north. Evidence supports the existence of an historic mesquite bosque on the site and, therefore, the plant species recommendation is a re-creation of the bosque. This will be, in other words, a more intensely cultivated version of the existing native population.

A cost/benefit analysis was prepared based upon assumptions from the planting and management plan. The analysis concluded that future revenues from wood harvested on a sustained-yield basis from the proposed greenbelt would exceed the marginal costs of establishment, maintenance and harvesting of the woodland. The bulk of the funds needed for 10 square miles of reforestation (estimated at about $13 million over a period of fourteen years) is expected to come from several federal agencies and national foundations that have expressed initial enthusiasm for the project concept. Such funding could partially defray the costs of floodway construction and maintenance.

Moreover, it is projected that $13 million (current dollars) in revenues will have been realized, by the 15th year, from the sale of wood products derived from planned thinning of the reforested area. Additional revenues are projected from sustained-yield harvesting of the more valuable hard wood beyond the 15th year.

[1] *Poster design and presentation by Jan Dillon, Graduate Research Assistant and Project Coordinator, Arizona State University, College of Engineering, School of Agribusiness & Environmental Resources*

Exotic Species and Sustainable Ecosystem Management

Tom L. Dudley[1] and Carla M. D'Antonio[2]

The goal of ecosystem management is to simultaneously promote ecological integrity and sustainable resource production; however, one factor often not considered in resource planning is the influence of introduced species on natural ecosystems. While some exotic species are relatively benign invaders, those of concern reproduce and disperse readily, but also exhibit one or more of the following traits: (1) directly interfering in survival of native species, with repercussions for other interacting species; (2) changing the rate of resource supply to the native community; (3) altering natural disturbance regimes. We use the examples of bermudagrass (*Cynodon dactylon*) and tamarisk (*Tamarix* spp.) in desert riparian areas, and perennial grasses in Hawaiian forests, to illustrate why these invasions have important implications for biodiversity, ecosystem productivity, and/or human welfare.

The grasses *Schizacrium condensatum* and *Melinus minutiflora* were introduced to Hawai'i as forage, but invaded dry forest areas and the added fuel changed the fire record from 27 fires of an average of 4 hectares in the 47 years prior to grass invasion of Hawai'i Volcanoes National Park, to 58 fires of 205 hectares in the 22 years after invasion. Native trees and shrubs are virtually eliminated, and total soil nitrogen declines 40% after two burns. Fire control expenses and the costs of exotic eradication efforts comprise over 80% of the Park Resources Management budget.

Bermudagrass can form mats on sandy stream and bank substrates in the Sonoran Desert, and is resistant to natural flooding. It also provides a refuge to macrophytes from such disturbance, altering the benthic successional sequence and reducing open sand substrates available for native invertebrates and fish, particularly *Agosia chrysogaster* which builds nests in clean sand. Tamarisk also invades desert riparian areas, and is known to increase soil salinity, reduce surface water and run-off for downstream uses due to high transpiration rates, and provide poor habitat for insects and native birds. In some streams which experience frequent natural flooding, however, it may not dominate because it appears to be more susceptible to such disturbance than some native riparian species but in regulated streams and those with infrequent flooding, it remains a major problem.

In many cases, control of exotic species is prohibitively expensive or impossible. Their influences are with us indefinitely because their propagules are widespread, and invasions tend to be promoted by continuing human land-use changes. These problems are global in nature, and rival other global concerns such as climate change in terms of their impacts on natural ecosystems.

[1] *Pacific Institute, 1204 Preservation Park Way, Oakland, CA 94612.*

[2] *Dept. of Integrative Biology, University of California, Berkeley, CA 94720.*

Relationships Between Forest Songbird Populations and Managed Forests in Idaho

Diane M. Evans[1] and Deborah M. Finch[2]

Abstract — Many species of songbirds have experienced population declines in the eastern U.S. in recent years, but conclusive data on population trends and factors affecting populations in the West are lacking. Few studies have evaluated the importance of surrounding land configuration to songbird abundances. In 1992, we initiated a study in mixed conifer forest in west-central Idaho to compare songbird composition and abundances among two untreated watersheds and three watersheds having clearcuts. Watersheds were selected on the basis of their large size, accessibility, dominant tree type, and timing and extent of management. Based on 1992 point counts of 29 selected bird species, we identified four species that had significantly lower mean birds/count station in managed study areas than in untreated areas. These were hermit thrush (*Catharus guttatus*), Swainson's thrush (*Catharus ustulatus*), pileated woodpecker (*Dryocopus pileatus*), and warbling vireo (*Vireo gilvus*). Twelve species were significantly more abundant in clearcut watersheds than in untreated watersheds, whereas abundances of 13 species did not differ between treated and untreated study areas. Variation in bird species richness among study areas may have been influenced by sampling intensity. Negative or positive responses to management were not clearly associated with migratory status. We discuss 1993 modifications to our study design and future use of a Geographic Information System (GIS) to measure landscape characteristics.

INTRODUCTION

Many forest-dwelling birds, particularly those that breed in North America and winter in Central and South America (neotropical migrants) have shown population declines over the last 10-20 years (Robbins et al. 1989a). Declining trends of songbird populations are best documented in the eastern U.S., where habitat fragmentation is proposed to be a primary contributor to species loss (Robbins et al. 1989b, Faaborg and Clawson 1991, Freemark and Merriam 1986, Whitcomb et al. 1981). However, no single contributing factor explains declines for all species in all geographic areas (Finch and Stangel 1993). Because data on population trends are lacking for many western songbird species, and relatively little is known of songbird habitat relationships in this region, comparisons to eastern studies are difficult.

Large areas of intact forest remain in the western U.S., but management is changing the landscape configuration. The resulting pattern is extensive forested areas increasingly containing more clearings. This differs from the predominant pattern in the east and mid-west where isolated forest patches are surrounded by clearings. Nevertheless, this alteration introduces habitat characteristics related to fragmentation (Van Dorp and Opdam 1987, Franklin and Forman 1987). Despite conflicting trends from Breeding Bird Survey data (reviewed by Finch 1991), evidence suggests that many forest-dwelling birds are negatively affected by reductions in habitat area due to increased nest predation and parasitism (problems associated with edge), and that a high proportion of these species are long-distance migrants (Temple 1986, Terborgh 1992, Askins et al. 1990, Whitcomb et al. 1981). In 1992 we initiated a study in mixed conifer forests in west-central Idaho to examine the

[1] *Wildlife Biologist, Payette National Forest, McCall, Idaho; and School of Forestry, Northern Arizona University, Flagstaff, Arizona.*

[2] *Research Wildlife Biologist, Rocky Mountain Forest and Range Experiment Station, Flagstaff, Arizona.*

composition of forest bird communities in managed and relatively intact (untreated) landscapes. In the context of this study, landscape pattern is the variation in vegetation cover resulting from natural influences (topography, fire, soils) and management activities (primarily timber harvest). Patterns vary by the size, shape, and distribution of forest and nonforested stands.

This paper reports on the study design, methods, and preliminary results based on the first year's (1992) data. The objectives were to: 1) determine if the sampling scheme was appropriate for a landscape analysis, 2) analyze overall bird species compositions and abundances in relation to general landscape pattern (managed vs. untreated), 3) evaluate differences in abundances of selected species among managed and untreated landscapes, and 4) determine if neotropical migratory birds or other bird species should be highlighted as species of special management concern in interior mixed conifer forests.

The results of our study will ultimately be used to predict species occurrences in managed and untreated western forests, and to offer recommendations to land use agencies for managing avian abundance and diversity in commercial forests of the western United States.

METHODS

Study Areas

Our study area includes the western half of the Payette National Forest in west central Idaho. Study areas were selected using a hierarchical process. A map of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) forest coverage was generated for the Payette National Forest from the University of Idaho Gap Analysis database (completed for the state of Idaho at a scale of 1:200,000). Douglas-fir forest is 1 of 33 groupings of broad vegetation categories designated as "wildlife habitats" by the Gap Analysis mapping system (Scott et al. 1990). Based on this vegetation group's distribution, potential study areas were then examined on 7.5' ortho-quadrangle maps. Using aerial photographs, timber inventory stand maps (Payette National Forest files), and ground verification, potential study areas were corrected for actual vegetation types and stand configuration.

Five study areas were selected, including 3 managed areas and 2 untreated areas. These were delineated from previously mapped watershed boundaries and contained primarily Douglas-fir/mixed conifer forests that represented a gradient of management. The managed areas were located in the upper Bear Creek drainage on the Council Ranger District (RD); the Price Valley area on the New Meadows RD; and the Bear Basin area on New Meadow and McCall ranger districts. Control areas included the French Creek roadless area on the McCall RD and Ponderosa State Park adjacent to the McCall RD (fig 1). The study areas ranged from 1376-1922 m in elevation.

Sample Stations and Census Methods

Sample Point Distribution

Sample points were located in forested stands only. Stand delineations were determined from timber inventory classification maps and photos, on which stands were classified by strata (age class and density). To minimize variability introduced by sources other than the treatment, stands were selected based on the following criteria: 1) forest types and range of age classes should be similar among treated and nontreated areas, 2) the range of elevations among all areas should vary less than 350 m, 3) slope should be less than 30% on average, and 4) stands should be accessible by road, mountain bike, or minute hike. Ponderosa State Park, a large block of relatively homogeneous habitat, was not classified into stands. This study area was divided into six geographic areas for placement of sample stations.

Sample stations were distributed in each stand along a transect or in a grid pattern. Transect bearings were determined from aerial photographs and established on the ground with a field compass. Within each stand, stations were paced to 225-250m apart. The number of stations/stand varied with size of the stand, ranging from 2 to 7. A total of 151 census stations was established among the 5 study areas, ranging from 24 to 32 stations per area.

Bird Count Techniques

Birds were counted using the variable circular-plot method (Reynolds et al. 1980). Observers visited each point three times from mid-May through mid-July 1992, recording all birds seen and heard during an eight minute sample period. Time of visit and observer were rotated per visit to reduce bias (Verner 1985). Distance from the station center (point) to the initial detection of each bird was estimated in meters. Counts were not conducted on days with wind 10 km/hr or during moderate to heavy precipitation.

Analysis

Mean numbers of birds per point for each species were tallied for each of the five study areas. These values were standardized by the number of visits per point. Oneway ANOVA's were used to compare abundances of counted species among study areas. We selected 29 bird species and 1 mammal species (red squirrel, a mammalian nest predator) based on their presence and abundance on three or more watersheds. We estimated point count abundance per species by averaging count data across all points and visits/point for each watershed. Levene's test was used to test for homogeneity of variances of mean counts among

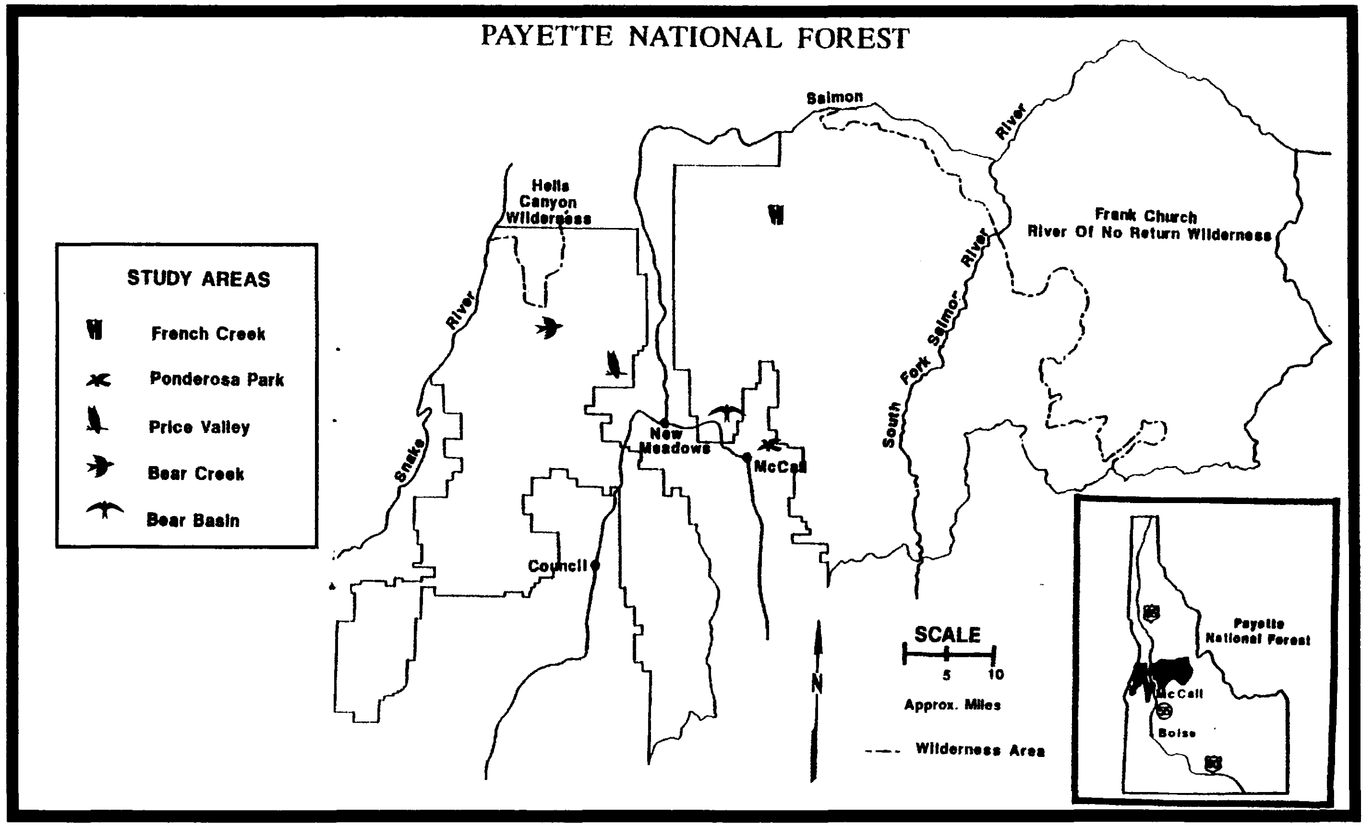

Figure 1. — Location of study areas on Payette National Forest in west-central Idaho.

study areas. For those species for which the assumption of homogeneity was violated, we computed Welch's statistic for assessing variation in mean counts among areas.

Contrast tests were used to determine if mean counts of pooled species and mean counts of each of 30 selected species varied between managed versus untreated watersheds. Levene's test for homogeneity of variance was used to select appropriate test statistics (pooled or separate t probabilities). We used multiple comparison tests (Tukey-HSD multiple range test, p = 0.05) to isolate differences in mean numbers of birds among specific pairs of study areas. We also used oneway ANOVA's to compare mean numbers of birds among and within stands within each watershed.

RESULTS

A total of 52 avian species was recorded across the five study areas. Of these, 19 were long-distance (neotropical) migrants, 17 were short-distance migrants, and 16 were residents. Total species richness (avian plus red squirrel) per study area ranged from lows of 30 and 34 in the untreated areas to 35, 37, and 43 in the managed areas (Table 1). Red squirrel presence was recorded during point counts in all areas. Twenty three species occurred in numbers too low for statistical analysis. These included three woodpeckers, three diurnal raptors, one owl, two corvids, one grouse, and thirteen passerines. Three species were recorded in only one study area: winter wren in Bear Creek, varied thrush in French Creek, and Townsend's solitaire in Price Valley. Brown-headed cowbirds were recorded in three of the five areas: Ponderosa Park, Bear Basin, and Bear Creek.

Mean numbers of birds/point significantly varied among all five areas ($F_{1,146}$=14.75; p=0.0000) and also varied between untreated and treated areas (Contrast tests: t_{146}=2.20, p=0.03). French Creek, an untreated area, had the lowest mean number of birds per point (10.46), significantly differing from two managed areas, Price Valley (12.75 birds/point) and Bear Basin (12.84) ((Tukey-HSD multiple range test, $p < 0.05$)), as well as from untreated Ponderosa Park (13.10) ($p < 0.05$). In addition, Bear Creek (managed) had significantly lower mean numbers of birds per point (11.57) than Bear Basin (managed) and Ponderosa Park (untreated) ($P < 0.05$). Mean bird counts/point were as similar among stands as within stands in each of the three managed watersheds (Table 1). In contrast, both untreated watersheds showed significantly more variation in total numbers of birds among stands than within stands (Table 1).

Of the 29 bird species which were statistically analyzed, 10 were long-distance migrants, 11 were short-distance migrants, and 8 were resident species (Table 2). Red squirrel had a higher overall abundance than any bird species. The most abundant bird species, based on mean numbers/point across all study areas, were red-breasted nuthatch (see Appendix A for scientific names), Swainson's thrush, and western tanager, respectively (Table 2).

Table 1. — Total numbers of species/area, mean number of birds/point, and stand variation in numbers of birds/point in untreated and managed study areas.

WATERSHED	NO. POINTS	TOTAL NO. SPECIES/ AREA	NO. BIRDS/ POINT Mean	SD	STAND VARIATION[1]
Untreated					
French Creek	32	34	10.46	1.43	0.048
Ponderosa Park	24	30	13.10	1.57	0.050
Managed					
Price Valley	21	37	12.75	1.55	0.831
Bear Creek	30	35	11.57	1.94	0.237
Bear Basin	44	43	12.84	1.50	0.234

[1] *Probability values from analysis of variance tests comparing bird numbers/point among and within stands.*

Table 2. — Analyses of variance, contrast tests, and trends in mean numbers of birds per point among untreated and managed watersheds.[1]

SPECIES	UNTREATED FC[2]	PP	MANAGED PV	BC	BB	F PROBABILITIES ANOVA	CONTRAST	CONTRAST TREND
Long-Distance Migrants								
Olive-sided flycatcher	0.00	0.04	0.14	0.27	0.09	0.002	0.000	+ (***)
Hammond's flycatcher	0.01	0.03	0.14	0.05	0.01	0.154	0.067	+ (*)
Dusky flycatcher	0.05	0.29	0.36	0.12	0.25	0.000	0.163	ns
Swainson's thrush	1.23	1.99	1.08	1.48	0.95	0.000	0.002	- (**)
Solitary vireo	0.07	0.04	0.28	0.05	0.18	0.003	0.004	+ (**)
Warbling vireo	0.21	0.18	0.11	0.08	0.08	0.258	0.039	- (*)
Townsend's warbler	0.39	1.06	1.02	0.37	1.21	0.000	0.171	ns
MacGillivray's warbler	0.04	0.19	0.23	0.27	0.21	0.000	0.014	+ (*)
Western tanager	0.95	1.14	1.27	1.37	1.14	0.068	0.020	+ (*)
Chipping sparrow	0.12	0.50	0.72	0.50	0.71	0.000	0.000	+ (***)
Short-Distance Migrants								
Northern flicker	0.00	0.06	0.10	0.10	0.06	0.089	0.021	+ (*)
Brown creeper	0.08	0.22	0.11	0.25	0.20	0.062	0.455	ns
Golden-crowned kinglet	0.09	0.22	0.11	0.23	0.12	0.378	0.971	ns
Ruby-crowned kinglet	0.21	0.26	0.68	0.13	0.67	0.000	0.000	+ (***)
Hermit thrush	0.53	0.01	0.16	0.03	0.16	0.000	0.001	- (***)
American robin	0.02	0.08	0.12	0.17	0.23	0.000	0.001	+ (***)
Yellow-rumped warbler	0.47	0.47	0.83	0.92	0.57	0.013	0.001	+ (***)
Dark-eyed junco	0.57	0.17	0.26	0.33	0.83	0.000	0.129	ns
Brown-headed cowbird	0.00	0.12	0.00	0.00	0.12	0.001	0.366	ns
Cassin's finch	0.02	0.00	0.05	0.07	0.01	0.243	0.146	ns
Pine siskin	0.00	0.12	0.19	0.08	0.08	0.099	0.205	ns
Residents								
Hairy woodpecker	0.07	0.00	0.03	0.05	0.06	0.364	0.613	ns
Pileated woodpecker	0.08	0.15	0.06	0.07	0.01	0.009	0.061	- (*)
Steller's jay	0.00	0.03	0.03	0.02	0.03	0.549	0.376	ns
American crow	0.00	0.00	0.00	0.02	0.08	0.087	0.028	+ (*)
Common raven	0.02	0.00	0.03	0.10	0.04	0.240	0.049	+ (*)
Mountain chickadee	0.58	1.24	0.93	1.08	0.89	0.000	0.604	ns
Red-breasted nuthatch	1.27	1.35	1.16	1.33	1.35	0.845	0.819	ns
Red crossbill	0.16	0.40	0.13	0.10	0.29	0.069	0.115	ns
Red squirrel	2.71	2.11	1.70	1.47	1.66	0.000	0.000	- (***)

[1] *ANOVAs compare mean numbers of birds across all five watersheds, and contrast tests compare mean numbers in untreated vs. managed watersheds. Contrasts indicate significant (* p0.10; ** p0.01; *** p0.001; ns = not significant) positive ("+") or negative ("-") trends of bird numbers in relation to fragmentation.*

[2] *FC=French Creek, PP=Ponderosa Park, PV=Price Valley, BC=Bear Creek, BB=Bear Basin.*

Seventeen species had significant differences in mean numbers/point between managed and untreated study areas (Table 2). Four bird species had lower mean numbers in managed watersheds than in untreated areas. These were hermit thrush ($p = 0.001$), Swainson's thrush ($p = 0.002$), warbling vireo ($p = 0.039$), and pileated woodpecker ($p = 0.061$). Red squirrel also showed a highly significant decline in managed areas ($p < 0.001$). Twelve species were more abundant in managed areas than in untreated ones: northern flicker, olive-sided flycatcher, Hammond's flycatcher, American crow, common raven, ruby-crowned kinglet, American robin, solitary vireo, yellow-rumped warbler, MacGillivray's warbler, western tanager, and chipping sparrow.

Thirteen species had no significant difference in mean numbers in managed vs. untreated areas. Of these, five species showed no significant difference in mean numbers across all areas (refer to ANOVA probabilities, Table 2). The remaining eight species did not show a significant difference in contrast tests of managed and untreated areas, but showed a significant difference in mean numbers among the five study areas (ANOVA probabilities, Table 2). These were dusky flycatcher, mountain chickadee, brown creeper, Townsend's warbler, dark-eyed junco, brown-headed cowbird, red crossbill, and pine siskin.

DISCUSSION

It is important to recognize that these results are preliminary, based on point counts conducted in 1992. Total numbers of species in each study area may have been influenced by sampling intensity, which varied among watersheds. Bear Basin, with the highest number of recorded species, also had the greatest number of sampling points (44). Ponderosa Park had the lowest number of recorded species (30) and the second lowest number of sampling points (24).

Mean total bird detections differed significantly between the two untreated watersheds, French Creek and Ponderosa Park, which in fact had the low and high extremes, respectively. However, for 15 of the 30 species, mean birds/point were similar between the two untreated areas, suggesting that these two areas shared certain habitat attributes to which these species showed a common response. These attributes possibly were related to lack of treatment.

Mean counts differed between French Creek and treated areas for more species than between Ponderosa Park and treated areas. A better understanding of the history of these two untreated areas helps to interpret these trends. French Creek, an extensive roadless area, is a large block of interior forest where timber has not been harvested. Ponderosa Park is a state park that also has not been actively managed for timber, but unlike French Creek, it is situated on a relatively narrow (range 500-1200 meters) peninsula extending into Payette Lake. In addition, it is close to the town of McCall, and it supports recreational use on trails and in a campground which may create disturbances not found in French Creek. We hypothesize that overall bird abundance may be higher in Ponderosa Park than in French Creek because it has increased edge and disturbance effects that attract certain species whose numbers may increase in relation to these habitat features. Hejl et al. (1993) report that dusky flycatcher, chipping sparrow, and brown-headed cowbird are more abundant in disturbed second growth forests with clearcuts than in uncut old growth in the western United States. Thus, species whose abundances in Ponderosa Park were similar to those in managed areas (e.g., dusky flycatcher, chipping sparrow, brown-headed cowbird, pine siskin, and mountain chickadee (Table 2)) may be responding to edge and disturbance effects. Landscape measurements and tests are needed to confirm this hypothesis.

