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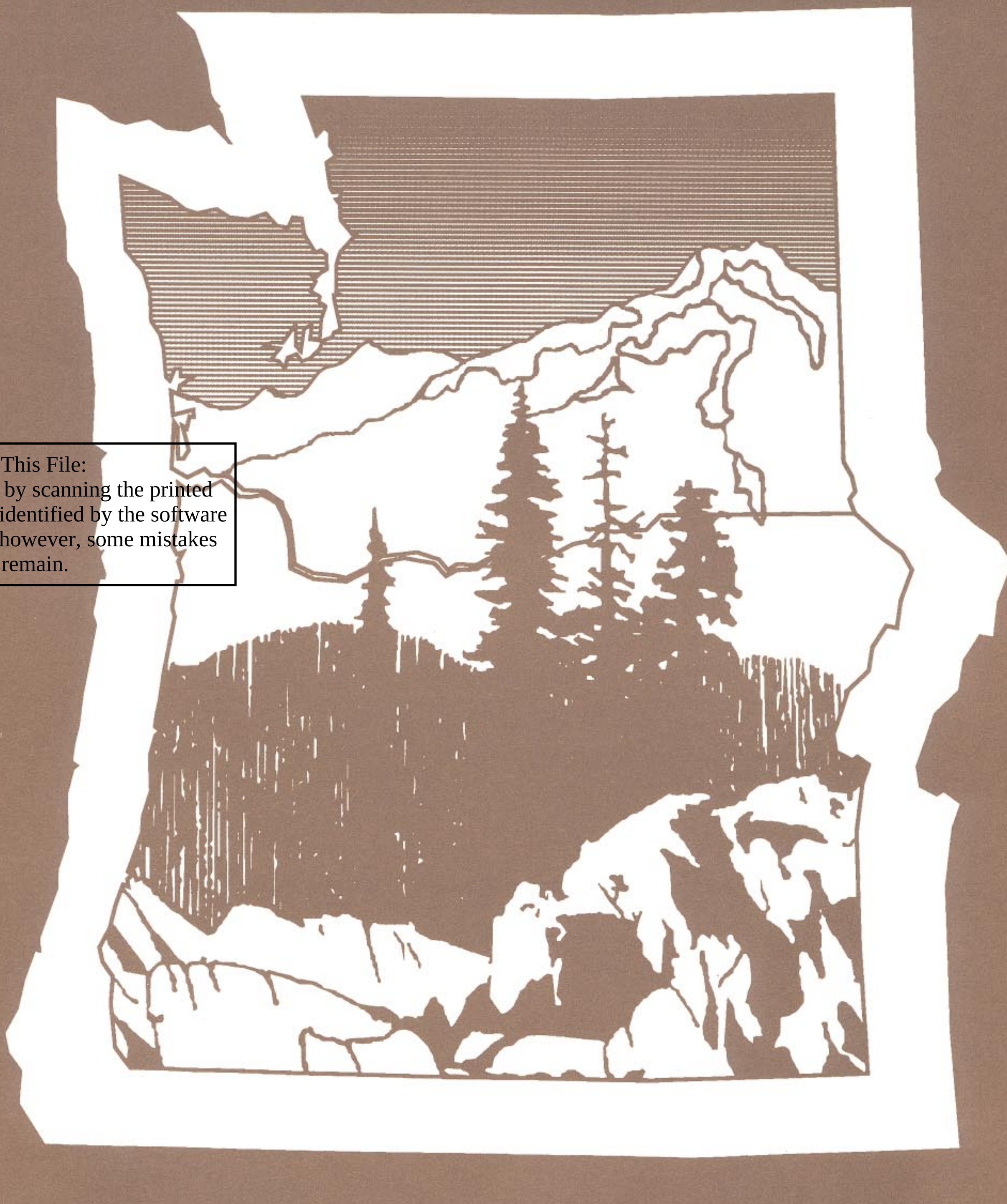
# Volume II:
# Ecosystem Management: Principles and Applications



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# Volume II: Ecosystem Management: Principles and Applications

M.E. Jensen and P.S. Bourgeron

Technical Editors

# Eastside Forest Ecosystem Health Assessment

Richard L. Everett, Assessment Team Leader

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## ABSTRACT

**Jensen, M.E.; Bourgeron, P.S., tech. eds. 1994.** Volume II: Ecosystem management: principles and applications. Gen. Tech. Rep. PNW-GTR-318. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 376 p. (Everett, Richard L., assessment team leader; Eastside forest ecosystem health assessment).

This document provides land managers with practical suggestions for implementing ecosystem management. It contains 28 papers organized into five sections: historical perspectives, ecological principles, sampling design, case studies, and implementation strategies.

Keywords: Ecosystem management, landscape ecology, conservation biology, land use planning.

# Ecosystem Management: Principles and Applications

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# ECOSYSTEM MANAGEMENT: PRINCIPLES AND APPLICATIONS

## INTRODUCTION

This document provides land managers with practical suggestions concerning the implementation of ecosystem management. It was developed in response to congressional direction for an independent, scientific evaluation of forest health in eastern Washington and Oregon (i.e., Eastside Ecosystem Health Assessment). Three science teams were formed in September 1992 to address this task: the Implementation Framework Team, the Eastside Assessment Team, and the Broad, Strategic Framework Team. The papers contained in this document represent the efforts of the Implementation Team. They were developed to: (1) document existing science principles appropriate to ecosystem management that would provide a foundation for the efforts of the Eastside Assessment Team, and (2) provide generic guidelines for ecosystem management that would apply to most land management needs.

The primary audience for this document are land managers who are increasingly being challenged to produce goods and services to society while managing for ecosystem sustainability (e.g., USDA Forest Service, USDI Bureau of Land Management, and Champion International). Accordingly, we have provided general summaries of appropriate science principles as well as recommendations for implementing those principles in management.

Most of the papers in this document emphasize Forest Service examples because of the congressional direction for this document (i.e., provide a framework the Forest Service can use in sustainable ecosystem management). The topics covered, however, are also applicable to other public and private land management organizations. The papers presented also vary in technical detail. Some of the topics covered are of value to a general nontechnical audience; while others (e.g., ecological theory) are appropriate to natural resource specialists and scientists. All papers are self-contained to facilitate review of this document by a varied audience (i.e., papers may be read independent of each other). They are cross-referenced, however, to display relations among the topics covered.

This document consists of five sections that address major ecosystem management issues. The first section (Historical Perspectives) presents some basic principles of ecosystem management and the historical context of their development. This section also provides general information that will be of interest to most land managers. The discussions by Jensen and Everett (ecosystem management overview), Kennedy and Quigley (social perspectives), and Shepard (political perspectives) are of particular importance because they review events that have influenced the Forest Service transition to an ecosystem management philosophy.

The second section of this document (Ecological Principles) provides a summary of important theoretical concepts for ecosystem management. This section provides information for a more technical audience and emphasizes the use of landscape ecology principles in land management. The topics addressed in this section (e.g., hierarchy theory, conservation biology, and ecosystem dynamics) are appropriate to ecosystem management implementation. Most of the concepts presented are not new; however, they have not commonly been explicitly addressed in previous land management efforts. We believe that if the theoretical concepts presented in this section were implemented, sustainable ecosystem management could be achieved. We do not believe that the development of new science theory is a prerequisite to ecosystem management. Instead, we emphasize the importance of using existing science principles (in more creative ways) to meet our evolving land management needs.

The third section of this document (Sampling Design and Data Analysis) presents various concepts useful to the characterization and evaluation of ecosystems. Some of the papers in this section are technical and may be appropriate only to specific resource disciplines. The topics covered, however, are important to the development of cost-effective survey designs for ecosystem characterization. The reviews of data analysis methods emphasize current techniques useful in describing ecosystem process and pattern relations. The ideas presented in this section facilitate improved ecosystem description and evaluation, thereby providing a more solid foundation for ecosystem management.

Planning efforts that use various aspects of the landscape ecology principles, sampling design, and data analysis methods discussed previously are presented in section 4 (Case Studies). This section provides practical examples of ecosystem management implementation that will be of interest to most readers. The new approaches to land evaluation and forest planning described in these case studies are appropriate to other land management efforts that emphasize the maintenance of sustainable ecosystems.

The last section of this document (Implementation Strategies) provides suggestions about the application of ecosystem management concepts. Major topics covered include the role of forest planning in ecosystem management, the use of ecological theory in land management, socioeconomic factors of ecosystem management, the role of incentives in economic tradeoffs, conflict resolution techniques, and adaptive management approaches to “experiments” in ecosystem management. The nine papers of this section offer straightforward suggestions to land managers and are appropriate to a general audience.

The authors of this document believe that ecosystem management is important and can be implemented based on our current scientific knowledge and land management experience. Experiments in ecosystem management, however, will need to be reconsidered, as new experience and knowledge is accumulated. Accordingly, we do not provide “cookbook” strategies for ecosystem management implementation in this document because they would not apply to all situations and would soon be obsolete. Instead, we review relevant scientific principles of ecosystem management and offer general recommendations to assist land managers in developing improved (area specific) land management plans and project designs.

M.E. Jensen and P.S. Bourgeron (editors)

# SECTION 1 - HISTORICAL PERSPECTIVES

## Summary of Historical Perspective Papers

The U.S. Department of Agriculture, Forest Service (Forest Service) has redefined its land management mission in “ecosystem management” terms (Overbay 1992). This new philosophy emphasizes ecosystem sustainability while providing for a wider array of uses, values, products, and services from the land for an increasingly diverse public. It reflects the agency’s commitment to its “Caring for the Land and Serving People” vision statement (USDA Forest Service 1986), and reflects an attempt by the agency to be more responsive to society’s changing concerns and expectations for Federal land management. Issues such as biological diversity, ecological function and balance, commodity production, and social values have required the Forest Service to adopt a new management direction that will change its traditional focus of sustaining yields for competing resource outputs to a more holistic vision of ecosystem sustainability (Kessler et al. 1992). This section contains a series of papers that describe fundamentals of ecosystem management, the historical context of their development, and the ability of Forest Service culture to embrace these concepts. The historical perspectives offered in these papers provide insights useful in developing strategies for ecosystem management.

Jensen and Everett open this section with a general overview of ecosystem management principles. Emerging definitions and concepts of ecosystem management are reviewed, and a conceptual framework for ecosystem management is provided. This framework is based on a land evaluation method of the Food and Agriculture Organization developed for international use in the 1970s.1n this approach, integrated land inventories are used to answer the following four questions: What is there? Where is it? When is it present? and How does it work? This information is then used to provide answers about “what, where, when, and how” in evaluation of alternative land uses and management practices. Factors such as anthropology and sociology (human wishes and requirements), land ecology (biological and physical relations), technology, and economics are used in this method to define optimal land use planning (i.e., ecosystem management).

The conceptual framework for ecosystem management described by Jensen and Everett suggests that the following steps be used in land evaluation and planning: (1) human requirements and desires must be considered initially in ecosystem management design; (2) such values are then compared with the land’s ecological potentials, to determine management potentials given long-term sustainability objectives; (3) available technology is then used to refine the list of management potentials by identifying limitations to moving the current landscape to a proposed desired condition; and (4) economic factors are used to determine what parts of the human requirements and desires (refined in steps 2 and 3) can be fulfilled. The authors advocate that an adaptive management approach (Walters and Holling 1990) is to be used in implementing this framework through land-use planning.

The second paper of this section (Kennedy and Quigley) reviews how the Forest Service did and did not adapt to the needs of American society in its industrial (1900-1960) and postindustrial (1961-present) stages of socioeconomic development. The authors suggest that the traditional, machine-model management paradigm of the agency was appropriate to an industrial society of the 1950s, which endorsed and budgeted a transformation of National Forests from an inaccessible, extensively managed, native forest system into a more roaded, intensively used and managed multiple-use forest estate, This traditional machine-model view of forest management was manifested in (1) narrow forest ecosystem perceptions (simple site productivity models), (2) forest and fire management (intensively managed plantations and trees-good-fire-bad mentality), (3) agency or organizational structures (line-staff, generalist-specialist, strict functionalism), (4) organizational processes (benign, educated, Forest Service professionals managing National Forests for the uniformed, self-centered public and future generations), and (5) functional, reductionist research. Control-oriented people and organizations find comfort in a machine-model world view; however, complex postindustrial societies have made such thinking obsolete.

Kennedy and Quigley indicate that more complex, diverse, and interrelated organic models (such as ecosystem management) are needed to understand and adapt to today’s world. Failure to adopt such ecosystem management models could result in a shift from multiple-use to dedicated use on Federal lands. For example, in 1984 about 75 percent of New Zealand national forest lands were removed from New Zealand Forest Service stewardship. In this situation, the New Zealand society judged that a machine-minded forestry profession and agency was only trustworthy to manage their machine-model conifer plantations. Kennedy and Quigley suggest that the following items be considered by the Forest Service to embrace ecosystem management values and methods (1) develop interdisciplinary classifications and training that transcend traditional functional boundaries by emphasizing an integrated ecosystem management approach; (2) shift RPA and forest planning from an output-driven exercise to a desired sustainable ecosystem model that secondarily estimates output endowments; (3) replace the current machine-model budgeting process with an organic-model process that enhances ecosystem management values and goals; (4) shift from a machine-model, output and loyalty-oriented reward system to a system that supports the creation and enhancement of diverse, adaptable, and sustainable ecosystems, organizational cultures, and user services; and (5) consider changes in the organizational structure of the agency to dilute the intellectual, power, and budgetary myopia of current Forest Service functionalism.

The third paper of this section (Shepard) discusses critical political implications, impediments, and imperatives that the Forest Service must acknowledge in developing ecosystem management strategies. Shepard states that traditional "old forestry" tenets (i.e., forest management as applied science, timber primacy, and decentralization) served the Forest Service well in its previous conservationism era. These same tenets, however, are now major political liabilities in a changing country that appears to be demanding more organic-based approaches to ecosystem management (Kennedy and Quigley). Shepard also indicates that the traditions of "old forestry" left the Forest Service ill-equipped to deal with both the harvest controversies of the 1970s and implementation of the National Forest Management Act of 1976. He suggests that if ecosystem management is to succeed, it must involve more than better applied science; it must also embrace the political responsibilities of the land manager.

Shepard states that major stresses and breakdown in the national political framework for Federal land management must be recognized and accommodated. Examples of these stresses and breakdowns include (1) a lack of linkage between local and national politics (e.g., national timber targets that cannot be implemented locally); (2) national politics that commonly rely on fantasy instead of vision and make promises that are really pipe dreams (e.g., we can cut taxes and dramatically increase spending without paying a price); and (3) a political agenda that relies increasingly on symbolic issues rather than on significant material problems.

Accordingly, Shepard recommends that the political aspects of forest management be renewed and reinvigorated (i.e., the problem is not too much politics, but too little good politics). Shepard suggests that the following items be considered by the Forest Service in developing a new political recipe for ecosystem management (1) accept the fact that timber primacy is gone and such interrelated concepts as sustainability, biodiversity, and health of the soil are taking its place; (2) embrace the political role as something necessary and positive; (3) use the expertise of natural resource managers to counter the politics of fantasy by educating the public and political leaders to the real choices, costs, and consequences that must be faced; (4) move decisionmaking away from national levels by increasing local initiatives for management; and (5) use more creative and hybrid organizations that are neither private nor public in the decisionmaking process (i.e., deemphasize formal political processes).

Ecosystem management requires a balance between human desires and the biological and physical capacities of ecosystems (Jensen and Everett). Accordingly, land managers must develop an improved understanding of the spatial and temporal relations of ecosystem processes and patterns if sustainable ecosystem management is to be achieved. Landscape ecology represents a scientific discipline that evolved to meet such management needs. The final paper of this section (Golley) describes the historical development of landscape ecology and its relation to environmental management. The historical perspective presented by Golley is appropriate to this document given that most of the papers in sections 2, 3, and 4 are based on landscape ecology theory and application.

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# An Overview of Ecosystem Management Principles

**M.E. Jensen and R. Everett**

## ABSTRACT

Ecosystem management is an emerging philosophy that the USDA Forest Service has embraced in its multiple-use, sustained-yield management of National Forest System lands. The primary objective of this philosophy is to sustain the integrity of ecosystems (i.e., their function, composition, and structure) for future generations while providing immediate goods and services to an increasingly diverse public. This objective can be achieved through integrated land evaluation, optimal land-use planning, and the creation of landscape structure and process that meet society's expectations but also consider the constraints of the land's ecology. Sociological, ecological, technological, and economic information must be integrated to identify optimal land uses and to describe the spatial relations of commodities and values across the landscape. A balance is needed among demands for resources, the maintenance of ecosystem integrity, and the conservation of options for future generations. We present a conceptual framework for ecosystem management implementation that incorporates these basic principles of integrated land evaluation.

## INTRODUCTION

Ecosystem management requires the maintenance of sustainable ecosystems while providing for a wider array of uses, values, products, and services from the land to an increasingly diverse public (Overbay 1992). In many respects, ecosystem management represents a refocusing by the USDA Forest Service, on the "sustainable" part of the Multiple-Use, Sustained-Yield Act of 1960. Decisions that emphasize a view to the future, promote sustained production over the long run, and maintain all the pieces of ecosystems are characteristic of this emerging management philosophy (Risbrudt 1992).

Overbay (1992) proposes that the following six principles be used to describe the initial components of ecosystem management:

- Multiple-use, sustained-yield management of lands and resources depends on sustaining the diversity and productivity of ecosystems at many geographic scales.
- The natural dynamics and complexity of ecosystems means that conditions are not perfectly predictable and that any ecosystem offers many options for uses, values, products, and services, which can change over time.
- Descriptions of desired conditions for ecosystems at various geographic scales should integrate ecological, economic, and social considerations into practical statements that can guide management activities.
- Ecosystem connections at various scales and across ownerships make coordination of goals and plans for certain resources essential to success.
- Ecological classifications, inventories, data management, and analysis tools should be integrated to support integrated management of lands and resources.
- Monitoring and research should be integrated with management to continually improve the scientific basis of ecosystem management.

We provide further discussion of these ideas and offer suggestions that will assist ecosystem management implementation. A review of legislative directives and management philosophies is presented to describe the historical development of ecosystem management. We also provide a conceptual framework for ecosystem management implementation based on principles of integrated land evaluation.

## Historical Development of Ecosystem Management

### Legislative Directives

The legal precedence for ecosystem management originated with the Organic Administration Act of 1897. This act stated that: “No National Forest shall be established, except to improve and protect the forest within the boundaries, or for the purpose of securing favorable conditions of water flows, and to furnish a continuous supply of timber for the use and necessities of citizens of the United States.” This decree was more specifically interpreted by Congress through the Multiple-Use Sustained-Yield Act (MUSYA) of 1960, which states: “The Secretary of Agriculture is authorized and directed to develop and administer the renewable surface resources of the National Forests for multiple use and sustained yield of the several products and services obtained therefrom.”

The MUSYA refined the “improve and protect the forest” provision of the Organic Act through multiple-use and sustained-yield concepts. These concepts provide the legal foundation for the ecosystem management philosophy, and it is through the implementation of their intent that ecosystem management will be achieved. The definitions of these terms (section 4, MUSYA) are provided below:

“Multiple use” means the management of all the various renewable surface resources on the National Forests so that they are used in the combination that will best meet the needs of the American people; making the most judicious use of the land for some or all of these resources or related services over areas large enough to provide sufficient latitude for periodic adjustments in use to conform to changing needs and conditions; that some land will be used for less than all of the resources; and harmonious and coordinated management of the various resources, each with the other, without impairment of the productivity of the land, with consideration being given to the relative values of the various resources, and not necessarily the combination of uses that will give the greatest dollar return or the greatest unit output.

“Sustained yield of the several products and services” means the achievement and maintenance in perpetuity of a high-level annual or regular periodic output of the various renewable resources of the National Forests without impairment of the productivity of the land.

Refinement of the intent of the MUSYA is evident in the Wilderness Act of 1964 which dictated that some areas within the National Forest System would be “used for less than all of the resources.” According to Overbay (1992), “all lands within the National Forest System are used for less than all the resources; always have been and probably always will be.” This practice occurs because different areas of land have different potential for resource use and resiliency to disturbance. Accordingly, commodities and values are emphasized differently across the landscape based on their potential and on societal demands. For this reason, multiple use is a concept that works best at landscape and larger geographic scales where the patterns or mosaics of land uses may be inspected and integrated (Overbay 1992).

The National Environmental Policy Act (NEPA) of 1969 directed that Federal lands be managed to “encourage productive and enjoyable harmony between man and his environment; to promote efforts which will prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of man; (and) to enrich understanding of the ecological systems and natural resources important to the Nation,” The emphasis placed on ecosystem management in the language of NEPA was further refined in the Endangered Species Act of 1973, which dictated that all Federal agencies conserve endangered and threatened species and the ecosystems they depend on. The Forest and Rangeland Renewable Resource Planning Act (RPA) of 1974 required descriptions of the potential National Forest System lands offered for public forest and rangeland resources, goods, and services. This Act was followed in 1976 by the National Forest Management Act which reemphasized the importance of multiple-use, sustained-yield management and directed the Forest Service to develop long-term plans to describe how they would meet the intent of the MUSYA. Accordingly, the Agency entered into an era of forest planning that included plan development, implementation, monitoring, and revision (Grossarth and Nygren 1993, Morrison 1993, and Shepard 1993).

### Management Philosophies

The Forest Service is responsible for implementing the intent of legislative direction, Accordingly, the agency has changed management philosophies to ensure that congressional intent involving public desires was met in its stewardship of National Forest System lands (Kennedy and Quigley 1993). The agency has historically embraced a mechanistic, reductionist world view (Botkin 1990) in its management philosophy (Kennedy and Quigley 1993). This philosophy is evident in the agency's use of agricultural-based production models in wildland management (NRC 1990). Such models emphasize the identification and minimization of limiting factors to management. They also treat the relation among different resources (multiple uses) as constraints on the dominant use (e.g., timber production and cattle grazing) (Behan 1990).

This interpretation of multiple use is explicit in many forest plans (Morrison 1993) which were commonly developed following "old forestry" concepts such as timber primacy (Shepard 1993). Additionally, much of the analysis used in forest plan development (e.g., FORPLAN) was rather simplistic and did not consider ecosystem dynamics and spatial patterns (Grossarth and Nygren 1993, Morrison 1993). The traditional multiple-use philosophy of the Forest Service views the land as a place to produce commodities while maintaining other amenity values by identifying optimum yields of desired (often competing) uses.

Kessler and others (1992) suggest that this interpretation of multiple use may not be appropriate to a society that increasingly demands the maintenance of healthy, diverse, and productive wildlands. These authors also suggest that a philosophy is required that recognizes that forest lands (as living systems) have importance beyond traditional commodity and amenity uses (i.e., they are important life-support systems, Dawkins 1972). This philosophy should emphasize developing management objectives that relate to ecological and aesthetic conditions of the land, and implementing practices that maintain resource values and yields compatible with those conditions (Kessler et al. 1992).

The increasing number of appeals and litigation of forest plans indicate that the agency's production-oriented, multiple-use paradigm no longer reflects public opinion, and that new strategies for land management need to be developed that better reflect the opinions of society (Kessler et al. 1992). Various professional societies and groups have also emphasized the need for natural resource managers to take a more holistic, ecosystem-based approach to land management (Lubchenko et al. 1991, NRC 1990, SAF 1993, USDA Forest Service 1990).

## A Conceptual Framework for Ecosystem Management

Ecosystem management reaffirms the intent of the MUSYA by the Forest Service. Accordingly, the agency must develop strategies for land management that meet everchanging public desires and needs while maintaining the sustainabiiity of ecosystems. Overbay (1992) suggests that such strategies must consider: (1) the fact that people want a wider array of uses, values, products, and services from the land than in the past; especially, the amenity values and environmental services of healthy, diverse lands and waters, (2) biological diversity is a key factor in sustaining the health and productivity of ecosystems, (3) integrated ecological inventories are required to support ecosystem management, (4) people outside the Forest Service want more direct involvement in the decisionmaking process, and (5) the complexity and uncertainty of natural resource management require stronger teamwork between scientists and resource managers.

The land evaluation method of Beek and Bennema (1972) provides a conceptual foundation for ecosystem management that incorporates Overbay's suggestions. This system has been adopted by the Food and Agriculture Organization for use in all its projects (FAO 1976) and by the International Society for Soil Sciences in engineering, agriculture, rangeland, and forestry land-use planning (Zonneveld 1988). In the following discussion, we provide a brief overview of this land evaluation system as described by Zonneveld (1988).

Land evaluation includes inventory, classification, and analysis to determine optimal land uses. Inventory requires data from relevant land properties that describe: What is there? Where is it? When is it present? and How it functions? Analyses of such data addresses questions of "what, where, when, and how" in relation to the alternative land uses considered or the management actions to be implemented.

Zonneveld (1988) suggests that this process is appropriate to both internal land evaluation (i.e., one individual holding) and integrated external land evaluation (i.e., regional land-use planning). In the latter situation, he recommends that land evaluation consider sociological (human wishes and requirements), ecological (ecological possibilities), technology (tools available to management), and economic (available funds) factors. Simultaneous synthesis, of these factors provides for optimal land-use planning.

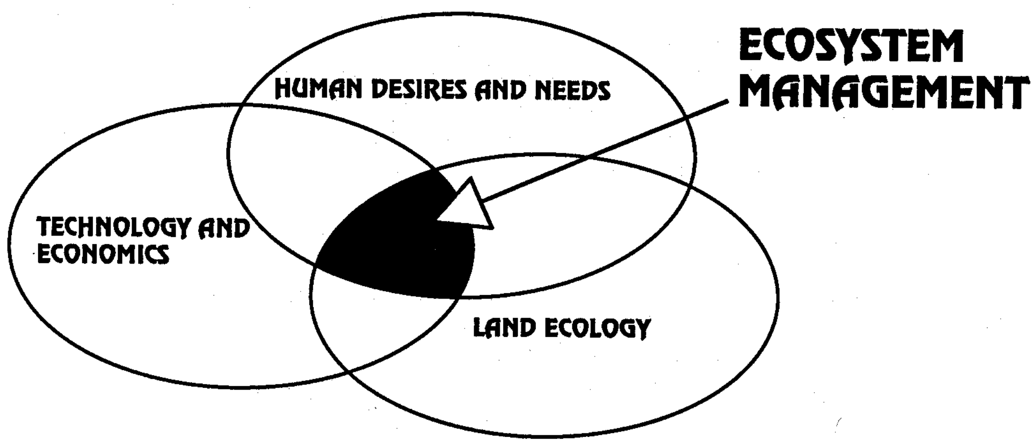


**Figure 1--Conceptual framework for ecosystem management (adapted from Zonneveld 1988).**

We present a modification of Zonneveld's concepts (fig. 1) where ecosystem management is displayed as the optimum integration of societal desires and requirements, ecological potential of the landscape, and economic plus technological considerations. The following steps describe how the land evaluation process may be used to achieve ecosystem management:

1. Determine the desires and requirements of people who will be influenced by the planning outcome.

2. Describe the ecological potential of the land for meeting stated societal needs. Such descriptions must include a description of the range of conditions required to maintain long-term system sustainability, a description of current conditions, and a description of desired landscape conditions that achieve societal needs.

3. If desired landscape conditions fall outside the range of conditions required for long-term system sustainability, inform the people who will be affected. Public awareness of ecosystem potential is critical in developing achievable "desired future condition" strategies for land management. Public desires are refined through this process, based on an understanding of sustainable ecosystem criteria.

4. Once a socially acceptable, sustainable vision of the landscape is achieved, it is then contrasted against available technology to determine if it can be implemented. For example, in many instances the desired landscape condition may differ from existing conditions. In these situations, factors such as system design and equipment availability must be considered to determine if the existing landscape can be changed to some desired set of conditions.

5. Determine what parts of the stated human desires can be fulfilled given economic factors. If resources (economic and technological) are not available to construct the desired landscape, the public should be notified and alternative strategies developed. In most situations, short-term economic reasoning and large management impacts contribute to situations that violate land

ecological and human values (Zonneveld 1988). Accordingly, these factors should be avoided in the development of strategies for ecosystem management.

These steps refine human desires based on land ecology, technology, and economic considerations. Such refinement requires that the public be informed of land evaluation findings and that public opinion be solicited throughout the process. The maintenance of sustainable ecosystems (as a basic tenent of ecosystem management) requires constant public input; however, ecosystems (in and of themselves) do not require management. The ability of our planet to sustain itself through periods of major climate change (glaciation), tectonic activity, and other disturbance events (biblical floods) indicates that the earth is quite capable of maintaining itself without our assistance. Instead, we manage ecosystems to ensure that desires and requirements of people are met now and in the future. Managers must understand the ecological potential and interactions of the land if they are to provide sustainable ecosystems for future generations. We provide general recommendations for the social, ecological, and economic-technological components of land evaluation in the following discussion:

**Social Consideration**

Human desires and requirements (as reflected in public laws, forest plans, and project decisions) define the goals of ecosystem management (Overbay 1992). One of the largest challenges to ecosystem management, therefore, is to ensure that public desires are compatible with ecosystem potentials. Accordingly, the expertise of natural resource managers and scientists must be used to educate the public and political leaders to the real choices, costs, and consequences of public land management.

Perceptions and preferences of risk are important factors that influence public demands on forest management policy. The uncertainty of ecosystem management (both in product outputs and public demands) requires that land managers design flexible policies to ensure that changing public perceptions of risk may be accommodated (Montgomery 1993).

Public participation has always been a major component of forest planning (Morrison 1993, Grossarth and Nygren 1993). New approaches to Forest Plan implementation, such as integrated resource analysis, improve information-sharing with the public regarding ecological conditions and trends affecting public lands. The integrated resource analysis process may also be used to minimize conflicts between different user groups through mutual development of ecologically based, desired-future-condition landscape descriptions (O'Hara et al. 1993). Additionally, collaboration (i.e., a process in which interdependent groups work together to affect the future of an issue of shared interests) should increasingly be used in the resolution of social conflicts that may arise from ecosystem management (Daniels et al. 1993).

**Land Ecology Considerations**

Determining ecological interactions and land potential is a major component of sustainable ecosystem management. The theory and principles developed in the academic disciplines of landscape ecology (Golley 1993) and conservation biology (Bourgeron and Jensen 1993) provide a solid foundation for experiments in ecosystem management. These principles and theories have been embraced by various professional societies and agencies (Lubchenko et al. 1991, NRC 1990, USDA Forest Service 1990, 1992) in their attempts to develop strategies for the maintenance of healthy ecosystems during a period of rapid human population growth and associated resource demand.

Some of the major landscape ecology and conservation biology principles applicable to ecosystem management are summarized below:

- Hierarchy theory--the development and organization of landscape patterns (e.g., vegetation communities) is best understood in the context of spatial and temporal hierarchies (Bourgeron and Jensen 1993). Disturbance events that maintain landscape patterns and ecosystem sustainability are also spatial-temporal scale dependent phenomena (Turner et al. 1993). Acknowledgment of these facts is critical to the development of management strategies for ecosystem sustainability (USDA Forest Service 1992). Applying these principles requires that land evaluation be conducted at multiple scales of ecological description rather than at traditional detailed scales such as stands or stream reaches (Milne 1993, Glenn and Collins 1993, Hann et al. 1993a, Bailey et al. 1993). The temporal variability (e.g., vegetation succession dynamics) of landscapes also needs to be addressed in land evaluation (Hann et al. 1993b, O'Hara et al. 1993, Shlisky 1993).

- Natural variability--all ecosystems vary across time and space, even without human influence. Knowledge of this variability is extremely useful in determining if the current condition of a landscape is sustainable given historic pattern and process criteria. Descriptions of historic landscape disturbance regimes (e.g., fire magnitude and frequency) and the ecosystem component patterns they maintained (e.g., vegetation composition) provide an initial template for assessing ecosystem health (Bourgeron and Jensen, Swanson et al. 1993). Such descriptions are useful in broad-level resource analyses of risk (Hann et al. 1993a) as well as in more detailed identification of watershed restoration treatment needs (Shlisky 1993). These descriptions also provide information for forest planning and monitoring (Morrison 1993, O'Hara et al. 1993).

- Coarse-filter conservation strategy--the conservation of diversity (e.g., species, ecosystem processes, and landscape patterns) is the primary method for maintaining the resilience and productivity (health) of ecological systems. Traditional approaches to conserving diversity have relied on a species-by-species approach (i.e., fine filter) which emphasized maintaining habitat for threatened, endangered, and sensitive species. A more proactive approach to species conservation is the "coarse-filter" approach to biodiversity maintenance (Bourgeron and Jensen 1993, Hunter 1991). This approach assumes that if landscape patterns and process (similar to those that species evolved with) are maintained, then the full complement of species will persist and biodiversity will be maintained. Application of this concept requires an understanding of the natural variability of landscape patterns and processes. Landscape ecology principles provide this understanding and are the foundation for experiments in ecosystem management (USDA Forest Service 1992). Such experiments are effectively implemented through an adaptive management approach to land management (Everett et al. 1993).

**Economic and Technology Considerations**

Nonmarket ecosystem products (biodiversity) are not well considered in most traditional economic systems (Montgomery 1993). Recent developments in the field of ecological economics, however, offer promise for ecosystem management (Common and Perrings 1992, Pearce and Turner 1990). Specifically, much work is now being directed to developing common units of measure that integrate renewable (market) and nonrenewable resources in the description of "natural capital stock" (Pearce and Turner 1990). Suggestions for applying these concepts in ecosystem management are provided by Ervin and Berrens (1993).

Ecosystem management goals may be facilitated through regulation or economic incentives. Regulations have traditionally been used to emphasize nonmarket (and to a lesser extent market) values in land management; however, such regulations often stifle economic growth and may be inappropriate to local conditions (Lippke and Oliver 1993). Given this fact, economic incentives may be required if nonmarket ecosystem values are to be maximized on both public and private lands (Lippke and Oliver 1993). New technologies (e.g., geographic information systems, remote sensing, and harvesting systems) must also be used in ecosystem management efforts (Oliver et al. 1993).

## CONCLUSION

Ecosystem management represents a new approach by the Forest Service to implement the intent of the Multiple-Use, Sustained-Yield Act through its stewardship of National Forest System lands. To successfully use this approach, land managers should recognize basic principles of land evaluation as described by Zonneveld (1988). Optimum land-use planning (i.e., ecosystem management) is achieved through synthesis of sociological, ecological, technological, and economic information. The conceptual framework for ecosystem management presented in this paper is based on land evaluation techniques and is useful for developing strategies for ecosystem management. Refinement of such strategies is best achieved through an adaptive management approach (Everett et al. 1993, Walters and Holling 1990).

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# Evolution of Forest Service Organizational Culture and Adaptation Issues in Embracing Ecosystem Management

**J.J. Kennedy and T.M. Quigley**

## ABSTRACT

This paper examines how the USDA Forest Service did and did not adapt to the needs of American society in its industrial (1900-1960) and postindustrial (1961-present) stages of socioeconomic development. Several marker events in Forest Service adaptation to a postindustrial American society are examined (e.g., Bitterroot controversy, advent of unit planning). These events illustrate cultural changes that have moved the agency toward is current ecosystem management era of organizational evolution. Shifts in agency values, policies, structures, and operation necessary to embrace and implement ecosystem management are examined and applied to current forest health issues in eastern Oregon and Washington.

## INTRODUCTION

By most power, size, and budget measures, the USDA Forest Service has been a very successful organization (Clarke and McCool 1985, Gold 1982). This was especially true for the first two-thirds of this century, when the agency's conservation era mission and management style were so compatible with an urbanizing, industrial nation immersed in three major wars and a great depression (Gulick 1951, Hays 1959, Kaufman 1960). The National Environmental Policy Act of 1969, however, signaled the arrival of the environmental movement and more democratic public involvement. This proud and nationally respected agency, that in the 1960s starred in the prime-time TV series *Lassie*, was about to write its own chapter of *Future Shock* (Toffler 1970) in adapting to the values of an urban, postindustrial America (Kennedy 1985, 1988). Despite is bureaucratic inertia and traditional client loyalties, external sociopolitical change (Kennedy 1985) and agency diversification (Kennedy 1991) has moved the Forest Service toward is current era of ecosystem management, announced by Overbay (1992). Our paper describes several stages of U.S. and Forest Service sociopolitical evolution in this century, examines how Forest Service culture is and is not positioned to embrace and implement ecosystem management values, and suggests several organizational changes necessary to implement this new ecosystem management paradigm.

### Developmental Stages of Forest Service Organizational Culture

Any attempt to characterize 100 years of Forest Service history in three to four pages will frequently overgeneralize. Nevertheless we will attempt an overview of U.S. socioeconomic change and Forest Service organizational evolution in three stages, summarizing with a 1950s versus 1990s comparison of internal and external Forest Service changes.

### Stage I. Forest Service Birth and Establishment (1880-1909)

In the last two decades of the 19th century, the American frontier closed and the United States became an urbanizing, industrial nation. Amid growing political concerns about forest fires, flooding, wood scarcity and long-term social risks of the free enterprise system, an organization emerged that was to become the Forest Service in 1905. The National Forest System was our nation's biggest experiment with socialism at the time. This system was designed to be a multivalue, socially oriented (vs. profit-oriented), natural resource trust fund for the nation and for future generations (USDA Forest Service 1907). National Forests were to be an insurance policy or alternative to free enterprise values and methods of forest management. In this initial Gifford Pinchot-era, the agency adopted the forest and organizational models, values, and management styles of the German Forest Service (Twight 1983, 1985). The agency's regimented campaign against the evils of forest fires, short-run greed, and natural resource exploitation, plus its promises of long-term sustainable commodity flows for an emerging industrial state, gave it noble purpose and broad sociopolitical appeal (Hays 1959, Steen 1976). The Forest Service lead the American people and politicians into the conservation era. It was a lean, righteous, radical, organization confronting frontier era and laissez-faire natural resource values that were no longer appropriate for a modem, industrializing America.

## Stage II. Forest Service Adolescence and Young Adulthood (1910-69)

The United States became an urban industrial state with a great depression and three major wars which provided a clarity of purpose for the nation and its Forest Service--a national clarity of purpose that would evaporate in the 1970s. The Forest Service and is employees were seen as cleancut heroes, fighting forest fires and natural resource ignorance or exploitation (Frome 1962). The organization changed from being a rebel against the system to becoming a proud and powerful part of that system. The Forest Service was no longer a small fraternity of dispersed and independent tough guys, but an elite (and often aristocratic) professional forestry bureaucracy that responded harshly to outside criticism (e.g., Reich 1962).

Just as the United States was becoming an urban, postindustrial nation (with emerging environmental values), powerful and legitimate socioeconomic forces were to lock the Forest Service on an industrial trajectory for the next 2 to 3 decades. In the 1950s, post-World War II economic demands had liquidated enough private softwood supplies to create a social need for Forest Service timber. Additionally, there was legitimate sociopolitical desire to provide the housing part of the American dream for GIs and for a rapidly expanding population. Consistent with its German intensive forest management values and desire to contribute to rural growth and national prosperity (Clary 1986, McGee 1910, Twight 1983), the Forest Service shifted from forest protection and custodial management to becoming a major provider of softwood timber supply. Its timber harvest jumped almost 800 percent between 1941 and 1971--from 1.5 to 11.5 billion board-feet per year (Steen 1976:314). This shift of western National Forests from a resource trust fund to a regional employment lunch bucket (within sustained yield and multiple use limits, of course) had its organizational rewards. For example, the agency budgets and work force grew rapidly (a 40 percent increase in employees between 1958 and 1963; Aiken et al. 1982). But this new organizational wood production priority (Alston 1972) put the agency on a collision course with the environmental values of a postindustrial and postmodern American society (McQuillan 1992). In those heady developmental years of the 1950-60s, the Forest Service, the administrations that guided it, and Congress would often forget the original National Forest role for American society and generations yet to be born. Rather than functioning as a trust fund and counterbalance to private forest and grasslands management, the Forest Service often looked and behaved like an echo of private management (such as Weyerhauser Corporation).

## Stage Ill. Mid-Life Crisis and Struggle for a New Forest Service Maturity (1970-Present)

The public shift toward valuing National Forests for recreation, wildlife, and landscape values was first indicated in the Wilderness Act of 1964. The legislative last hurrah for intensive timber management of National Forests came in that period as well, when Congress rejected a proposed National Timber Supply Act. This law would have allowed the Forest Service to keep much of its timber sale receipts to invest in more intensive, scientific timber management.

In 1969, the National Environmental Policy Act (NEPA) marked the beginning of the environmental era by requiring development alternatives and justification of Federal actions that would substantially affect the environment. These alternatives were to be analyzed by an interdisciplinary team of professionals and the process was to include public participation. Other environmental legislation (e.g., Endangered Species Act of 1973), plus affirmative action and equal employment opportunity policies also affected the Forest Service in this era.

In the next 2 decades, the Forest Service went through stages of denial, confusion, and mourning for the good old days when it was an elite forester fraternity with a clear purpose and a national mystique (well described by Kaufman 1960 and Frome 1962). It also received mixed messages during this time from conservative administrations, commodity-oriented budgets, a postindustrial American society with growing environmental demands, and from its own employees. Despite these mixed messages, the agency has moved unevenly but inexorably toward environmental values and a new maturity signified in the ecosystem management paradigm (Overbay 1992)--an organizational evolution well documented by traditional client groups, such as the National Forest Products Association (Gladics 1991). These agency changes are highlighted in the next section.

## Forest Service Paradigm Shifts: A 1950 Versus 1990 Snapshot

A comparison of events, values, and Forest Service management paradigms in the 1950s and 1990S is presented in table 1. This table contrasts today's complex world and the 1950-60s era of Forest Service "manifest destiny" --when western National Forests were to be transformed from an inaccessible, extensively managed, native forests into the triumph of the conservation era (a roaded, intensively used and managed multiple-use forest estate that would approach the vision of an original Forest Service prophet, W. J. McGee (1910). Underlying these 1950s conservation era values, images, and metaphors was a fascination with the machine-model that influenced National Forest management, the agency's functional organization, and employee administration (table 1, element E). This simpler, machine-model view of reality is challenged today by the diverse, complex organic-models of Work Force 1995 (USDA Forest Service 1987), NEPA processes, and the new ecosystem management paradigm.