Mean numbers of birds differed among stands in each of the untreated areas but not in the three managed areas. Only forested areas were sampled in each study area; thus, this analysis compared bird numbers among and within forested stands, but not between clearings and forested stands. We interpret stand variation in bird numbers within untreated areas as a product of clumped distributions of birds. Aggregations of bird abundance in unmanaged forests may be related to natural clumping of habitat features that vary among stands. Further investigation of the relationships between bird abundances and habitat characteristics within unbroken forest blocks is needed to verify this interpretation. Lack of significant differences in mean bird numbers among stands within managed areas may indicate that these landscapes were relatively homogeneous in their degree of fragmentation and/or that remaining stands were similar in habitat features, so that birds responded similarly in abundance among stands. A dominant attribute of remaining forested stands in managed watersheds is their association with clearcuts. The shared feature of clearcut/forest borders may outweigh effects of natural habitat variation among stands. This could explain similarities in bird abundance among stands in treated watersheds.

ANOVAs indicated variation in mean numbers of birds/point among all five study areas and were useful in interpreting differences in avian use of these areas. For example, 13 species showed no significant differences in mean numbers between managed and untreated areas, yet abundances of eight of these species differed significantly among all five areas. This suggested that some species were selecting certain habitat characteristics regardless of whether the landscapes were managed or unmanaged. Pairwise comparisons of study areas helped to pinpoint the areas where bird numbers of each species differed. This information highlights the need to evaluate how specific habitat features influence bird distributions independently of, or in concert with, management effects.

A negative response to fragmentation did not appear to be correlated with migratory status in this preliminary assessment. More long-distance migratory species showed a positive response than a negative or nonsignificant response. More short-distance migrants showed a nonsignificant response than either positive or negative. Responses for individual species need to be evaluated on a case-by-case basis. Our results showed, however, that more species responded positively to treatment than negatively, suggesting that treatment created more habitat for more species. However, since the majority of species that responded favorably to treatment are already common or abundant, we do not believe that management benefitted species of concern. Species of most concern to managers are those whose populations are listed as threatened, endangered, sensitive, or declining, or whose populations are of economic importance (e.g., game species) (Finch 1992). Three of the four species that responded negatively to management in our study - hermit thrush, Swainson's thrush, and pileated woodpecker - have been identified as potentially sensitive to forest management (Raphael 1987, Hejl et al. 1993). Increases in local species richness due to forest management must be considered in relation to changes in regional biological diversity, especially if regional changes are due to losses or population reductions of sensitive species (Finch and Ruggiero 1993).

Based on these preliminary results, the study has been modified to more completely assess relationships between landscape features and bird abundances. The primary modifications for the second year include 1) division of watersheds into 300 ha landscapes to standardize comparison of areas, and 2) extension of sampling into all habitats on the landscape in the proportion in which habitat patches occur to better describe variability within watersheds. Sampling intensity will remain consistent across all landscapes. A GIS will be used to characterize landscapes based on selected fragmentation measures, including size and spatial distribution of forest and nonforest, distance to nearest similar habitat (contagion), total length of edge, and perimeter-areas relationships (Ripple et al. 1991, Rex and Malanson 1990, O'Neill et al. 1988).

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APPENDIX 1 - SCIENTIFIC NAMES OF BIRD SPECIES

COMMON NAME	SCIENTIFIC NAME
Hairy woodpecker	*Picoides villosus*
Northern flicker	*Colaptes auratus*
Pileated woodpecker	*Dryocopus pileatus*
Olive-sided flycatcher	*Contopus borealis*
Hammond's flycatcher	*Empidonax hammondii*
Dusky flycatcher	*Empidonax oberholseri*
Stellar's jay	*Cyanocita stelleri*
American crow	*Corvus brachyrhynchos*
Common raven	*Corvus corax*
Mountain chickadee	*Parus gambeli*
Red-breasted nuthatch	*Sitta canadensis*
Winter wren	*Troglodytes troglodytes*
Brown creeper	*Certhia americana*
Golden-crowned kinglet	*Regulus satrapa*
Ruby-crowned kinglet	*Regulus calendula*
Townsend's solitaire	*Myadestes townsendi*
Swainson's thrush	*Catharus ustulatus*
Hermit thrush	*Catharus guttatus*
Varied thrush	*Ixoreus naevius*
American robin	*Turdus migratorius*
Solitary vireo	*Vireo solitarius*
Warbling vireo	*Vireo gilvus*
Yellow-rumped warbler	*Dendroica coronata*
Townsend's warbler	*Dendroica townsendi*
MacGillivray's warbler	*Oporornis tolmiei*
Western tanager	*Piranga ludoviciana*
Chipping sparrow	*Spizella passerina*
Dark-eyed junco	*Junco hyemalis*
Brown-headed cowbird	*Molothrus ater*
Cassin's finch	*Carpodacus cassinii*
Red crossbill	*Loxia curvirostra*
Pine siskin	*Carduelis spinus*

Distribution and Abundance of Plants in Colorado Plateau Hanging Gardens

James F. Fowler and Nancy L. Stanton[1]

Abstract — Hanging gardens are rare, mesophytic/hydrophytic habitats associated with springs on xeric canyon walls of the Colorado Plateau. We surveyed regional distribution and community importance of vascular plant taxa in 48 hanging gardens of Zion and Capitol Reef National Parks, and Glen Canyon National Recreation Area (GLCA). Regional metapopulation distributions of 144 plant species are not bimodally distributed as predicted by Hanski's (1982) core-satellite hypothesis but instead have a log series distribution (Chi square = 12.31, df = 143, P = 1.0). Distribution and abundance of ten species endemic to the plateau ranged from one plant at one site for *Sphaeromeria ruthiae* to a mean of 22% canopy coverage at 18 sites for *Cirsium rydbergii*. Species area curves for ZION hanging garden plant assemblages are significantly different from GLCA's ($t_{(.05)3,36}$=2.12, P=.045) due to Zion's higher species richness; however, the slope values (z= 0.17-0.18) are not significantly different. A higher number of endemic taxa and lower species richness indicates that GLCA's hanging gardens are more isolated biogeographically than ZION's. Since these habitats are rarely directly disturbed, management objectives in maintaining biodiversity at these sites should be directed toward groundwater recharge area and long term monitoring of plant distribution and abundance.

INTRODUCTION

The Colorado Plateau geologic province harbors isolated hanging garden habitats within its entrenched drainage system (Welsh 1989a, Welsh and Toft, 1981). A suite of sedimentary geologic formations consisting of alternating sandstones and silty-sand to mudstones is exposed over much of the Colorado Plateau. This characteristic stratigraphy, exposed through dissection by the incised drainages of the plateau, creates the world-famous aesthetics of the canyon country. Hanging gardens have been described as part of canyon country since the earliest recorded explorations of the region; e.g., Glen Canyon was named by Powell for the "oak glens" surrounding hanging gardens on the Colorado River canyon walls. The contrast between a hanging garden and the surrounding xeric habitat or plantless slickrock can be striking, not only in terms of vegetation, but also in ambient air temperature and humidity. One can pass from the stark reflection of heat and glare off slickrock into a shaded, cool, moist habitat of trickling water, maidenhair fern, columbine, orchids, and lush grasses and sedges. The source of water is small springs and seeps that issue from sandstone aquifers intersected by the canyon system of the plateau. Most hanging gardens are afforded at least some protection from wind and sun by their location in depressions or larger alcoves on canyon sidewalls or deep within canyon headwalls. Several Colorado Plateau endemic plant species have been found in these habitats (Welsh 1984, 1989a, 1989b). Our research objectives were to compare the distribution and abundance of vascular plants in these herbaceous hanging garden plant communities which develop around seeps and springs in Zion and Capitol Reef National Parks (ZION and CARE) and Glen Canyon National Recreation Area (GLCA).

METHODS

We define hanging gardens by the predominance of mesophytic and hydrophytic herbaceous vegetation (Tuhy and MacMahon 1988, Stanton et al. 1992) growing on wet rock

[1] *James F. Fowler is Biology Instructor at State Fair Community College, Sedalia, MO and Nancy L. Stanton is Professor of Zoology, University of Wyoming, Laramie, WY.*

walls and on wet colluvial soils with moisture supplied by aquifer water via wide seep(s) (Stanton et al. 1992). They are apparently microclimatically isolated from the surrounding slickrock and xeric vegetation.

Data collection on the physical parameters of each hanging garden included mean aspect, elevation, size, and UTM coordinates. Measurements taken to determine size were part of the vegetation sampling grid. Abundance of plant species was determined by Daubenmire (1959) canopy cover classes (1-6) using a 20 cm x 50 cm sampling frame. A systematic sample (n=10) with random starts (Manly 1989, Krebs 1989) was taken on each hanging garden microhabitat accessible by non-technical climbing. Gardens were also searched for plant species which may not have been detected in quadrat sampling. Voucher specimens for each plant species present on each hanging garden were collected and identified by the staff at Rocky Mountain Herbarium in Laramie, WY.

RESULTS

Herbaceous hanging garden vegetation develops either directly in cracks on moist rock surfaces, or on colluvial soils supplied with subsoil moisture from the seep. Many seeps issue from the transition between the Navajo sandstone and the Kayenta Formation in these three parks. Others issue from impermeable facies within the Navajo sandstone. Each hanging garden was visually separated into the following microhabitats: wet walls, ledges, ledge-soil complex, and seeplines. Wet walls include slopes and floors covered with thin sheet flow of water as well as vertical walls and are dominated by ferns, prokaryotic and protistan communities. Ledges are of sufficient width and length to support linear hydrophytic plant communities in cracks and in narrow strips of wet, saturated soils. The vegetation is intermixed with thin sheet flow of water over wet rock surfaces. Ledge-soil complexes are composed of wet colluvial soils located just up slope of stabilizing ledges, including wet soil underneath an alcove. Since these soils develop directly from the weathering of sandstone, they are obviously very sandy and may be virtually saturated near seeps. Much of the characteristic herbaceous hanging garden vegetation is found on these wet colluvial soils. Seeplines are drier, linear microhabitats that develop at fractures in the sandstone or on impervious bedding planes on canyon walls and at the back of drier alcoves. Virtually all water in this microhabitat is used by the vegetation or lost to evaporation. Excess water for sheet flow on rock surfaces is not available. The distribution and areal extent of wet rock and colluvial soils on a garden depend on the physical attributes of the site including relative discharge of the seep or spring, facies characteristics, and location within the drainage. The resulting microhabitats fall into four major categories as listed in the methods section: wet wall, ledge, soil-ledge complex, and seepline.

Regional distribution of the 144 species of vascular plants found on 48 hanging gardens in ZION, GLCA, and CARE fit a log-series distribution pattern (fig.1) (Williams 1964 as cited in Krebs 1989) (Chi square = 12.31, df = 143, P = 1.0). In addition, the log-series pattern fits all within park distributions (Chi square = 12.41-GLCA, = 7.17-ZION, = 1.76-CARE, P = 1.0). Most vascular plant species were found on only one or two sites; for example, 56 of the 144 species were found in only one hanging garden and 15 were found on only two. At the other extreme, only the maidenhair fern, *Adiantum capillus-veneris*, was found on most hanging gardens (47 out of 48).

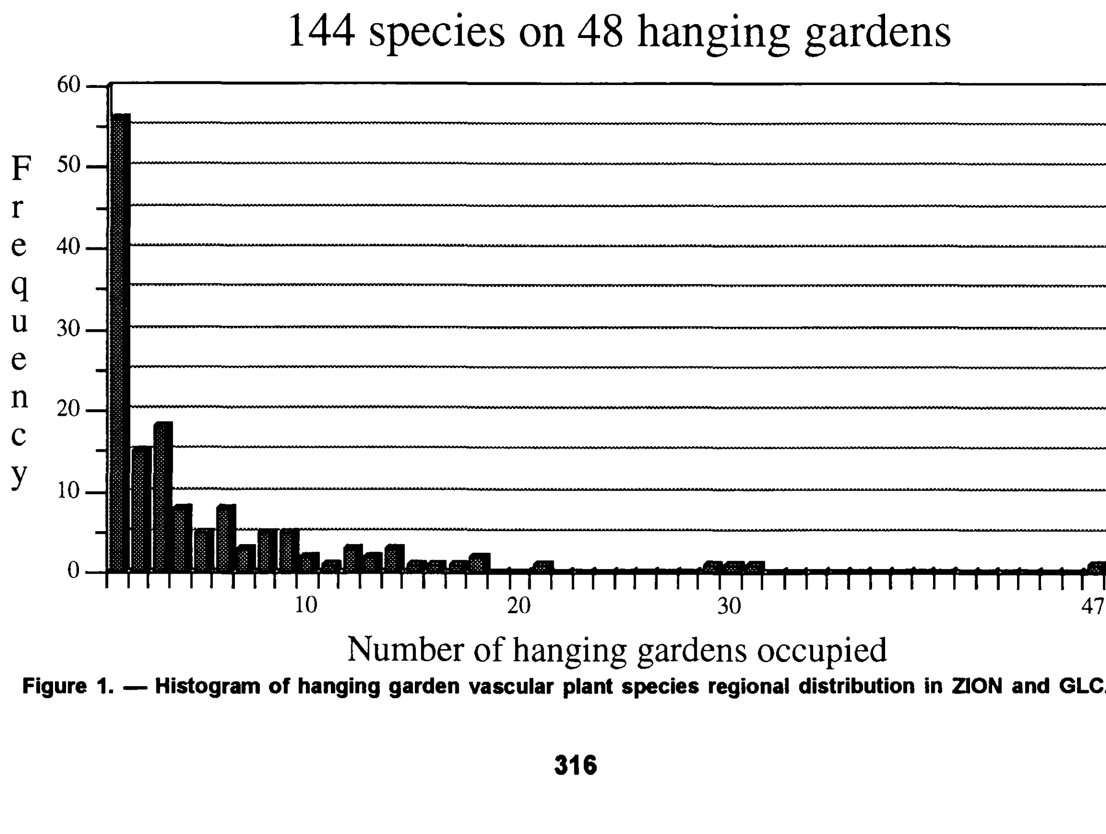

Figure 1. — Histogram of hanging garden vascular plant species regional distribution in ZION and GLCA.

Table 1. — Plant species richness and physical parameters of hanging gardens in Zion National Park: size in m^2, aspect in degrees, elevation in ft., location in UTM coordinates. R = Richness.

HangingGarden	Size	R	Aspect	Elevation	UTM X	UTM Y
Pine Creek		6	344°	4200	325849.8	4120485.0
Upper Emerald	1170	33	120°	4700	326061.4	4125225.0
Lower Emerald	131	12	158º	4300	326166.9	4124882.0
Grotto	628	28	16°	4600	327434.5	4124914.0
Menu Falls	190	23	218°	4500	324416.6	4127546.0
Fall	274	14	31°	4500	328178.1	4126268.0
Falling Water	420	21	318°	4600	326982.4	4123753.0
Narrows Trail	383	26	259°	4500	327489.1	4128541.0
Trail's End	226	13	303°	4500	327397.1	4128715.0
Canyon Overlook I	4	14	238°	5250	327979.7	4120384.0
Canyon Overlook II	70	26	124-242°	5250	327878.7	4120338.0
Court Patriarchs	99	20	177°	4750	325531.3	4123556.0
Snail	61	13	270°	4500	327713.9	4127987.0
Kaye's	124	29	192°	4600	327134.3	4126287.0
Weeping Rock	812	28	196°	4500	328346.6	4126609.0
Hailstone	28	13	100°	4700	327052.6	4126318.0

Table 2. — Plant species richness and physical parameters of hanging gardens in Glen Canyon National Recreation Area: size in m^2, aspect in degrees, elevation in feet, location in UTM coordinates. R = Richness.

HangingGarden	Size	R	Aspect	Elevation	UTM X	UTM Y
Dune	173	18	90°	4000	525718.3	4124789.0
Crossbed	115	22	201°	4040	525855.6	4124822.0
Rattle-snake	836	20	356°	3800	525142.8	4122410.0
Hardwood	1177	29	155°	3840	526840.1	4125133.0
Pedestal	269	15	160°	3800	511965.1	4127387.0
Zephyr	81	16	174°	3800	512511.3	4127486.0
Graffiti	44	7	175°	3800	512332.5	4127398.0
Upper Three	825	15	230°	4100	510509.6	4116716.0
Lower Three	162	6	230°	3840	510372.5	4116613.0
Surprise	150	9	120°	3880	527149.5	4125485.0
Ivy	70	7	233°	3840	527418.3	4125696.0
Baby	35	10	263°	3760	527633.3	4125514.0
Baby Too	38	10	205°	3920	527706.3	4125909.0
Zigy	1215	30	186°	3880	514304.1	4123519.0
Hook	351	27	212°	4160	515549.0	4123938.0
Hawk	193	12	243°	4160	516034.8	4122757.0
Swallow	52	11	7°	3880	514926.3	4122935.0
Ice	893	18	238°	4050	526533.3	4127594.0
Corner	249	13	112°	4050	526583.3	4127625.0
Channel	714	11		3920	526583.4	4127412.0
Barbara	344	21	156°	3980	506610.6	4139277.0
Maria	86	14	60°	3850	505060.1	4139649.0
Stone Basin	115	16	138°	3950	505925.5	4141012.0
Wrong	35	11	109°	3840	504665.9	4139096.0
Boon-doggle	17	11	195°	3800	538226.2	4159463.0
Camp	341	19	5°	3800	537350.4	4158251.0
Pyro	37	21	202°	3800	568255.1	4159430.0

Plant species richness varied from 6-30 per site while size varied from 4 to 1215 m^2 (Tables 1, 2). The area-species relationships (MacArthur and Wilson, 1967), $S = cA^z$ where S = number of species, A = area, are $S = 0.889A^{.17}$ for ZION and $S = 0.733A^{.19}$ for GLCA where S is species richness and A is area of each hanging garden (fig. 2). The slopes (z) of the two regression lines (.17 vs .19) are not significantly different ($t_{(.05)2,39} = 2.02$, $P > .50$). However, the constant values (0.889 vs .733) are significantly different ($t_{(.05)2,36} = 2.12$, $P = .04$) primarily due to the higher average number of species per hanging garden in ZION. The area-species relationship for CARE was not calculated due to small sample size (n = 5).

We have analyzed the distribution and community importance of endemic taxa, disjunct populations of more widespread species, and possible endangered species (Table 3). Federal and state status was taken from Atwood et al. (1991). Quantitative measurement of hanging garden vegetation communities indicates that maidenhair fern, *Adiantum capillus-veneris*, the most widely distributed hanging garden species on a regional basis, was also the most abundant species in wet wall and seepline microhabitats in most hanging gardens. Grasses, especially Jones reedgrass, *Calamagrostis scopulorum*, and sedges, *Carex*, tended to be abundant on ledge and ledge/soil complex microhabitats. Zion daisy, *Erigeron sionis*, was found on only two hanging gardens in ZION as a very small part of the community canopy coverage. There are, however, many individual plants on the wet wall at Canyon Overlook II HG within one meter of the trail on which thousands of tourists walk each year. Alcove bog-orchid, *Habenaria zothecina*, was found on wet ledge/soil complex microhabitats on four hanging gardens in GLCA where it occupied 2-5% of the canopy. Three hanging gardens have a north aspect (Table 3): Rattlesnake-356°, Swallow-7°, and Camp-5°. These sites represent new distribution records for the species and for GLCA. They are isolated locations in Pictograph, Ribbon, and Knowles Canyons respectively which show little, if any, sign of visitation. Each of the stands had many individual plants of this species and appear to be vigorous. Disjunct populations of American spikenard, *Aralia racemosa*, and cliff jamesia, *Jamesia americana* var. *zionis*, were found on two ZION hanging gardens: Falling Water and Grotto. American spikenard had an 8% average canopy cover on ledge microhabitats as well as several individual plants on both hanging gardens. Cliff jamesia had much fewer individuals and did not show up in community samples. Zion tansy, *Sphaeromeria ruthiae*, was collected only at Falling Water HG. This rare species is endemic to Washington County and previously noted only in ponderosa pine communities (Atwood et al. 1991). At the other extreme are a few hanging gardens in GLCA such as Hawk HG (Table 3) in which the endemic Rydberg thistle, *Cirsium rydbergii*, covered almost all of the vegetated surface over large parts of the hanging garden.

DISCUSSION

Regional species distribution patterns have often been described as log series (Williams, 1964) or lognormal (May, 1975) for many different taxa. A recent hypothesis formulated by Hanski (1982) predicts a bimodal distribution pattern for similar sites within a region: a mode of rare species found at few sites, a mode of abundant species found at most sites, and very few species found at a moderate number of sites. Our distribution patterns did not support the bimodal pattern of Hanski's (1982) core-satellite hypothesis. Hanging garden vascular plants have a log series distribution pattern with many species found at only a few sites. Many of these are transients

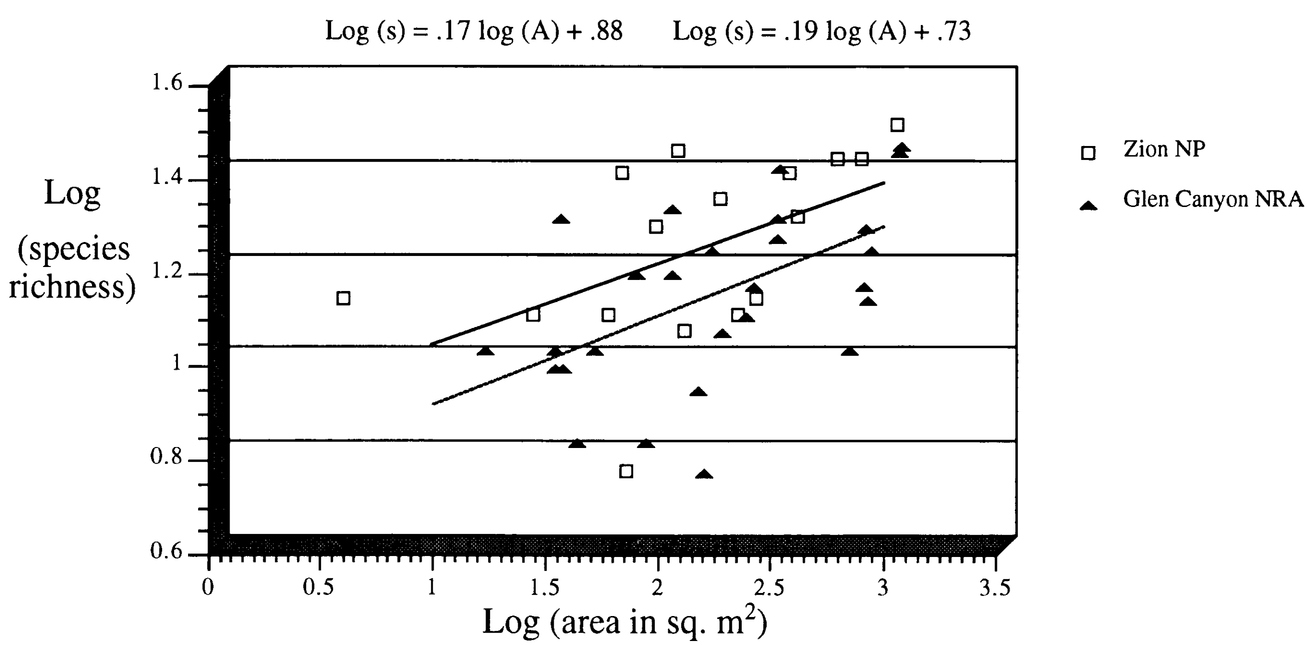

Figure 2. — Area-Species relationship linear regressions for ZION and GLCA hanging garden vascular plant species.

Table 3. — Protection status and average canopy coverage values for hanging garden vascular plants with narrow distributions: * = species endemic to Colorado Plateau, ( ) = coverage range, T = trace, [ ] = sample size. Microhabitat symbols: ww = wet wall, l = ledge, ls = ledge/soil, sl = seepline. Note: *Carex* sp. values not separable by species but ranged from (T-.37).

Species	Status		Microhabitat			
	Fed	St	ww	l	ls	sl
Jamesia americana zionis	C2	G5T1 /S1	-	T [2]	-	-
**Habenaria zothecina*	C2	G1 /S1	.05 [1]	-	.10 (T-.26) [3]	-
**Erigeron sionis*	C2	G2 /S2	.01 (T-.01) [2]	-	-	T [1]
**Carex curatorum*	-	G5Q /S1	-	-	[8]	-
**Carex haysii*	-	-	-	-	[5]	-
Aralia racemosa	-	-	-	.08 (T-.15) [3]	.04 [1]	-
Dodecatheon pulchellum zionense	-	-	.03 (.03-.04) [3]	.11 (.08-.13) [3]	.20 (.06-.35) [8]	-
**Sphaeromeria ruthiae*	C2	G2 /S2	T [1]	-	-	-
**Zigadenus vaginatus*	-	-	-	-	.13 [1]	T [1]
Rubus neomexicanus	-	-	-	-	.04 [2]	-
**Cirsium rydbergii*	-	-	.03 (T-.13) [7]	.05 [1]	.22 (T-.79) [14]	.01 (T-.02) [2]
Cladium californicum	-	-	-	.037 [1]	.063 [1]	-
**Primula specuicola*	-	-	.01 (T-.08) [8]	.07 (.03-.15) [3]	.02 (.02-.03) [5]	-
**Mimulus eastwoodiae*	-	-	.08 (T-.20) [5]	T	T	-
**Aquilegia micrantha*	-	-	.10 (.01-.20) [4]	-	.03 (T-.06) [9]	-

from nearby xeric or riparian vegetation and probably not a stable component of hanging garden species assemblage. Other rare species such as Alcove bog orchid and Zion daisy are restricted to hanging gardens and seeps and should be monitored for population size changes.

The area-species relationship Z values for ZION (0.17) and GLCA (0.19) are intermediate to MacArthur and Wilson's (1967) predicted values for continental (0.12-0.17) and island (0.20-0.35) biota. While these intermediate values for insular continental areas were also predicted by MacArthur and Wilson (1967), empirical studies of plants on soil islands in granite outcrops have found higher values: 0.566 in the Southeast U.S. (Houle, 1990), and 0.59 in Oklahoma (Uno and Collins, 1987). Additionally, Bond et al. (1988) found a Z value of 0.43 for fynbos shrublands surrounded by Afrotemperate evergreen forests in South Africa. In this context, our Z values are more continental. The 56 species which occurred in only one hanging garden (fig. 1) also supports MacArthur and Wilson's (1967) hypothesis that habitat islands receive large numbers of propagules from adjacent mainland areas.

Many Colorado Plateau endemic species found in hanging gardens are widespread regionally within the hanging garden habitat. In addition, Rydberg thistle and Zion shooting star can be locally abundant within a hanging garden. It is the habitat that is rare and finite. Except for Weeping Rock hanging garden at ZION, these hanging garden habitats are rarely impacted by human visitation. Since the first requirement for hanging garden existence is the presence of a perennial seep, conservation priorities should be placed on managing the Navajo sandstone aquifer and its recharge areas.

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Inonotus (phellinus) weirii: Origins, Nomenclature, and Pathogenesis in Western Ecosystems

M. J. Larsen, G. I. McDonald and A. E. Harvey[1]

Filamentous fungi, as saprophytes and as partners in mutualistic and parasitic relationships with higher plants, are very old and probably contemporary with the first terrestrial plants. Origin and early development of wood rotting fungi was synchronous with the early evolution and migrations of both the Pinidae and Angiospermae. Thus, wood rotting fungi (saprophytes and parasites) were intimately associated with the formation and function of forests similar to present ones. During the early to mid-Cretacean periods the advent of continental drift created effective barriers against continued migrations of both plants and fungi. Thus, existing data on the distribution and the prehistoric origin of *Inonotus sulphurascens* support the view that the species had its origins in the temperate forests of the early Cretacean period in Siberia. As plants migrated to the east and across the Beringial formation, this fungus co-migrated with its hosts. Thus, the closely related *I. weirii* appears to have evolved from *I. sulphurascens* in North America.

We have shown that *Inonotus weirii* actually represents two intersterile species in North America. *Inonotus sulphurascens* occurs in Siberia and in northwestern North America solely on conifers other than western redcedar; while *I. weirii* has evolved and occurs almost solely on western redcedar. Thus, both are indigenous pathogens, widely distributed in the Northwest, and cause a disease of the site. *Inonotus sulphurascens* is the fungus that causes most of the mortality due to Inonotus (Phellinus) root-rot. Continuous and repetitive disturbances, white pine blister rust and fire control have increased the amount of soil inoculum, thus intensifying the amount of tree mortality in subsequent forests. In addition, these site phenomena have shifted the prevailing cover types to more disease susceptible species, e.g., shallow-rooted and thin-bark less resistant species that prevail in the absence of fire rather than deep-rooted and thick-bark more resistant species. Though prehistory of this root-disease is virtually unknown, it appears that Inonotus root-rot may become a classic example of the results of man's intervention, through harvesting and concomitant activities, leading to a decline in site-health and possibly non-sustainability within time frames as defined by current standards.

Management of co-existing species (host:pathogen) must take its cues from the patterns and processes that are observable now in naturally developing and existing old growth ecosystems. Our efforts in the future should simulate natural processes and functions on both temporal and spatial scales or at least create disturbances that mimic those imposed during the formation of highly diversified forested ecosystems.

[1] *USDA Forest Service, Intermountain Research Station, Moscow, Idaho, USA.*

Maintaining and Restoring Aquatic Habitats to Benefit Water Quality

Michael D. Marcus[1] and Clayton S. Creager[2]

ABSTRACT — The objective of the Clean Water Act "is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Historically, water quality improvements under CWA have minimally emphasized needs regarding the physical integrity of waterbodies. Yet, water qualities in many previously degraded waterbodies have improved to where degraded habitat factors now primarily limit their successful restoration. EPA's emerging Watershed Protection Approach (WPA) and its total daily maximum load (TMDL) framework provide increasing recognition of physical habitat restoration as important for achieving this CWA objective. Habitat restoration in many waters can produce large improvements in the integrity of sustainable aquatic resources beyond those gained by improving water quality alone. Habitat improvements also can enhance the abilities of many waterbodies to process contaminants, making habitat restoration and preservation cost-effective supplements to traditional point and nonpoint source controls. This refined interpretation of CWA requires best management practices (BMPs) addressing waterbody integrity, including links to terrestrial systems, and better coordination with other resource management agencies. We will review our recent work with EPA advocating the importance of maintaining and restoring habitat in achieving the goals of CWA. We will also describe strategies for restoring and maintaining riparian and instream habitat qualities to benefit water qualities and sustainable ecological resources in streams and rivers.

WATERSHED PROTECTION APPROACH (WPA)

WPA describes comprehensive efforts by U.S. EPA and other Federal, State, and local agencies to use a watershed-oriented approach to meet water quality goals necessary to address all threats to human health and ecological integrety within specific watersheds.

[1] *The Cadmus Group, Inc., Flagstaff, Arizona.*

[2] *The Cadmus Group, Inc., Petaluma, California.*

THREE KEY ELEMENTS OF WPA

1. RISK-BASED GEOGRAPHIC TARGETING — All significant watershed problems are identified. Watersheds at risk are ranked and one or more selected for cooperative integrated assessment and protection.
2. STAKEHOLDER INVOLVEMENT — Working as a task force, stakeholders reach consensus on goals and approaches for addressing the watershed's problems, specific action taken, and how actions will be coordinated and evaluated.
3. INTEGRATED SOLUTIONS — The selected tools are applied to the watershed's problems, as established by the stakeholders. Progress is evaluated periodically using ecological indicators and other measures.

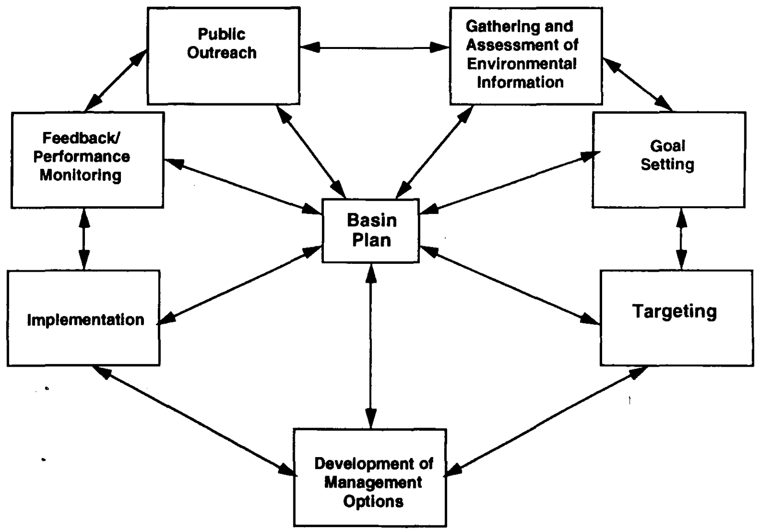


Figure 1. — Selected Components of a Watershed Protection Approach

POTENTIAL STAKEHOLDERS IN WPA PROJECTS

- State environmental, public health, agricultural, and natural resource agencies
- Local and regional boards, commissions, and agencies
- EPA water and other programs
- Other Federal agencies (e.g., FS, SCS, FWS, BLM, BIA)
- Indian tribes
- Public representatives
- Non-governmental wildlife and conservation organizations
- Industrial, agricultural, and other water user representatives
- Academic representatives

POTENTIAL HEALTH OR ECOLOGICAL RISKS IN WATERSHEDS

- Industrial wastewater discharges
- Municipal wastewater, stormwater, and combined sewer overflows
- Waste dumping and injection
- Nonpoint source runoff or seepage
- Accidental leaks and spills of toxic substances
- Atmospheric deposition
- Habitat alteration, including wetland loss
- Stream flow alteration

EXAMPLE OF COORDINATED WATERSHED ACTIONS

- Voluntary source reduction programs (e.g., BMPs, waste minimization)
- Permit issuance and enforcement programs
- Standard setting and enforcement programs (nonpermitting)
- Direct financing
- Economic incentives
- Education and information dissemination
- Technical assistance
- Remediation of contaminated soil or water
- Emergency response to accidental leaks or spills

TOTAL MAXIMUM DAILY LOAD (TMDL) PROCESS

Section 303(d) of the Clean Water Act established the TMDL process for determining the allowable loadings or other quantifiable parameters for waterbodies and provided the basis for States to establish water quality-based controls.

TMDL = LC = WLA + LA + MOS

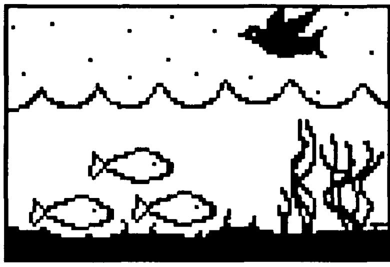
LC = LOADING CAPACITYthe greatest amount of loading the water can receive without violating water quality standards (also sometimes called "assimilative capacity")

WLA = WASTE LOAD ALLOCATIONproportion of LC allocated to existing or future point pollution sources

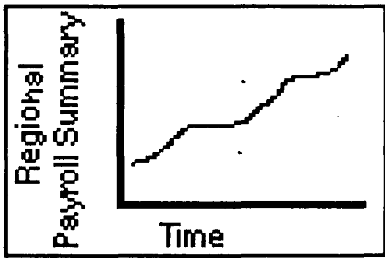
LA = LOAD ALLOCATIONproportion of LC allocated to existing or future nonpoint pollution and/or natural sources

MOS = MARGIN OF SAFETYa required estimate for the uncertainty between pollutant loads (WLA + LA) and receiving water quality

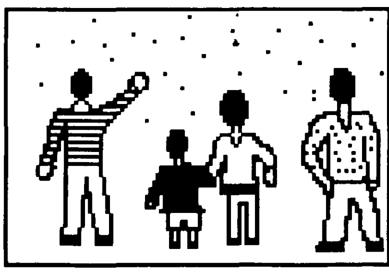
EXAMPLE WPA GOALS



Protect and expand of physical habitat (e.g., forest cover, wetland) throughout the watershed and create contiguous habitat corridors along the margins of streams and rivers.

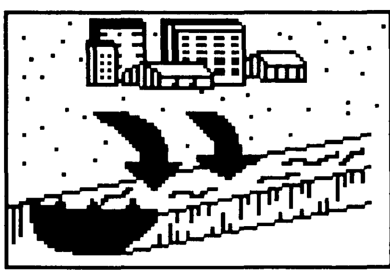


Ensure that natural resource mangement programs are consistent and supportive of local economic development programs.

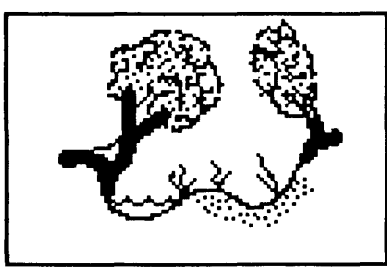


Increase public awareness of its key role in the watershed planning and efforts to cleanup the river; increase volunteer participation in watershed restoration activities.

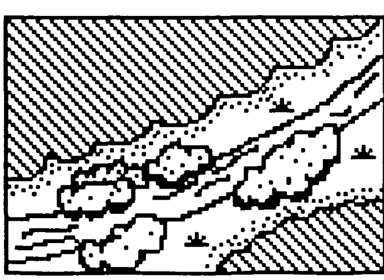
EXAMPLE WPA GOALS



Dramatically reduce pollutant loads delivered to the receiving waters to improve water quality conditions by the turn of the century.



Protect and restore the ecological integrity of streams to enhance aquatic diversity, provide for a quality fishery, and other recreational opportunities.



Restore biological integrity to historical conditions, including aquatic species, terrestrial fauna, and vegetative communities.

EMERGING EPA ROLE IN ECOLOGICAL RESTORATION

- Establish the significance of the relationships between ecological restoration and water quality within the overall TMDL process
- Demonstrate the utility of ecological restoration to encourage wider use of ecological restoration techniques and principles to improve water quality
- Develop tools and methods to integrate ecological restoration into the TMDL development process
- Foster the continued development of a network of ecologists and other scientists within and outside EPA to continue to interject ecological values into ongoing water quality management programs
- Investigate the use of biological criteria and indicators, Rapid Bioassessment Protocols, etc., as practical ways to incorporate general ecological concerns into the TMDL process and water programs

SELECTED EXAMPLES OF PAST AND ONGOING PROJECTS

- WHOLE BASIN APPROACH TO WATER QUALITY MANAGEMENT FOR NORTH CAROLINA, WASHINGTON, DELAWARE, TEXAS — led or is leading to statewide frameworks for permitting, monitoring, modeling, NPS assessments, and planning.
- WATER QUALITY REGULATIONS AND APPROACHES TO SUPPORT ECOLOGICAL PRESERVATION AND RESTORATION — a literature review and synthesis on program approaches, restoration techniques, and methods identifying eligible systems.
- IMPROVEMENT STUDIES FOR SEGMENT 15 OF THE SOUTH PLATTE RIVER — assessment of use limitations due to past NH_3 and chlorine impacts and ongoing habitat and DO impacts.
- ASSIMILATIVE CAPACITY ASSESSMENT FOR THE SANTA MARGARITA RIVER — evaluation of point and nonpoint impacts on nutrient and aquatic community dynamics in this southern California river-estuary system.

RELATIVE BENEFITS OF TYPICAL STREAM HABITAT RESTORATION PRACTICES ON SELECTED WATER QUALITY CONDITIONS

Restoration Practice	Minimum DO	Temperature	pH	NH_3	Suspended Solids	Toxic Metals
Build Drop Structures	↑	O	↕	↓	O	O
Increase Channel Depth, Narrow Stream Width, Increase Undercut Banks	↑	↓	↓	↓	O	O
Plant Riparian Vegetation	↑	↓	↓	↓	↓	O
Augment Wetlands	↕	O	↕	↓	↓	↓
Build Settling Ponds on Tributaries	↑	↓	↓	↓	↓	↓

Evaluation of a Restoration System for Sandhills Longleaf Pine Communities

K.W. Outcalt[1]

Abstract — Longleaf pine (Pinus palustris Mill.) communities, which once covered 60 to 80 million acres, have declined to near 3 million acres. To reverse this trend will require restoration of damaged areas. The study purpose was to evaluate the effectiveness of hexazinone for restoring longleaf pine communities on dry sandhills sites. Three areas were treated with hexazinone (2 lbs ai/acre) by spot gun application followed by v-blade planting of containerized longleaf pine. Response of the plant community was monitored by collecting cover data from transects in treated and untreated control plots before and after treatment application. The hexazinone reduced turkey oak (Quercus laevis Walt.) cover from 6 to 0.4 percent. All areas were dominated by wiregrass (Aristida stricta Michx.) prior to treatment. The mechanical disturbance caused by v-blade planting reduced wiregrass cover from 51 to 37 percent and created bare soil on 32 percent of the area. Wiregrass recovered the following season to 49 percent cover while bare soil declined to 7 percent. There were species gains and losses on control and treated plots, but there were no species which were specifically eliminated from the community. Thus, the treatment was successful at reducing the woody component of the community and promoting the successful re-establishment of longleaf pine without undue long-term harm to the other species of the plant community.

INTRODUCTION

Longleaf pine (Pinus palustris Mill.) ecosystems once occupied perhaps as much as 80 million acres in the Southeastern United States, stretching from southeastern Virginia south to central Florida and west into eastern Texas. These fire-dependent ecosystems have been intensively exploited since colonial times, with little regard for regeneration, resulting in a decline to less than 3 million acres today. The continuing reduction of this important forest type carries with it a risk to the myriad of life forms characteristic of, and largely dependent on, longleaf pine ecosystems. Extreme habitat reduction is the main cause for the precarious state of at least 191 taxa of vascular plants. A committed effort to restore and manage longleaf pine ecosystems will help insure a future for an important part of this nation's natural heritage. The purpose of this study was to assess the effect of a restoration system, using hexazinone and V-blade planting of longleaf pine, on the understory plant community.

METHODS

- Study Location - Ocala National Forest, central highlands region of Florida, USA.
- Chose three stands (i.e. blocks) dominated by scrub oaks for restoration treatment.
- In each block a square 0.6 acre control plot was established.
- All but control area was treated with spot gun application of hexazinone.
- Application rate 2 lb a. i./ acre on 6 by 6 foot grid in late May, 1991.
- Plant cover assessed along 50 foot line transects.
- 10 transects in treated part and 2 in control portion of each stand.
- Data collected pretreatment (May, 1991), 1st season (Oct., 1991), 2nd season (Oct., 1992).

[1] *USDA Forest Service, Gainesville, FL., USA.*

RESULTS

- There was a significant increase in wiregrass (Aristida stricta) on control sites the 1st season.
- There were no other significant changes in plant species cover on control sites.
- A significant reduction in wiregrass occurred on treated sites the 1st season.
- By 2nd season wiregrass had recovered to pretreatment level. Wiregrass declines resulted mostly from scalping during planting as shown by corresponding increase in bare soil.
- Broomsedge (Andropogon virginicus) also declined initially but then recovered.
- Hexazinone reduced turkey oak (Quercus laevis) cover. Gopher apple (Licania michauxii) cover declined but not by a significant amount.
- Cover of Balduina angustifolia, Eupatorium compositifolium, Pityopsis graminifolia, Polygonella gracilis, and Sorghastrum secundum increased the second season on treated areas.
- There were no significant changes in cover for any of the other plant species in the community.
- Hexazinone treatment reduced the importance of turkey oak in the community.
- Soil disturbance from the V-blade increased bare soil the 1st season but it declined rapidly and by the 2nd season wiregrass again dominated all areas.
- Plant species richness increased on treated areas from species invading bare soil microsites.

Sustaining Rangelands: Application of Ecological Models to Evaluate the Risks of Alternative Grazing Systems

Mark E. Ritchie and Michael L. Wolfe[1]

Abstract. — Sustaining natural ecosystems requires evaluating the consequences of unpredictable environmental events, e.g. precipitation, human disturbance. On North American rangelands, managers are concerned with sustaining plant communities in the face of grazing by livestock and wild herbivores and unpredictable precipitation. We present a model for evaluating the probability that a given rangeland plant community can be sustained over a specified time period while subject to grazing. The model describes the population dynamics of herbivore and plant species in terms of their mechanisms of resource acquisition, growth, and species interactions. We then input randomly varying annual precipitation, a livestock grazing strategy, and a wildlife harvest strategy to project the future dynamics of herbivore and plant species. Iteration of model projections for different random sequences of annual precipitation calculates the probability that a particular grazing system will produce unacceptable consequences (biological or political). As an example, we apply the model using data from our current study of plant-herbivore interactions at Desert Land and Livestock in northern Utah. We show that the modeling approach can provide valuable insights for the management of herbivores to sustain ecosystems.

INTRODUCTION

To sustain ecosystems, managers need to know how ecosystems respond to manipulations (active management practices) and unpredictable environmental events (e.g. weather, human disturbances). More specifically, managers need to evaluate risks, or the probability of undesirable responses to their management practices (Loucks 1985). Such evaluation is called ecological risk assessment (Bartell et al. 1992). While empirical responses of organisms have been used by toxicologists to assess risk (e.g. Hendrix 1982, Suter et al. 1983), complexity and the lack of good experimental data has discouraged such approaches for whole ecosystems (Giesy 1980).

An alternative to using empirical responses to measure ecological risk in ecosystems is to model (Bartell et al. 1992). Modeling provides several powerful advantages: (1) complex interactions among organisms can be considered, (2) long-term responses can be explored, and (3) the consequences of alternative management plans can be compared. However, modeling has a significant weakness: model predictions may not reflect reality (Caswell 1975). One way to improve the match between empirical data and modeling is to use "mechanistic" ecological theory based on the known biology of the organisms within ecosystems (Schoener 1986, Tilman 1980, 1987). Mechanistic models can predict real dynamics of organisms, e.g. population growth (Schoener 1973), competition (e.g. Tilman 1976, Rothhaupt 1988), and predation (Werner and Hall 1988). Consequently, simulation models of ecosystems that are based on mechanistic models of population growth and species interactions may be useful tools for evaluating ecological risks in managing ecosystems. In this paper, we demonstrate how such a modeling approach might work by considering a specific management problem and constructing an example simulation model.

A central problem in sustaining North American rangelands is evaluating the impacts of grazing by livestock and wild herbivores in the face of unpredictable annual precipitation

[1] *Mark E. Ritchie and Michael L. Wolfe are faculty members in the Department of Fisheries and Wildlife, Utah State University, Logan, UT 84322-5210.*

(Heitschmidt and Stuth 1991). On most rangeland, livestock grazing is a major land use with important economic implications. Livestock producers have traditionally perceived competition for forage from wildlife as a threat to their livelihood (Bastian et al. 1991). However, environmental groups increasingly perceive livestock as a threat to sustaining biodiversity on rangelands (Ferguson and Ferguson 1983). To resolve this conflict, managers choose from different livestock grazing strategies and wildlife harvest strategies to maintain desired (acceptable) populations of plants and other animals. Because animal and plant production is often highly variable, making these choices depends on some type of risk assessment, i.e. evaluating which grazing strategy is most likely to produce the desired goal. Such evaluation from existing empirical information is difficult because the interactions of multiple species of rangeland plants and herbivores are not yet well understood (Coughenour 1991). Risk assessment in this case involves understanding complex interactions, long-term results, and the consequences of many possible management alternatives, so modeling may be the only reasonable way to find a solution to the problem.

In this paper, we develop, validate, and explore a simple simulation model to address the implications of different livestock grazing and wildlife harvest strategies on rangeland ecosystem sustainability. The model uses simple equations that describe the population dynamics of herbivores and plants, where herbivores are limited by plant abundance and plant production is limited by water availability. Our goal was to describe these dynamics as simply as possible with the fewest data inputs, since managers are unlikely to ever have extensive, detailed data sets with which to model. In addition, we made the model as general as possible, but left room for site-specific inputs of herbivore and plant species as well as precipitation. We tried to take typical manager's viewpoint of having a vexing problem but scarce resources, little time, and few data with which to address the problem.