It is not surprising that a young industrialized nation and its Forest Service would be fascinated with a machine-model view of reality. This model sees the world in simple, compartmentalized, cause-effect, goal-oriented, and mechanistic terms that can be understood separately by standard efficiency or optimization analysis (Schiff 1966, Taylor 1957). Such Forest Service machine-model thinking was manifested in: (1) narrow forest ecosystem perceptions (e.g., simple site-productivity models), (2) forest or fire management (e.g., intensively managed plantations, forest pest wars, or out before 10 AM fire rules), (3) agency organizational structures (e.g., line-staff, generalist-specialists, or strict functionalism), (4) organizational processes (e.g., Kennedy and Thomas' 1992 "dog" loyalty to line, mechanistic employee-spouse-children response to Forest Service transfers), (5) public relations (e.g., benign and educated Forest Service professionals managing National Forests for the uninformed, self-centered public and for future generations), and (6) functional, reductionist research scientists and their projects. Control-oriented people and organizations (table 1, element F) find comfort in a machine-model world (Schiff 1966).

Ironically, complex postindustrial societies, created by the simpler industrial eras of the first two-thirds of this century, have made machine-model thinking obsolete (table 1, element A). More complex, diverse, and interrelated organic-models are necessary to understand and adapt to today's world. This is true for public and private organizations, including machine-model institutions in eastern Europe, and corporations such as Chrysler, Sears, and IBM, which have not adapted well to the changes and challenges of our modern world (Bennis 1966). This sociopolitical change also requires National Forest managers to discard simple machine-models of reality for organic-models such as ecosystem management, and to resurrect Forest Service employees Leopold and Marshall as respected role models along with Pinchot (Robertson 1991). Given that many of us were taught to become rational, knowledgeable adults, in control of our internal and external world, this invitation to an organic-model of reality does not come without challenge, threat, and uncertainty (Magill 1988, Twight and Lynden 1989).

Refusal to make such an organic-model adaptation has a price. For example, the New Zealand Forest Service, so successful with the machine-model of conifer plantation silviculture until the 1970s, found it difficult to accept organic-models that would have helped them adapt to the environmental values of an urban, postindustrial New Zealand society (Clawson 1988, Kennedy 1981). This inflexibility contributed to over 75 percent of New Zealand Forest Service lands (most of its native forests) being removed from Forest Service stewardship in the mid-1980s. New Zealand society judged a machine-minded agency trustworthy to manage only their machine-model conifer plantations. As with Chrysler Corporation or the Russian Communist Party, U.S. and New Zealand Forest Services must escape the anchors of their history and reinvent a future relevant to a changing sociopolitical environment--or die.

### Forest Service Adaptation to Some Wake-Up Calls in the Last 25 Years--<br>Prologue to Ecosystem Management

For most of its history, the Forest Service was internally directed and a leader in the conservation movement (1890-1970). The agency required external sociopolitical pressure to adapt from is conservation and resource development values to those associated with the environmental movement (e.g., wilderness, biodiversity, or outdoor recreation values). Several social or legislative wake-up calls (and cross-body blocks) are highlighted below, with implications for ecosystem management and eastside Oregon and Washington forest health issues.

**Table 1--Forest Service (FS) and National Forest (NF) Environment, Images, Values, and Management Paradigm Shifts: 1950 Versus 1990 Snapshots.**

| | Contrasting Decades | |
|---|---|---|
| Elements | 1950s | 1990s |
| A) Stages of U.S. socioeconomic development | Triumph of the U.S. industrial state in post-WWII euphoria | Urban, postindustrial state in a competitive global economy and complex, sobering times |
| B) FS mottos | "Land of Many Uses"--but timber usually major use and ROTT (Resources Other Than Timber) a secondary consideration or constraint | "Caring for the Land and Serving People" (USDA 1986)<br><br>Shift from ROTT to legitimate multiple use and EM management |
| | Sustainable goods and service flows (system output focus) | Sustainable ecosystems (focus on health and uniqueness of the ecosystem itself) |
| | Heavy road development for user access and fire control | Road and infrastructure developmental era waning |
| | Intensive timber management focus | Shift from single resource focus to multiple social values (Kennedy 1985). |
| | Fascination with technology (machines, chemicals, genetics) | Questioning dominance of technology in management innovation and efficiency |
| C) Respected NF management models | Adopted European intensive forest management paradigm | Search for own, new, unique NF management paradigms and identities |
| | Primary focus on maximum output effficiency, within sustainability and multiple-use constraints | Focus first on healthy, diverse, sustainable ecosystems, then estimate output possibilities |
| | A multiple-use conifer plantation often the vision for many NFs | Desired future NF conditions that balance and complement public and private lands in an ecoregion context |
| D) Respected NF managers and FS role models | Era of great men and benign professional aristocrats | Era of interdisciplinary teams; team leader, public and partner facilitators |
| | John Wayne action and achievement-oriented, omnipotent forester (Behan 1966) | Specialized expert, capable of inter-disciplinary or public communication and power sharing |
| E) Dominant models and metaphors | Dominance of the simple, compartmentalized, machine-model on land and in organization | More complex, inclusive, inter-related organic-model of ecosystems and organization |
| | Simple, homogeneous, well-managed forest stands and hard working loyal "Forest Rangers" (e.g., Kaufman 1960) | Respect for diversity and uniqueness of land, individual employees, user groups or partners |
| | Fascination with machine-model plantations, road networks, developed campgrounds | Birth of new perspectives era and evolution to ecosystem management (Kessler et al. 1992) |
| F) Dominant FS values | Action, can-do, development-oriented mythic heros | Can't do and shouldn't do many things--let's think, plan, seek consensus |
| | Must dominate and control forests, self (especially emotional self), family (e.g., transfer-at-will), and the public (educate them if they don't support us) | Codependence and mutuality with nature (Rolston and Coufal 1991), expanded image of self and family (multifaceted lives, dual careers), and the public (public servant and partnership era, Magill 1988) |
| | "Dog" loyalty to line and the agency (Kennedy and Thomas 1992) | FS organizational loyalty counterbalanced with loyalty to land, to profession, to working spouses, and so on. |
| G) Space focus | Focus on forest stands (or the research project) | Broad, inclusive landscape- and ecosystem-scale focus |
| | Local and regional focus | Regional-national-global thinking |
| H) Time focus | Annual reports of specific target accomplishments | Movement toward achieving long-term desired future conditions |
| | Short-run economic and project efficiency focus | Decadal focus needed; annual target myopia questioned |
| I) Land, labor, and capital conditions | Public land per U.S. and global population more abundant and less developed | Public land per U.S. or global population more scarce and more developed |
| | Abundant capital from old-growth forests and a deficit-naive Congress | Capital scarcity in second-growth and multiple use-oriented forests, plus a deficit-burdened society |
| J) User fee | Low, often subsidized, fees restricted to a few users (e.g., ranchers and some recreationists) | Likely to rise for traditional users and expand to others |
| K) Patron Saints | Gifford Pinchot, FS employee: 1898-1910 | Aldo Leopold, FS employee: 1909-1928 |
| | St. George the Dragon-Killer | St. Francis of Assisi |

## Case 1. Oops, We're Not as Technically Competent and Omnipotent as We Once Thought: Monogahela and Bitterroot National Forests Controversies

In the Kaufman (1960) era, Forest Service professionals (that is, foresters) were educated in universities and socialized in the agency to view themselves as stewards of the public interest and broadly enough trained to manage all National Forest issues (e.g., Behan's 1966 "omnipotent forester"). Conventional wisdom held that all professional foresters were competent silviculturalists. The Bitterroot (U.S. Senate 1970) and Monogahela (Fairfax and Achterman 1977) controversies provided several lessons for the Forest Service, including the need for sensitivity to diverse recreational, wildlife, or landscape values and to new public demands for shared decisions. The Forest Service also learned that all foresters were not, ipso facto, competent silviculturalists. Although the dark side of the Forest Service "can-do" culture helped cause these two controversies, the bright side of that "can-do" attitude allowed the agency to quickly initiate a graduate-level training program to position and empower certified silvicufturalists to guide, approve, and monitor silvicultural practices on most National Forests.

The cumulative factors that have created eastside Oregon and Washington forest health issues (e.g., climate, fire management, and silvicuftural practices) are less simple and obvious, and require a more complex wake-up call than the Monogahela and Bitterroot issues. Similar agency responses, however, are required. Specifically, the Forest Service now needs to: (1) consider functional budgets and specialist roles in an organic-model context; (2) recognize that ecosystems are composed of structures and processes that do not stop at public or private ownership boundaries; (3) accept that many line and staff specialists might not initially have the understanding and vision to adequately plan, manage, and monitor more demanding and sophisticated ecosystem management organic-models; and (4) consider that "certified ecosystem managers" might be needed to direct and monitor landscape-scale ecosystems, cumulative effects, and progress toward the establishment of more stable, healthy, desired future ecosystem conditions.

## Case 2. Okay NEPA (1969) Includes Us, So Let's Settle All National Forest Conflicts at the Planning Stage.

In 1971, the Forest Service embarked on a new "unit planning" program to identify and resolve natural resource issues at the planning stage (versus on-the-ground management stages). In many ways, it was an initial Forest Service organic-model adaptation to an increasingly diverse and complex world. Planning units were set by landscape ecology criteria and often crossed over district, National Forest and political boundaries. Planning units also were referenced to larger National Forest and Regional policies, and to socioeconomic and ecological conditions of adjacent public and private forests. For example, the planning units for the Ouachita National Forest, Arkansas, were directed by national (USDA Forest Service 1970) and regional goals. The Guide for Managing the National Forests in the Ozark Highlands (USDA Forest Service 1974) was an advanced and enlightened document that placed the Ozark and Ouachita National Forests in a large regional ecological and sociopolitical context. It proposed guidelines emphasizing "vegetative diversity" (p. 38), favoring hardwoods over softwoods on appropriate sites (because adjacent private lands had a bias to pine plantations, p. 39), promoting undeveloped recreation and depending on adjacent private lands to provide more developed opportunities. If the Ouachita National Forest had followed this direction and not been driven by other targets, it might have avoided much of the legal and Congressional conflict it experienced in the last 15 years. It would have been more of a forest social value alternative and less of an echo to private forest land management surrounding it.

As with ecosystem management, unit planning in the early 1970s focused first on analyzing the sustainable capabilities of landscape-scale ecosystems. It provided a bottom-up estimate of National Forest output capabilities secondarily. Concerns with initial unit plan output declines, the many plans involved, and new legislation (e.g., the Forest and Rangeland Renewable Act of 1974) largely reversed this bottom-up, ecosystem management-type planning process. Output targets regained leverage and mechanistic, optimization machine-models (e.g., FORPLAN) drove the new forest planning paradigm. Current regional ecosystem issues (such as eastside Oregon and Washington forest health), court discussions, regional studies of spotted owls or salmon, recent Congressional Forest Service studies (Office of Technological Assessment 1990 and 1992), and the advent of ecosystem management may now send the agency "back to the future" to incorporate more bottom-up, landscape-scale, ecosystem (unit) planning philosophy and methods. For the planning process is the most essential, pivotal entry point for ecosystem management values and methods to impact Forest Service employees and lands.

## Case 3. Forest Service Soul Searching at Snowbird (1985) and Sunbird (1989)

In 1985, Forest Supervisors, Regional Foresters, Chief and Deputies all met for the first time at Snowbird Ski Resort, Utah. A major goal of this meeting was to develop a new Forest Service vision statement. The result, Caring for the Land and Serving People (USDA Forest Service 1986), incorporated verbs (e.g., “caring” vs. the more traditional, clinical and macho “management”) and concepts (e.g., diversity) more consistent with current American social values. The ecosystem management-type goals of this vision statement are more difficult to quantify and target than the “Land of Many Uses” values of earlier production and development-oriented goal statements (e.g., USDA Forest Service 1970).

At the second (Sunbird) meeting of Forest Supervisors, Regional Foresters, Chief and Deputies (Tucson, AZ 1989), Kennedy and Quigley (1989) examined whether top line officers at that conference and recent professional recruits endorse these caring-serving values and if the Forest Service reward system supports the agency vision statement. This, and an expanded followup study (Kennedy et al. 1992), found that employees from top line officers to recent agency recruits believe that professional competence, care and concern for healthy ecosystems, and care and concern for future generations should be the most rewarded Forest Service values. Most believed, however, that the following standard bureaucratic values were actually the most rewarded by their agency: (1) loyality to the Forest Service, (2) meeting targets, (3) promoting a good Forest Service image, (4) following rules and regulations, and (5) working well in teams. The people questioned in these Forest Service samples and those surveyed by Quigley (1989) generally believed the Forest Service valued timber and grazing (versus recreation, water, or wildlife) more than Forest Service employees or the general public, a trend the new RPA Strategic Plan (USDA Forest Service 1990) seeks to reverse. The Sunbird survey and an open letter to the Chief from several Forest Supervisors indicated broad internal concern that the Forest Service become more true to its stated caring and serving mission.

## CONCLUSIONS: A FOREST SERVICE ORGANIZATION TO EMBRACE THE LETTER AND THE SPIRIT Of ECOSYSTEM MANAGEMENT

Overbay (1992) sets forth comprehensive, diverse, organic-model direction for ecosystem management of eastside Oregon and Washington and all National Forests or Grasslands. It is a management paradigm worthy of: (1) a similarly organic-model “Caring for the Land and Serving People” vision statement (USDA Forest Service 1986), (2) a more democratic organizational culture envisioned in the new Forest Service management charter (USDA Forest Service manual, title 1300, 1992), (3) the innovation in many regional initiatives (e.g., USDA Forest Service 1985), and (4) much new Forest Service ecological thinking (e.g., Botkin 1990, Diaz and Apostol 1992). Given the sociopolitical change in American society, plus the promises and expectations of the Forest Service New Perspectives Movement (Kessler et al. 1992), the agency had no choice but to proclaim ecosystem management as its new management paradigm. It also has no choice but to embrace and implement its spirit--for Overbay (1992) spoke what was in the hearts of many Americans inside and outside the Forest Service.

Attempts to achieve and reward organic-model ecosystem management goals with machine-model planning systems, traditional organizational structures, current targeted budgeting, or the existing reward system are likely destined for frustration and failure. Overnight, revolutionary change cannot be expected in the organizational culture of such a large bureaucracy as the Forest Service. But a clear strategy for organizational change and movement in that direction can maintain public and employee faith in the organization as it attempts to respond to the diversity and complexity in National Forests and Grasslands, in American society, and in its own work force. We offer some initial direction for such Forest Service cultural evolution.

National Forests and Grasslands, the public, and Forest Service employees have,always been complex and diverse. Viewing them in a machine-model context resulted from deficiencies in knowledge and sensitivities of the past, and unintentionally contributed to issues such as eastside Oregon and Washington forest health. Ecosystem management represents an organic-model maturity that honors complex, diverse, evolving, and interrelated ecosystems and is consistent with the democratic and land values of an urban, postindustrial American society. Our changes and additions to Overbay's (1992) ecosystem management proclamations in table 2 expand his organic-model thinking from its current ecosystem focus to incorporate the equally complex, diverse, and interrelated nature of Forest Service employees (expanding it to incorporate U.S. and international publics and future generations was not addressed in as much detain. National Forest ecosystems, Forest Service employees, and the public deserve better than machine-model values and action. By adding 20 percent more words [in brackets] to Overbay's (1992) critical concepts of managing land in the spirit of ecosystem management (table 2), we have expanded these ecosystem management concepts to include Forest Service employees and the public.

**Table 2--An Organic Model "To Care For The Land" (As Quoted from Overbay 1992) And "Care For Forest Service Employees" [As Added by Kennedy and Quigley, In Brackets].[1]**

| | |
|---|---|
| The Ecosystem Management Charge (Overbay 1992:1)[1] | It is time to embrace the concept of managing ecosystems to sustain both their diversity and productivity and to chart a course for making this concept [,and other organic-model versus machine-model concepts,][2] the foundation for sound multiple-use sustained yield management [of Forest Service land and its employees.][2] |
| Ecosystem Management Frameworks and World Views (Overbay 1992:2) | An ecosystem [and the Forest Service organization] is a community of organisms and its environment that function as an integrated unit. Ecosystems [and the Forest Service organizational structure] occur [and function] at many different scales, from micro site [or work unit][2] to the biosphere [and regional, national or international levels]. |
| Principles for An Ecosystem Management Approach to Land [and Forest Service People] (Overbay 1992:1-5) | Diversity and Sustainability: Multiple - [value], sustained-yield management of lands and resources [,and Forest Service people,] depends on sustaining diversity and productivity of ecosystems [, and employee teams,] at multiple geographic [and organizational] scales.<br><br>We need to understand natural events and the effects of humans in our management [of the land, Forest Service people and public use].<br><br>The best [organic] model for operating [with the land, Forest Service people or the public] in this fashion is called adaptive management. It means that research and monitoring [of ecosystems, our employees and the public] will play substantial roles in ecosystem management [approaches to understanding ourselves, as an agency, the people we serve, and the land we are privileged to manage.]<br><br>Desired Future Conditions: Descriptions of desired future conditions for ecosystems [and a Forest Service organizational culture] at various geographic scales should integrate ecological, economic, and social [-psychological] considerations that can guide management activities.<br><br>Integrated Management and Research: Monitoring and research should be integrated with [our land and human] management to continually improve the scientific basis of ecosystems, [our work force, or general people] management.<br><br>A bias for diversity is a good watchword for ecosystem [, Forest Service work force or user] management [-- and for democratic principles applied everywhere in our nation].<br><br>But ecosystem [and other organic-models of] management by itself [themselves] will not improve our performance unless we follow through on the new understandings we gain in how ecological [,work force and user] systems function. It cannot be just another name for the same old priorities and operating principles [on the land, in the office, with the public, or with future generations who depend on our values and actions.]<br><br>Whatever the goals, the benefits of ecosystem [and other organic-models of] management come from the spirit of [diversity and] integration. |

[1]All statements are quoted from Overbay (1992) and identified by page number.
Our additions to expand Overbay's organic-model ecosystem management paradigm to a comparable organic-model of Forest Service employees and agency structure/processes are in brackets.

A broader, more diverse team is required to develop a strategy for changes in Forest Service organization and operation to accommodate the scientific and technical ecosystem management suggestions in many eastside forest health papers of this report, in Overbuy (1992), and in other “new perspectives” and ecosystem management-type innovations throughout the agency (e.g., Diaz and Apostal 1992). We will highlight only a few strategies, some of which might be applied in an experimental basis on eastside Oregon and Washington National Forests and Grasslands. The initial suggestions which follow are those which require immediate attention:

1. Employee Classification and Training. Develop interdisciplinary classifications and training that transcend traditional range, recreation, or hydrology functional boundaries.

    a. Ensure that before any specialized training is undertaken by Forest Service employees on specific ecosystem functions or output endowments (e.g., fisheries; soils, or range), a series of general courses should be taken that address socioeconomic, planning and management, and ecosystems in a broad, integrated ecosystem management manner. Advanced training in certain ecosystems (e.g., stream ecology) or output and user delivery and management systems (e.g., recreation, fisheries, or range output services) could then be offered.

    b. Develop ecosystem management certification with the rigor, respect, and responsibility of the Forest Service certified sitvicufturalist program.

2. Planning Systems. Shift RPA and forest (the current “unit”) planning from its output-driven focus (within sustained-yield constraints) to a desired sustainable ecosystem model that secondarily estimates output endowments.

    a. Design National Forest planning units on landscape ecosystem criteria and reference such units to political and administrative parameters and trends.

    b. Develop desired future conditions for landscape-scale ecosystems as the initial, pivotal planning activity.

    c. Estimate National Forest output capability from a bottom-up approach, centered at the District and Forest levels. Estimation should be based on ecological and socioeconomic analysis of sustainable desired-future-conditions determined at the National Forest or Grasslands level.

3. Budgeting. Reconsider the machine-model, output-targeted budget system (of the “Land of Many Uses” era) for organic-model budgeting that enhances ecosystem management values and goals.

    a. Shift from a line-item budget to a more end-state system (Office of Technological Assesssment 1992:8) of achieving and maintaining desired conditions of ecosystems, user systems, or output systems.

    b. Increase sensitivity of budgets and accountability to the decadal timeframe of ecosystem adaptation and change.

    c. Allow a small percentage of budgets (say 10 percent) to be used for innovative, experimental options (fully documented), without traditional sanctions for failure to efficiently achieve stated objectives.

4. Reward System. Shift from a machine-model, output-oriented reward system to one that creates and enhances diverse, adaptable, and sustainable ecosystems, organizational cultures, and output and user services. Such an organic-model reward system in an ecosystem management era would accommodate risk-taking, entrepreneurship, and team processes and would pivot on the core-value of enhancing diverse, sustainable ecosystems, user systems, and organizational cultures. For example, the organic model would reward movement toward desirable future conditions as well as output endowments along the way.