MODELS OF POPULATION DYNAMICS

To describe the dynamics of plants and herbivores, we used simple, previously established mechanistic population growth models from the ecological literature (Schoener 1973, Tilman 1980). In doing so, we made several simplifying assumptions. First, we assumed that herbivores and plants were resource limited, i.e. herbivores were limited by plant abundance and plants were limited by a single resource (e.g. water, nitrogen, light). Second, we assumed that populations had no age or size structure. Third, we assumed that there was no physical disturbance to the community, e.g. fire, soil disturbance, etc. Fourth, we assumed that dynamics could be described with difference equations, i.e. population changes occurred in discrete intervals or "pulses". We made these assumptions to keep the model simple and data inputs to a minimum. Population growth for a given plant group i (e.g. grasses, forbs, shrubs) was described in terms of the change in biomass (N_i, g/m^2) from one growing season to the next (time $t+1$ - t):

$$N_{i,t+1} - N_{i,t} = C_i N_{i,t} [(S_N / \sum_{i=1}^{n} N_{i,t}) - M_i] - \sum_{j=1}^{x} [h_j(N_{i,t}) H_{j,t}] \quad (1).$$

C_i is the nutrient-use efficiency of plant group i (g tissue produced per g nutrient). M_i is the maintenance nutrient requirement for plant group i per unit above-ground biomass during its growing season. S_N is the supply rate of nutrient (g/season). The function $h_j(N_{i,t})$ is the consumption rate (g/season) of plant group i by an individual of herbivore group j as a function of plant biomass. $H_{j,t}$ is the density (#/m^2) of herbivore group j during time t. The variable n is the number of plant groups and x is the number of herbivore groups. Thus, plant dynamics depend on nutrient availability, their efficiency at utilizing nutrients, and the intensity of herbivory. Note that the plant groups compete exploitatively for the limiting resource.

The population growth of each herbivore species j was described as a function of the species' dry-matter intake of each plant group, its ability to utilize that intake, and its rate of harvest:

$$H_{j,t+1} - H_{j,t} = G_j H_{j,t} \{\sum_{i=1}^{n} [D_i h_j(N_{i,t})] - R_j\} - P_j H_{j,t} \quad (2).$$

G_j is the conversion efficiency of energy into new offspring for herbivore group j (offspring/kJ). D_i is the dry-matter digestible energy content of plant group i (kJ/g). R_j is the energy requirement (kJ/season) of herbivore group j. Finally, P_j is the proportion of herbivore group j harvested each season (a different harvest function could easily be used). Other parameters and functions are the same as in Eqn. 1. Herbivore groups compete exploitatively by indirectly reducing the biomass of plant groups.

Consumption of each plant group by a given herbivore group is a complex function that depends on plant biomass, Time available for foraging, proportion of the plant group in the diet, herbivore bite size and herbivore movement rate (Spalinger and Hobbs 1992).

$$h_j(N_{i,t}) = \frac{T a_j q_i N_{i,t}}{1 + b_j (\sum_i q_i N_{i,t})} \quad (3).$$

T is the time the forager spends foraging (min/season). The variable q_i is the product of the diet proportion and aboveground biomass proportion of the plant group i . The variable a_j reflects the herbivore group's search capability (area/min). The variable b_j reflects the herbivore group's handling cost (area/g), and reflects the time required to bite all the food items encountered per unit area. This variable is a function of bite size and search capability (Spalinger and Hobbs 1992).

EXAMPLE SIMULATION

These population dynamics models are useful to managers only when parameter values, nutrient inputs, and initial conditions are specified. To demonstrate how these models can be used, we will perform an example simulation using specific data from a field study site, Desert Land and Livestock (DL&L), a 911 km^2 ranch in northern Utah (elevation 1900 - 2600 m). The results of this analysis should be viewed as an example of how modeling can be used to address management problems, rather than as a general statement about plant-herbivore interactions.

Rangelands at DL&L consist of two types, winter range (sagebrush grassland) and summer range (montane meadows interspersed with timber). Mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*) and cattle are the major livestock species on the ranch. Competition among wildlife and cattle for spring and summer range is often most controversial (Bastian et al. 1991. Consequently, we analyzed the effects of different grazing strategies and wildlife harvest rates on the long-term impacts of elk, deer, and cattle on summer range vegetation. To simplify the model, vegetation was grouped as grasses, forbs, and shrubs. For the intermountain West, water is the nutrient most likely to limit plant production, even on summer range (MacMahon and Schimpf 1991). Consequently, we used water as the nutrient limiting plant growth in our simulation.

Our simulation attempted to capture the natural timing and use of summer and winter range by these herbivores. We divided each year into two seasons: summer (150 days) and winter (210 days), and calculated changes in plant biomass and herbivore densities in each season. We assumed that cattle density changed only with stocking rate. Elk and deer densities were assumed to change with plant biomass, with summer range biomass affecting reproduction and winter range biomass affecting mortality. Consequently we modeled the dynamics of plants on both summer and winter range as well as the dynamics of deer and elk.

For plants, we obtained average parameters for each plant group from the literature (Table 1), including water-use efficiency and seasonal water requirements. For each plant group, average dry-matter digestible energy content for deer and elk as well as diet proportions for cattle, deer, and elk were also obtained from the literature (Table 1). Proportions of above-ground biomass were 0.3 for grasses, 0.5 for forbs, and 0.25 for shrubs (MacMahon and Schimpf 1981). For herbivores, we estimated average energy conversion efficiency, maintenance energy requirements, search ability, and handling costs from herbivore body mass using allometric relationships (Peters 1983, Calder 1984) (Table 2). Daily feeding time was approximately 300 min/day for all three herbivores (Belovsky and Slade 1986).

Except for cattle densities, initial conditions were kept constant in all simulation runs. For winter range, initial biomasses (g/m^2) of plants were: grasses, 25; forbs, 10; shrubs, 100. For summer range, biomasses (g/m^2) were: grasses, 100; forbs, 10; shrubs, 300. Deer and elk densities each began at $10/km^2$.

Table 1. — Average plant characteristics used in the population dynamics equations for plants and herbivores in the simulation model.

Characteristic	Grasses	Forbs	Shrubs
Water-Use Efficiency (g tissue/g H_2O)[1]			
Summer Range	0.0018	0.0020	0.0028
Winter Range	0.0023	0.0031	0.0040
Water Requirements (g H_2O-g $tissue^{-1} \cdot season^{-1}$)[2]			
Summer Range	68.7	100.4	77.7
Winter Range	53.8	64.8	54.4
Dry-Matter Digestible Energy Content (kJ/g dry mass)[3]			
Elk			
Winter	7.02	NA	4.38
Summer	12.2	13.1	8.77
Deer			
Winter	5.2	NA	7.89
Summer	9.1	15.9	10.5
Diet Proportions[4]			
Cattle	1.00	0	0
Elk			
Winter	0.60	0	0.40
Summer	0.40	0.35	0.25
Deer			
Winter	0	0	1.00
Summer	0.12	0.46	0.42

[1] *During growing season, Refs: Miller 1988, Romo and Haferkamp 1989, Warne et al. 1990, Singh et al. 1991.*
[2] *During growing season, Refs: Detling et al. 1979, Atkinson 1986, Irving and Silsbury 1987, Miller 1988.*
[3] *Refs: Robbins 1992, Frank and Karn 1988.*
[4] *Refs: Mackie 1970, Belovksy 1986.*

Table 2. — Allometric body mass relationships used in the population dynamics equations for herbivores in the simulation model[1].

Parameter	Equation[2]
Energy Requirements KJ/season	$1.2 \times 10^5 M^{0.72}$
Conversion Efficiency offspr/KJ	$0.000153 M^{-1.33}$
Search Ability M2/min)	$0.1 M^{0.54}$
Handling Cost (area/g)	$1.3 M^{-0.37}$

[1] *Refs: Peter 1983, Calder 1984.*
[2] *M = body mass in kg.*

Because water was assumed to be the major limiting nutrient for plants, we used precipitation to measure water availability. Annual and even seasonal precipitation in the intermountain West is unpredictable, so we treated it as a random variable. We used crop-year (April - September) precipitation measured for 1950-1990 at the two closest weather stations to DL&L (summer range: Monte Cristo ranger station, Utah; winter range: Woodruff, Utah). To generate a random sequence of annual water availability, we picked random values from the distribution of crop-year precipitation at each weather station. Thus, each year of a simulation run differed in precipitation, and each run differed in its sequence of annual precipitation. To estimate the actual water available during the growing season, crop-year precipitation was then multiplied by 0.29 to account for run-off, evaporation, and percolation below the rooting zone (Johnson and Gordon 1988).

To measure effects of management strategies, we simulated the population dynamics resulting from each strategy for twenty years (a typical target time frame for management decisions). Management strategies were implemented in the form of different cattle stocking rates (grazing strategies) and different harvest proportions (harvest strategies). Effects of strategies on biomass of each plant group and herbivore densities were estimated from the mean biomass in year 20, based on 50 runs. The probability that a plant group would go extinct was calculated as the frequency of 1000 runs in which that plant group was reduced to zero biomass.

MODEL VALIDATION

Any simulation model requires validation to be useful. As a preliminary validation of the model presented here, we chose to compare the densities of elk predicted by the model for different cattle stocking rates with elk densities observed in summer range pastures stocked with different cattle densities. We chose this test because predicted herbivore densities are most likely to reflect compounded errors or incorrect assumptions in the population dynamics equations (Caswell 1975). Ken Clegg (unpubl. data) provided ground counts of elk and cattle using different 5-10 km^2 drainages within 400 km^2 of summer range at DL&L in 1992.

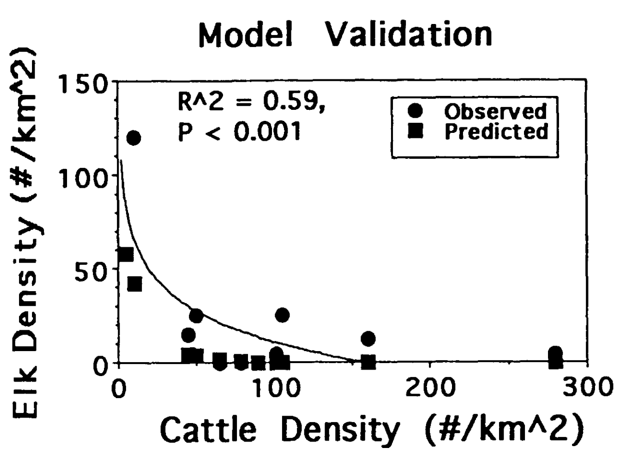


Figure 1. — The relationship between elk density vs. cattle density observed in different watersheds on Desert Land and Livestock summer range in 1992 (closed circles). Regression line is y = 120.4 - 12.7 log(x). Elk densities predicted with the simulation model for the same cattle densities are also shown (solid squares).

Elk densities declined negatively and non-linearly with cattle densities (Fig. 1). Using the observed cattle densities as inputs, we predicted elk densities with our model (Fig. 1). The predicted densities tend to underestimate observed elk densities, but the same negative, non-linear relationship with cattle densities is predicted. Predicted (P) and observed (O) elk densities are also positively correlated (r^2 = 0.66, O = 9.6 + 1.45 P, P < 0.001) and the slope of the relationship is not different from one (t = 0.92, df = 9, P = 0.36). These results suggest that our model may be useful for describing the qualitative relationships among herbivores and, by inference, the effects of these herbivores on plant biomass. From this comparison, we argue that the simulation model can provide some insights into the effects of potential management strategies at DL&L.

MODEL PREDICTIONS

We predicted sustainability of plant groups in two ways: (1) mean biomass, and (2) the probability of extinction. Greater biomass is often used (directly or indirectly) as a measure of land "health" or "condition" (Heitschmidt and Stuth 1991). Probability of extinction is the chance that the density of a group or species is reduced to zero within a specified time. In variable ecosystems, this probability is always greater than zero, since there is always some chance, however small, that a population will go extinct (Goodman 1987).

For these measures, we addressed three important questions about herbivores and ecosystem sustainability.

(1) What is the effect of increasing herbivore density on plant production and biomass?
(2) Does a mixture of livestock and wildlife have less impact on plant biomass and diversity than livestock alone?
(3) At what densities do herbivores begin to reduce biodiversity or degrade land?

We used our model to provide answers about the DL&L system; the results may apply to other systems as well, but such generality awaits future tests.

Different herbivore densities were produced by altering cattle stocking densities and wildlife harvest rates. We simulated the effects of grazing strategies by stocking 0-100 cattle/km^2, while allowing wildlife to attain unharvested densities. The typical pattern for summer range in northern Utah is a 2 or 3 pasture rotation, i.e. cattle are moved at a density of 20-30/km^2 through three pastures during the course of the summer and each pasture is grazed only once (Heitschmidt and Stuth 1991). Thus, a typical stocking rate produces an overall density of 7-10 cows/km^2. However, cattle densities in preferred-use areas (e.g. riparian areas, wet meadows) may greatly exceed the overall rate. To simulate wildlife harvest strategies, we used harvest rates ranging from 0-0.5 of the density of deer or elk, while holding cattle densities at 9/km^2. The typical harvest rate for these species ranges from 0.05-0.15 (Utah Division of Wildlife Resources harvest books). A harvest rate of 0.5 approximates a maximum sustained yield harvest (Getz and Haight 1989).

Herbivore Effects on Plant Biomass

Because we input randomly varying precipitation, simulation runs with the same initial conditions produced different results. Consequently, we analyzed effects of grazing and harvest strategies on the <u>mean</u> response of plants and herbivores and tested the statistical significance of differences in responses with standard analysis of variance.

Increased cattle densities significantly reduced grass biomass but increased shrub biomass (Fig. 2). Intermediate cattle densities significantly increased forb biomass. Increased wildlife harvest rates had less dramatic effects (Fig. 3). Grass biomass decreased significantly at intermediate harvest levels, while forb biomass increased significantly at only the highest harvest level. Harvest rates had no significant effects on shrub biomass. These results suggest that plant biomass is more sensitive to cattle stocking rates than to wildlife harvest rates. The results also suggest that indirect effects can be as important as direct consumptive effects. For example, increased cattle densities led to increased shrub biomass because cattle grazing reduced competition between grasses and shrubs, thereby increasing shrub vigor.

Effects of Single vs. Multiple Herbivores

We tested whether wildlife species could affect the impact of cattle grazing on ecosystems. Specifically, we compared grass biomass predicted for three different cattle densities under two types of simulations (Fig. 4). First, we kept wildlife density at zero (No Wildlife). Second, we began with 10/km^2 each of deer and elk and allowed them to undergo simulated dynamics with no harvest (With Wildlife). With no cattle stocked, adding wildlife did not affect grass biomass. As cattle density increased, however, adding wildlife increased grass biomass, and the magnitude of increase grew with increasing cattle density. This pattern was due to indirect effects, namely wildlife reducing shrub biomass and competitive effects on grasses, thereby increasing grass vigor. These results suggest that multiple herbivore species, which are likely to consume a variety of plant groups or species, may improve ecosystem sustainability.

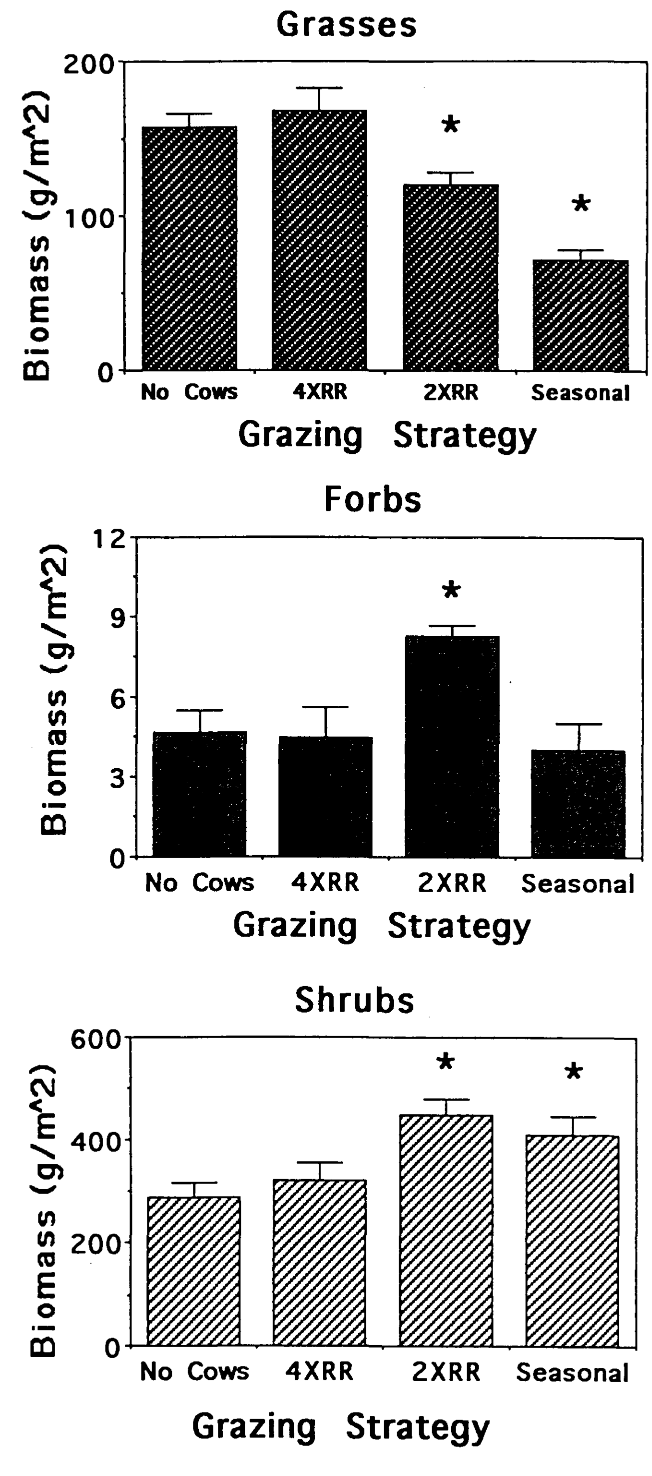


Figure 2. — Predicted effects of different cattle grazing strategies on standing crop biomass of three principal plant groups from the simulation model. The four grazing strategies tested were, in increasing order of grazing intensity, no cattle, 4 pasture rest-rotation (4XRR) (9/km^2), 2 pasture rotation (2XRR) (18/km^2), and seasonal (cattle stocked in a single pasture for 180 days, 36/km^2). Asterisks indicate significant differences from the no cattle treatment.

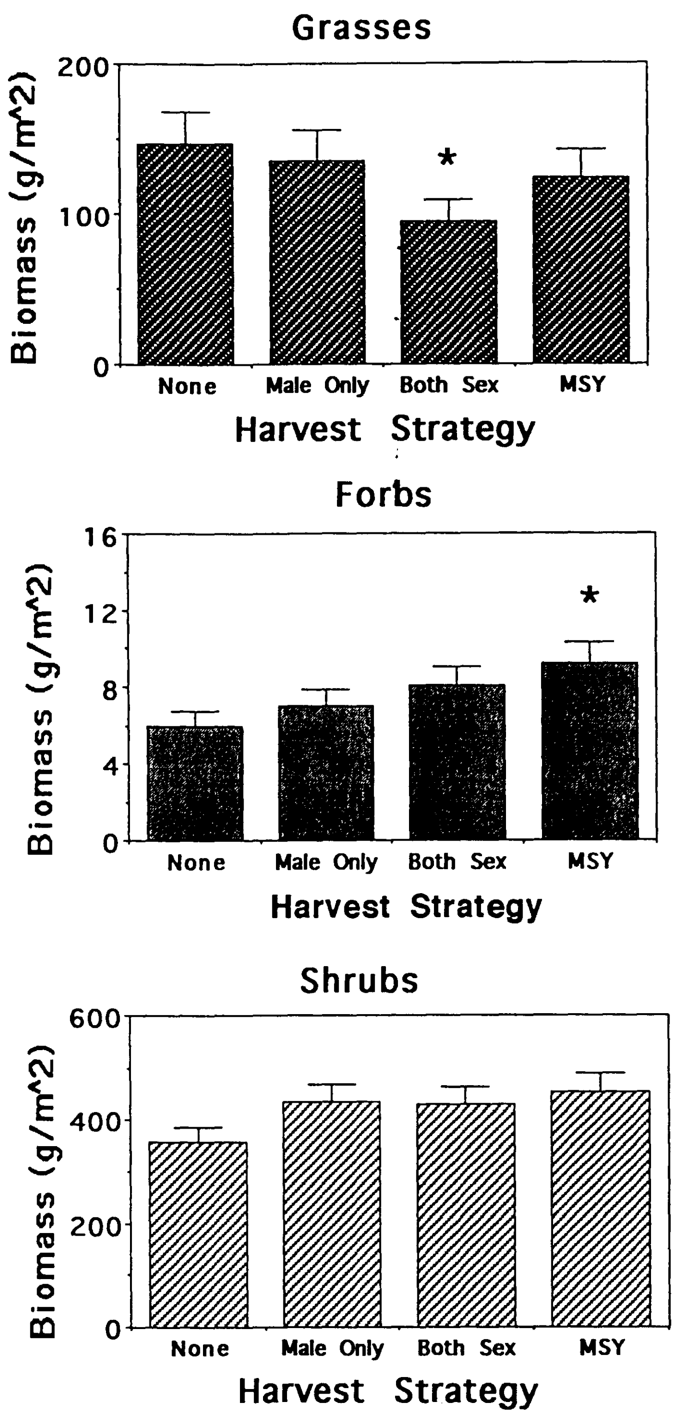


Figure 3. — Predicted effects of different wildlife harvest strategies on standing crop biomass of three principal plant groups from the simulation model. The strategies tested were, in increasing order of harvest intensity, no harvest, all-male harvest (10% of density), harvest of both sexes (20%) and maximum sustained yield (50%). Asterisks indicate significant differences from the no harvest treatment.

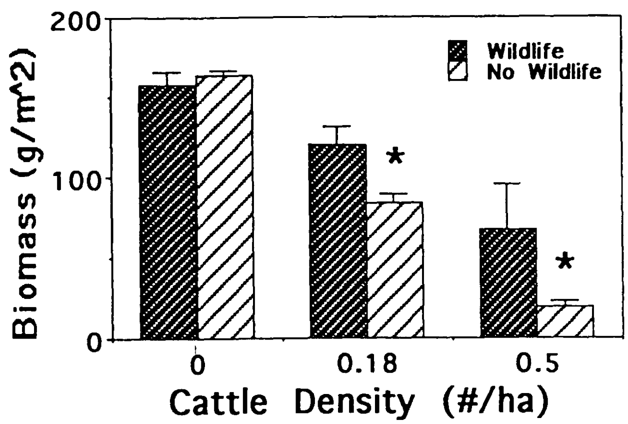


Figure 4. — Predicted effects of the presence of wildlife on the impact of cattle on grass biomass. Predictions are shown for three different cattle stocking densities (0, 18, and 50 cows/km^2) and for no wildlife vs. wildlife occurring at densities predicted for each cattle stocking density (no harvest). Asterisks indicate significant differences for a given cattle density.

Herbivore Effects on Extinction

We addressed the possibility that management strategies might result in the extinction of plant groups or species, and thus fail to sustain the original ecosystem (Fig. 5). We calculated probabilities of extinction within twenty years for grasses, forbs, and shrubs. We estimated two types of extinction: (1) probability of diversity loss (one or more plant groups going extinct), and (2) probability of land degradation (grasses going extinct). Diversity loss occurred primarily but not always from forbs going extinct.

Without cattle and at maximum wildlife harvest rate, probabilities of diversity loss and land degradation in twenty years were less than 1×10^{-6}. With zero wildlife harvest but no cattle the chance of diversity loss increased to 27%. Stocking cattle with unharvested wildlife further increased the chances of diversity loss. Typical cattle stocking densities with unharvested wildlife produced a 30-40% chance of diversity loss. With no cattle, the probability of diversity loss declined rapidly with increasing wildlife harvest. Overall, wildlife harvest rates had a larger impact on reducing diversity loss than stocking fewer cattle. This result is due to wildlife feeding preferentially on the rarest plant group, forbs.

The chance of land degradation changed only with increased cattle density; it was unaffected by wildlife harvest rate. Typical cattle stocking densities produced a low chance of degradation (< 0.5%). Chances of degradation increased rapidly, however, for cattle densities in the range of 50-100/km^2. Such densities are typically observed in riparian areas (Ferguson and Ferguson 1983). At low cattle densities, grasses are able to sustain a large enough biomass to avoid extinction in low precipitation years. There appears to be a threshold, however, where cattle reduce grasses to a level where they are vulnerable to extinction by drought.