5. Forest Service Organizational Structure. Legislative trends (Multiple-Use, Sustained-Yield Act of 1960, National Environmental Policy Act of 1969, National Forest Management Act of 1976) and court decisions (e.g., Craig 1987) have been a consistent invitation to dilute the intellectual dominance and budget myopia of Forest Service functionalism. Ecosystem management is another call to reconsider the machine-model, output-focused functionalism for a Forest Service organized around the soul and substance of the agency, namely organizational divisions for terrestrial and aquatic ecosystem(s). Then consider divisions for output services (e.g., timber, range, mining, or water), user services (e.g., recreation and education), and adminis trative services. Even with this administrative restructuring, appropriate ecosystem management adaptation is not assured, for budget forces seem to have dominated planning (Office of Technological Assessment 1992:12) or legislative decisions (Alston 1972) in the last decades.

The changes proposed are founded on the need for the Forest Service to evolve from an output- and target-focused agency (within long-run productivity constraints) to an organization that enhances diverse, sustainable forest and grassland ecosystems (for regional-scale biological and socioeconomic balance), and apply similar ecosystem management awareness, sensitivity, and skills to its own employees and to the public.

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# Ecosystem Management in the Forest Service: Political Implications, Impediments, and Imperatives

**W.B. Shepard**

## ABSTRACT

For more than half a century, "old forestry" was a highly successful blend of three elements: forest management as applied science, timber primacy, and decentralization. In a changing country, all three of these political assets became major political liabilities. The traditions of old forestry left the. USDA Forest Service ill-equipped to deal with both the harvest controversies of the 1970s and implementation of the National Forest Management Act of 1976. Ecosystem management, if it is to succeed, must involve more than better applied science; it must embrace the political responsibilities of the land manager. Major stresses and breakdowns in the encompassing national political framework must be recognized and accommodated.

## INTRODUCTION

Ecosystem management joins a lengthening list of terms that have gained currency in the discussion of forest management. Public attention was captured several years ago by a grab bag of research findings and hypotheses labeled "new forestry." We have since heard of "new perspectives," and "adaptive forest management," among other new terms. This litany should not suggest, however, a haphazard or frenetic attempt to find management answers or a public relations cover; ecosystem management should not be viewed as the latest flavor-of-the-month. Rather, these terms capture a clear evolution of thinking and on-the-ground management as the silvicufturaf implications of 20 years of scientific findings are joined with emerging appreciation of changed sociological, political, and economic circumstances.

Ecosystem management will evolve as challenges are confronted and as understandings improve. Kennedy and Quigley (1993), identify the forces of organizational culture that will shape--indeed, that must be allowed to shape--the further development of ecosystem management in the USDA Forest Service. I will develop the political considerations that must be embraced if the hands-on, day-to-day, on-the-land, and among the-people application of ecosystem management is to be effective. Like Kennedy and Quigley (1993), I will first look back in order to look forward.

Many of us think of politics as a dirty word. With that observation, my central proposition may strike some readers as odd. My thesis is that, for forest management to succeed, the political aspects of forest management must be renewed and reinvigorated. The problem has not been too much politics, it has been too little politics.

So, how did we get where we are?

## Old Forestry

Controversy in the management of our country's forests is nothing new. The origins of the National Forest System can be directly traced to two radical social movements that coincided at the start of the 20th century: one was the reaction to the dominance of government by business interests that came to be known as the progressive movement; the other was the conservation movement: a reaction to the wide-scale, unregulated exploitation and destruction of increasingly valued natural resources. In addition to the collision of the progressive movement and the conservation movement with the status quo, there also was a raging division between the conservationists and the preservationists (McQuillan 1990).

When an assassin's bullet put Theodore Roosevelt in the White House, the victory of the conservationists over the preservationists was assured (Nash 1967), and Gifford Pinchot, the first Chief of the Forest Service, had the political security fie needed to address the social and political turmoil surrounding the unregulated use of American forests in the early 1900s. He developed a brilliant political solution that had three elements: scientific management, timber primacy, and decentralization.

Conservationism had, as a cornerstone, the belief that scientific management could be substituted for politics (Hays 1959), and there were attempts to apply this belief in many areas, for example, to the management of cities. The attempts generally failed in areas outside forestry (Banfield and Wilson 1963). But, with the political triumph of the conservationists over the preservationists, there was clear agreement on what the benefits should be of managing the National Forests: an assured supply of water and timber. Agreement on ends provided the consensus Pinchot needed to define forest management as the application of the emerging science of forestry: problems were subjected to technical solutions and, where solutions failed, one looked for more data, deeper understandings, better machinery, and improved forest practices.

The emphasis on scientific forestry imbued the Forest Service with an image of technical competence and credibility that, for many decades, could be used to increase budgets and protect the agency from outside political threats. The profession of forestry evolved along with the Forest Service (West 1992), and the availability of a pool of employees with common values and training added greatly to the internal strength of what some have referred to as a “paramilitary organization” (Gulick 1951, Kaufman 1960, Schiff 1966, Wondolleck 1988).

Scientific management was coupled with what has been called “the rule of timber primacy” (Clary 1986). Trained in biological relations rather than economic principles of supply, demand, and substitution, American foresters had been predicting for the last 80 years that a timber famine was only a decade-or so away (Bennett 1968). It is easy to see, with such a creed, why timber management has been a priority. For decades, the emphasis on timber was also a political asset: furnishing goods and services to people at below cost is an old recipe for maintaining political good will, for stimulating bureaucratic growth, and for doing the professionally “right thing” for well-managed forests.

Pinchot introduced decentralization in the organization of the Forest Service to a degree unusual for a Federal bureaucracy. Decentralization served the agency well during a period when the politically significant users of National Forests lived in proximity to the forests, and when most users probably knew the forest ranger in their area on a first name basis. For much of its existence, this sensitivity to largely rural interests allowed the Forest Service to adapt to local changes. An effective presence in many congressional districts also did not hurt when budget-time rolled around. In fact, the Forest Service is one of those agencies to which Congress wanted to give more money than the agency, under Presidential orders, was allowed to request.

While timber primacy, scientific management, and decentralization persisted, the country-and the world-was changing (Costain 1992, Inglehart 1990). The first Earth Day occurred about 20 years ago. It is no coincidence that the modem problems of forest management exploded onto the political agenda in the years immediately after that first Earth Day. An agency emphasizing the production of wood fiber was not in tune with a growing environmental movement. An agency with its ear to the ground in 156 National Forests and many more local Ranger Districts found its head was resting in the path of a truck--or, perhaps, a Volvo--with an urban vehicle registration. And, an agency that defined forest management in terms of science and technology lacked weapons when it needed to figure out what to use the forests for.

The first salvos resulted from harvest controversies, and the shots were fired by good ol’ boy turkey hunters rather than condo-dwelling members of environmental organizations. The controversies arose because local constituencies felt that the Forest Service, with its belief in “scientific management” and “timber primacy,’ was not listening to “unprofessional” local views on how the forests should be managed. These were the harvest controversies at the Monongahela National Forest in West Virginia and the Bitterroot National Forest in Montana. Both conflicts provoked years of state and congressional investigations, and after a lawsuit used the Organic Act of 1897 to successfully challenge clearcuttirig in the Monogahela National Forest, Congress was forced to act (Weitzman 1977).

## Planning: The Solution Becomes The Problem

Congress responded to the Bitterroot and Monongahela controversies with the National Forest Management Act of 1976. As is typical in American politics, the National Forest Management Act (NFMA) was put together by the major affected interest groups, including the Forest Service. The NFMA required a planning process for the National Forests that included interdisciplinary teams, economic analysis, and citizen participation. The act papered-over the harvest practice questions that led to the Monongahela and Bitterroot controversies but did delete the troublesome language in the Organic Act that had threatened to halt clearcutting in the National Forests.

With the passage of NFMA, the Forest Service, and we who use and own the nation's forests, entered a lengthy period of struggling with forest planning. That struggle amply illustrates the limits of old forestry. The NFMA required a difference in the way the Forest Service did business. The problem was not glitches in the tools of scientific forestry; the problem was reliance on scientific forestry itself. The questions the Forest Service increasingly faced in the 1970s were how the forests should be managed and for whom the forests should be managed. Answers to those questions come not from science but from values and interests.

Put simply, those responsible for the management of our National Forests in 1976 were faced with a public that had two concerns: clearcuts were ugly, and tree plantations were not forests. In 1976, it was as simple as that; NFMA was asking land managers to address some basic political concerns.

Forest management as applied science, however, had worked political wonders for the Forest Service for over 60 years, and that is how the agency approached implementation of NFMA. A committee of scientists (created by the NFMA legislation) offered recommendations that turned into planning regulations, Forest Service Handbook materials, and Forest Service Manual chapters that boggle the mind in their detail and complexity. What had been a simple political warning from the U.S. Congress to do a better job of listening to people was fumed into a nightmare. Millions of dollars-the last estimate that i have seen was two billion dollars were spent on planning (Behan 1990); an agency that rightly prided itself on 'getting the cut our missed its initial planning deadlines by a decade; and as the plans finally hit the street, the street fumed out to run straight to the courthouse.

The Forest Service was facing questions about what ought to be. Some of the questions were fairly commonplace, pragmatic considerations of who is going to win and who is going to toss. Some of the questions tapped deeply held moral concerns. The Forest Service was teaming that, in addition to sterile matters like board feet, recreation visitor days, animal unit months, and acre feet of water, they were, by their actions, giving or withholding public recognition to particular moral positions on the question of how humans relate to their planet. In this respect, the Forest Service was joining the company of school teachers, public health workers, and art museum directors.

The early and persistent dedication to the use of linear programming models as planning toots epitomizes the inadequacy of old forestry. Adapting tools that had been appropriate in earlier years when there was consensus on timber primacy, the Forest Service turned harvest scheduling models into forest planning tools. Year after year, as it became clear that these computer models were not doing the job, this approach was not abandoned. Rather, fancier models were developed, matrices were expanded, additional programming modules were added, and larger and faster computers were sought. FORPLAN was the primary computer model used in the planning process, and a review of that effort concluded: "it appears that undue reliance has been put on the optimizing feature of FORPLAN without seriously pondering the more important question of 'what should I optimize?'" (Alston and Iverson 1987).

So, how did we end up in the current situation? Old forestry failed because what had been political assets became, in a changing country, major political liabilities. The emphasis on timber did not mesh with the interests of emerging groups. Decentralization created a rural bias in an increasingly urban country. And, as amply illustrated by the difficulties of implementing NFMA, forest management that relied largely on applied science was not up to the political challenges of the last 2 decades. What lessons can we draw from this?

## Ecosystem Management: Will It Evolve or Devolve

The causes of the current problem are clear; the solutions are hazy. At least one inference jumps out: politics must become an integral part of on-the-ground forest management.

As the Forest Service embraces ecosystem management, I believe the picture is clear enough to support the assertion that “timber primacy” is ending. This solves part of the political problems with o’d forestry. To the extent that ecosystem management rejects the traditional emphasis on the production of wood fiber and substitutes sustainability, the approach of ecosystem management will make significant political gains.

Much of ecosystem management seems to follow the tradition of Pinchot in seeking improved forest management through better scientific understanding. When there was agreement on ends, that approach was very successful; however, forest management as applied biological or physical science is a politically inadequate response to today’s challenges. If ecosystem management evolves to be simply better science applied to forest management, its use will be limited. Ecosystem management would become as irrelevant to the issues of the 1990s as old forestry was to the issues of the 1970s and 1980s.

There is another evolutionary path, one clearly provided for in Deputy Chief Overbay’s description of ecosystem management (1992). In Overbay’s formulation, ecosystem management begins with the assumption that current crises are largely political and social in origin, that people inside and outside the agency seek more involvement in decisionmaking, and that forest management today is about who gets what, winners and losers, and politics. If the political role is recognized--indeed, welcomed--and incorporated as a major and necessary component of ecosystem management, then there is a brighter future. Intense, stressful, and challenging--but brighter.

The new scientific understandings that are a part of ecosystem management are challenging enough to grasp (see sections 2 and 3 of this document) and they require significant change. My analysis suggests, though, that such change is only half the need, and it is the easier half to deal with. Complete ecosystem management must incorporate both changed scientific concepts and changed views of the importance of politics in land management. We must recognize that acceptance of this second element-political responsibility-goes against the grain of both agency history and personal predilection. Its acceptance contradicts almost a century of agency tradition and success. Perhaps more important, it goes against the grain of foresters who are comfortable with questions that can be answered on the basis of “facts,” but who are uncomfortable with questions that require understanding of values, interests, and influence.

Although there are impediments to a full development of ecosystem management, there are also reasons for optimism. One study (Clarks and McCool 1985) of Federal natural resource agencies identifies the Forest Service as a “bureaucratic superstar” and observes: “the history of the Forest Service... reveals a remarkable ability to sense changing public priorities and to adapt its mission to meet those demands.”

There is another factor that must be considered: increasing diversity within the agency (Kennedy 1991). With this diversification, there is risk that a once high morale, “can do” outfit with common values will become a missionless bureaucratic basket case. There is another possibility, though. With diversity comes openness to change. Ecosystem management offers both a conceptual framework and a management orientation for recognizing that long-term viability and success of an interdependent system (e.g., the Forest Service) rests fundamentally an maintaining diversity. A fully developed ecosystem management plan, that includes both the scientific and political responsibilities of a manager, also could provide the sense of renewal and new mission necessary for the increasingly diverse Forest Service to continue its earlier record of success.

## Can You Fix It On The Ground When The Sky Is Falling?

The need to embrace the political responsibilities inherent in ecosystem management is clear. But there are even greater challenges to be faced. Suppose the Forest Service accepts the challenge and goes about listening and leading and crafting compromises on the ground. Indeed, many managers have succeeded in doing this. Then what happens? The efforts of the natural resource manager are embedded in the larger political sphere, and the capabilities of that political system are under very serious strain.

There are breakdowns in our systems of governing. It is not the personalities-the Clintons, Bushes, and Perots. It goes deeper and is much more troublesome: it rests in the form of democracy we inherited from James Madison. Madison crafted a delicate balance of relations among the branches of government (the legislative, executive, and judicial) and between the levels of government (national and local), and there are now serious tears in the warp and weave of that political fabric.

Those breakdowns could be the subject of another chapter. Ross Perot could probably explain it in a 30-minute TV commercial. But I will inventory what I see as several of the most serious challenges:

- There is a chasm between what is happening at the local level and what is happening at the national level. The link between the national and local levels is broken. You can look at it as forces at the national level interfering with desires at the local level to make the most productive use of Pacific Northwest forests. Or, you can see that, even as local managers figure out how to merge technical and political information, the resulting plans do not “fly” politically at the national level. Congress continues to set unrealistic Allowable Sale quantities, and the fallout can be found in whistleblowers, organizations of Forest Service Employees for Environmental Ethics, letters from Forest Supervisors, and a contested resignation of a Regional Forester. Local and national politics do not mesh.

- In part, this political chasm is the result of another breakdown: a national politics that relies on fantasy instead of vision and that promises pipe dreams. We have come to expect that we can have all that we want without having to make hard choices. Beginning 12 years ago--and the blame rests in the then-Republican White House, the Democratic Congress; and our own gullibility as citizens--we believed we could cut taxes and dramatically increase spending without a price. In the environmental area, we believe that if we do not like certain tradeoffs, we do not have to have them. Absurd promises are made, immediate benefits consumed, hard issues avoided, and costs postponed.

- This leads to a third breakdown: ours is a politics in which symbolic issues are an increasingly important component of the political agenda. Lacking the resources or the will to address our significant material problems, vulnerable to cheap symbolic distractions offered by such issues as prayers in school, flag salutes, or sexual orientation, and confronted by single-interest groups uninterested in substantive trades and compromises, we conduct a politics that is more and more characterized by images, rituals, and myths. Leaders simply joust symbolically with problems. When the issues are politically too costly to be settled by the legislative branch, symbolic legislation is passed that seems to address the problem but, in fact, simply passes the political hot potato to the bureaucracy and the courts. The bureaucracy is similarly stymied as it casts about for public relations solutions to what may be politically unwinnable situations. As symbolic politics incapacitate the legislative and bureaucratic decisionmakers, the branch of government least suited to policy-making, the courts, play a larger and larger role and are forced (by default) to shift from procedural to substantive matters.

## CONCLUSIONS

At the start of the 20th century, Pinchot solved the crisis of his time with a political solution having three ingredients: scientific management, timber primacy, and decentralization. We need a new solution, a new political recipe. I have outlined some elements of a potential recipe:

- Accept that timber primacy is gone and such interrelated concepts as sustainability, biodiversity, and the health of the soil are taking its place.

- Embrace on the ground, the political role as something necessary and positive: listening to and leading people.

- My inventory of broader, encompassing challenges hints at other elements: use the expertise of natural resource managers to counter the politics of fantasy by educating the public and political leaders about the costs and consequences that must be faced; take more initiatives at the local level and move decisionmaking away from the national level; and, move more decisionrnaking away from the formal political processes and into more creative and hybrid organizations that are neither strictly private nor strictly public (the “Salmon Summit” being one example).

Whether you buy the recipe that I suggest is unimportant. That we all be thinking hard about what the recipe should be is very important. It is clear that what we have now is not satisfactory.

Whatever happens, perhaps both citizens and practitioners can benefit by remembering one of the emerging principles of the ecosystem approach: change is healthy, and major disruptions--even catastrophes--are necessary agents for change. When society changed, forest management had to change. The disruptions that have occurred in forest management are not symptoms of a failure; rather, they are the unavoidably tumultuous forces of regeneration and renewal.

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# Development of Landscape Ecology and Its Relation to Environmental Management

**F.B. Golley**


## ABSTRACT

This paper describes the development of landscape ecology in response to social and political desires for sustainable ecosystems. Landscape ecology has been used in Europe for planning and land management since the term was coined in 1939. In this role, landscape ecology was an integrating field among hydrology, geomorphology, vegetation science, soil science, economy, sociology, and land planning. Internationally, landscape ecology grew rapidly in the 1980s. This expansion, as represented by the creation of national and international research programs, is an example of the shift from an older scientific scheme of centrally defined subjects organized around one or more paradigms, to the current mode of working in networks. This network pattern is driven by the recognition that current societal, environmental, and economic problems are multidimensional and are the consequences of phenomena occurring at many spatial and temporal scales. The focus of landscape ecology on scales and integration of complex processes is an expression of the growing desire by the public, politicians, land managers, and scientists to focus on linkages among human and all other components and processes of the global ecosystem.


## INTRODUCTION

This paper describes the development of landscape ecology and thereby provides a background for ecosystem management. Historical comments will be relatively brief because of limited space.

Landscape ecology in the United States grew rapidly circa 1980 when it burst on the scene with an international congress, a technical review, and several books on the topic. Obviously, a gestation period preceded this emergence, but like the proverbial unmarried maiden, even the parents of landscape ecology (the subjects of geography and ecology) were generally unaware of its imminent appearance. The appearance of this discipline is an example of a scientific development pattern common among scholarly disciplines at the end of the 1900s. The older scheme, which has not entirely disappeared, consisted of subjects defined centrally, organized around one or more paradigms, with a well-defined method of research. These central subject areas were surrounded by a margin of out-of-date, less focused, and syncretic approaches to the topic. Any practitioner could define the center and the margin of their field.

This pattern has gradually been replaced by networks of research and practice. In the network pattern, interactions among subjects become more important and the central nodes less so. Driving this change is the wide recognition that current problems in the environment, society, and the economy are multidimensional. None of the former centralized disciplines is adequate to address, let alone solve, current problems. When putting together approaches and materials to address speck questions, the problem solver ignores the older boundaries between disciplines (Zonneveld 1988). The result is an enormously stimulating and exciting change in which one is free to burst out of the straightjacket of the classical subjects and explore any topic relevant to the question. Landscape ecology is one of many expressions of this current phenomenon.

### Definition of Landscape Ecology

Landscape ecology can be defined in several ways, but probably the most useful definition is that landscape ecology is the study of how land patterns influence processes. Processes may include flows of water, soil, chemicals, or energy; movements of organisms and humans; and the movement of products, resources, or capital. The ecologist tends to approach landscape ecology from a biological perspective and defines it spatially and temporally within ecological theory. The spatial pattern of the ecologist is often less than a few kilometers, sometimes only a few meters. The geographer, on the other hand, is frequently concerned about spatial patterns of human activities, and the landscape scale of the geographically oriented landscape ecologist can be many kilometers. Temporal scales used by ecologists and geographers are different as well.

## Historical Background

An awareness of landscape ecology developed well before the discipline's growth in the 1980s. The phrase was coined by a German geographer, Carl Troll, in 1939. Troll applied aerial photography to the theoretical study of land patterns. Troll's choice of the word "ecology" was probably an expression of a holistic orientation to spatial patterns. At that time in Germany, holistic thought was widespread in biology, forestry, and other subjects (Friederichs 1927, Thienemann 1939, Weber 1939), primarily because it was congruent with the political philosophy of the National Socialist party, which controlled the German government. The term "ecology" would distinguish this new approach to use of aerial photographs from routine photographic interpretation and cartography. Troll proposed to go further in interpreting his photographs by identifying surface objects that represented the interaction of water, land surface, soil, vegetation, and land use.

After World War II, several countries in Europe used land planning as a way to organize use of natural resources. These countries formed a band across northern Europe with The Netherlands on the west, the two Germanies in the center, and Czechoslovakia in the east. For different reasons in each country, landscape ecology became a specific discipline that organized information about the land surface for land planning. In this role, landscape ecology was an integrating subject that could translate between the hydrologist, geomorphologist, vegetation scientist, soil scientist, economist, and sociologist and the land planner (Last et al. 1982). Landscape ecology commonly has been used in Europe for planning and land management for many years. Indeed, the Dutch society of landscape ecology has hundreds of members, and in Slovakia, the government declared that all land planning would be grounded in landscape ecology.

The former Soviet Union occupies a special position in this story. In the USSR, geography is defined differently than it is elsewhere (Anonymous 1981, Isachenko 1973). Geography covers all subjects involving land. Russian geography began early with the exploration of the east and south and the need to understand how to manage soils, vegetation, and animals in vast forest and steppe regions (Fortescue 1992). Vasilii Dokuchev's contributions to soil geography and Vladimir Vemadsky's concept of biogeochemistry are known to many Western scientists, but these are only two of many scientists who worked in the discipline now called landscape ecology. Because of the Cold War, however, these scientists never made the contribution to the development of landscape ecology that we would expect.

In the United States, landscape ecology was essentially unknown until the 1980s. Here a holistic approach to the problems of land management had been fractured into competing subjects, thereby fitting the American culture. Nevertheless, several attempts were made by ecologists and geographers to bring these subjects together. For example, one attempt was made by Frederich Clements in the early 1900s under the concept of the biome (Clements 1916, Clements and Shelford 1939). Clements' biome became the conceptual device American ecologists used to structure the Analysis of Ecosystems Program of the U.S. contribution to the International Biological Program (IBP) (1967 to 1974). Biomes were defined as large spatial units, such as the arctic tundra, the boreal forest, the grasslands, Eastern temperate deciduous forests, and the coniferous forests of the western United States. These biomes were large enough that scientific activity could be organized and disputes avoided about who was in or out of a particular program, but biomes were too large to provide an integration of data and theory. Scientists who studied the grassland biome made the greatest effort to take an integrated biome-wide approach and had some modest success (French 1979), but scientists studying most other biomes were not able to express their results in a spatial context. American ecologists, frustrated by this experience, were ready to embrace a landscape ecology approach.

The IBP was organized in the mid-1960s as a theoretical and applied research program. The IBP was based on ecosystem science, which was dominant in American ecology by this time. The IBP plan in the United States was to integrate ecological data in ecosystem models. The model parameters were supposed to be adjusted for varying environmental conditions and different structural conditions throughout a biome, and were to be used to predict the response of the ecosystem under different management scenarios. This goal could not be achieved for several reasons. The technology of modeling complex systems had not advanced far enough, For instance, an ecosystem model of the grassland was built but not implemented because its structure made it obsolete before it could be used by land managers. Most IBP projects used models, but these models usually were focused on processes occurring on small areas. Ecologists also found it difficult to incorporate ecological information on species and individuals in ecosystem models.

Rules for aggregation of ecological groups were disputed and no consensus was obtained during IBP. Indeed, these arguments mirrored a division within the discipline of ecology between those ecologists focused on species and communities and those focused on ecosystems.

A second set of large-scale, complex system models was developed in the United States as part of the Research Applied to National Needs (RANN) project of the National Science Foundation. The RANN attempted to apply ecological modeling of ecosystems to problems in cities, landscapes, and large development projects. Brian Mar led a review of RANN for the National Science Foundation (NSF) in 1977 and showed that both modelers and practitioners failed to communicate and understand procedures, goals, and results. Modelers tended to focus on interesting questions and techniques from the perspective of technical modeling, which fit their professional goals. Practitioners tended to be captured by immediate events in their political and social context. Mar concluded that regional environmental systems analysis also required substantial input from the social sciences and the humanities before it could be successful.

American ecologists were concerned about the temporal and spatial scale of their studies and began after the IBP to lobby the NSF for a program that would provide support for long-term research. Their requests addressed a peculiar U.S. pattern of research funding whereby money was given for studies lasting 2 or 3 years, rarely 4 or 5, and then the investigator had to justify the need to continue the study. Natural processes in the ecosystems studied in the IBP operated at various time scales, and many of these time scales were longer than either 3 or 5 years. The NSF responded by organizing a special program (Long-Term Ecological Research) to continue and expand the IBP activity but in a more integrated and critically reviewed form. The NSF program fit into a general pattern in the U.S. Government where agencies, such as the Environmental Protection Agency (EPA), created national networks of study areas. All these efforts required a spatial system to integrate the results of research.

Government management agencies increasingly recognized these needs. In the 1970s, EPA's need for a national spatial information system to integrate environmental disturbances into a national environmental management network was discussed at the meetings of the Hazardous Materials Advisory Committee. Rather than having each region of the EPA be independent and develop its own procedures and approaches, the agency needed a hierarchical model with linked scales from the national to the local level (Patrick and Golley, unpublished). This idea was rejected as utopian. Similarly, in the early 1980s, the utility of parkwide information systems was discussed with the U.S. National Park Service southeastern regional science staff. These systems were to provide a superintendent with instantaneous data on species distributions, land use patterns, visitor use and similar information that could be used to make management decisions. As in the case of EPA, the suggestion was premature because some scientists claimed that political issues overrode technical information in decisionmaking. But now EPA has developed a national hierarchical model for monitoring (EMAP), and national park information systems are being developed and used in many places. These are merely two examples of a wide trend among the agencies to develop systems to manage and analyze spatial information over time.

Finally, a growing concern was expressed internationally and nationally about global pollution. In the U.S., scientists recognized that global models of atmospheric physics and chemistry required knowledge of exchange processes operable at the soil-vegetation and water surfaces of the Earth. Again, information on the surface of the Earth needed to be integrated in a spatial scheme. Large sums were being spent on collecting data by satellite and other forms of remote sensing, and this monetary support created further interest in landscape ecology.

Thus, there are many reasons why landscape ecology emerged as a discipline in the 1980s. The internal organization of science, the experience of science programs, social needs, and environmental concerns all played a role.

## Emergence of Landscape Ecology

Landscape ecology became widely known throughout the scientific community within a few years of its emergence in the early 1980s. An international congress was held, several books defining the subject were written, an international organization was developed, and a technical journal was published. Each contributed to recognition of landscape ecology as a new subject of wide importance for solving land planning issues.

It is fitting that this newly found recognition of landscape ecology started at an international congress organized by one of the oldest groups involved in that field, The Netherlands Society of Landscape Ecology. The congress consisted of a series of lectures and posters, workshops, and a closing discussion on politics (Tjallingii and de Veer 1982). The topics covered by the congress reflect the integrative aspect of landscape ecology, as they include theory, rural problems, rural-urban relations, natural areas, and methods. The theoretical part of the congress mainly tried to express the integrative approach of landscape ecology conceptually or verbally. For example, Veen (1982) stated that there was a basic cell of landscape analysis, called the ecotope and Phipps (1982) applied information theory to landscape analysis. It was noted that ecotopes could be aggregated into higher level units. Although this might seem unimportant, the reader should note that landscape ecologists often use “ecotope” for the smallest manageable land unit and not “ecosystem”. Ecosystem is used for all kinds of ecological systems, ranging from the Earth to the individual organism, and emphasizes energy flow, material cycling, and dynamic processes. Ecotope describes an object in the landscape; ecosystem refers to an orientation to research and management.

It was clear from applied sections of the congress that landscape ecologists were successful in studying relatively small areas, such as coastal dunes exposed to intense recreation pressure or a rural area whose water level was changing because of groundwater irrigation or subsurface construction. At this scale of planning, scientists had substantial experience with landscape ecology. At larger scales, there was less or no experience.

It was also was clear that the goal of landscape ecology extended beyond the traditional disciplines. The titles of the workshops at the Congress indicate this intention: terminology; theoretical aspects of ecological relations; theory of island biogeography; stability; rural problems in developing countries with emphasis on tropical rain forests; nature, agriculture and recreation in rural areas of industrialized countries; urban-rural relations; pollution and degradation; conservation aims and management; new human-made nature; the role of water in the landscape; inventory, classification and evaluation; databanks; stratification and sampling procedures; modeling; the visual landscape; landscape architecture; environmental planning; environmental impact assessment; landscape ecology and environmental education; and landscape ecology and politics. This list covers almost every topic that would be included in landscape ecology today! But note that the ecosystem concept that would motivate and guide many American ecologists was not included in the topic headings.

The most important consequence of the international congress was that ecology of the landscape was recognized as a research subject that applied to large scale land planning problems. Participants of the Congress wanted to meet again; consequently, a second meeting was held 2 years later in Slovakia and then 4 years later in Munster, Germany. At these meetings, the International Association for Landscape Ecology (IALE) was formed.

Landscape ecology also became formalized through two books published in 1984 and 1986. The first was written by an Israeli landscape architect, Zev Naveh, and an American landscape architect, Arthur Lieberman, under the title *Landscape Ecology: Theory and Application*. This book had been introduced in a long paper by Naveh (1982) in which he stressed that landscape ecology had evolved in German-speaking parts of Europe, and because most American ecologists did not read German, they were unaware of the subject. He proposed to summarize the advances made by German and other landscape ecologists for the wide audience of English-speaking ecologists. But Naveh had another motive. He is a holist and he used his book to advance the concept of the total human ecosystem, a phrase which emphasizes the integrated land-human system of environmental interactions. His experience has been with Mediterranean landscapes, and he convincingly described the role of human history in producing the landscape patterns and processes of this region. The perspective of Naveh and Lieberman was strongly conceptual and verbal, and as such, it provided a bridge from landscape ecology to many other subjects, especially the social sciences and humanities. Because their approach is not strongly quantitative, however, it has had less impact on the development of the scientific form of landscape ecology, which characterizes American landscape ecology.

The second book was also titled *Landscape Ecology* and was written by the American ecologist, Richard T. T. Forman, and the French ecologist, Michel Godron (1986). Their book follows the conventional order of modern ecology books, with introductory chapters on principles and concepts followed by sections on structure, dynamics and heterogeneity, and management. The chapters on landscape structure have been especially influential because they introduce the concepts of patches, corridors, and the landscape matrix. The chapters on landscape dynamics are less well developed and have been less influential, although the authors anticipated most of the topics that have since interested landscape ecologists. In the area of management, however, the two books deviate especially strongly. Forman and Godron, repeating the approach of the typical ecology text (for example, Odum 1953), treat the human species and its activities in about 30 pages, which cover production, planning, quality, and modeling. In contrast to Naveh and Lieberman, this text includes quantitative data and use of equations.

It is appropriate to throw the net a little wider and note the publication in 1982 of another book that has strongly influenced landscape ecology but that did not recognize the subject directly. This book was titled Hierarchy: Perspectives for Ecological Complexity and was written by T. F. H. Allen and Thomas B. Stan (1982), environmental scientists at the University of Wisconsin. This book made ecological hierarchies explicit and, therefore, provided the conceptual basis for linking ecosystem studies, which were well developed through the IBP and other programs, with biomes and other spatial units of intermediate scale. The concept of an ecological hierarchy was not new; indeed, it has been used as a planning instrument by Environment Canada (Northern Ecological Land Survey Map Series, No Date). But the concept was new in providing a theoretical basis for connecting separate approaches to form a single system. After Allen and Stan's book, the idea of spatial-temporal hierarchies became commonplace.

These three books illustrate the approaches of landscape ecology and potential avenues of application. First, the fundamental concept of spatial-temporal hierarchies, although not new, was cast in terms of nested series of ecological systems with linkage up and down and across subsystems. Linkage is through process and provides stability or control of function. Second, Foreman and Godron provided landscape ecology with a structural system that accurately represented our experience with land. Patch, corridor, and matrix are real elements of land to most people, and this language links highly technical and common experience. Finally, Naveh and Lieberman contributed the human dimension to landscape ecology. Naveh and Lieberman emphasized human needs and purposes and the linkage of humans and nature. If we are to have success in adapting to our environment and building sustainable societies in the United States, we must incorporate human needs and philosophies into land management and planning. Otherwise, we have a tyranny of the technician. Naveh and Lieberman identity a missing link in ecosystem management, and fortunately for the ecosystem manager, there is no lack of activity in the humanities and social sciences. In the past 10 years, society has seen the emergence of human ecology, ecological economics, cultural ecology, social ecology, environmental history, ecological engineering, environmental ethics, political ecology, environmental law, and more. Every field now has a focus on the environment, and students are being trained in these special subjects with added emphasis on the environmental linkages with the traditional disciplines. Land management agencies have a unique opportunity to bring this expertise into the planning and management process.

Every organized scientific subject has a technical journal to publish the reports and analyses of its practitioners. In 1987, the first issue of Landscape Ecology was published. The new technical journal was guided by an editorial board of 18 distinguished scholars and practitioners and now has formal connections to the International Association for Landscape Ecology. Seven volumes of Landscape Ecology have been published. The titles of papers in this journal reflect the current emphasis of an entire field, its success and its shortcomings. John aliens (1992) recently reviewed the papers published in Landscape Ecology. Wiens found that half the papers were concerned with landscape structure and were predominantly descriptive or conceptual. Although one-fifth of the papers dealt with ecosystem studies, only one was concerned with biogeochemistry. Only two papers employed experimental techniques, and very few included mathematical expressions. Wiens was concerned about the descriptive character of the subject and its lack of a theoretical base. Wiens' concern is justified. The problem he identifies is caused by inadequate support for this type of research, the organization of science management, and the problems of coupling theory to practice.

## Topical Development

Wiens' review is useful to landscape ecology because it focuses on several key problems in the development of an integrated, hierarchical approach to land-human systems. If we are to understand how landscape pattern interacts with, controls, and shapes landscape processes, we must have a way of moving from description of static structure to quantitative expression of dynamic action. Models must be systems of equations describing flow over a surface, modified by the surface itself. What are some of the foundations for such models?

There are two major techniques that support modern landscape ecology: remote sensing and ecological landscape modeling (Turner and Gardner 1991). Remote sensing represents a broad range of techniques in which physical signals are recorded by a camera in a satellite, balloon, or airplane, and translated into quantitative data that can be converted into pictures of the Earth's surface. Repeated photographs permit a record of change and therefore of process. This subject is enormously complex. The problems of adequately sensing radiation reflected or emitted from a surface, distinguishing wave lengths of energy that have meaning about a surface, decoding the information, and correlating the signal consistently to phenomena at a surface are all difficult. Yet, remote sensing offers an extremely powerful tool for landscape ecology because it potentially permits repeated interrogation of any surface, no matter how remote. The problem is that the physics far surpass the ecology, and many more studies of the relevant processes are required before we can apply this technology routinely.

The second technique is modeling. Modeling allows researchers to calculate the interactions of a cell, based on geographic information system data, with all adjoining cells, and to sum these interactions for all cells in the landscape over time. This simple language conceals great complexity. If the landscape is large or the number of units is large, the data require a supercomputer. Input-output models may not be adequate because cells store information and materials. Further, the exchange may be in many forms--energy, materials, money, ideas, rules, cultural attitudes, social organizations, and so on. It is not yet clear how to alter a set of equations that describe water flow across a landscape to account for economic exchange, technological advance, and traditional constraints. Again, this is a topic for advanced research. A second form of ecological modeling is of interest to ecologists and biologists who study and manage a single species. In this form, the models describe how individuals use and occupy space. Recent advances couple use of space with demographic parameters that measure fitness of the organism.

Wiens observed that few landscape ecologists use an experimental approach. It is unlikely that researchers could experiment with systems of many square kilometers in which humans live, or that small landscapes of a few meters can do more than give us ideas about human-sized landscapes. Yet, it is important not to pass off Wiens' comment too quickly. Certainly, a rigorous structural analysis of landscapes should permit researchers to trace processes of change and to take advantage of human-caused change. These may not be designed experiments, but they provide abundant opportunity to observe "natural experiments." Repeating these observations in space and time permits identification of mechanisms that explain patterns of behavior. Being effective requires a proactive rather than a reactive research approach.

## CONCLUSION

Landscape ecology is of substantial, practical importance to environmental management, which justifies its further development. Landscape ecology is an integral part of land planning, and as long as humans increase their demands on the land and water resources of the planet, planning for these demands will be critical. To plan, we need effective methods to integrate information on, the capacity of natural and managed systems to provide services, resist and recover from disturbance, and sustain function over space and time scales appropriate to social and political needs. The sustainability of the forest ecosystems of eastern Oregon and Washington depends on how effectively information on biological diversity, ecological function and balance, product output, and social values of the study area are integrated. Ecosystem management principles must be formulated based on this integration. Implementation strategies should be developed to incorporate these principles into the planning process of the USDA Forest Service.

Ecosystem studies have made advances in integrating processes across spatial and temporal scales. Throughout the country, biome-type studies are underway and are providing baseline information, technical skills, and models that can be used to compare systems managed for specific purposes. It is necessary to view land management at regional scales, in which local needs and differences are recognized and respected. A hierarchical system allows managers to understand the consequences of rules and decisions at multiple levels of scale. This approach can show how the internal logic of the subsystems after the higher scale rules and produce outcomes contradictory to that expected. It can show how fine-scale, local systems may operate counter to the regional system, yet not disturb neighboring systems; that is, the approach has the potential of enhancing biological and human diversity.

At the global level, managers of change also will find landscape ecology useful. To study and manage very large ecoregions, or the globe itself, it is necessary to aggregate information gathered at finer scales The technical problems of aggregating and integrating small-scale processes at multiple scales seem overwhelming. A major challenge for landscape ecology is to discover the assembly rules for aggregation of small landscape units and processes into units of large scale.

Landscape ecology represents an interest in large-scale phenomena among a growing number of scientists, land managers and decisionmakers. The twin focuses on scale and on integration of complex processes are technical expressions of a growing desire to incorporate human interactions into studies of ecosystems. Thus, the academic development of the subject is increasingly driven by practical need. As we recognize the impact of human life on the planet, more people from all parts of society express their concern about what may happen to the planetary environment. Landscape ecology is linked through this concern to biological and environmental conservation, human ecology, environmental engineeering and management, ecological economics, and other efforts to focus on linkages between human and other components and processes of the global system.

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# SECTION 2 - ECOLOGICAL PRINCIPLES

## Summary of Ecological Principles Papers

This section describes key principles of landscape ecology that contribute to improved ecosystem management. The maintenance of healthy ecosystems on regional to local scales requires an understanding of their composition, structure, and function (Franklin 1988). Additionally, the relation between ecosystem patterns and the processes that maintain such patterns (e.g., fire) must be understood. Accordingly, landscape ecology theory offers promise to land managers because of its focus on the development and dynamics of pattern in ecological phenomena, the role of disturbances in maintaining ecosystems, and the spatial and temporal scales of ecological events (Urban et al. 1987). The following discussion provides a brief overview of papers contained in this section and suggests how various aspects of landscape ecology theory may be incorporated into ecosystem management.

A basic overview of key landscape ecology principles is provided in the first paper of this section (Bourgeron and Jensen). The authors review the apparent complexity of landscape dynamics (pattern and process relations) and illustrate how hierarchy theory can simplify such complexity. In this approach, the development and organization of landscape patterns (e.g., vegetation) are described in the context of spatial and temporal hierarchies. Disturbances (e.g., fires and floods) that influence landscape development and pattern, biotic processes (e.g., species migration and extinction), environmental constraints (e.g., microclimate and global climatic change), and vegetation patterns (from stands to biomes) are better understood when viewed as phenomena dependent on the spatial and temporal scale of study. The concept of landscapes as hierarchically organized systems is central to efficient landscape evaluation and ecosystem characterization. Bourgeron and Jensen also describe how understanding ecosystem composition, structure, and function (in a hierarchical context) is useful to the development of conservation strategies. The coarse-filter strategy for maintaining biodivers'ity (Hunter 1991) is reviewed in this context and suggestions are offered about the use of landscape ecology principles in land evaluation and monitoring.

Applying the principles outlined above in landscape characterization and evaluation involves data collection and interpretation. Information concerning biotic and abiotic components of the landscape and their interactions are critical to such efforts. This information, in turn, is commonly used in land management to describe spatial and temporal patterns of species and communities, patterns of bioenvironments and physical environments at different spatial scales, and spatial and temporal patterns in ecosystem composition, structure, and function. Models developed from such information are routinely used by land managers to predict biotic responses to management. The second paper of this section (Bourgeron et al.) addresses these issues by discussing theoretical concepts pertinent to data collection and interpretation. The authors suggest that landscape evaluation and ecosystem characterization are dependent on pattern recognition and environmental correlations at various scales. These factors are, in turn, greatly influenced by the type of survey data used in analysis and the mental model of ecosystem relations used in survey design. The authors recommend that: (1) the community concept of species distribution is appropriate to general characterization and mapping; however, the continuum concept of species distribution should be acknowledged when describing the response of species to environmental gradients; (2) testable ecological relations of biotic-abiotic interactions must be incorporated into the design and delineation of ecosystem bioenvironment and physical environment mapping units; (3) the temporal and spatial variability of landscape patterns must be considered in survey design; and (4) predictive models of species and community response to the environment may require complex curvilinear functions that vary by bioenvironmental setting.