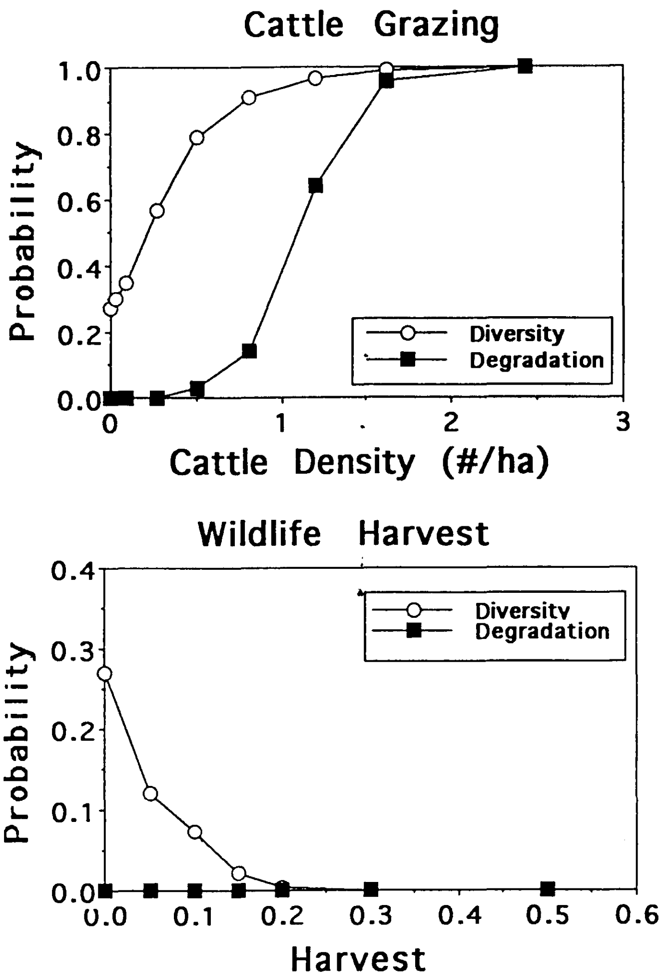


Figure 5. — Predicted effects of cattle stocking and wildlife harvest on the sustainability of a grass/forb/shrub ecosystem. We calculated the probability that diversity would fail to be sustained, i.e. at least one functional group (grasses, forbs, or shrubs) would go extinct in twenty years (open circles). We also calculated the probability that land degradation would occur (grasses would go extinct) (solid squares). Probabilities of extinction were calculated by repeating simulations 1000 times, each time with a different random sequence of annual precipitation, and calculating the proportion of runs resulting in extinction.

DISCUSSION

The example simulation model we present illustrates how models can be used to evaluate alternative management decisions for ecosystems. Specifically, we show that basic ecological theory can be put into practice with a four step process: (1) consider mechanisms of population growth and species interactions, (2) find data for these mechanisms for the appropriate species at a given site, (3) validate the model, and (4) apply different management strategies by altering model inputs. Such an approach does not produce a single, general model that "will work anywhere"; rather the approach defines an organized way to synthesize information and make better guesses about how the ecosystem of interest works.

For our example system at DL&L, the simulations suggest that different management strategies should be used for different types of sustainability goals. If the management goal is to produce cattle but also maximize "condition" or grass biomass, then wildlife should be either unharvested or harvested at a low rate and cattle should be stocked in a rotational grazing scheme. If the management goal is to maximize plant diversity, then wildlife should be harvested at a high rate and cattle should not be stocked. On the other hand, if the management goal is to maximize wildlife density (as in a camera park), then cattle should definitely not be stocked and some plant diversity should be expected to be lost.

DL&L has a goal of maintaining range "condition" and avoiding land degradation, while simultaneously making some economic profit. Interestingly, their management strategies are to use an extensive rest-rotation cattle grazing system and harvest 5-10% of wildlife density each year. These would be the management strategies predicted to be "best" for the management goal by our simulation model. DL&L uses extensive, long-term, empirical data collected at the ranch to make their decisions; it is reassuring that our model predictions, which actually use no data from the ranch, make similar recommendations as the ranch managers.

The modeling results predict that some management strategies may be mutually exclusive, or trade-off. For example, maximizing plant diversity is best achieved by heavy harvesting of wildlife, which may risk population crashes or extinction of herbivores. Thus, improving plant diversity is likely to be incompatible with sustaining large herbivore populations. Likewise, improving range condition may be incompatible with increasing or sustaining plant diversity. For example, cattle densities might be increased without reducing diversity if wildlife harvest rates are also increased, but this is likely to lead to reduced grass biomass and poorer range "condition". Such trade-offs in the consequences of different management strategies are often the source of intense controversies in natural resources management (e.g. Wagner 1978, Singer and Schullery 1989). These trade-offs require the use of optimization techniques to decide which combinations of management strategies will achieve biologically or politically acceptable criteria (Bastian et al. 1991, Loomis et al. 1991). Perhaps the use of models based on ecological theory may help managers to understand and solve these conflicts better.

The model predictions are driven mainly by two assumptions: (1) plants compete, and (2) wild herbivores reduce the biomass of both the dominant plant group (shrubs) and the rare poorly competitive plant group (forbs). The first assumption is likely to be true, as plants have been shown to compete in most environments (Grace and Tilman 1990). The validity of the second assumption depends upon the relationship between plant competitive ability and its palatability to herbivores (Pacala and Crawley 1992). In the model, the most palatable plants to wildlife, forbs, are the poorest competitors, the rarest, and the most vulnerable to extinction. Consequently, increased wildlife densities increase the chance of forb extinction.

Competition among plants produces indirect effects among herbivores and plants, such as positive interactions between herbivores and non-forage plants and between herbivores (Grace and Tilman 1990). Such indirect effects have been documented in the literature (e.g. Urness 1975, Reiner and Urness 1982, Brown and Heske 1990). Indirect effects were crucial in determining sustainability in our model system. For example they explain why the presence of wildlife increases grass biomass (Fig. 4). The importance of indirect effects is consistent with the idea of "holistic" management (Savory 1988), in that sustainable ecosystems incorporate many interacting processes and managers must maintain a "balance" of these processes or risk a break-down of the system. A model, such as ours, can provide valuable clues as to how to maintain such an ecosystem.

The model predictions also indicate that considering variability is crucial in determining sustainability. For example, the effects of herbivores on mean biomass of different plant groups suggested that increasing wildlife density should improve sustainability (in terms of biomass) (Figs. 3, 4). However, calculating probabilities of extinction, which incorporated variability in precipitation and wildlife density, revealed the opposite prediction: increasing wildlife density should decrease sustainability (in terms of diversity). Thus, risks of undesirable outcomes to management may be independent of the mean outcome. Too often, land managers and ecologists have examined only average responses of plants and animals to management strategies (Chesson 1985). Risk management incorporates this variability (Fleisher 1990) and is an alternative to traditional problem-solving management techniques that may prove invaluable for sustaining ecosystems. The consequences of variability and risk can usually only be evaluated with many repetitions of experiments or calculations (Goodman 1987, Belovsky 1987); models may be indispensable for such analyses.

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The Potential Effects of Increased Temperatures and Elevated Ambient Carbon Dioxide on Loblolly Pine Productivity: Results From a Simulation Model

David Arthur Sampson[1]

Abstract — Loblolly pine forests of the southeastern United States represent a vast, economically and biologically important land base. General circulation models predict increased temperatures for this region. We used the process model BIOMASS in conjunction with empirical field data to explore "potential" loblolly pine productivity simulated under increased temperatures, increased CO_2 concentration, and two treatments in low and high productivity sites.

Simulation output suggested a net increase in stand productivity under a doubling of ambient CO_2 (700 ppm) and a four degree Celsius increase in daily temperatures. Low productivity sites increased from 3.5 to 5.7 Mg C ha^{-1} $year^{-1}$ while high productivity sites increased from 7.7 to 11.6 Mg C ha^{-1} $year^{-1}$ in control plots. This represented a 63% and 51% increase in net carbon flux for low and high sites, respectively. A doubling of CO_2 under ambient temperatures in control plots increased net carbon gain by 93% and 52% for low and high sites, respectively. Maintenance respiration (Rm) accounted for a 26% loss in net carbon available for growth for low sites. Gross carbon fixed increased by approximately 18% for high sites in fertilized plots resulting in a 14% increase in net carbon gain.

INTRODUCTION

Models incorporating theoretical and empirical algorithms to simulate growth are used to examine the processes influencing forest productivity. These process models may be mechanistic; rate equations that characterize the biophysics of carbon fixation and carbon partitioning are used in a time-step model based on field experiments (e.g. McMurtrie et al. 1992). Factors that control or influence photosynthesis and respiration determine the amount of carbon fixed (Larcher 1983), while differences in partitioning of the fixed carbon depend at least in part on sink strength (Cannell 1985) and resource availability (Gholz et al. 1986; Keyes and Grier 1981). Biochemically sensitive algorithms that interpret photosynthesis based solely on the kinetics of Rubisco are available (Farquhar et al. 1980) and must be used if growth models are to be capable of prediction under elevated CO_2 (Reynolds and Acock 1985). Additionally, tissue respiration has been modeled (Ryan 1990; Kinerson 1975). However, the mechanisms determining carbon partitioning are still poorly understood (Cregg et al. 1993, Sprugel et al. 1991), and the influence of resource availability on carbon assimilation and allocation has not been congruently elucidated (Nadelhoffer et al. 1985, Keyes and Grier 1981).

Quantifying carbon partitioning among tissue components remains a major impediment to modelling forest productivity. The formidable task of assessing and incorporating the role of sink strength on carbon partitioning at the biochemical level makes empirical surrogates to these biochemical processes

[1] *David Arthur Sampson is a research associate in forest ecophysiology in the Department of Forestry, North Carolina State University, Raleigh, N.C.*

practical. The role of resource availability on carbon assimilation and partitioning can be examined in empirical investigations. Data are available on the growth response of young and mid-rotation loblolly pine (*Pinus taeda*) stands to nutrient amendments (NCSFNC 1993, NCSFNC 1991). The effects of soil water availability on loblolly pine growth and phenology remain unknown, however studies are underway to address these uncertainties.

Nutrient availability and soil water deficit are the primary resource-limiting factors influencing loblolly pine production in the Southeast (NCSFNC 1993, Teskey et al. 1987). Low nutrient availability and soil water stress are key factors causing suboptimal levels of leaf area index (LAI) (Colbert et al. 1990; Gholz 1986; Vose and Allen .1988). Nutrient amendments increase LAI and canopy N content (Vose and Allen 1988). Higher leaf area increases the interception of photosynthetically active radiation (PAR) and, therefore, the amount of carbon fixed (Cannell 1989), reflected in increased stemwood growth increment (Vose and Allen 1988). Elevated canopy N content would be expected to increase photosynthesis (Zhang and Allen, in review) and, therefore, production per unit LAI. In addition to limiting LAI, water stress may reduce loblolly pine production by promoting early stomatal closure (Teskey et al. 1987; Bongarten and Teskey 1986). The role of resource availability on carbon allocation to branches and roots at the stand level remains unknown.

Carbon allocation to foliage, stems, branches, and roots will determine the relative contribution of these components to total stand autotrophic maintenance respiration (Rm). Rm may account for almost 60% of gross carbon fixed in loblolly pine forests (Kinerson 1975). For loblolly pine the order of contribution to total Rm has been estimated as; branches > foliage > stems > roots (Kinerson 1975).

At present no data are available on the effect of chronic, elevated CO_2 and elevated temperatures on the growth and phenology of mature trees. Short-term exposure to a doubling of CO_2 may decrease stomatal conductance by 40% (Morison 1985). Increased ambient CO_2 does significantly increase photosynthesis in loblolly pine branch chamber experiments when water is not limiting (Teskey, personal communication). Increased temperatures will increase dark respiration. Unfortunately, the complexity of these interactions cannot be easily resolved in standard factorial experiments which makes simulation modelling necessary.

The objective of this paper is to examine the potential effect of increased ambient CO_2 and increased temperatures on loblolly pine productivity in a high and low site under two treatments using computer simulations. Questions addressed include: 1) can we expect increased net C assimilation if a 4^{o} C increase in mean annual temperature and a doubling of ambient CO_2 occurs?, 2) to what extent will maintenance respiration offset any expected gain due to elevated CO_2?, and 3) will fertilization decrease, maintain, or increase forest production over control sites in a hotter, higher CO_2 environment? We used the process model BIOMASS parameterized for loblolly pine to address these questions.

METHODS

We parameterized the process model BIOMASS version 12.0 for loblolly pine forests. A complete review of the model has been described elsewhere (McMurtrie et al. 1992). Model descriptions included in this paper represent source code changes made to BIOMASS version 12.0 during model parameterization, and specific model characterization to clarify process level interactions.

BIOMASS was written to explore the mechanistic factors influencing radiata pine (*P. radiata*) growth response to various water and fertilization treatments at a physiological process level (McMurtrie and Landsburg 1992). BIOMASS was developed using empirical data from the Biology of Forest Growth (BFG) experiment (see McMurtrie and Landsburg 1992; Benson et al. 1992; Linder et al. 1987).

Study Locations

Simulations used in this analyses were based on empirical data from two fertilization trials of mid-rotation loblolly pine plantations of the North Carolina State Forest Nutrition Cooperative (NCSFNC). The low site was established on the Piedmont of South Carolina on a Cecil soil series coinciding with a low site potential and the high site was established on the upper coastal plain of North Carolina on a Leaf soil series corresponding to a high productivity site (Table 1). Two treatments (control and fertilized) were replicated twice with fertilized plots receiving a one-time application of 200 Kg N ha^{-1} + 50 kg P ha^{-1} in 1987. Simulations presented are for the 1988 growth year (1 January through 31 December).

Table 1. — Initial stand characteristics for two mid-rotation loblolly pine plantations. Projected peak leaf area index data are for control plots of the growth year.

Site	Age (years)	Basal Area m^2 ha^{-1}	Stand Density (stems ha^{-1})	Site Index (m) Base age 50	Peak LAI (m^2 m^{-2})
Low	14	21	238	18	2.0
High	10	20	244	21	2.4

Model Parameterization

We parameterized BIOMASS using data from several sources. Input parameters were defined as model run-time conditions, initial stand characteristics and growth parameters that vary in time and space, and process parameters. Run-time conditions set run-constant model parameters. Initial stand structure parameters were derived from stand inventory data. Published equations were used to derive estimates of, for example, initial standing branch and bole biomass and soil water content. We obtained process parameters (eg. tissue respiration rates, maximum photosynthesis, and optimum temperatures for photosynthesis) and growth parameters from unpublished data, from the literature, or from personal communication.

Each simulation was run on a daily time step. Daily mean soil and air temperatures were estimated from daily minimum and maximum air temperatures. Leaf area index (LAI) for each plot was estimated from litter-trap data (Vose and Allen 1988). The maximum minus the minimum LAI, converted to mass units, provided an estimate of the yearly foliage production. The empirical estimate of foliage production was used to partition simulated net carbon assimilated more precisely.

Relative growth rate, and carbon partitioning and storage modules were written to model the seasonal patterns in loblolly pine growth phenology. Empirical data from the Southeast Tree Research Education Site (SETRES) were used to develop the phenology routines (SETRES 1993).

Phenological Rates

The closed form logistic equation was fit to the growth data and scaled to sum to one for initiation and cessation of stem and branch diameter growth, and leaf area development. The form of the equation is:

$$RGR = ((e^{(B0+B1*T)})/(1+e^{(B0+B1*T)})) \quad (1)$$

Where:

e = 2.71828,
RGR = Relative growth rate,
B0 = Scaling parameter,
B1 = Inflection parameter, and
T = Year day (1 to 366).

The B0 and B1 model parameters are estimated from foliar nutrient concentration at the beginning of the growth year. A hypothetical model of the same form was used to simulate root activity. The timing of root growth initiation and cessation was approximated using data from Harris et al. (1977).

The first derivative of equation 1 provided daily growth rate functions for the foliage, stem, branch, and root phenologies. Day length determines the initiation of foliage development. A threshold sum of consecutive daily mean air temperatures beginning with the first day of the growth year, along with day length, determines the commencement of stem and branch rate functions. In a similar fashion a threshold sum of mean soil temperatures determines the initiation of the first flush of root development. A maximum threshold for stemwood growth rate determines initiation of the second flush for roots. A minimum threshold in the rate change from time t, to t + 1 controls the termination of each rate function.

Daily relative growth rates for foliage, stems, branches, and roots are termed component tissue activity levels (surrogate for sink strength). All component tissue activity levels are zero during the dormant season. During a growth period one or more of the activity levels will be greater than zero.

Assimilation

BIOMASS can use either an empirical model of assimilation based on light absorption, or a biochemical model based on enzyme kinetics. For these simulations we used the biochemical model (Farquhar et al. 1980). This model interprets C_3 photosynthesis from the kinetics of Rubisco. The rate of carboxylation obeys Michaelis-Menten kinetics, and depends on the partial pressures of the competing gaseous substrates, CO_2 and O_2, and on the ratio of ribulose-1,5-bisphosphate (RuBP) concentration to enzyme active sites. This structure makes the model sensitive to changes in CO_2 concentrations. Net carbon assimilation is predicted from gross photosynthesis minus construction and maintenance respiration.

Carbon Pools

Daily net carbon assimilation enters either active or passive labile carbon pools. The component activity level and net carbon assimilated determines movement of carbon into or out of these pools. For example, when the activity level is zero and net assimilation is greater than zero, carbon enters the passive pool to be stored in component tissue. Carbon storage begins with foliage. When the foliage storage reaches maximum capacity, carbon is stored in roots. This process continues with the remaining tissue components and the hierarchy of storage is: foliage > roots > branches > stems. If daily net carbon is negative, an equal amount of carbon is removed from storage beginning with foliage. The carbon removal hierarchy follows the carbon storage ranking.

During an active growth period net carbon assimilated enters the active pool. If net carbon assimilated is negative during an active growth period, carbon is removed from storage. Additional carbon proportional to the maximum activity level must be removed from storage to meet the growth demand. Available carbon is partitioned among the tissue components during positive net carbon availability. During an active growth period with positive net carbon availability, carbon is removed from storage at a rate proportional to the sum of the tissue component rates. The foliar nitrogen concentration modifies this carbon flux.

Carbon Partitioning

The relative component tissue activity levels, when expressed as a fraction of one, determine the partitioning of net carbon to foliage, stems, branches, and roots. Daily partitioning rates must therefore sum to zero or one. Carbon flux to foliage must be met first before carbon can be made available to other tissue components. If the demand for foliage production is less than daily net assimilated, carbon is removed from storage in an amount equal to the deficit. Carbon storage occurs when daily production become less than net carbon assimilated.

Model Assumptions

The following assumptions pertain to source code changes made during model parameterization. General model assumptions can be found elsewhere (McMurtrie et al. 1992).

- Daily root production cannot exceed one-half of current standing root carbon (see Gholz et al. 1986).
- Maximum storage of carbon for stems and branches is 4% of current standing carbon.
- Maximum carbon storage in foliage is 14.5 % of current standing carbon (Birk and Matson 1986).
- Maximum carbon storage in roots is 14.0 % of current standing carbon (Adams et al. 1986).
- Initial root biomass is equivalent to initial foliage biomass (see Gholz et al. 1986).
- Root biomass and production refers to fine roots (1 mm).
- Root sloughing is proportional to needle litter-fall.
- No internal acclimation to elevated CO_2.
- A four degree increase in minimum and maximum daily temperatures approximates a four degree increase in mean annual temperatures.

RESULTS AND DISCUSSION

Simulated net carbon production was comparable to the literature for southern pine species (Table 2). Additionally, net carbon allocated to stemwood growth was similar to the empirical estimates (Figure 1). Carbon budgets presented here are likely feasible given the parameterization procedure.

Simulation results indicated a net increase in stand productivity under a doubling of ambient CO_2 (700 ppm) and a four degree Celsius increase in daily temperatures (Figure 2). Low productivity sites increased from 3.5 to 5.7 Mg C ha^{-1} $year^{-1}$ while high productivity sites increased from 7.7 to 11.6 Mg C ha^{-1} $year^{-1}$ in control plots. This represented a 63% and 51% increase in net carbon flux for low and high sites, respectively (Figure 2).

Table 2. — Comparison of net carbon production for temperate coniferous forests from this study with published literature[1].

Source	Net Carbon production (Mg C ha^{-1} $year^{-1}$)
Simulated carbon from this study	3.5 and 11.6
Seven-year-old *P. elliottii* stand from Florida	2.4
Twenty seven-year-old *P. elliottii* stand from Florida	8.2
Sixteen-year-old *P. taeda* stand from North Carolina	9.8

[1] *Vogt, K. 1991.*

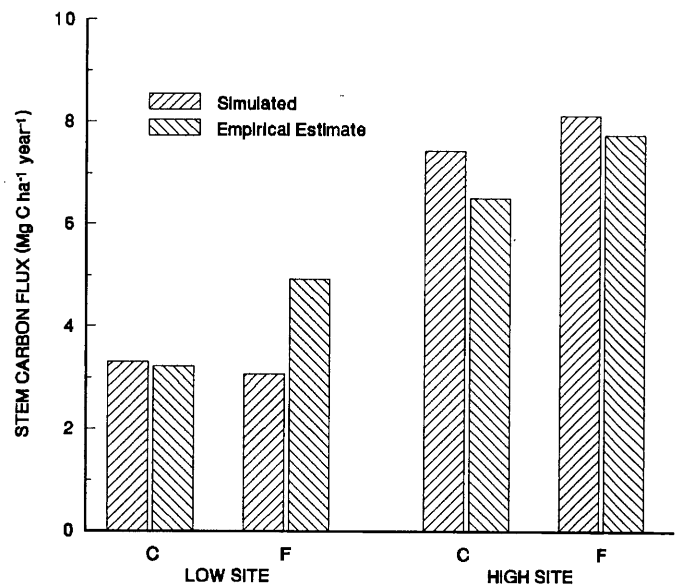

Figure 1. — Comparison of simulation output and empirical estimates for annual stem carbon production (Mg C ha^{-1} $year^{-1}$) for two mid-rotation loblolly pine stands of the southeastern United States. C and F designate control and fertilizer (one-time application of 200 kg N ha^{-1} and 50 kg P ha^{-1}) plots for low and high productivity sites.

A doubling of CO_2 under ambient temperatures in control plots increased net carbon gain by 93% and 52% for low and high sites, respectively (Figure 2). The differences in net carbon gain in the 2x CO_2 simulations are not maintained when increased temperatures are imposed. Maintenance respiration (Rm) accounted for a 26% loss in net carbon available for growth for low sites. High sites under elevated CO_2 did not change appreciably in net carbon gain with increased temperatures.

An increase in gross carbon fixed accounted for the negligible effect of increased temperatures on net carbon gain for high sites in control plots. Gross carbon fixed increased by 8% in these plots. The 8% increase off-set an almost identical increase in Rm between the 2x CO_2 and the 2x CO_2 with the imposed four

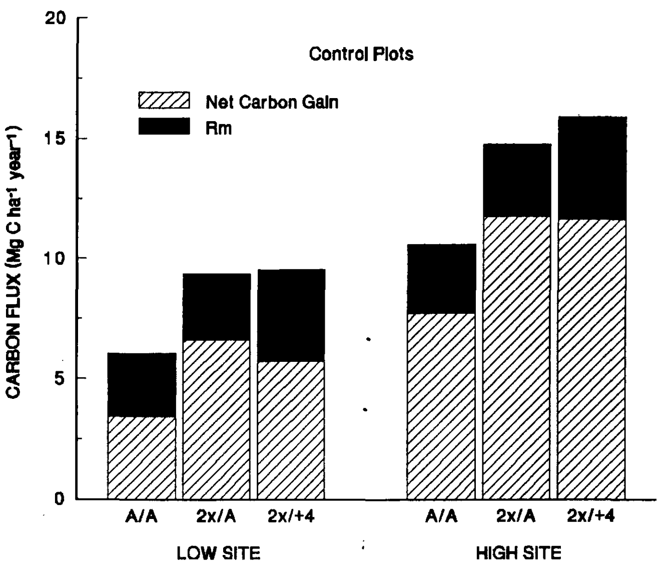

Figure 2. — Simulation output of annual carbon flux (Mg C ha^{-1} year^{-1}) for two mid-rotation loblolly pine stands of the southeastern United States. A/A represents ambient CO_2 concentrations (350 ppm) and ambient temperatures; 2x/A represents twice ambient CO_2 with ambient temperatures; and 2x/+4 designates twice ambient CO_2 and a plus 4 degree Celsius increase in mean annual temperatures for low and high productivity sites.

degree Celsius increase in daily temperatures (Figure 2). Gross carbon fixed remained unchanged for the 2x CO_2 and 2x CO_2 plus increased temperature scenario for low sites.

The effect of fertilization on total carbon flux and carbon partitioning varied by site. Control and fertilized plots for low sites did not differ in either gross carbon fixed or net carbon gain (Figure 3a). Carbon partitioning among foliage, branches, stems, and roots remained unchanged with treatment for these sites. Conversely, gross carbon fixed for high sites increased by approximately 18% in fertilized plots resulting in a 14% increase in net carbon gain. The net result of increased carbon availability was increased foliage, stem, branch, and root production when compared to control plots in high productivity sites. Foliage production did not increase in fertilized plots for low sites which can explain the lack of growth response to treatment in these sites (Figure 3a).

On a mass basis, foliage, stems, branches, and roots contribute disproportionately to Rm. For instance, foliage may represent 4 to 6% of the standing biomass yet may contribute > 34% of Rm (Kinerson 1975). Stem mass may exceed 65% of standing biomass, and, if we assume a live cell volume of 8 to 10% for bole wood (Ryan 1990), live stem tissue may represent 5.2 to 6.5% of standing biomass yet may contribute only 13% to Rm (Kinerson 1975). Increased foliage and root production rather than increased stem and branch production explained the roughly 4% increase in total Rm in fertilizer plots for high sites (Figure 3b). Data suggest that for these stands the order of contribution to total Rm for loblolly pine systems would be; roots > foliage > branches = stems.

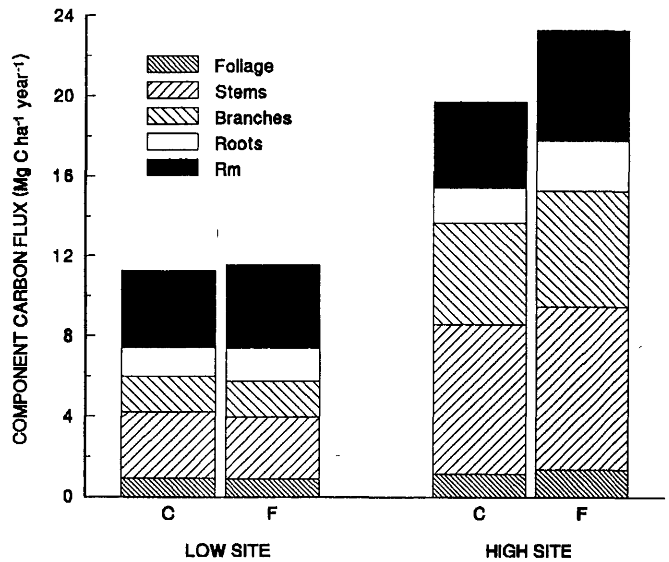

Figure 3a. — Simulation output of annual component carbon flux (Mg C ha^{-1} year^{-1}) for two mid-rotation loblolly pine stands of the southeastern United States. C and F designate control and fertilizer (one-time application of 200 kg N ha^{-1} and 50 kg P ha^{-1}) plots for low and high productivity sites.

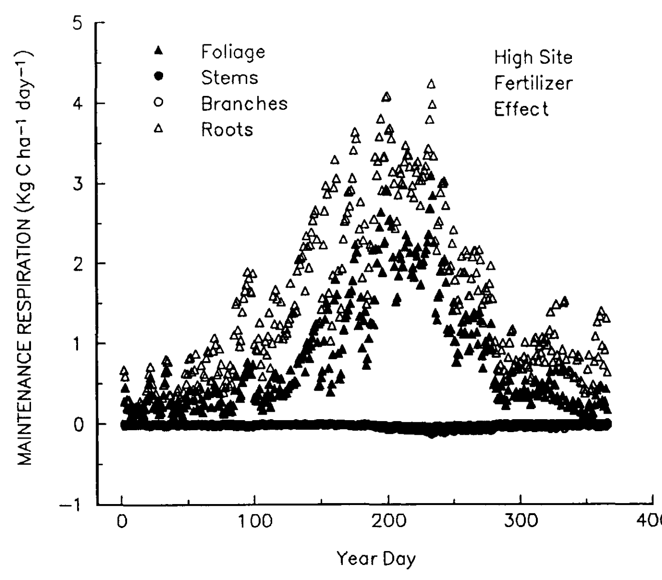

Figure 3b. — Simulation output of the component maintenance respiration (kg C ha^{-1} day^{-1}) difference between control and fertilized plots at 700 ppm CO_2 concentrations and a plus four degrees Celsius increase in temperature for two mid-rotation loblolly pine stands of the southeastern United States. Simulation data are for the fertilizer (one-time application of 200 kg N ha^{-1} and 50 kg P ha^{-1}) plots of the high productivity site.

CONCLUSIONS

Simulation output suggested a net increase in stand productivity under a doubling of ambient CO_2 (700 ppm) and a four degree Celsius increase in daily temperatures for loblolly pine stands of the southeastern United States. Site potential will likely effect the response of trees to these perturbations, with low sites responding greater than high sites. If fertilizer treatments are used, the response of trees to treatment under elevated CO_2 and temperatures may also depend on site potential.

Maintenance respiration comprises a large portion of the carbon budget for loblolly pine systems. Because foliage, stems, branches, and roots due not contribute proportionately to total Rm, estimates of component biomass production will strongly influence simulated net carbon assimilated in these modelling exercises.

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Impact of Large Ungulates in Restoration of Aspen Communities in a Southwestern Ponderosa Pine Ecosystem

Wayne D. Shepperd and M.L. Fairweather[1]

Abstract — Experience has shown that in some areas of the Southwest, regenerating aspen suckers require fencing to protect them from browsing elk. In October, 1991 we removed the fence surrounding a 6.5 ha aspen sucker stand north of Flagstaff, Arizona to test whether the trees were large enough to be out of reach of the animals. The site had been fenced for five years following clearfelling of several clones that comprised the original stand. The regenerated stand averaged 50,000 stems/ha with dominant stems over 3 m in height. By October, 1992, most stems in one clone had been severely damaged by elk. Animals broke many stems to reach the terminal foliage, often infecting the residual stem with Cytospora canker. Monitoring will continue to determine if the remaining clones will be browsed in future years. It appears that fencing must remain in place indefinitely in this ecosystem, given the demand for browse associated with current high animal populations.

INTRODUCTION

Aspen is currently a minor component of most southwestern landscapes. Large, stable aspen communities similar to those on the plateaus of Colorado and Utah (Shepperd 1990) do not exist in the Southwest. Instead, most aspen stands are in advanced stages of succession to conifers. This is especially true in the ponderosa pine type on the Coconino and South Kaibab National Forests, where the only remaining aspen clones consist of a few small scattered groups of declining and damaged stems in a sea of pine. Aspen is approaching "threatened and endangered" status in these situations, since the few remaining aspen genotypes will be lost if the surviving stems die without re-sprouting. The long term survival of aspen in some southwestern landscapes may be in doubt.

Although an earlier study of techniques to regenerate aspen in this area reported success (Larson 1959), attempts to regenerate small clones in recent years have invariably failed, not from lack of initial suckering, but from subsequent sucker mortality. Two factors have contributed to this situation: the absence of frequent fire in southwestern ecosystems since European settlement (Covington and Moore 1991, Dieterich 1980) and extreme browsing pressure from large ungulates.

The impacts of long-term fire control measures upon vegetation succession in Southwestern ponderosa pine ecosystems has been well documented (Cooper 1960,1961). Conifer understories have established under many aspen clones. Young conifers have grown into aspen canopies in many cases and are gradually crowding the aspen out.

Clearfelling isolated mixed conifer/aspen stands to re-establish aspen has not been successful in isolated clones. Fencing trials on both the Apache-Sitgreaves and Coconino NF's have verified that elk browse aspen suckers and have confirmed the need for elk-proof fencing to allow regenerating aspen suckers to establish and grow. Although there is no doubt that clearfelling and subsequent fencing will result in abundant stands of healthy aspen suckers (Schier et al. 1985, Shepperd and Engelby 1983), the question of when fencing can safely be removed remains. This study was designed to test if aspen regeneration can be certified as established if fencing is removed after five years. We had hoped that five-year-old aspen stems would be dense enough and tall enough to withstand animal browsing after fences were removed and not succumb to subsequent diseases introduced from animal wounds.

[1] *Research Forester, Rocky Mountain Forest and Range Experiment Station, USDA Forest Service, Fort Collins, CO; and Plant Pathologist, Southwestern Region, USDA Forest Service, Flagstaff AZ.*

METHODS

The study site was located on the Peaks RD of the Coconino NF in the NE 1/4 of Sec 17, T24N, R6E in an area known as the Hochderffer Aspen Regeneration Project. The 6.5-ha study area was one of several aspen stands that were clearfelled and fenced with 2 m hog-wire fencing in 1986. In 1990, a sample of ten 2.322-m^2 plots indicated that the area contained about 51,000 stems/ha with dominant stems averaging 2.5 m in height. No animal-related damage was evident inside the fence and no suckers survived outside the fence, although there was evidence that suckering had occurred.

In the fall of 1991 Forest Service crews removed fencing from the unit except for an exclosure at one end. Sixty 4.05-m^2 circular plots were then established in a uniformly stocked area in one genotype within the cut unit. Thirty of the plots were located in the area where fencing had been removed and 30 were inside the remaining fenced area. All plots were measured one year later, in September, 1992. Live aspen stems in each plot were examined for damage and tallied as undamaged, or assigned a damage code (based on the damage most likely to affect stem vigor). Dead stems were tallied by size class only.

RESULTS

In September, 1992, the fenced exclosure contained an average stocking of 50,000 live stems/ha while the adjoining area, where fencing had been removed one year earlier, contained only 30,000 live stems/ha (fig. 1). The greatest differences in live stem numbers occurred in size classes from 0.45 m in height to 2.5 cm diameter breast height (dbh) (fig. 2). Although this was a significant difference ($p = 0.05$) in overall live stem density (fig. 1), the data belies the serious browsing damage suffered by many surviving stems outside the fence. Nearly all stems less than 0.45 m in height were browsed, as were nearly half of the surviving mid-size classes (fig. 3).

Surprisingly, about 60% of large dominant stems (> 2.5 cm dbh) were also browsed. Many larger stems were stripped of lower branches, or were broken completely off by the elk (fig. 4). In addition, most of the severely wounded stems were infected with Cytospora canker [*Cytospora chrysosperma* (Pers.)] (fig. 5). Elk browsing was also manifested by a significant ($p = 0.05$) reduction in the average height of dominant stems in the unfenced, heavily browsed area. Dominant stem heights averaged 3.47 m inside the fence in 1992, but only 2.67 m outside the fence (fig. 6).

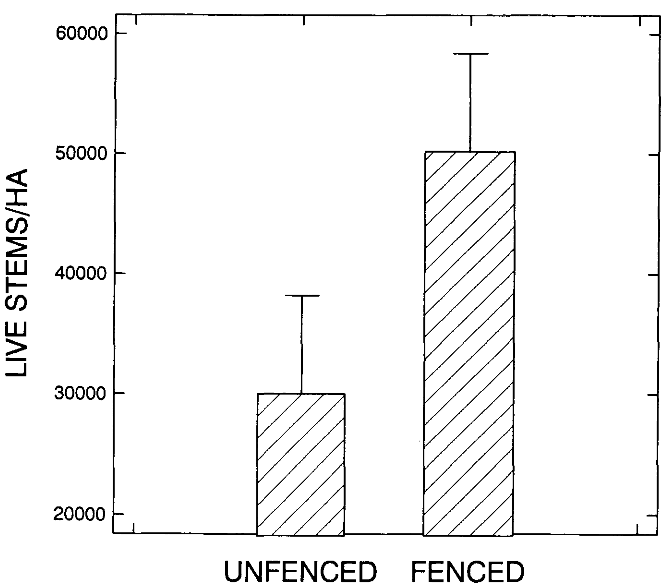

Figure 1. — Analysis of Variance of live aspen stem densities for six-year-old aspen sampled from 60 4.05 m^2 plots. Bars are 95% Tukey HSD intervals for stem density in unfenced and fenced treatments a year after fence removal.

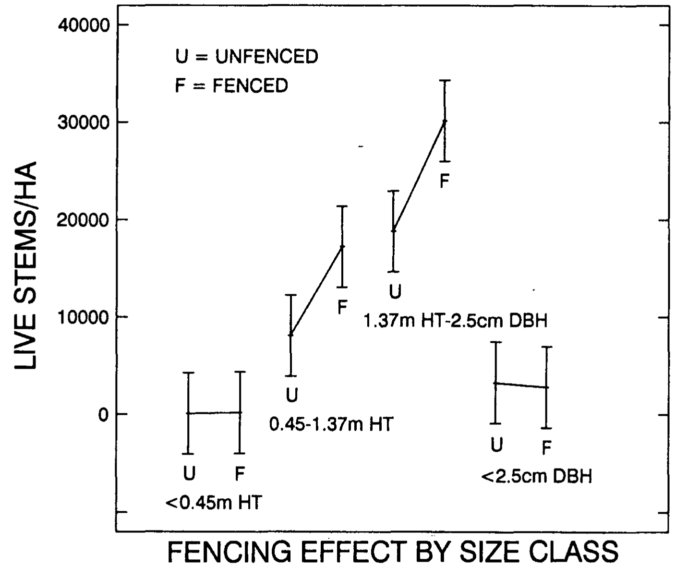

Figure 2. — Ninety-five percent Tukey HSD intervals for a Two-way Analysis of Variance of live six-year-old aspen stem densities grouped by stem size and fencing treatment.

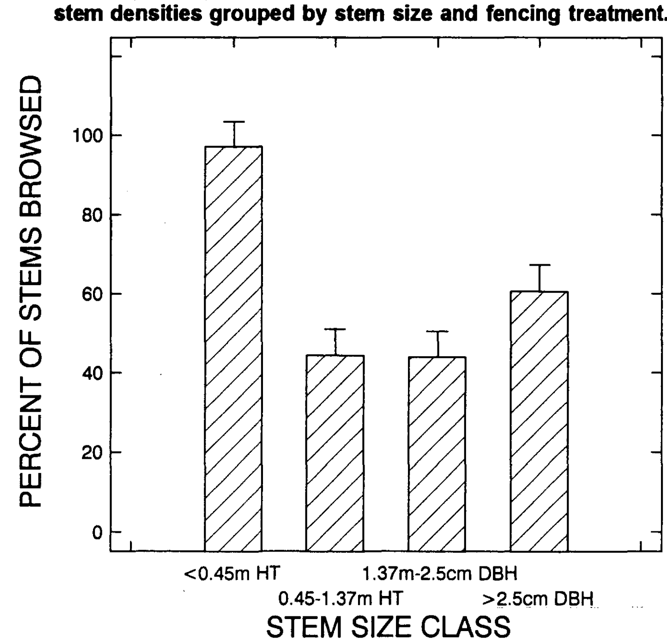

Figure 3. — Ninety-five percent Tukey HSD intervals comparing the percent of browsed aspen stems by size class.

Figure 4. — Effect of elk browsing in a six-year-old aspen population, one year after fence removal. Note the distinctive browse line in the background and the stems broken by elk to access upper foliage.

Figure 5. — Fruiting bodies of *Cytospora chrysosperma* canker. Many broken stems were infected with this disease.

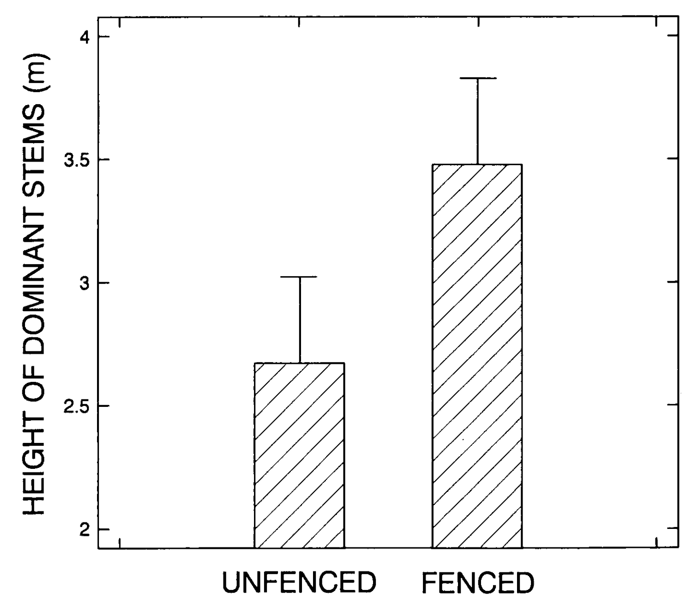

Figure 6. — Ninety-five percent Tukey HSD intervals comparing the height of dominant stems in unfenced and fenced treatments a year after fence removal.

DISCUSSION

It is clear from the striking change in appearance of the previously fenced area a year after fence removal that elk severely impacted the health and vigor of this five-year-old aspen sucker population. A sea of dense, vigorous aspen suckers was reduced to a scattering of severely damaged stems in the most severely affected area (fig. 4). The high percentage of large dominant stems that were browsed by elk (fig. 3) is disturbing. Most of these stems must survive intact if a new generation of aspen is to survive and prosper. Heavy browsing, destruction of terminal leaders, and canker infections in the largest stems only forecast regeneration failure. As long as elk go to these extremes to reach live leaves, aspen stems will have to be much larger to resist breakage and foliage browsing. Ten, or perhaps 15 years of continued fencing protection may be necessary.

One encouraging factor is that not all genotypes in the study area were heavily browsed the first year after fence removal. Ironically, the clone where the plots were located received most of the damage. This suggests that elk have exhibited a genotypic taste preference for the foliage of one aspen clone over others in the study area. If so, there may be compounds present in the foliage of the other clones in the area that discourage herbivory. Remeasurement in future years should determine if this effect is permanent, or if the elk will browse less tasty genotypes once the preferred clone is depleted.

The existence of a genotype over a hectare in size that is highly preferred by elk indicates that browsing intensities observed here were not likely to have occurred in the past. This suggests that there is either a lot less aspen, or a lot more elk in the landscape today than in the past. Both cases are likely to be true.

Fire suppression has resulted in a reduction of openings within southwestern ponderosa pine forests (Cooper 1960,1961; Covington and Moore 1991). Accompanying the increased density of ponderosa pine has been a reduction in the light and fire-disturbance regimes favorable to the regeneration of aspen (Schier et al. 1985; Shepperd and Engelby 1983). This trend has also reduced other forage available to browsing animals and further exacerbated the impacts of elk upon regenerating aspen in this area.

This study has clearly demonstrated that fences will have to remain around aspen regeneration in this southwestern ponderosa pine ecosystem longer than five years to guarantee survival of genotypes that are preferred by elk. The severity and pattern of elk browsing observed here are indicate a larger underlying ecosystem problem. Restoring aspen may ultimately require reduced elk populations as well as altering forest conditions within the landscape to support remaining animals and favor aspen establishment. Otherwise, some aspen genotypes in this area may have to exist like other endangered species — behind zoo-like fences.

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Assessing the Impacts of Timber Harvest on a Northern Arizona Rare Plant, *Clematis hirsutissima* var. *arizonica*, Through Canopy Manipulation and Matrix Demographic Analysis

Edward Bennett Smith[1]

Abstract — In a study of the demographics of a rare plant, the Arizona Leather Flower (*Clematis hirsutissima* var. *arizonica*), we assessed the effects of canopy cover on Clematis survival and reproduction. Varying amounts of ponderosa pine canopy cover or shading were removed (branches, poles, and saplings) from some plots, while artificial shading was added to other plots in the Coconino National Forest. Results indicate that experimental removal of canopy had detrimental effects on seed production, while shade addition in previously low-shaded areas had a positive effect on seed production. These changes may be stochastic, but matrix analysis shows that all plots had eigenvalues below 1.0 (0.129-0.931), indicating that they are not growing, and are in long-term decline. These findings are important, because there are only 1500 extant plants of this southernmost subspecies, and more than half the population exists in areas of current or planned timber sales. Factors besides tree canopy that may be affecting Clematis' survival include climate, fire, insects, forest floor depth, introduced ungulates, and other ecosystem variables and their interactions. The U.S.D.A. Forest Service should continue to protect this plant's habitat and monitor its demography, and if possible, expand studies into other landscape processes that may have affected the reproductive biology of this plant.

INTRODUCTION

The process of listing plants as threatened or endangered takes up to ten years, compared to animals, which take 2-5 years (Phillips, pers. comm.). The scientific and political processing necessary for listing therefore may take too long to protect a plant before numbers of individuals drop below minimum viable populations. The Arizona leatherflower (*Clematis hirsutissima* var. *arizonica*, fig. 1) is a U.S. Fish & Wildlife Service category two (C2) plant, for which there is insufficient data to decide between listing or not listing. In the interim, the U.S. Forest Service manages this plant as 'sensitive' so as not to jeopardize its existence on lands under its control. The author and Dr. Joyce Maschinski continued a long-term demography study on plots of Arizona leatherflower within the Lake Mary timber sale area.

Demographic studies quantify the change of individually mapped plants from state to fate, so that comparisons can be made of the eigenvalues, fitnesses or growth rates of different samples over time intervals. Demographic analysis using matrices is very useful because the source of a plant's decline can be pinpointed very accurately. Plant populations are divided by age-, size- or stage class, and the relative contributions from these classes, or elasticity analysis, can be quantified, and the class or classes that are diminishing or not contributing to the plants overall fitness can be identified. If the particular class does not contribute to the overall fitness of the plant, for instance the seedling stage, then experiments can be designed to increase the survival of plants in this stage. Experimental results from

[1] *Master's degree candidate at Northern Arizona University, Flagstaff, Arizona, USA.*

Figure 1. — Clematis hirsutissima var. arizonica.

these manipulations could be quantified with the matrix model, and management decisions could be based on the analysis of these data. Examples of this type of model analysis have been done for several species, including the effects of prescribed fire on a tropical savanna grass, testing spatial and temporal variation of a perennial bunchgrass, predicting growth of temperate deciduous forest stands, and for conserving an endangered animal species.

MATERIALS AND METHODS

Forty 2.3 m2 (square meter) plots were established in the summer of 1991, encompassing 401 of the approximately 1500 extant plants in Arizona. Plants were identified on an x-y axis grid within each plot, and measurements taken on each plant for the number of stems, number of flowers, number of flowers eaten, and number of flowers that set seed. Plants were identified as seedling (1), juvenile (2), or reproductive development stage, based upon leaf morphology and presence of flowers. Seedlings and young ramets have characteristically wide, flattened !cotyledons, although it is very difficult to differentiate between them. Reproductive plants have flowers or seedheads, and juveniles have neither wide leaves nor reproductive structures. Canopy cover for the plots was determined with a spherical densitometer and a photometer, and plots were divided into subgroups of 'high', 'medium' or 'low' canopy, with the 'high' canopy intercepting the greatest amount of solar radiation. For each subgroup, half the plots were randomly designated as controls, and half as experimental, in which the canopy or degree of shading was manipulated. In early spring 1992, the plots with 'high' and 'medium' canopy levels were altered by removing branches or small trees surrounding the plots (reduced 5-25%), and during the summer of 1992, plots of 'low' canopy cover were altered by the addition of 85-95% shade structures (fig. 2 and fig. 3). Control plots provide baseline data and comparisons for the experimental plots.