The third paper of this section (Turner et al.) suggests that the environmental heterogeneity of landscapes is hierarchical and is controlled by different processes at different spatial and temporal scales. The authors demonstrate that landscape heterogeneity is often produced and maintained by ecosystem disturbance and recovery dynamics, and the resulting patterns (e.g., community distribution) have consequences for various ecological processes at the landscape scale. Recognition of heterogeneity patterns (as well as their causes and consequences) is critical to the development of management plans for ecosystem sustainability. The authors indicate that four factors can be effectively used to describe the scaled dynamics of landscapes: disturbance frequency, rate of recovery from disturbance, spatial extent of disturbance events, and spatial extent of the studied landscape. These factors may be reduced to two key parameters representing time and space in the description of potential and current landscape disturbance dynamic states (e.g., equilibrium-steady state systems and stable-low variance systems). The effects of management practices on ecosystem sustainability may be analyzed by describing their relation to shifts in landscape level disturbance dynamic states.

Turner and others also state that land managers must recognize the hierarchical structure of landscapes, the effects of disturbances at different spatial and temporal scales, and the scale-dependent effects of heterogeneity in their ecosystem management strategies. They suggest the following items be considered in developing strategies for landscape maintenance across multiple spatial and temporal scales: (1) landscape-level indices should be used to measure pattern at multiple scales (instead of single, simple concepts like patches and corridors); (2) natural levels of landscape heterogeneity in space and time should be maintained by allowing natural processes (e.g., fire and flooding) that create and maintain heterogeneity to occur, or management practices should be used that mimic such processes; (3) connectivity should be maintained in the landscape by keeping the amount of native habitats above potential connectivity thresholds; and (4) managers should be aware of the potential importance of crossing critical thresholds; for example, small changes in habitat abundance and pattern can suddenly fragment an otherwise well-connected landscape at some (but not all) resource-use scales.

The fourth paper of this section (Swanson et al.) builds on the discussions of Bourgeron and Jensen and Turner and others by specifically suggesting how an understanding of the natural (historic) variability of the composition, structure, and functional components of an ecosystem may be used in ecosystem management. The description of historical processes (e.g., fire) which maintained ecosystems patterns (e.g., community distribution) is stressed in this paper. Historical ranges in disturbance regimes (i.e., their magnitude and frequency) are correlated with landscape patterns and may be used to provide an initial description of desired conditions for land management planning in this approach. The authors consider an understanding of historical processes and interactions to be a logical starting point for adaptive management approaches to ecosystem management. The method described by Swanson and others is also consistent with the coarse-filter approach to managing biodiversity (Hunter 1991) and some regional strategies for sustaining ecological systems (USDA Forest Service 1992). These management strategies assume that if communities and their processes are similar to those that occurred naturally (i.e., before European settlement), then conditions are similar to those under which species evolved; consequently, the full complement of species will persist and biodiversity will be maintained. The validity of such assumptions and the limitations of natural variability descriptions in ecosystem management are addressed by the authors.

The final paper in this section (Bailey et al.) describes the design and use of ecological mapping units in land management planning. Such mapping units delineate similar biophysical environments for land evaluation planning and may be defined at various hierarchical scales depending on management needs. Ecological mapping units also delineate areas with similar potentials for management based on landscape components (e.g., soils, landform, and climate) that change slowly. Ecological mapping units are commonly used to describe how the landscape could look or function under natural processes as well as under different management scenarios. Most other resource maps describe ecosystem components that display high temporal variability (e.g., vegetation) and are used to describe what the landscape currently looks like. Both types of maps are required to describe ecosystem health (i.e., what the landscape currently looks like relative to what it could or should be, given management objectives for sustainability); however, availability of well-designed ecological unit maps is limited. Accordingly, the authors provide basic theoretical and practical design considerations useful to future ecological mapping efforts.

Bailey and others suggest that the boundaries between ecological mapping units should be based on semipermanent landscape components important in differentiating ecosystems at various scales (e.g., landform) to help recognize ecological units regardless of present land use or existing vegetation. The authors also suggest that differentiating criteria used in ecological map unit design will commonly vary by mapping scale, given that different variables exert greater control on biotic patterns and processes as mapping scale changes. For example, at broad (macro) scales, vegetation biomes are primarily controlled by regional climate processes. At meso and micro scales, however, landform and topographic settings are primary determinants of biotic pattern. The authors suggest the primary challenge of ecological classification and inventory is to distinguish natural associations of ecological factors at different spatial and temporal scales. Additionally, they suggest that the differentiating criteria for ecological map unit design must reflect those factors that exert primary control on the hierarchies of organization contained within ecosystems.

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# An Overview of Ecological Principles for Ecosystem Management

**P.S. Bourgeron and M.E. Jensen**

## ABSTRACT

The study of spatial and temporal patterns across landscapes is central to formulating ecosystem management principles. The hierarchical structure of ecological systems allows the characterization of ecosystems and the identification of patterns and processes at different scales. Ecosystem composition, structure, and function determine diversity patterns across a range of spatial-temporal scales. There is no single correct scale at which to study and manage ecological patterns, processes, and diversity. The ecological hierarchy of interest is determined by the purpose of each project. Hierarchical monitoring schemes must be formulated that consider all scales of ecological organization, Patterns of natural variability across a range of scales must be defined if ecosystems are to be sustained at all relevant scales.

## INTRODUCTION

Landscape managers seldom know how management scenarios affect treated landscapes, their component ecosystems, the movement of animals, the distribution of plant species, disturbance regimes, and biogeochemical cycles. The study of spatial and temporal patterns of landscapes is part of the discipline of landscape ecology, which is central to ecosystem management (Jensen and Everett 1993). Understanding landscape patterns in terms of the processes that generate them drives landscape ecology (Golley 1993) and is key to the development of principles for land management.

The concepts of scale and pattern are interwoven (Hutchinson 1953, Levin 1992). This paper outlines a hierarchical approach to the study of landscapes that focuses on: (1) quantifying ecological patterns and processes in space and time; (2) understanding pattern changes with scale; (3) understanding the causes and consequences of patterns; (4) the consequences of the hierarchical structure of ecological systems for ecosystem management; and (5) defining hierarchical monitoring schemes for biological patterns, processes, and diversity. This approach considers and simplifies landscape dynamics (Forman and Godron 1986, Urban et al. 1987, Levin 1992) to make it amenable to ecosystem management.

### Landscape Pattern Formation

Landscapes are heterogeneous mosaics of patches (Forman and Godron 1986, Urban et al. 1987). Describing these mosaics requires the identification of pattern. Pattern recognition is the description of variation, and it requires the determination of scale (Levin 1992). Once ecological patterns are characterized, the agents of pattern formation (sensu Urban et al. 1987) must be identified. The agents of pattern formation have been grouped into three categories (Levin 1978, Urban et al. 1987): biotic processes (e.g., migration and extinction), disturbances (e.g., fires and floods), and environmental constraints (e.g., landforms and soils). Ecological relations are defined by matching ecological patterns with their relevant agents of formation.

Complex landscape patterns and the many processes that form them exist within a hierarchical framework (Allen and Starr 1982, Allen et al. 1984, O'Neill et al. 1986). In recent years, attention has been directed toward describing the formal hierarchical organization of ecological systems. Hierarchy theory (Allen and Starr 1982, O'Neill et al. 1986) is concerned with multiscaled systems, in which an upper level of organization provides to some extent the environment that lower levels evolve from. A critical characteristic of a hierarchical system is the "whole/part" duality of its components (Koestler 1967, Allen and Starr 1982, Allen et al. 1984). Every level is a discrete functional entity and also part of a larger whole. As applied to landscape ecology, hierarchy theory allows the definition of the components of an ecosystem or set of ecosystems and the linkage between the different scales of ecological organization. Both the object of study (Rosen 1975) and the ecological pattern of interest (Puttee 1978) are defined by the observer.

For example, if one is interested in the composition and structure of the high-elevation subalpine fir forests of the Rocky Mountains, the ecological pattern of interest is the vegetation pattern which can be described at six scales: (1) the individual plant; (2) the individual mature canopy tree; (3) the stand or community; (4) the cover type; (5)

the physiognomic formation; and (6) the biome (fig. 1a). Each of these scales spans a certain spatial and temporal range. At each scale, the vegetation pattern produces patchiness. This patchiness can be related to specific scales of biotic processes, disturbances, and environmental constraints.

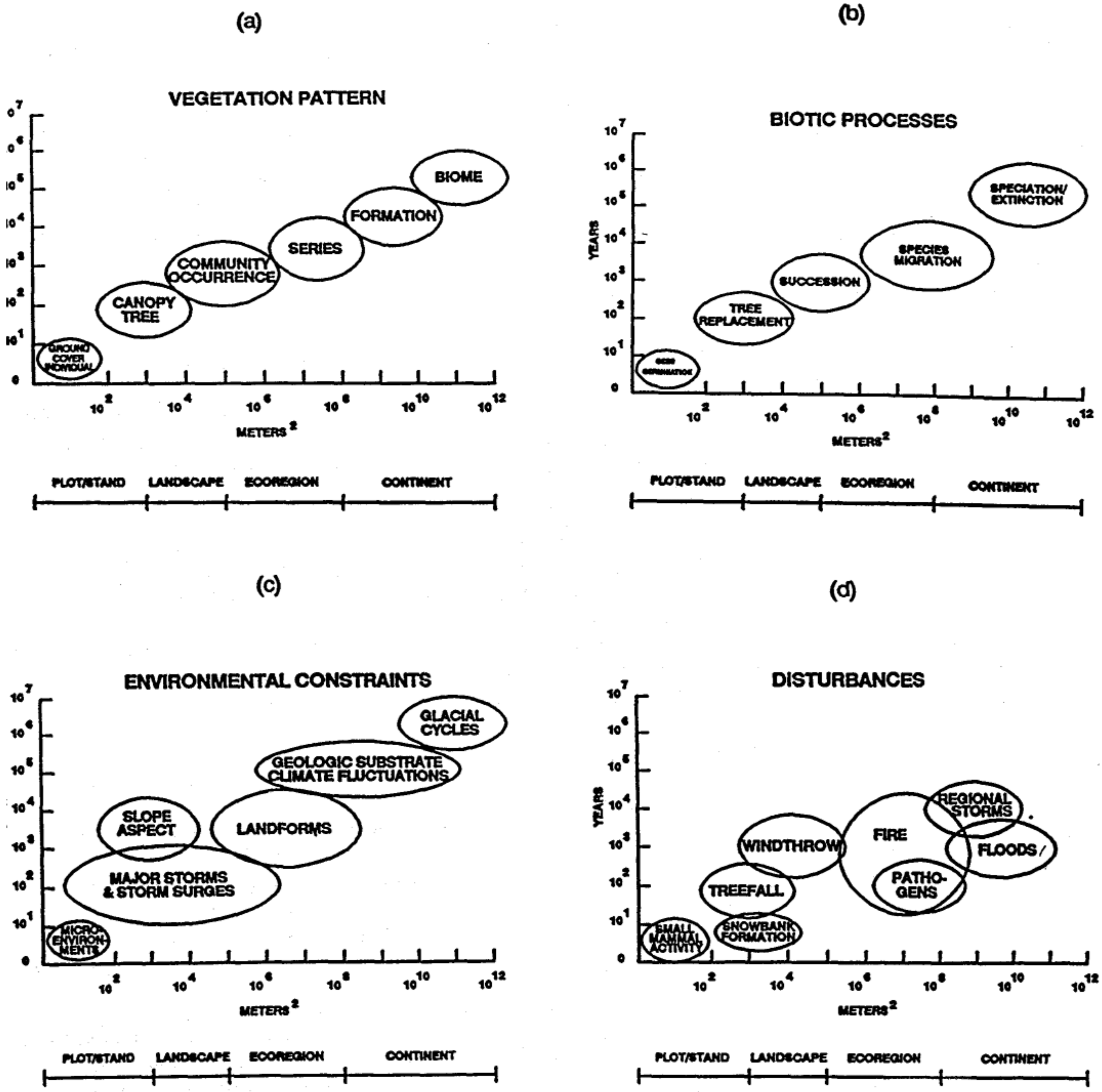


**Figure 1--Spatial-temporal scaled patterns for (a) vegetation, (b) biotic processes, (c) environmental constraints, and (d) disturbances (modified from Urban et. al 1987).**

For the Rocky Mountain high-elevation subalpine fir forests, biotic processes exhibit scaled patterns (Watt 1947, Urban et al. 1987) (fig. 1b). Environmental constraints can be decomposed as shown in figure 1c. Information about the scales of each constraint can be found in the literature. For instance, detailed information on landforms has been published for the northern Rocky Mountains area (Donahue and Holdorf 1990). Mitchell's (1976) climatic regions of the western United States provide the necessary information for the climatic phenomena of interest. Finally, disturbances affecting vegetation can also be arranged hierarchically (Pickett et al. 1989). A review of the literature allows the construction of the hierarchy shown in figure 1d for subalpine fir forests. Information can be readily found in the literature concerning fire regimes and generalized successional models (Fisher and Bradley 1987, Arno et al. 1985, Romme and Knight 1982), snowbanks, pathogens, activities of small mammals, and tree dynamics (Benedict 1983).

The overlay of the four hierarchies provides a conceptual picture of ecological relations at different scales (fig. 2). This process characterizes the composition, structure, and function of high-elevation subalpine fir ecosystems across a range of spatial and temporal scales (fig. 2 and table 1). Vegetation pattern at different scales can be considered in light of biotic processes, disturbances, and environmental constraints. Mechanisms that generate the pattern of interest can then be formulated. For example, let us assume that the pattern of interest is the species composition of the subalpine fir/whortleberry Abies lasiocarpa/Vaccinium scoparium, stands in a forested landscape of southwestern Montana. As shown in figure 2, subalpine fir/whortleberry communities are part of a broader high-elevation forest type, the subalpine fir forests, that has a larger geographical and environmental distribution. A given stand of subalpine fir/whortleberry is constrained by landform (broad ridge) that also affects other environmental gradients such as the frequency and intensity of disturbance by fire and wind (Swanson 1981, Swanson et al. 1988). The composition, structure, and function of a subalpine fir/whortleberry stand is constrained by specific patterns of fire frequency and intensity (Fisher and Bradley 1987). The subalpine fir/whortleberry stands of southwestern Montana are part of a well-documented generalized successional sequence (Fisher and Bradley 1987). Within a stand, various components can be identified. For example, a tree seedling germinates in a favorable microsite, but its success is constrained by the spatial distribution of snowbanks. Early in succession, lodgepole pine Pinus contorta will generally dominate the canopy. As lodgepole pine trees die and produce gaps, they will be replaced by subalpine fir trees. Gap dynamics will contribute to stand dynamics, and the stand will succeed from a lodgepole-dominated canopy to a subalpine fir-dominated canopy. Strong windstorms are likely to produce stand-level treefalls that also contribute to stand dynamics.

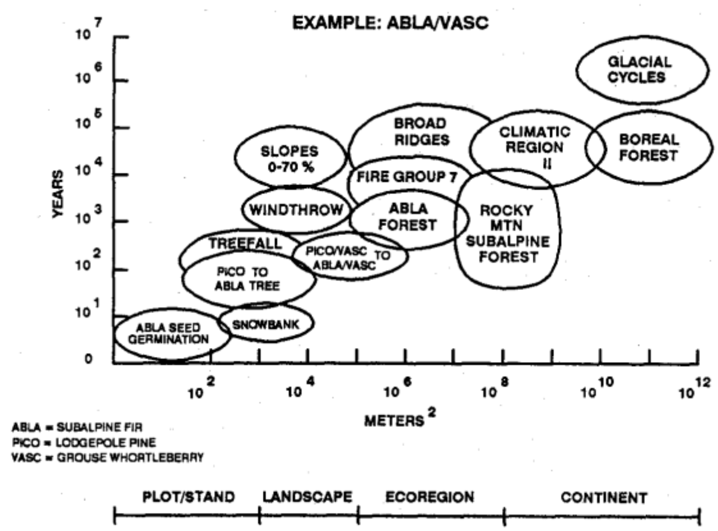


**Figure 2--Hierarchical spatial-temporal representation of ecological relationships.**

**Table 1--Example of Ecosystem Characterization at Different Scales**

| Scale | Vegetation patterns | Biotic processes | Environmental constraints | Disturbances |
|---|---|---|---|---|
| Continent | Biomes/formations | Speciation/extinction | Climatic region | Glaciation |
| Ecoregion | Series | Species migration | Geology | Fire |
| Landscape | Communities seral or climax | Succession | Landforms major storms | Windthrow |
| Plot | Community/ tree dynamics | Tree replacement | Soils | Treefall |

The hierarchical approach described above allows the characterization of ecosystems and the identification of patterns and processes at different scales (table 1). Variability can then be quantified within each level and the emerging pattern can be related to its causes and consequences. Correlation analysis usually provides an initial understanding of which mechanisms generate patterns (Austin 1985, 1991, Levin 1992). For example, within the broad category of high-elevation subalpine fir forests, there is a definite substructure of spatial vegetation pattern that is primarily correlated with moisture and temperature gradients, mineralizable nitrogen and phosphorus, organic matter, and spatial variation in nutrient content. Landforms interact with spatial patterns of ecosystems directly through control of nutrient cycles and water flows at a particular scale and indirectly through control of fire and wind regimes at another scale (Swanson et al. 1988). The hierarchical approach leads to such a listing of possible mechanisms (Levin 1992) in the relevant spatial and temporal context (Urban et al. 1987).

Identification of landscape hierarchies can be applied to a wide range of ecosystems and management goals. For example, using 23 managed ecosystems, Holling (1992) hypothesized that landscapes form a hierarchy containing discontinuous spatial structures at specific scales (see also Holling 1986). These 23 managed ecosystems fall into four broad categories: forest insect pests, forest fire, semi-arid grasslands, and fisheries. Holling (1992) concluded that “the landscape is hierarchically structured by a small number of structuring processes into a small number of nested levels, and that those form physical textures and temporal frequencies specific to each level” and that “the processes that generate discontinuous time dynamics also generate discontinuous physical structure.”

## A Hierarchical Approach To Biodiversity

“The problem of ecological pattern is inseparable from the problem of the generation and maintenance of diversity” (Levin 1981, 1992). The description of environmental patterns and ecological heterogeneity is a description of patterns of diversity. Composition, structure, and function are the three primary attributes of ecosystems (Franklin et al. 1981, Franklin 1988). These attributes determine the diversity patterns of a specific area (Franklin 1988, Noss 1990). Therefore, an understanding of landscape patterns, their components, and the processes that generate them is central to the understanding of diversity.

The consequences of the hierarchical structure of ecological systems for diversity patterns are many. One consequence is that different patterns of diversity are exhibited at different scales of organization (Norton and Ulanowicz 1992, Noss 1990). For example, one aspect of biotic diversity is the diversity of physiognomies. A series of studies of the climatic controls of vegetation across large areas (Neilson 1986, 1987, Neilson et al. 1989, 1992) demonstrated that the physiognomy of a biome is related to large-scale patterns of climate and ecosystem processes and functions, such as carbon and nutrient cycles (Burke et al. 1990). The evolutionary adaptation of different life forms to different carbon and nitrogen flows is expressed in their life history strategies (Schultze 1982, Stearns 1976). Differences in life forms and life history strategies generate the distinct physiognomy of each biome.

Another consequence of the hierarchical structure of ecological systems for diversity is that since there is no single scale at which landscape and ecosystem pattern should be described (Levin 1992), there is also no single scale at which diversity should be described (Noss 1990). For example, regional species diversity patterns can be related to large-scale climatic factors (Currie 1991, Currie and Paquin 1987, Neilson et al. 1992) and therefore can be used to characterize ecoregions. Local factors, such as soil types, slope, and aspect, however, modulate the coexistence and distribution of species at landscape scales. Thus, the landscape is partitioned into different patches or suitable habitats that will result in varied diversity patterns. These patterns depend on the number, size, frequency, and spatial distribution of suitable habitats. The hierarchical structure of the environment has been used successfully to account for community level phenomena such as species diversity (Kolasa 1989). The geometry of the landscape across a range of scales, which is the result of the distribution of suitable habitats, determines species distribution to a large extent (Milne 1993), and, hence, diversity patterns as well (Palmer 1992). The relative role of regional and local constraints and processes on community diversity at the site level (Ricklefs 1987) differs depending on the location of a site within a region (Neilson et al. 1992).

## Natural Variability

A third consequence of organizing landscapes and ecosystems into hierarchies is that patchiness and heterogeneity can be found at a broad range of scales. The spatial and temporal variability in the environment provides biota (plants and animals) with diverse resources, thereby allowing the coexistence of species that would not coexist in a nonhierarchical environment (Levin 1992). Therefore, spatial-temporal variability affects the persistence and coexistence of species, thereby creating various biotic communities and increasing biotic diversity.

Climate shifts (Delcourt and Delcourt 1988), changes in temperature and precipitation, which are modulated by landforms and other environmental constraints, produce different patterns of suitable habitats at the landscape scale. Each resulting landscape pattern provides a different environment for the biota. Some species may find more suitable habitats after climate change, whereas others may find less or none at all. The frequency of climate changes will dictate whether an organism might be carried over from an unfavorable cycle to a favorable one (Neilson 1986, Neilson et al. 1992). If the lifespan of a species is much longer than the frequency of climate change, it can become established during one climatic regime, and merely survive during a succeeding regime (Neilson et al.1992). Temporal variability superimposed on spatial variability shapes patterns of diversity.

Fire dynamics also change in response to climate changes (Overpeck et al. 1990). Therefore, the mosaic created by wildfires will also exhibit spatial-temporal variability (Jensen et al. 1991). These observations have important implications for predicting ecosystem responses to global or land management-induced changes. Such changes can potentially alter the pattern of natural variability at many scales. Species respond individually to change, and new communities might be formed in newly defined habitats (Bourgeron et al. 1993). Patterns of species distribution and abundance could change (Glenn and Collins 1993) in response to changes in regional patterns of suitable habitat (Neilson et al. 1992). To study landscape patterns and ecosystem composition, structure, function, and diversity, managers must quantify the patterns of variability in space and time, and they must understand patterns of change (Levin 1992).

## Coarse- and Fine-Filters and System Management

The concepts of coarse and fine filters are essentially based on the concept of scale. In its simplest form, the coarse-filter concept states that if aggregates are managed (e.g., communities, ecosystems, and landscapes), the components of these aggregates will be managed as well. For example, if a conservation strategy is designed by using plant communities as the coarse filter at the landscape scale, it is assumed that the species, which constitute the fine filter, will be protected as a consequence of the plant communities persistence in the landscape. It has been argued that designing conservation strategies for all species individually is impossible because scientific data do not exist at present for all species, and the cost would be prohibitive. The coarse-filter concept has been used mostly for maintaining common species over large areas (Scott et al. 1990). The fine-filter concept has been used to formulate management strategies for rare species, communities, or ecosystems that would fall through the cracks of the coarse filter (Jenkins 1976).

The coarse- and fine-filter concepts are management consequences of the hierarchical structure of ecological systems. The identification of a coarse filter for an ecological system involves the identification of an appropriate scale of ecological organization for a given purpose. The fine-filter elements are the components of the higher level selected. In the example of the high-elevation subalpine forests, the plant community (e.g., subalpine fir/ whortleberry) is the appropriate coarse-filter for managing the vegetation at the landscape scale (fig. 2 and table 1). The individual plant (species level) then becomes the fine filter. If a land manager works at the scale of an ecoregion, an appropriate coarse filter is the cover type (e.g., subalpine fir forests). The plant community would be the relevant fine filter if interest is in vegetation mosaics. The selected hierarchical scale of organization defines the type of patterns, ecological processes, and environmental constraints of interest and thus defines the appropriate coarse and fine filters. The choice of a scale is based on the management objective and the ecological system of study.

## Equilibrium and Pattern Persistence: Some Ecosystem Management Principles

Because the goal of ecosystem management is to maintain natural ecological patterns over time (sustainability), pattern persistence becomes the focus of management. One of the major consequences of the hierarchical organization of ecological systems is that nonequilibrium dynamics or spatial heterogeneity at one scale can be translated into equilibrium at a higher scale (O'Neill et al. 1986, Urban et al. 1987, Levin 1992). Patterns persist within a hierarchical framework; a pattern may be stable at one scale but not at another (Rahel 1990). Therefore, ecological pattern should be analyzed at more than one scale (Rahel 1990) and land management planning should consider all scales of ecological organization (Baker 1992a, Levin 1992, Urban et al. 1987). The first issue to address in defining any management strategy, after the definition of the pattern and processes of interest, is whether all processes and environmental constraints that generate the pattern are incorporated within the managed area. A landscape will be in a shifting mosaic steady state (sensu Borman and Likens 1979) if the area is large enough to encompass all aspects of the processes (including disturbances such as fire) that generate the landscape mosaic (Baker 1989, 1992a, Shugart and West 1981, Urban et al. 1987). The principle of defining landscape boundaries that include the full regime of disturbances and processes is called "incorporation" by Urban et al. (1987).

In practice, this problem of scaling landscape boundaries to reflect the size of disturbance regimes requires knowledge of temporal and spatial fluctuations in key processes and disturbances (Baker 1989, 1992a and 1992b). For example, in a large conservation area which is a subset of a larger fire-dominated landscape in northeastern Minnesota, Baker (1989) found that there was no spatial scale at which the environment within the study area would be in a temporally stable patch-mosaic because of temporal fluctuations in the fire regime (i.e., landscape structure fluctuates significantly over time). A consequence of this observation for ecosystem management is that management for patterns and processes requires knowledge of historical natural variability in the disturbance regime (Baker 1989, 1992a and 1992b, Swanson et al. 1993, Turner et al. 1993). For example, a management plan to re-establish fires should consider the frequency, intensity, and timing of historical fires as well as the size, shape, and location of burned areas and the distribution of these attributes over large areas (Baker 1992a; Hann et al. 1993). Any strategy to restore fire regimes to presuppression levels, however, should consider how much the present landscape structure deviates from presuppression times (Bonnicksen and Stone 1985) and the possible impact of unusually large fires (e.g., Yellowstone National Park) which could have a detrimental effect on the restoration project (Baker 1992b). Baker's (1992b) simulation of fire dynamics in the northeastern Minnesota landscape shows that in his study area, the re-establishment of presuppression patterns of natural landscape variability did not require structural restoration of the landscape. Furthermore, presuppression patterns could be re-established faster with large fires than with small fires. With this knowledge, ecosystem management can focus on perpetuating patterns of natural variability.

From a regional perspective, some processes will never be included in the boundaries of an analysis area. For example, global climatic changes might induce changes in biome boundaries and also in the frequency and size of suitable habitats (Neilson et al.1992). Global changes and their impact on landscape patterns, processes, and disturbances are often not included at the highest levels of the ecological systems used by land managers. Historically, some land managers believed that the best philosophy for resource management and conservation was that of no interference (e.g., the National Park Service policy). But because human activities are likely to have altered the structure of the environment and of biotic processes (e.g., migration of species), it is necessary to consider active manipulation of patterns and processes and the impact of such manipulation on species, communities, and ecosystems. As Neilson and others (1992) have pointed out, land managers will need to become observers and facilitators of change to stay within the range of patterns of natural variability.

## A Hierarchical Approach to Monitoring Patterns, Processes, and Diversity

One of the most natural ways to describe changes occurring in a system ,is to monitor such changes for a long period. A direct consequence of the hierarchical organization of ecological systems is that there is neither a single appropriate scale at which to monitor nor a single attribute to monitor. Noss (1990) described a 10-step method to monitor diversity in an entire region. Noss also suggested a list of indicators at four levels (landscape, community, species, and genetic) to monitor changes in ecosystem composition, structure, and function. An indicator is defined as a surrogate measure for assessing the patterns and processes at a given scale.

Two critical questions regarding monitoring patterns and processes are: Is there a hierarchical structure within the ecological system of study? and Are the hypothesized mechanisms valid for explaining the patterns in terms of processes? Because there is no single hierarchy fitting all purposes, there is also no single monitoring scheme. Therefore, an ecological hierarchy should be defined for a particular management purpose and ecological system, and a scheme should be designed to monitor all appropriate attributes of patterns and of the key mechanisms at relevant scales.

For example, in the case of the northeastern Minnesota conservation area, Baker (1989) noted that active management for re-establishing natural fire regimes could result in landscape structures that are not adequate for maintaining viable moose populations. If maintaining natural fire regimes and viable moose populations are management objectives, then two ecological hierarchies (one for each purpose) must be defined across their respective ranges of spatial and temporal scales, and two different hierarchical monitoring schemes should be designed. As Baker (1989) noted, if fire management is aimed at producing landscape mosaics within the range of natural variability but monitoring shows the landscape mosaic is no longer suitable for viable moose populations, action needs to be taken. Action could consist of temporary changes in the fire regime to re-establish a “moose-suitable” landscape mosaic. Appropriate monitoring of the two ecological hierarchies can also indicate whether both objectives can be met at the same time within the study area, as well as the management implications and cost. The important point of this example is that two different monitoring schemes are needed to make any ecologically meaningful and informed decision. The fire-moose mosaic example in the Minnesota landscape is similar to the investigation conducted in Yellowstone National Park on the impact of patch mosaic structure on elk populations..

These examples demonstrate the complexity of monitoring changes in ecosystem development. This complexity is confounded by three characteristics that have puzzled ecologists and land managers, and hindered the development and implementation of useful monitoring schemes. First, processes such as fires, floods, and insect outbreaks affect patterns at different scales and often interact with each other in a nonlinear fashion (i.e., they express relations that are not strictly proportional). Patterns constrained by these processes will also exhibit nonlinear behavior (Holling 1992). Second, ecosystem development may be discontinuous; that is, the ecosystem goes from one state to a very different state, sometimes without warning. A good example is the irreversible change from a savanna to woody vegetation induced by cattle grazing (Walker et al. 1981). When the change is abrupt, it is called a catastrophe. Third, ecosystems can develop along multiple pathways. This fact is well documented in forest succession (e.g., McCune and Allen 1985a, 1985b). These three characteristics often make ecosystem development unpredictable. These observations on ecosystem development are in agreement with major theoretical developments in the study of nonlinear systems that include chaos theory (e.g., May 1976).

Recent advances in nonequilibrium thermodynamics, or dissipative structure (Nicolis and Prigogine 1989, Kay 1991), provide two insights relevant to ecosystem management and the monitoring of ecosystem changes. First, the concept of integrity (i.e., the ability of an ecosystem to maintain its organization) cannot be captured by a single characteristic (Kay 1991). Integrity must be recognized as a multidimensional and multiscaled concept, and monitoring schemes must reflect this fact. The definition of integrity must also include an anthropocentric component. This component sets bounds to the type and amount of change that are acceptable to society (Kay 1991).

Second, a monitoring scheme must also monitor change in the attributes of interest "at a rate that is significantly faster than the rate at which the effect occurs" (Kay 1991; see also Holling 1986). As theory and observations indicate, rates of change in ecosystems can "accelerate or decrease very dramatically with little or no warning" (Kay 1991). As these rates change, ecosystems can move from one state to another in a seemingly unpredictable or catastrophic way. Spruce budworm outbreaks provide a good example of such behavior (Holling 1988, 1992). Holling (1986) defines surprise as something big happening between two sampling periods (i.e., the monitoring rate is too slow). Monitoring schemes based on multiple attributes at different scales provide a basis for assessing ecosystem changes (i.e., the loss of integrity). Knowledge of the historical ranges of natural variability can also help reduce surprises, but will not eliminate them. As Kay (1991) states, "any human systems that are meant to deal with ecosystems (or any dissipative systems) must be adaptive in their response, that is able to cope with surprise."

## CONCLUSIONS

The major consequence of the hierarchical nature of ecological systems is that any management decision is likely to have an effect at several scales of ecological organization. Ecosystem management recognizes the multiscale nature of ecosystems and uses this knowledge to ensure the persistence of ecological patterns at all relevant scales. The process of formulating management guidelines for sustaining ecosystems should be guided by eight central principles:

- Management goals must be defined precisely.
- Ecological hierarchies must be defined according to management goals.
- Ecological patterns and diversity must be understood in terms of processes and constraints generating them, as well as in terms of their possible impact on other components of ecosystems.
- The implications of management practices on patterns and processes must be understood at all scales of the hierarchies.
- Management for sustainability of ecological patterns and diversity must include maintenance of all ecosystem attributes across their natural ranges of spatial-temporal scales.
- Ecosystem management must be concerned with the sustainability of patterns and processes together rather than merely the maintenance of existing patterns.
- The historical range of natural variability across a range of spatial-temporal scales must be defined if patterns and processes are to be maintained at all appropriate scales of organization (e.g., ecological and evolutionary). The role of natural variability should be recognized in the development of management plans.
- Monitoring schemes must be designed that explicitly recognize the hierarchical nature of ecological systems. Monitoring multiple attributes at all appropriate ecological scales can provide a basis to assess ecosystem change.

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# Ecological Theory in Relation to Landscape and Ecosystem Characterization

**P.S. Bourgeron, H.C. Humphries, R.L. DeVelice, and M.E. Jensen**


## ABSTRACT

Maintaining healthy ecosystems is a priority management objective. This paper highlights applications of ecological theory to predict ecosystem responses to management treatments. We propose to characterize the following four ecosystem components: the biotic component; the abiotic component; biotic-abiotic interrelations; and ecosystem properties. These characterizations provide, respectively: definition of spatial and temporal patterns of species and communities; definition of patterns of bioenvironments and physical environments; correlations between biota and bioenvironments or physical environments; and definition of spatial and temporal patterns in ecosystem structure, composition, and function. The models developed through these characterizations will allow land managers to predict biotic responses to management treatments.


## INTRODUCTION

Ecosystem management relies on knowledge of the relations among the biota--individual species or communities of plants or animals--and environmental factors (such as climate, soil, and water). This knowledge is used to make predictions about the response of ecosystems and landscapes under various management scenarios. When data are adequate, ecosystem management should be guided by a specific ecological model relating the biota to the environment for the problem at hand.

Two important characteristics of landscape evaluation and ecosystem characterization are that the recognition of patterns and environmental correlation analysis depend on the survey data used, and process modeling and experimentation depend on the hypotheses generated by pattern analysis. Although the notion of survey may seem irrelevant to ecosystem management, it is how data are collected for identifying relevant patterns, ecological scales, and disturbance regimes (Milne, Minshall, Turner et al. 1993). Recognition of temporal and spatial scales (Allen and Starr 1982, Bourgeron and Jensen 1993, Levin 1992, Urban et al. 1987, Wiens 1989) depends on survey data (see Milne 1993).

The idea that surveys are objective or void of assumptions, or both, about the phenomena that the data seek to explain is unrealistic. Scientists have an implicit mental model of the relations between the different components of the systems they study (Austin 1991). This knowledge is used in the routine procedures followed by the field practitioner, whether explicitly stated or implicit. Cost-effective procedures for data collection need to take into account ecological interactions at various scales (Austin 1991, Bourgeron et al. 1993, Gillison and Brewer 1985, Mackey et al. 1989). Therefore, a few basic theoretical concepts of ecosystem structure, composition, and function influence every step of the process of characterizing ecosystems and ecological relations, from data collection to interpretation (Austin 1987). These theoretical concepts need to be clearly defined and continually reassessed in light of new developments.

For ecosystem management, issues in the following four areas of ecological theory should be considered: the biotic component of ecosystems; the abiotic component; biotic-abiotic relations; and ecosystem properties. The treatment of issues in these four areas has a serious effect on selecting the best sampling design and the most appropriate measurements for landscape evaluation, the best techniques for correlating the ecosystem and its component species to the environmental factors, and the best methods for predicting ecosystem properties and their responses to various scenarios of management. This paper reviews specific issues in these four areas of ecological theory and their effects on ecosystem management.

## Biotic Component

The biotic component of ecosystems is defined by species patterns over time and space. For the dual purpose of clarity and relevance to broadscale landscape evaluation, the following discussion will use vegetation as an example. For specific projects, however, the theoretical issues of species distributions and assemblages also apply to animal patterns. The following questions must be explicitly addressed in land evaluation and ecosystem characterization efforts: Are species assemblages temporary and fluctuating phenomena along regional gradients

--the so-called “continuum” concept? Or, can the vegetation be summarized by discrete entities, repeatable species assemblages found and maintained in discrete habitats with characteristic properties--the so-called “community” concept?

Although viewed as irrelevant by some, this conflict in theory has far-reaching implications for ecosystem management. If the continuum concept is correct, the problem of characterizing the biotic component of ecosystems is to identify which model of distribution each species follows and to define its particular set of responses to the environment. If the community concept is correct, the problem revolves around identifying the full set of environmental factors shaping the community. Additionally, if distribution data for all species in an area were available, would communities still need to be delineated? If communities do not need to be delineated, should the biotic component of ecosystems be characterized species by species? Also, what procedures should be followed to characterize species distribution patterns? Finally, what would be the management value of existing vegetation, vegetation-site, ecological, or ecosystem maps used worldwide for landscape evaluation and ecosystem management (e.g., Pojar et al. 1987, Zonneveld 1988b; see also the literature review and examples in Kuchler and Zonneveld 1988).

The present position adopted by scientists seems schizophrenic. Most researchers implicitly accept the continuum concept, even avoiding the term community and referring to the more neutral term, “species assemblage” (Austin 1991). The same researchers, cartographers, and other practitioners continue to recognize homogeneous areas of vegetation, implicitly using the community concept for pragmatic and practical purposes (such as fieldwork and delineation of mapping units). The most detrimental aspect of this practice is that surveying the land with such an attitude does not lead to the formulation of useful ecological models.

Austin (1985, 1991) and Austin and Smith (1989) review two aspects of the topic and offer conclusions pertaining to landscape evaluation and ecosystem characterization. They conclude that the continuum view of vegetation includes the following three general alternative distributional models of individual species: species are individually distributed without any pattern (the original individualistic concept of Gleason 1926), major species are regularly distributed along a complex environmental gradient, and other species are individualistically distributed (Gauch and Whittaker 1972); and when the vegetation is stratified, each stratum partitions the gradient with species regularly distributed but each stratum varies independently of the others (Austin 1985, Goodall 1963). The last two models lead to identifiable community patterns.

Austin and Smith (1989) also state that communities often are recognized at different spatial and temporal scales by different criteria. For example, at the landscape scale, the concept of spatially delineated plant communities or associations made up of co-occurring species on a specific site is commonly used; at the regional scale, the concept of floristic or biotic province--defined by the distribution of species with similar evolutionary history--is used. Furthermore, classification systems have been built by using different criteria for the vegetation (floristic, physiognomic), by considering to varying degrees the dynamics of the vegetation and its relations with the environment, and by spanning a range of scales from a stand to a region (Ellenberg 1956, Kuchler 1988).

In defining the biotic component of ecosystems, finding a practical approach is hindered by two limitations: existing evidence does not allow testing among the various continuum alternatives (Austin 1991); and very weak and ambiguous evidence for community characterization (from an ecosystem perspective) has resulted from unspecified or inconsistent criteria, vague definitions of key concepts, unspecified minimal areas of references, and undocumented sorting strategies (see various chapters in Kuchler and Zonneveld 1988, Whittaker 1978).

At the continental-global scale, the concept of the floristic or biotic province has been used to recognize large land units. This scale (among regions) leads to the delineation of regions assumed to be internally homogeneous. Biotic provinces (e.g., Dice 1943) are reasonably discrete large areas, usually at the scale of a region, with characteristic physiography, climate, vegetation, flora and fauna. This category includes ecoregional frameworks (e.g., Bailey 1976, Bailey et al. 1993, see discussion in Kuchler 1988, Omernik 1987, Walter 1979, Zonneveld 1988a) that use climate and landscape characteristics (e.g., landforms) combined with vegetation data to produce delineations of regional ecosystems. These regional ecosystems provide the context for ecosystem characterization at the landscape scale.

At the same continental-global scale, floristic provinces or elements are defined by recurrent patterns of plant distribution that reflect similarities among species in their evolutionary histories and ecological tolerances

(McLaughlin 1986, 1989). In contrast to biotic provinces, floristic elements are not discrete. Each area (plot, stand, landscape) is likely to contain representative species of several elements. The change in the percentage of representation of each element is similar to the change in species composition along geographic or ecological gradients, or both, whether regional or local. The interpretation of these assemblages of various types is used to document the range of past environments and ecological events, usually in a large area.

Some systems use concepts borrowed from both biotic and floristic provinces. Such hybrid systems include the biome-based system of Brown and others (1979) and the regional vegetation scheme that Kuchler (1967) used for a map of the United States. Both are employed by government agencies for landscape assessment of terrestrial regional ecosystems.

At the landscape scale (within region), landscape characteristics and individual species distributions along gradients combine to produce species assemblages. Distributions of species along a mountain transect are shown (fig. 1); their distribution along the one-dimensional elevational gradient is in figure 1A (data adapted from the transect study of Whittaker and Niering 1965), and the frequency of communities determined by the co-occurrence of the species along the transect is shown in figure 1B. Of the nine species combinations (figs. 1A and 1B), five (BC, QA, QA-ME, ME, and ME-PP) could be defined as communities (or assemblages or associations) because they occur with reasonable frequency; four (PJ, PJ-BC, BC-QA, and PP) might be considered ecotones or insignificant because of their very small frequency of occurrence. In terms of the elevational gradient, the species form a continuum of individualistic and overlapping distributions, regularly replacing each other along the gradient.