Matrices

The transition matrices were constructed from the raw data collected over the two field seasons. The columns represent the 'state' or condition of the plants in 1991, while the rows represent the 'fate' or condition of the plants in 1992. The entries

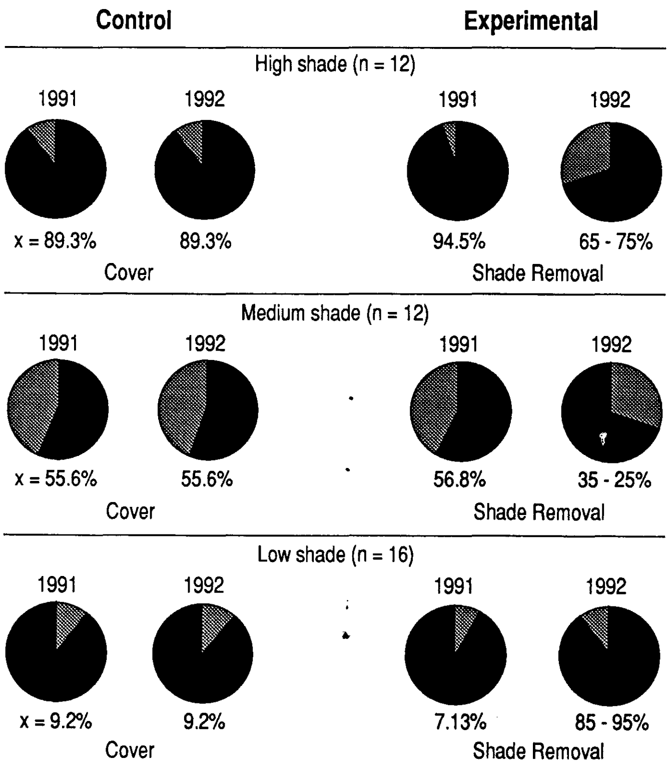

Figure 2. — Experimental design for canopy manipulation (Maschinski, 1990).

within the matrix are proportions of plants that went from one state to the corresponding fate. The stages for the plants were delineated as follows:

Stage 1	**Seedlings or new ramets**
Stage 2	**Juveniles, nonreproductive**
Stage 3	**Reproductive, <9 stems**
Stage 4	**Reproductive, 9-25 stems**
Stage 5	**Reproductive, >25 stems**

The transition matrices were analyzed using Eigenfinder in the MacMath™ package. The eigenvalues (λ) give the growth rates for the group of plots as indicated. This is a complex distillation of how each stage class contributes to the overall fitness of the group of plants (Caswell,1989). Unfortunately, it is not possible to compare the eigenvalues from only one transition matrix to another, rather two or more years are necessary for comparisons of spatial, temporal or experimental variability (Moloney, 1988). Sensitivity analysis is used to identify the most important stage(s) changes that are contributing to a plant's growth or decline (λ).

Figure 3. — Shade addition was accomplished with portable structures which were covered with pine boughs or wood lath.

RESULTS

There were 401 plants sampled from the 40 plots, with 283 adults (70.57%), 84 juveniles (20.95%), and 34 seedlings (8.47%). As is true for many rare plants, this demographic distribution indicates very low rates of regeneration. Maschinski found that the number of flowers that set seed in low canopy cover experimental plots increased from 162 in 1991 to 385 in 1992 (q'= 7.48, <p0.05), indicating that the addition of shade significantly improved the potential for sexual reproduction. Also, there was a high canopy removal treatment effect on the number of flowers that set seed (fig. 4). Plots that had high canopy cover reduced through thinning had significantly lower numbers of flowers that set seed (100 in 1991 decreased to 44 in 1992), compared to controls (37 in 1991 increased to 79 in 1992; F=2.76. p<0.1).

The eigenvalues presented in figures 5 and 6 indicate that all sampled plots are in general decline ($\lambda < 1.0$), which is further evidenced by the lack of plants in the earlier stages, and general lack of larger, reproductive plants. An eigenvalue of 1.0 indicates a group of plants that is neither declining nor increasing, whereas an eigenvalue greater than 1.0 would indicate growth.

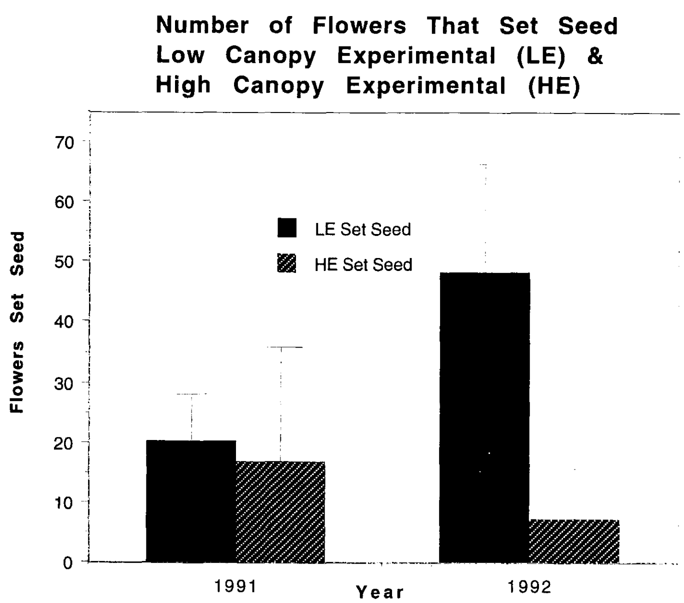

Figure 4. — Plots that had high canopy cover reduced through thinning had significantly lower numbers of flowers that set seed compared to controls, and the addition of shade significantly improved seed set in low canopy cover plots.

Low Control	λ = 0.931	93.1%	
State	1	2	3
Fate 1			
2			
3			0.93103

Low Experimental	λ = 0.646	93.8%	
State	1	2	3
Fate 1	0.04167		
2	0.08333	0.16667	
3			0.65483

Medium (	λ = 0.711	84.4%	
State		2	3
Fate 1			
2		0.04444	
3		0.04444	0.71111

Medium Experimental	λ = 0.539	88.0%	
State	1	2	3
Fate 1			
2	0.06667	0.09333	0.01333
3		0.17333	0.53333

High Control	λ = 0.662	96.4%	
State	1	2	3
Fate 1	0.03667		
2	0.09091	0.01818	0.03636
3		0.12713	0.65454

High Experimental	λ = 0.632	88.4%	
State	1	2	3
Fate 1			
2	0.02326	0.11628	0.02326
3		0.09302	0.62791

Figure 5. — These transition matrices were constructed from data for the 1991 (state) and 1992 (fate) data. The numbers within each cell represent the proportion o fplants from each specified canopy class that went from one state to another. Each matrix represents plants from 6 - 8 plots, and does not include any recreitment, either sexual or aseuxal. In these 3 x 3 matrices, stages 1 and 2 represent the seedling and juvenile stages, respectively, as in the 5 x 5 matrices, but stages 3, 4, and 5 have been lumped into one reproductie stage, 3 (as explained in text). the λ values are the dominant eigenvalues, or overall growth rates for the group of plots. The overall percentage of living plants remaining in 1992 is given in the upper right had corner. For ease of comparison, and to keep as many cells as possible filled with values, these 3 x 3 matrices are preferable to the 5 x 5 matrices.

Low Control				λ = 0.624	93.1%
State	1	2	3	4	5
Fate 1					
2					
3			0.62069	0.03448	
4			0.03448	0.20689	
5					0.03448

Low Experimental				λ = 0.563	93.8%
State	1	2	3	4	5
Fate 1	0.04167				
2	0.08333	0.16667			
3			0.5625	0.02083	
4			0.02083	0.04167	
5					

Medium Control				λ = 0.578	84.4%
State	1	2	3	4	5
Fate 1					
2	0.04444	0.04444			
3		0.04444	0.57778	0.08889	
4			0.02222		0.02222
5					

Medium Experimental				λ = 0.499	88.0%
State	1	2	3	4	5
Fate 1					
2	0.06667	0.09333	0.01333		
3		0.17333	0.49333	0.01333	
4				0.02667	
5					

High Control				λ = 0.563	87.3%
State	1	2	3	4	5
Fate 1	0.03637				
2		0.01818	0.03637		
3		0.10909	0.56364	0.03637	
4		0.01818		0.05455	
5					

High Experimental				λ = 0.494	83.7%
State	1	2	3	4	5
Fate 1					
2	0.02326	0.11628	0.02326		
3		0.09302	0.48837	0.02326	
4			0.02326	0.02326	
5					0.02326

Figure 6. — Transition matrices for the data presented in a 5 stage format, which is how the data were collected. The λ values are the dominant eigenvalues, or overall growth rates for the group of plots. The overall percentage of living plants remaining in 1992 is given in the upper right had corner. Note that many cells are empty below the diagonal, indicating a severe lack of plants in these transition stages.

DISCUSSION

The low eigenvalues, and paucity of plants in the earlier stages of growth indicate that the plant is having trouble reproducing, sexually and asexually. It has been previously noted that presumably mammalian herbivory accounts for the loss of significant numbers of flowers and seeds within some plots. It has also been noted that many insects frequent these plants, and that up to 90% of once-viable seeds are parasitized by some unknown weevil or beetle (Maschinski, 1989).

The plant's decline may be in response to the periodic drought of 1988-89, but one would expect a quick rebound to such a transient change in precipitation. With the record-breaking rainfall we have had over the last two years, observations from another field season should test this hypothesis.

Fire

Clematis grows under a ponderosa pine canopy, which is fire-tolerant. Perhaps Clematis is fire-tolerant or even fire-dependent, relying on periodic removal of the chaff, composed of old stems, to allow a higher rate of photosynthesis, and to eliminate competition from grasses, forbs and pine trees. A similar demographic study on a tropical Andropogon grass in Venezuela showed dramatically different eigenvalues for unburned (0.2762) versus burned plots (1.2524). Most of this discrepancy was shown to be due to the growth, survival and reproduction in the two smallest size classes, which were shown by elasticity analysis to be the two most important classes to population growth (Silva et alia, 1991). Analysis of fire scars in northern Arizona has shown that some forests experienced burn frequencies of 2-15 years (Covington and Moore, 1992), indicating that fire may have played a critical role in maintaining this ecosystem. Fire played a role in the thinning of trees, so that there were fewer but larger trees per acre. This could have affected water relations in the soil, as fewer young, fast-growing trees may have reduced competition for available soil water. With more trees in the older age classes, the forest would have had interlocking canopies providing more shade with fewer trees. The character of ponderosa canopy changes with age, to a more patchy, heterogeneous spatial arrangement, allowing light to pass through while maintaining leaves at lower temperatures, thereby reducing evapotranspiration. There probably exists an ideal canopy closure level that maximizes plant fitness by maintaining high photosynthetic rates, while minimizing water loss through evapotranspiration and evaporation from the soil and litter. This idea of there existing an ideal overstory canopy composition that is neither too dense nor too sparse is supported by preliminary analysis of 1993 field data (Maschinski, pers. comm.).

Depth of Forest Floor

Logging and fire exclusion have drastically changed the character of the canopy of these forests. The litter layer has also presumably changed, the depth of which is a function of leaf deposition rates, fire frequency, and decomposition rates. Natural fire frequency disruption through suppression could have slowed the litter decomposition and nutrient mineralization processes, limiting the amount of nutrients available to growing and mature plants, and making it more difficult for seedling establishment. While a thick litter layer has the 'mulch effect' of moderating moisture loss and slowing soil temperature change, it also has a significant effect on the amount of precipitation available to plant roots by interception and subsequent evaporation.

Insects

Since there is an abundance of insects that utilizes these plants, and the number of seeds parasitized by insects may be significant, the suppression of fire may have a positive impact on any populations of insects that overwinter in the soil, allowing them to maintain imbalanced, epidemic numbers. Perhaps frequent fire in the past maintained insect populations at stable levels that allowed higher levels of Clematis reproduction than is observed today. Further study is warranted in this area.

Elk tracks were noticed around some of the mammalian-herbivore impacted plots. The management of this introduced species remains controversial, as some believe elk numbers are out of control. While there was a natural population of Merriam's elk, many believe its range was closer to the Mogollon Rim, and did not frequent the areas now inhabited by the larger herds of Rocky Mountain elk. This increased grazing pressure from large herds of elk (and cattle) may result in elk having to consume less desirable plants like Clematis, which is in the Ranunculaceae family. This family contains several poisonous genera (Cimcifuga or bugbane, Aconitum or monkshood, Delphinium, Anemone and Columbine), which are normally avoided by ungulates.

CONCLUSIONS

It is interesting to note that the effects of the experimental canopy manipulation were statistically significant within one season. The number of flowers that set seed increased in low canopy plots where shade was added, and the number of 'seedlings' decreased in high canopy plots that had canopy removed. Since this variation may be due to other factors, such as El Niño weather patterns, this demographic study should be continued. Because the eigenvalues are so low, and comparisons among treatments require more years' data, the intensity of this study should be maintained, and possibly expanded to include some of the Clematis populations in Walnut Canyon, Rio de Flag drainage, and Volunteer Canyon.

Logging activity has changed the Southwest landscape to such a degree that sensitive areas, such as Clematis habitat, should be protected until enough information has been gathered to delineate a clear picture of all the long term effects of management activities.

Priorities for Further Study

1. Maintain or expand present demography research.
2. Begin prescribed fire or simulated fire studies along with ungulate-exclusionary fencing.
3. Capture and identify insects that impact the plant. Emphasize those insects that forage on seeds - quantify samples (% seed herbivory) from different canopy, experimental, and fire regime plots.
4. Develop stem map from stumps and current stems. With tree coring, timber sale history and regression from current canopy and site index, develop fire and stand density history to determine forest structure in Clematis habitat for the last 100+ years.
5. Measure predawn and noon water potential values under different canopy levels and treatments to elucidate water stress relations.

ACKNOWLEDGEMENTS

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Preservation of a Rare Annual Grass, *Puccinellia parishii*, in Native American Agricultural Fields

M. Tremble and B. Hevron[1]

Abstract — Parish alkali grass, *Puccinellia parishii*, is a rare annual grass known from seven small populations. Two populations are known from southern California, six from northeastern Arizona, and one from southwestern New Mexico. All populations are known from saline, perennial springs or cienegas, a habitat type that is rapidly disappearing throughout the southwest. The largest population is found in Pasture Canyon on the Navajo Nation. The canyon has been farmed by native americans for over 400 years. *P. parishii* will soon be proposed as Endangered under the Endangered Species Act and Pasture Canyon may be designated as critical habitat. We have initiated a research project, of which one objective is to determine sustainable agricultural practices that will preserve this species. Fifteen Navajo and Hopi farmers have answered questionnaires regarding historical and current agricultural activities. This information includes methods of plowing, crops, water diversions, fertilizers, herbicides and burning. We will also collect data on population biology of this rare grass. Data collected will include seedbank density, seed dormancy, seedling emergence, and the environmental variables that regulate these factors. An ecological model and management plan will be developed in order to preserve the species and the cultural practices of the native american farmers.

Efforts to conserve biodiversity, particularly in North America have focused on "natural" rather than non-native or artificial ecosystems (Katz, 1991). At least 95% of the terrestrial environment is affected by human activities, including agriculture; and terrestrial habitats provide over 98% of human food (Paoletti et. al, 1992). The concept of "naturalness" may sometimes stand in the way of conserving biodiversity (Wedin, 1992). Anthropogenic landscapes may be managed for high biodiversity. For example, the ancient grassland communities of Europe have a high species richness that often reach 30-40 species per meter (Bakker, 1989).

In another study, the number of arthropod species in soil and litter in a forest and corn ecosystem were compared and found to be nearly equivalent (Paoletti et. al, 1988). There are farming systems which favor sustained biodiversity. These include minimum or no tillage, a mosaic landscape structure, biological pest control, polyculture, and rotation (Paoletti et. al, 1992). It has also been pointed out that biodiversity conservation and the objectives of sustainable agriculture are economically compatible (Altieri et. al, 1987). More research is needed to compare agroecosystems to undisturbed ecosystems.

In comparison to Europe, however, North American efforts to preserve biodiversity and make agriculture more sustainable have not been connected; this may due to the fact that those working on sustainable agriculture have emphasized ecosystem functions such as reducing erosion or restoring hydrologic regimes whereas conservation biologists have focused on fragmentation of communities or preservation of threatened species (Wedin, 1992). The lesson from European grassland conservation efforts is that "until we realize that finding sustainable agricultural practices and conserving threatened grassland biodiversity are intertwined problems, we may not find a solution to either" (Wedin, 1992).

[1] *Coordinator/Ecologist, Navajo Natural Heritage Program, Window Rock, AZ,; and Botanist, Navajo Natural Heritage Program..*

A long term objective of our research project will be an attempt to find sustainable agricultural practices that will preserve and enhance populations of a rare annual grass, *Puccinellia parishii*. This species is found exclusively associated with saline springs and cienegas, habitats which are threatened throughout the western United States. *P. parishii* is only known from seven small populations in: 1) Southern California in the vicinity of San Bernardino and Edwards Air Force Base; 2) Southwestern New Mexico approximately 30 miles south of Silver City; and 3) The Tuba City area of the western Navajo Nation and the Hopi Tribe (map. Fig. 1). An uncertain locality based on a 1948 collection from the northern Navajo Nation has not been relocated despite several attempts. The Tuba City populations were only recently discovered (Hevron, 1991).

The largest population of the grass near Tuba City occurs in Pasture Canyon which is farmed by Navajos and Hopis. The Southern California populations occur in an area of considerable population growth and on Department of Defense land. The New

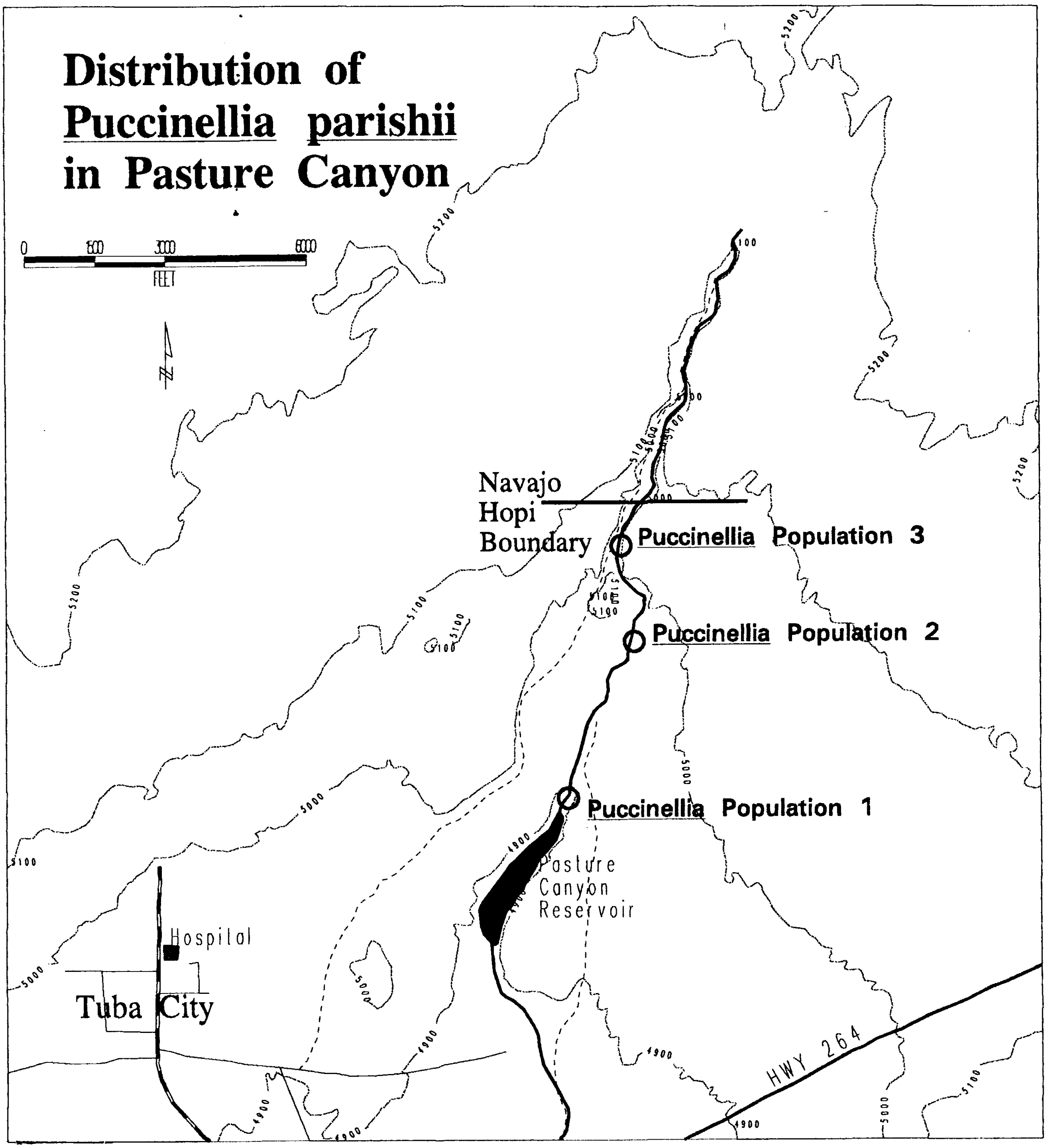

Figure 1. — Distribution of *Pucinellia parishii* in Pasture Canyon.

Mexico population is on land owned by the Phelps-Dodge mining corporation. Clearly threatened, the USFWS is proposing that Parish alkali grass be listed as Endangered under the Endangered Species Act. This paper will concentrate on research conducted in Pasture Canyon.

This rare alkali grass grows in three patches in Pasture Canyon. Two of these patches are in agricultural fields which are plowed and planted with crops each year. Pasture Canyon, nearly four miles in length, is bordered by sheer Navajo sandstone walls that exceed 100 feet in height. A perennial stream flows through the canyon and is fed by an extensive chain of springs and seeps. The canyon has been known as a oasis in the middle of barren rimrock and sand dunes for at least three hundred years. Use of Pasture Canyon by Navajo and Hopi farmers may antedate the Spanish invasion (Gregory, 1916). The canyon has been formally divided into a northern Navajo section and a southern Hopi section.

The crops grown in Pasture Canyon have significant cultural and economic value. Many of the plant crops are varieties indigenous to the Hopi lands. Of about 150 indigenous species, 144 plants were used for food or religious ceremonies (Gregory, 1916). We have observed Navajos collecting medicinal plants in the canyon. Much of the Hopi culture and religion is based upon corn which is the predominant crop grown in Pasture Canyon. Some Hopi farmers have stated that they depend upon the crops for income and food. Most of the Navajo traditional use area of the canyon, approximately one-half of the area is essentially fallow, whereas the Hopi use area is largely farmed.

Modern farming techniques are increasingly preferred over traditional techniques. Commercial fertilizer and herbicides are sometimes applied. Modern farming techniques appear to allow invasion of weedy species which may be altering the competitive relationship of *Puccinellia* within the plant community. The weedy invasive *Polygonum* sp. has formed a dense ground cover in one *Puccinellia* population, most likely in response to deep disc plowing.

It is possible to reconstruct the historic landscape ecology of the Canyon. The pre-historic vegetation was probably a dense cover of familiar marsh plants, *Typha*, *Scirpus*, *Juncus*, and *Triglochin*, with saltgrass *Distichlis* on the drier margins. Today there are small remnants of this native vegetation in fallow fields and along the margins of other fields. There are no records of the past abundance and distribution of *Puccinellia* in Pasture Canyon. None of the Navajos or Hopis interviewed have recognized the grass. Several farmers have asked "what is this plant good for?"

In 1915, Pasture Canyon contained three lakes. One of these was a 15 acre reservoir formed by a dam while the other two lakes were formed by encroachment of wind-blown sand (Gregory, 1916). Since that time floods have undoubtedly altered the alluvium geometry, and attempts have been made to prevent sand from encroaching upon the canyon floor. In addition, a ditch has been constructed to channel the spring flow along the sides of the canyon wall. In the winter this water and melted snow inundate portions of the canyon floor.

The Navajo Natural Heritage Program has commenced a research project on the distribution and biology of *P.puccinellia* in Pasture Canyon. The three objectives are: a management plan to protect the rare annual grass; a special management area in which the biodiversity of the wetlands are enhanced, restored, and conserved; and introduction of *Puccinellia parishii* into fallow plots in the upper canyon.

There are two main elements of the study. One is gathering cultural information from the farmers on their farming practices and their knowledge of the canyon's historical condition. Some results of a questionnaire circulated among farmers are as follows.

1) There are 34 fields, 14 of which are farmed by Navajos, and 20 of which are farmed by Hopis.
2) Many plots are farmed by several families or individuals. One plot is farmed by 10 people.
3) Most people indicated that the plots would be passed on to younger family members, but they were aware that the younger people were not very interested in farming.
4) Among the current farmers, the farming tenure in Pasture Canyon ranges from 10 to 51 years.
5) Many farmers do not farm in certain years because the road is impassable or the fields remain too wet to plow. Some farmers pump water from their field in the springtime.
6) The earliest plowing time is in April and the latest is in July.
7) The crops include blue, white and yellow corn, several types of squash, watermelons, cantaloupes, sweet corn, varieties of chili, tomato, cucumber, zucchini, onions, and several varieties of beans.
8) While plots were historically plowed using horses, increasingly modern techniques are being utilized. Deep disc plowing is the preferred cultivation method over the rototiller or spade.
9) Fertilizer and herbicide applications are now more common.
10) Many farmers burn their fields in order to control weeds.
11) Farmers have observed many changes in Pasture Canyon over their lifetime. These include more weeds, bigger sand dunes, greater vandalism, more livestock in the fields, more water and fewer plants, and fewer cottonwoods. One farmer stated that the corn does not grow as high as it once did.
12) Most farmers would like to see better management efforts by the tribal and federal governments to resolve some of these problems.

The second part of the study is collecting biological and ecological information. Soil samples have been collected from all known populations, including Pasture Canyon. The soils were analyzed for the following parameters: pH, water soluble salts (EC), calcium, magnesium, sodium, sodium adsorption ratio (SAR), carbonate, cation exchange capacity (CEC), sulfate, selenium, % sand, % silt, % clay, and texture. Preliminary analysis indicates no significant differences between sampling sites with *Puccinellia* and those without. Germination studies using seeds and soils from Pasture Canyon have been initiated at the Flagstaff Arboretum, a Center for Plant Conservation cooperator. Initial studies indicate that the annual grass grows better in moister soils (Machiniski, 1993). However, these initial studies utilized commercial potting soil and the seeds were planted in the late summer. This study may indicate that photoperiod and soil chemistry are as important factors as the period of inundation.

With the cooperation of two farmers, seven exclosures were set up in two populations of the annual grass. However, the tenure of one field changed before the field was plowed; the new land user removed the exclosures. Two exclosures remain in the other field. We will be examining the effects of this no plowing regime on these *Puccinellia* populations next year. Photos were taken from established points in order to monitor land use and periods of water inundation. In addition, seeds were collected from all Arizona populations in 1993 for more germination studies and genetic analysis.

Studies are planned to test several hypotheses. One hypothesis is that plowing represents a disturbance that allows *Puccinellia* to persist. Disturbance is important in maintaining species diversity in grasslands (Bazzaz, 1983). Restoration of locally extinct forbs have failed when seeds have simply been added unless a disturbance has been created by grazing or some other process (Wedin, 1992). A heterogeneous matrix of species may coexist longer than when they occur in extensive monoclonal populations. Loss of environmental fluctuations in dune grasslands results in vegetation succession and dominance of perennial species (Van Andel et. al, 1991). Most annuals are adapted to environmental fluctuations. For instance we have been studying an annual saltplant, *Proatriplex pleiantha*, that relies upon the occasional year when precipitation is abundant enough to se a good seed crop. During dry years, the populations of the species are extremely small.

An alternative hypothesis is that plowing is a threat to this rare annual grass. In a study of the effects of tillage and mulch on the emergence and survival of weeds in corn, it was concluded that tillage had a consistent effect on annual weed species that maintain a soil seed bank; that is no tillage improved emergence and survival of the weeds (Mohler and Calloway, 1992). A model of the effects of tillage on the emergence of weed seedlings indicates that no tillage will have more seedlings than tillage in the first year following input of seeds to the soil, but no tillage will have fewer seedlings in later years unless innate or induced dormancy is high or seed survival near the soil surface is very good. No tillage or minimum tillage will have more seedlings perennially if seed return is allowed. It would appear that cases in which the persistence of seeds increases with depth are annual grasses with large short lived seeds (Mohler, 1992).

Another hypothesis is that however modern farming techniques may be incompatible with this rare annual grass. Deep disc plowing as well as fertilizer and herbicide applications may represent significant threats. Increases in fertilizer and herbicides have greatly increased productivity of grasslands; however this management has also led to a sharp decrease in species diversity (Wedin, 1992).

We also postulate that inundation of the fields may be significant for two reasons. Due to the inundation, a late plowing schedule (the latest plowing being in July) may allow *Puccinellia* to complete its life cycle. Extensive pumping of the fields could jeopardize these populations of *Puccinellia* by altering soil chemistry. Winter rains and snowmelt may push alkalinity ions deeper into the soil column, resulting in a lower EC of the soil moisture and the surface; *Puccinellia* germinates and grows during the spring when the soil is moist and the EC's are low. As the moisture decreases seasonally, the salts move up in the soil column and thereby prohibit *Puccinellia* from extracting moisture, in which case, the annual grass may die (Griggs, F. Thomas, personal communication). Alternatively, inundation may create high turnover in the seed bank by causing anaerobic conditions. Wet meadows are sensitive to hydrologic changes brought about by drainage of agricultural land (Baker, 1987). If the peaty soils of the meadow dry out, then turnover of nitrogen increases significantly; this compounds the eutrophication caused by fertilization.

In addition to testing these hypotheses, we are also examining management options whereby land users are compensated for practicing ecologically sound and sustainable agricultural practices. It is hoped that both the Navajo and Hopi tribes can sustain their traditional farming and preserve *Puccinellia parishii* at the same time.

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Characteristics of Managed Forest Habitat Selected for Nesting by Merriam's Turkeys

Brian F. Wakeling[1] and Harley G. Shaw[2]

Abstract — Nest site selection by Merriam's turkey (*Meleagris gallopavo merriami*) was studied on the Mogollon Rim, Arizona, from 1987 through 1991. Compared with measured habitat availability, nests had higher shrub and deciduous tree seedling densities ($\underline{P} < 0.001$). Nests also had more cover comprised of shrubs, deciduous trees, and rock ($\underline{P} < 0.001$, $\underline{P} < 0.001$, and $\underline{P} = 0.036$, respectively). This cover averaged a greater height than at random sites ($\underline{P} = 0.032$). Green foliage volumes at nests were greater ($\underline{P} < 0.05$) and horizontal visibilities were lower ($\underline{P} < 0.001$). Nest sites were selected in stands that had clumped understory ($\underline{P} = 0.032$) and overstory distributions ($\underline{P} < 0.001$), and patchy forest canopies ($\underline{P} = 0.003$). Basal areas were greater on nest sites ($\underline{P} < 0.001$). Steep slopes were selected ($\underline{P} < 0.001$) for nesting purposes and canyons were the selected landform ($\underline{P} < 0.001$). Although turkeys nested in logging slash, slash piles were avoided.

To date, little timber harvest has occurred on slopes >40%, inadvertently protecting much nesting habitat. Timber treatments that promote small scale patchiness, such as uneven-aged management or group selection harvests, can emulate vegetational characteristics of nesting habitat. Leaving loosely scattered slash, especially near the base of trees, can provide suitable nest sites in stands with $\geq$ 50% canopy coverage.

INTRODUCTION

Merriam's turkey populations in the southwest may have declined from historic levels (Shaw 1986, Green 1990). Alterations to forest habitat by land management practices probably contributed to the decline (Shaw 1986). Turkeys select habitat characteristics based upon specific behavioral activity such as feeding or loafing (Rumble 1990, Mollohan and Patton 1991). Manipulations to forest structure can reduce habitat suitability (Scott and Boeker 1977).

Not all timber treatments, however, negatively influence turkey populations. In some cases timber harvest improved the suitability of turkey brood range (Mollohan and Patton 1991). Additionally, turkeys have used second growth timber extensively for feeding in some locations (Rumble 1990).

Limited information was available on nest site selection in southwestern Merriam's turkey range. Because timber harvest did not affect habitat selection by turkeys in a consistent manner (Scott and Boeker 1977, Mollohan and Patton 1991), loss of nesting habitat was considered to be a potential cause of reduced turkey numbers. Nesting habitat may be especially critical in affecting southwestern turkey populations. Generally, only adult (2 year old) hens nest in the southwest (Goerndt 1983, Crites 1988, Wakeling 1991, Stone 1993). In addition, relatively few hens live beyond the age of 3 (Wakeling 1991). We studied nesting habitat selection by Merriam's turkey to determine how management activities affected habitat use.

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[1] *Brian F. Wakeling is a Research Biologist with the Arizona Game and Fish Department, 2221 West Greenway Road, Phoenix, AZ 85023.*

[2] *Harley G. Shaw is a Wildlife Biologist with General Wildlife Services, P.O. Box 370, Chino Valley, AZ 86323.*

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STUDY AREA

The 335 mi^2 Chevelon Study Area (CSA) was located approximately 40 mi south of Winslow, Arizona, along the Mogollon Rim. Elevations ranged from 5500 ft in the northern portion to 7900 ft in the southern portion. Annual precipitation averaged 18.6 in, with 2 concentrations, the first occurring during winter storms in January through March, and the second during summer storms in July through early September (Natl. Oceanic and Atmos. Admin. 1991). Five habitat associations were identified on the CSA based upon terrestrial ecosystem surveys (Laing et al. 1989). These associations were mixed conifer, ponderosa pine (*Pinus ponderosa*)-Gambel oak (*Quercus gambelii*), pinyon (*P. edulis*)-juniper (*Juniperus* spp.), aspen (*Populus tremuloides*), and forest meadow associations (fig. 1). Mixed conifer associations were dominant above 7600 ft, and extended along east facing slopes and drainages. This habitat included Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), limber pine (*Pinus flexilis*), and Rocky Mountain maple (*Acer glabrum*). Ponderosa pine dominated west facing slopes below 7600 ft and above 6000 ft. Below 6000 ft, pinyon-juniper was dominant with ponderosa pine stringers in drainages. Gambel oak occurred in all associations, in pockets in the mixed conifer and pinyon-juniper associations, and as a widespread conspecific with ponderosa pine. At elevations below 7000 ft, pinyon and alligator juniper (*Juniperus deppeana*) became increasingly abundant.

Logging and grazing have been and remain major commercial land uses on the CSA. Logging began in the late 1930's and most ponderosa pine stands on level terrain have been logged at least once. Little logging has occurred on steeper slopes of major canyons.

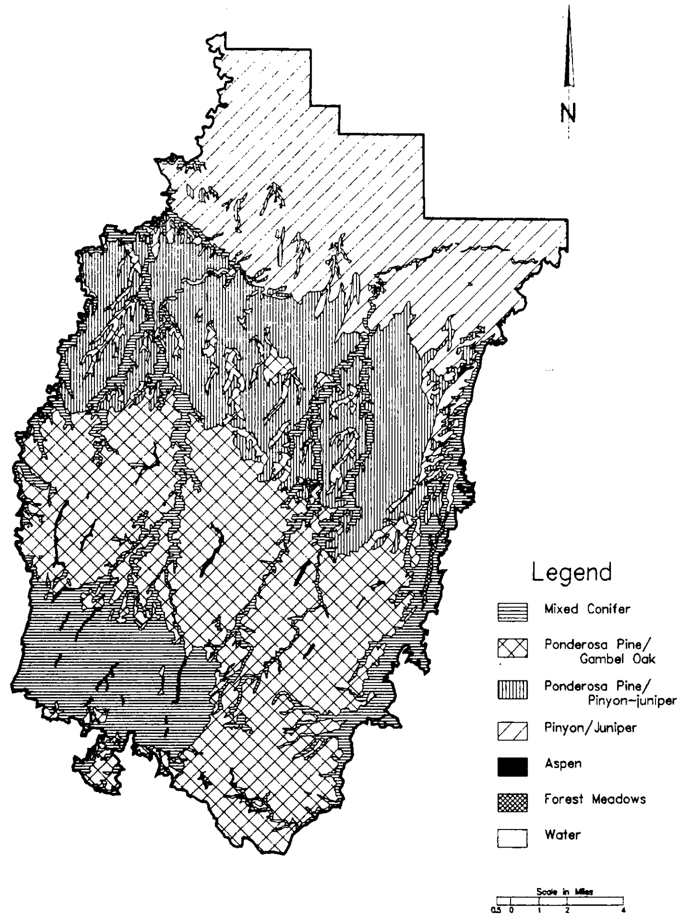

Figure 1. — Vegetation associations present on Chevelon Study Area, Arizona (based on Laing et al. 1984).

METHODS

Merriam's turkey hens were captured during winters of 1987 through 1990, using box traps, drop nets, and rocket nets (described by Wakeling 1991). Hens were equipped with motion-sensing backpack radio telemetry units (Telonics, Mesa, AZ and AVM Electronics, San Francisco, CA) and released at the capture site.

Nest sites were located by monitoring radio instrumented hens ≥2X weekly. Hens were suspected of nesting when >2 consecutive locations were within 1/4 mile of each other and motion-sensing transmitters indicated inactivity. Inactive hens were not approached for 1-2 weeks to prevent premature nest abandonment. Nests were then located and monitored daily to determine dates that hens left nests.

Habitat Mensuration

Habitat characteristics of each nest site were measured after the hen and brood abandoned the nest area following predation or hatching. Percent slope was measured with a clinometer. Landform was classified as minor canyon (<200 ft wide), major canyon (≥200 ft wide), or ridgetop-flat. Canopy structure was classified as single storied, multiple storied-uniform, or multiple storied-clumped (i.e. gave an uneven-aged appearance). Understory and overstory were classified as clumped or evenly distributed. Stem densities of shrubs and deciduous seedling (<1 in diameter breast height [DBH]), sapling (<1 to 5 in DBH), and mature (>5 in DBH) trees were determined on a 0.01 ac circular plot centered on the nest. Stem density of conifers was measured on a 0.1 ac circular plot, also centered on the nest. Basal area was determined from DBH measurements taken on all conifers encountered on this plot.

Four 25-ft line intercept transects were established, each radiating from site center at right angles to one another. The first transect was randomly oriented. These transects were used to determine canopy cover, within 18 in of ground, from rock, down wood, grass, forbs, shrubs, deciduous trees, and coniferous trees. The distance from which a turkey silhouette, placed at site center, could no longer be seen was determined in 4 directions, parallel to line intercept transects. Average height of cover was ocularly estimated. Green foliage volume was determined at each site according to MacArthur and MacArthur (1961).

Identical measurements were taken at random plots to represent habitat availability (Marcum and Loftsgaarden 1980). We generated random Universal Transverse Mercator coordinates by computer and plotted them on 7.5' USGS topographic maps. Once a point was located on the ground, we stepped off a random distance in a random direction. This endpoint was considered random plot center.

Data Analysis

The Mann-Whitney U test was used to test differences in non-normal continuous data (Zar 1984:138). We used Student t-tests to test for differences between normal continuous data (Zar 1984:126). Chi-square contingency table analysis was used to test categorical data (Zar 1984:62). Multiple categories were evaluated using Bonferroni simultaneous confidence intervals (Neu et al. 1974). All tests were considered significant at $P \leq 0.05$ with the exception of Bonferroni confidence intervals. Because these confidence intervals take into account simultaneous tests which affect individual alpha levels, the overall alpha level was set at $P \leq 0.1$ (Byers et al. 1984).

RESULTS

Habitat parameters were measured at 67 nest sites and 29 random plots. Shrub and deciduous seedling, sapling, and mature tree densities were higher at nest sites than at random plots (Table 1). Nest sites also had more 0-18 in cover comprised of shrubs, deciduous trees, and rock (Table 1). This cover averaged a greater height at nest sites (mean = 10.6 in) than at random plots (mean = 8.2 in) (t-test, $P = 0.032$). At nest sites, green foliage volumes below 15 ft were greater (Fig. 2) and horizontal visibility distances shorter (Table 1) than at random plots. Nest sites were selected in stands that had clumped understory and overstory distributions (Table 2). Multiple storied-clumped forest canopies were favored (Table 2). Basal areas were greater in stands at nest sites (mean = 96.1 ft^2/ac) than at random sites (mean = 57.0 ft^2/ac) (t-test, $P < 0.001$). Steep slopes were selected (Table 1) for nesting purposes and canyons were the selected landform (Table 2). Although turkeys frequently nested in scattered logging slash, slash piles were only used twice.

Less grass and forb cover was found at nest sites than at random plots (Table 1). No differences could be detected between nest sites and random plots in conifer tree densities or canopy cover from conifer trees or down wood.

Table 1. — Mann-Whitney U values, probabilities, and mean values for habitat parameters at nest and random sites on the Chevelon Study Area.

Habitat Parameter[a]	U	P	Mean Nest	Mean Random
Shrub Density	485.5	<0.001	5960	1017
Deciduous Seedling Density	524	<0.001	2228	169
Deciduous Sapling Density	726	0.006	300	10
Mature Deciduous Tree Density	728	0.006	120	7
Conifer Tree Density	808.5	0.193	50	60
Grass Cover	503	0.010	6.0	10.8
Forb Cover	545.5	0.031	4.4	7.3
Rock Cover	552.5	0.036	14.4	8.1
Down Wood Cover	755	0.896	12.7	12.4
Deciduous Tree Cover	408	<0.001	1.7	0.1
Conifer Tree Cover	718.5	0.621	2.0	2.0
Shrub Cover	394.5	<0.001	3.0	0.7
Silhouette Visibility	143	<0.001	49	111
Slope	187	<0.001	54	18

[a] *Densities are presented per acre, cover as percent canopy cover between 0-18 in of ground, visibility in ft, and slope as percent*

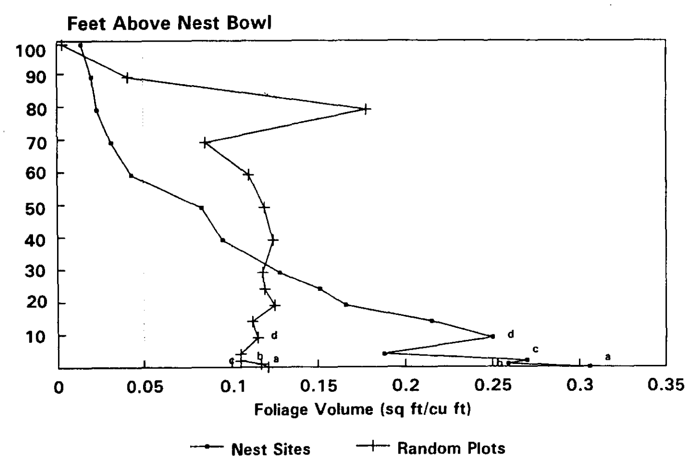

Figure 2. — Foliage volume (MacArthur and MacArthur 1961) on nest and random sites on the Chevelon Study Area, Arizona.

[abcd] *Points with the same letter are significantly different from each other (t-test, P 0.05).*

Table 2. — Nest and random site proportions, Bonferroni confidence intervals around nest site proportions, Chi-square values, and probabilities for habitat parameters on the Chevelon Study Area.

Habitat Parameter	Observed	Available	CI	X^2	P
Landform				37.07	<0.001
Minor Canyon	0.473	0.069	0.330-0.616		
Major Canyon	0.400	0.138	0.259-0.541		
Ridgetop-Flat	0.127	0.793	0.031-0.223		
n	55	29			
Understory Distribution				4.58	0.032
Even	0.222	0.448	0.111-0.333		
Clumped	0.778	0.552	0.667-0.889		
n	54	28			
Overstory Distribution				6.72	<0.001
Even	0.241	0.448	0.127-0.355		
Clumped	0.759	0.552	0.645-0.873		
n	54	29			
Canopy Structure				12.02	0.003
Single Story	0.000	0.071	0.000-0.000		
Multi-Uniform Stories	0.264	0.572	0.135-0.393		
Multi-Clumped Stories	0.736	0.357	0.607-0.865		
n	53	27			

DISCUSSION

In our study, steep slopes were selected for nesting. Merriam's turkey appear consistent in selection for steep slopes throughout their range (Goerndt 1983, Mackey 1984, Crites 1988, Hengel 1990). This selection may assist nesting turkeys in eluding detection by ground predators. Additionally, hens may take advantage of the incline to gain flight if disturbed.

Other characteristics selected at nest sites in our study seem to provide hens with hiding cover. Greater amounts of shrub and deciduous vegetation, rock, and green foliage volumes function to obscure nesting hens. Obscurity and camouflage would help hens avoid predator detection while nesting. Merriam's turkey consistently select nest sites that provide greater cover (Goerndt 1983, Hengel 1990, Rumble 1990).

Although turkeys selected nest sites in stands that had clumped multiple canopies, it is difficult to determine whether this was a true selection for canopy characteristics or selection for steep slopes, where these characteristics predominated. The importance of clumped overstories is not readily apparent. Clumped overstories may simply be indicative of suitable understories. Nest sites in our study not located in canyons were frequently in stands that had been treated with a group selection harvest, yielding a clumped overstory. While steep slopes were selected for, clumped overstories may also be important, although to a lesser degree.

Given adequate availability of other important features, such as water and suitable brood habitat, we believe Merriam's turkey select nesting habitat based upon 1) slope steepness and 2) suitable hiding cover. Turkeys select nest sites in habitats that are steep (>30% slope) and have short (<70 ft using a turkey silhouette) horizontal visibility distances.

MANAGEMENT IMPLICATIONS

Characteristics of habitat where nests are generally found include moderate to high basal area of conifer trees and high densities of shrubs and deciduous trees, arranged in a small scale mosaic. For management purposes, horizontal visibility distances is simpler to evaluate than cover composition and quantity. The distance at which a turkey silhouette is obscured is an appropriate rapid measure of habitat suitability for nesting purposes. Mollohan and Patton (1991) found a correlation (r = 0.853) between horizontal visibility distance of a turkey (HVD_t) and that of a person (HVD_p). The relationship between the 2 measures was $HVD_t = 0.39 \times HVD_p$. Thus, horizontal visibility distance at which a person is obscured can be used to evaluate habitat suitability for nesting. Obscuring cover may be comprised of herbaceous vegetation, shrubs, trees, steep slopes, or rocks.

Whenever possible, timber treatments should be avoided on slopes >30%. Timber treatments that leave moderate basal areas (70-90 ft^2/ac) in small (1-4 ac) patches favor retention of nest site characteristics. Scattering logging slash following harvest, rather than leaving slash piles, will result in habitat more suitable for nesting. Any treatment that increases horizontal visibility distance will reduce site suitability for nesting. Our results are consistent with nesting habitat recommendations suggested by Hoffman et al. (1993).

Timber treatments that result in characteristics described for nest sites can be used to retain or create suitable nesting habitat. Treatments such as group selection harvests or uneven-aged management appear to facilitate this management goal. Even aged management approaches appear less suitable.

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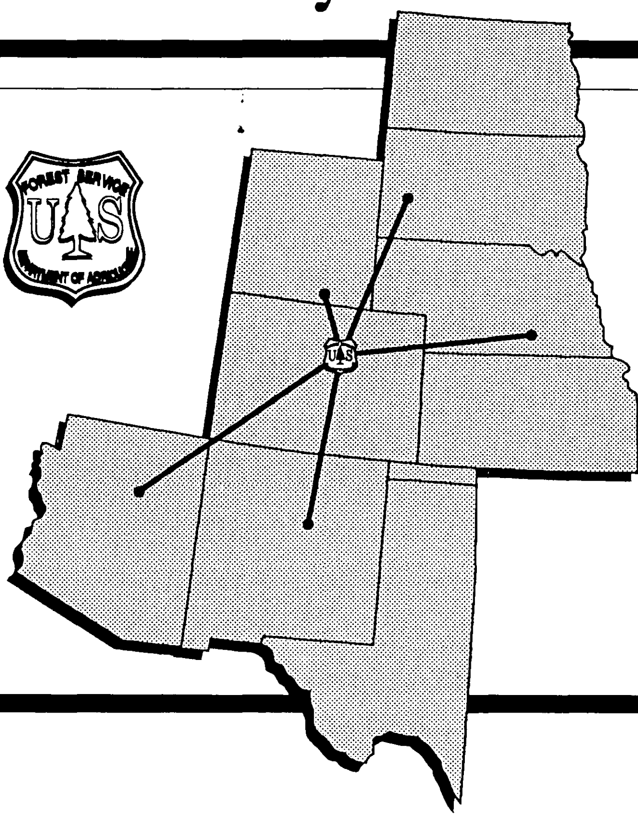

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