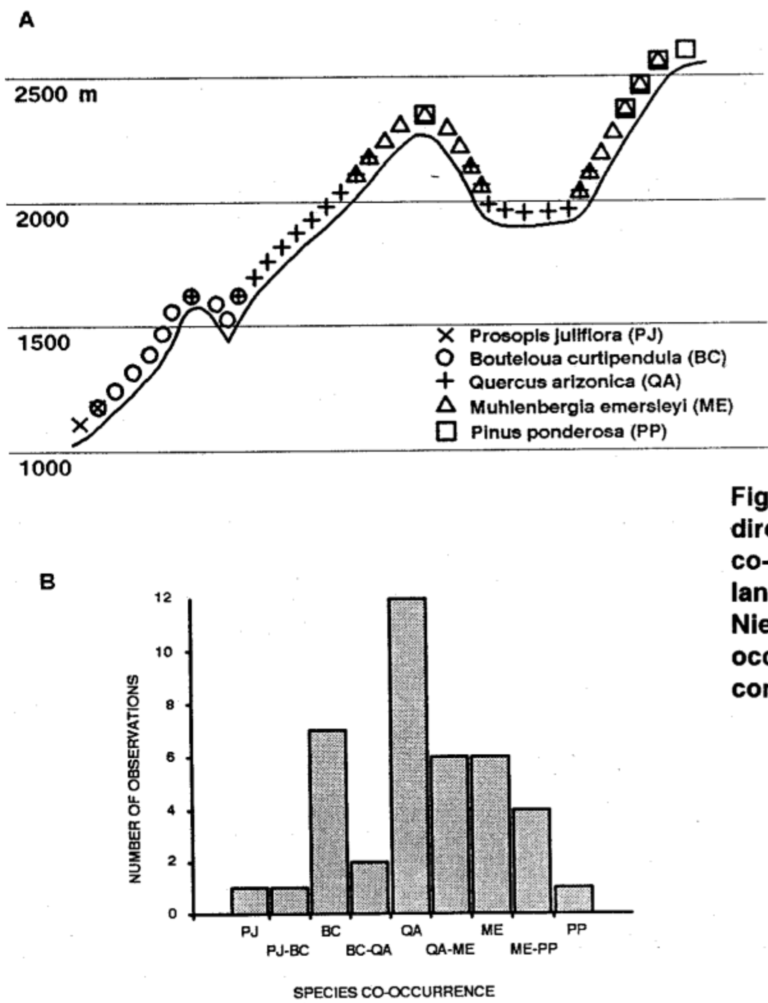


**Figure 1A--Species distribution along a directional transect showing the spatial co-occurrence of species as a function of landscape pattern (data from Whittaker and Niering 1965); Figure 1B--Frequency of occurrence of species groups used to define communities.**

Austin and Smith (1989) and Austin (1991) observed that the frequency of combinations of species often is used as a criterion to recognize communities. If the landscape configuration were as in figure 2A, which differs from figure 1A in the location and extent of mid-elevation peaks and plateaus, the frequency of species associations would change (fig. 2B). Combination QA-ME would now be considered an ecotone instead of a community, as in figure 1B; combination BC-QA would be considered a community instead of an ecotone. The salient features of this observation are that the occurrence of a species combination is a consequence of a particular landscape pattern, and that the frequency of species combinations is also a function of the landscape pattern.

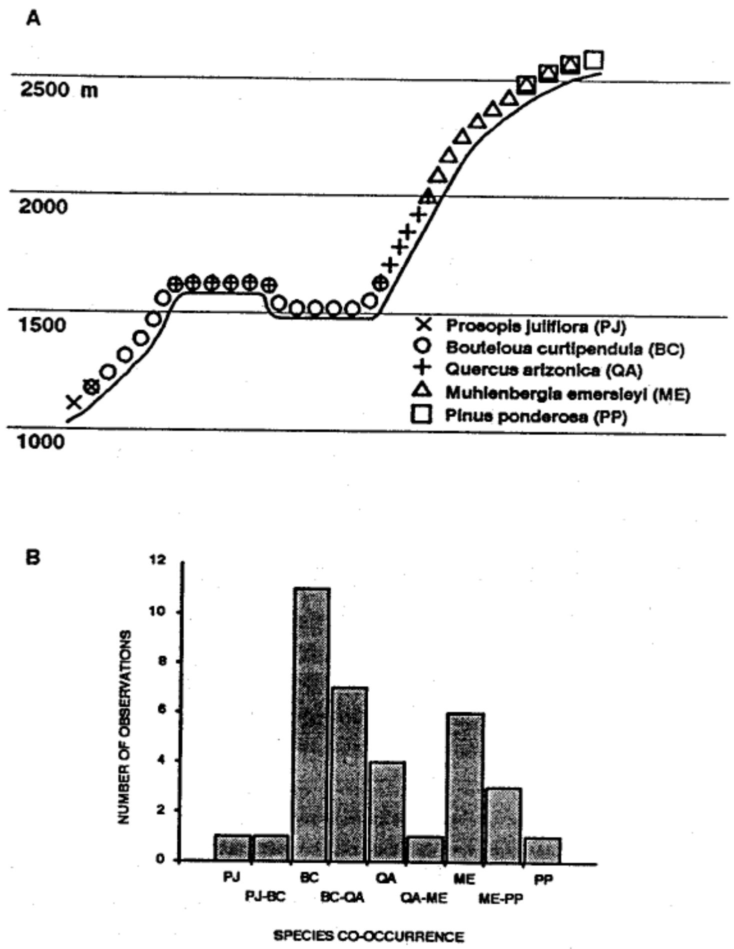


**Figure 2--Species along a transect. Species and the elevation gradient are the same as In figure 1. Landscape pattern is different (see text).**

The vegetation component of the ecosystem is thus characterized by the link between distribution patterns of individual species, their occurrence in landscape features, and the distribution of the landscape features. Therefore, various aspects of both the continuum and the community views of vegetation complement rather than exclude each other (Austin 1991, Westhoff and van der Maarel 1978). Species can be individually distributed along gradients (unidimensional or complex), following any of the possible models (Austin 1987, Austin and Smith 1989). The pattern of distribution of the landscape features controlling environmental factors constrain the pattern of species combinations, their distribution in the landscape, and their frequency.

The consequences of these facts for landscape evaluation and ecosystem characterization are as follows. The concept of a floristic or biotic province applies to a region with a characteristic pattern of climate and landscape characteristics (e.g., landforms), which combine with vegetation data to produce a delineation of regional ecosystems (Bailey et al. 1993). At the continental-global scale, attempts to define guidelines for terrestrial regional ecosystem management can be successful and useful only if vegetation units defined as biotic or floristic provinces (or any variant such as the systems used by Brown et al. 1979 and Kuchler 1967) are modified to correlate with regions defined by climatic and landscape patterns (Bailey and Hogg 1986, Burger 1976, Kuchler 1988, Rowe 1980). A primary purpose of such regionally defined ecosystems is to serve as a reporting structure for information about regional resources and environment (Bailey and Hogg 1986, Bailey et al. 1993). Another purpose is that these regional ecosystems define homogeneous regions within which finer scale ecosystems can be characterized.

As biotic components of finer scale ecosystems, plant communities or associations are landscape properties (Austin and Smith 1989, Westhoff and Van der Maarel 1978). The continuum concept applies to the resource-environmental space, not to a simple geographical space unless the two are highly correlated (Austin 1991). The concept of a community defined as a recurrent pattern of co-occurring species is relevant to a pattern of environmental variables or natural landscape features. Attempts to characterize landscape-scale ecosystems and conduct landscape evaluation by using such communities can be successful only if the intracommunity pattern of gradual changes is correlated with gradual changes in the environment. Effective landscape surveys need to take into account this dependence of biotic patterns on abiotic patterns (Bourgeron et al. 1993).

Communities may be distributed over large areas, sometimes in several regions (e.g., the spruce fir forests of the Rocky Mountains). Ecosystem characterizations should not assume that all occurrences of a community have identical properties. With numerous species in a community, the individual distribution of species ensures gradual intracommunity changes along regional gradients. Occurrences of the same community may have different species composition, and size of areas where it is found, thereby resulting in different properties important to ecosystem management. The same community found at different locations along a regional gradient or in different climatic regions would likely respond differently to a specific conservation-management practice. Species distributions need to be established with data spanning the range of environmental variability that they are distributed along. Communities should be defined by using data covering preferably the range of environmental variability in all landscapes in which they occur. Effective ecosystem characterization requires adequate replication within communities to allow for and detect geographical variability.

## Abiotic Component

To be functional, ecosystem characterization and landscape evaluation must include plant and animal habitats as well as the organisms themselves. Austin (1985), Austin and Smith (1989), and Austin and others (1984) make the point that three types of environmental variables or gradients influence species distribution: indirect factors that have no necessary physiological influence on the species components of the ecosystems, such as elevation; direct factors that have a direct physiological influence but are not consumed as a resource, such as temperature and pH; and resource gradients that can be used directly by species, such as nutrients. In principle, the definition of habitats for ecosystem management should be based on the most proximal variables that can be measured or estimated.

Theories and methodologies used to define the abiotic component (indirect factors, direct factors, and resource gradients) fall into three major categories: delineation of biological communities as surrogates for the environment; delineation of landscape units containing recurrent patterns of landforms and landscape characteristics; and identification of bioenvironments (i.e., classes in environmental variables that take into account key ecological interactions and processes).

The first approach, applied worldwide, uses the vegetation as a surrogate for the environment. It is based on the assumption that vegetation is a faithful expression of site characteristics (e.g., Troll 1941, 1943, 1955, 1956; see discussion in Kuchler 1988). Kuchler (1988) states that “indeed, mapping the vegetation is the only effective method to present the ecological order of our living space.” When testing the value of an ecological relation for ecosystem management, arguments about assessing the correlations between the biological units and the environmental factors can become circular, with the focus being switched between biotic and abiotic factors (Mackey et al. 1988).

The second approach has been to use broad environmental patterns alone or broad correlations between vegetation and environment to describe and delineate habitats of both plant and animal communities. Classifications have been developed from climatic attributes, either alone or in conjunction with other attributes (e.g., Austin and Yapp 1978, Bailey 1976, Bailey et al. 1993, Omernick 1987, Walter 1979). A great variety of systems based on combinations of soils, lithology, and landforms have been used alone or combined with vegetation data to produce classifications of biophysical regions or natural landscape units. In Canada, Rowe and Sheard (1981) detailed a landscape system for identifying units of lands that are meaningful at the ecosystem scale.

The main weakness of this approach is that it relies too heavily on indirect factors (such as soils and landforms) without explicitly stating the ecological relations between biotic and abiotic components. For example, landform patterns often have been used to stratify areas into natural landscape units on the basis of a single attribute. The units are argued to represent natural assemblages of ecosystem integration with respect to environmental regimes and key processes, for which compelling evidence exists (Swanson et al. 1988). Geomorphic pattern, through erosion and sedimentation processes, has been shown to control carbon, nitrogen, and phosphorus cycles in soils of riparian forests in southern France (Pinay et al. 1992). The actual test of the strength of the hypothesized relation between the biotic and abiotic components of ecosystems is not always performed. For example, Kolvachik and Chitwood (1990) used geomorphology in addition to a floristic classification of the vegetation of riparian zones in central Oregon but did not explicitly test the purported relation of vegetation to geomorphological processes.

On the other hand, the strength of this approach is that it allows a direct analysis of the spatial and temporal scales of landscape features (Delcourt and Delcourt 1988, Urban et al. 1987), which is necessary to match patterns and processes (Levin 1992). For example, in a mountainous landscape in northwestern Montana, Lathrop and Peterson (1992) tested whether watershed morphological characteristics and ecological processes exhibited the same basic properties at various spatial scales (self-similarity). They established structural self-similarity, but did not conclusively demonstrate self-similarity for ecological properties. Establishing a relation between landscape structure and ecosystem functional attributes across a range of spatial scales has important implications for the proper scaling of process models at landscape to regional scales (Turner et al. 1989).

A third approach recently has been developed in Australia out of concern for the problems outlined above. This approach is based on the argument that, to be meaningful, ecological evaluation--and its corollary ecosystem management--should be based on species' niche-habitat relations (e.g., Brown 1984, Hutchinson 1959, Nix 1982, Whittaker 1972). The aim of the methodology is to summarize environmental variability, identify the distribution of major environmental gradients, and indicate where significant shifts in ecological variability might occur (Mackey et al. 1988, 1989).

The need is, therefore, to estimate a species' responses to a limited set of dominant environmental variables comprising primary niche dimensions (Nix 1982), such as radiation, thermal, moisture, mineral nutrient, and biotic regimes (Mackey et al. 1988, 1989; Nix 1982). Site-specific data are used to generate classes of sites sharing similar ranges of values of the environmental variables. A map of these classes, or bioenvironments, can be used alone in the assessment stage of an area for given purposes (DeVelice et al. 1993), or in conjunction with vegetation data for quantifying biotic-abiotic correlations (Mackey et al. 1989).

Sampling ecologically significant factors in the physical environment at sufficient resolution ensures that key processes and interactions can be taken into account (Mackey et al. 1988). Estimating the key attributes involves modeling of terrain-climatic interactions, including simple surface-fitting procedures, as well as models that take into account known effects of physical processes. The accuracy of the results is limited by the extent to which the processes and interactions are known.

The consequences of these points for ecosystem management are as follows. Attempts to describe patterns in abiotic factors or habitats for ecosystem management can be successful only if the ecological meaning of the selected factors is understood. Natural landscape units and bioenvironments are different: the former refer to gegraphical phenomena, and the latter to an ecological space. An actual combination of natural landscape features (elevation class X landform X geologic substrate) may be found in two different regions. A change in regional climate may change the suitability of the habitat for particular species. Therefore, when the physical environments of a region are characterized and mapped, all occurences of a mapped environment should not be assumed to be identical. With changes in climatic-terrain interactions occurring over large areas, the values of the direct factors and resource gradients to which the biota respond also change. Similar physical environments may correspond to different bioenvironments.

The accuracy and utility of describing and mapping natural landscape features or bioenvironments for ecosystem management are functions of the environmental variables selected, the relation between indirect and direct factors, the estimation procedures and mapping scales used, and the strength of the purported relation between the biota and the selected environmental criteria. Consider an example from Mackey and others (1988). Solar radiation is important as the source of energy for photosynthesis; it also provides energy for evapotranspiration, hence modifying the water balance. The effect of local topography and landform on radiation is likely to have ecological significance in places with a water deficit during a period of the year, because sheltered slopes would maintain a more favorable water balance during the dry season. Ideally the interactions between landscape features, climatic factors, and ecologically meaningful variables should be obtained by using a combination of geographical information system and simple process models. This way, land managers could generate maps of bioenvironments that may change because of management or global climate change from maps of the landscape features that do not change rapidly. Application of this approach may be limited in the short term as a result of lack of appropriate environmental data, lack of explicit predictive models linking biotic and abiotic variables, or lack of rigor in defining what constitutes a suitable area for ecosystem management and conservation. Priorities need to be established to fill the gaps in these three areas.

## Biotic-Abiotic Relations

Ecosystem management requires the capability of predicting the response of the biota to large- and small-scale changes in habitat factors. Biotic-abiotic relations must be defined explicitly. Relations among regional gradients, local processes, and aspects of community structure have been discussed mostly from the standpoint of regional climatic control on local habitat (see Neilson et al. 1989). Three aspects characterize biotic-abiotic relations: the characterization and modeling of the differential response of species to environmental gradients; the development of predictive models of species distributions using the species' response to environmental factors; and the definition of whether the biotic-abiotic relations are the same at different scales, from the site to the region (Bourgeron and Jensen, Glenn and Collins 1993).

Developing models for predicting the pattern of species distributions historically has been the realm of ecologists and biogeographers. Land managers have neglected this area, focusing instead on the community concept. Defined communities have been used in conjunction with environmental features to characterize mapping units and related properties. The latter are used in turn to make predictions about target species distributions and for landscape evaluation. Three conclusions from the previous discussion make evident the possible weakness and circularity of this approach.

- Local community composition depends to a large extent on the individual distributions of species. Interpretation of community species composition depends on the knowledge of individual species distributions along regional gradients.

- No individual species prediction can be made accurately from community occurrence alone unless adequate data allow for characterizing intracommunity species distributions.

- The ecological relations between biotic and abiotic ecosystem components need to be explicitly stated and tested. Part of the problem is that a clear link among scales, patterns, and processes is generally not established a priori. Another problem is failure to take into account temporal variability (Swanson et al. 1993). For example, Levin (1992) points out that in systems with localized disturbances, local dynamics are predictable only in terms of long-term statistical averages.

Several models have been proposed to predict the distribution and abundance of species among similar communities. These models are based directly (e.g., Brown 1984) or indirectly (e.g., Hanski 1982, Levins 1969) on the concept of the niche (Hutchinson 1959). The models do not incorporate the direct analysis of the interactions between the biota and habitat factors. Instead, the expected response of species to environmental gradients (the so-called fundamental niche) is used to make predictions about where species will occur, and what would happen on a given site if site conditions change (e.g., as a result of management). Although much theoretical work has been conducted on the prediction of the occurrence and abundance of species using niche-habitat relations, relatively little work has addressed the actual shape of the response of the species to the environment. Usually this response is expected to be that of a bell-shaped curve (e.g., Brown 1984, Pianka 1981). Austin (1991) and Austin and others (1984, 1990) show, however, that the species response can be more complex (see also Bradshaw 1986). This problem is far from being academic. The shape of the biotic response is included in equations that predict the distributions of species or communities, or both, and the changes occurring on sites after various management scenarios. The validity of such predictions in the context of ecosystem management rests to a large degree on the validity of the models of biotic-abiotic relations.

Previous theory has upheld the use of bell-shaped curves for responses as appropriate (Brown 1984). Recent Australian work (Austin 1991, Austin et al. 1990, Margules and Stein 1989, Nicholls 1989, 1991a, 1991b) has shown that good results cannot be achieved from modeling species or community responses to the environment solely by using bell-shaped relations. In fact, for many tree species in Australia, the predominant shape is skewed (Austin 1985, Austin et al. 1984, 1990). Some statistical models predicting the occurrence of species in relation to suitable environmental variables include cubic terms and polynomial response surfaces (Austin et al. 1990, Nicholls 1991b). Such curvilinear models appear to generate accurate predictions (Margules and Stein 1989, Nicholls 1989, 1991b).

Ecosystem management relies extensively on the ability to predict the response of a species or group of species to spatial and temporal changes in biotic processes, environmental constraints, and disturbances. Usually land managers rely heavily on the properties of communities to make such predictions. This approach is seriously limited because occurrences of a community are not invariant. Testing differences in species composition of communities and distributions of species and communities in different management areas is needed. The common application of a simple bell-shaped curve for response of the species to environmental changes may not be satisfactory. Predictive models of distribution need to be developed case by case to reveal complex biotic-abiotic relations. Work is urgently needed in biotic-abiotic predictive modeling if such models are to become part of cost-effective analyses for ecosystem management. Clear relations between the environment and the biota should be established both spatially and temporally. Work is needed on the interactions between temporal and spatial scales (Levin 1992). Results could be used for evaluating areas for different purposes and for inclusion in ecosystem simulation models.

## Ecosystem Properties

Much work in ecosystem science has concerned the use of ecosystem attribute information (structure, composition, and function) to predict various system properties of interest. Two important properties in resource management and conservation planning are species richness (the number of species) and primary production. The ability to predict these properties is part of the landscape evaluation process (Zonneveld 1988b). The most common and practical approach is to include values of the properties in the description of mapping units (Bailey et al. 1993). Two untested assumptions are made: the attribute is a clearly defined and predictable property of the characterized ecosystem; and it is invariant for that ecosystem over its known range of distribution. As is true for predicting distribution patterns of species in communities, these assumptions may not be met.

Theory regarding species diversity patterns has limited predictive power. Empirical relations have been developed at one of three scales: global (Whittaker 1972), regional (Brown 1984, Pielou 1979), or local (Grime 1979, Woodward 1987). Models relating diversity to disturbance (Huston 1979) do not have site-specific predictive power. The cumulative effects of niche relations, habitat diversity, mass effect (the flow of individuals from favorable to unfavorable areas), and ecological equivalency (the fact that different species may be ecologically equivalent to each other) has been presented and summarized in a muitiscale context (Shmida and Wilson 1985). Recent work has focused on the relation among scales (Ricklefs 1987). The dependence of local diversity on regional patterns has been shown for some biota (Ricklefs 1987) but not for others (Jackson and Harvey 1989). Due to the multiscale nature of ecosystems (Bourgeron and Jensen 1993, Levin 1992, Milne 1993, Turner et al. 1993), relations at one scale may have complex relations to structure, processes, and disturbances at other scales. Neilson and others (1989) suggest that the prediction of local diversity patterns needs to be rooted in the understanding of the hierarchy of constraints imposed by regional and local factors, as well as their mutual interactions. A spatial and temporal context should be provided for analysis of biotic diversity (Bourgeron and Jensen 1993, Hoover and Parker 1991, Whittaker 1972).

Progress has been made in developing empirical relations between diversity and environment by using curvilinear statistical models. Useful relations (Austin 1991, Margules et al. 1987, Nicholls 1991a, 1991b) include more than one environmental variable. As in the response of individual species or communities to the environment (see “Biotic-Abiotic Relations”), these relations should be derived from survey data and probably cannot be extended beyond the bounds of the data (Margules et al. 1987). Site-specific predictions can be made for particular study areas (Margules and Stein 1989).

The same problems of scales, scale interactions, and model generalization beset predictions of ecosystem primary production. Spatially explicit models of biogeochemistry have been developed through a combination of geographic information systems and terrestrial regional ecosystem models (Burke et al. 1990, Houghton et al. 1983), but their widespread use is limited by gaps in soil, climate, and vegetation data bases (Stewart et al. 1989). Models like CENTURY (Parton et al. 1987) or LINKAGES (Pastor and Post 1986) attempt to explicitly link abiotic and biogeochemical factors with primary production and carbon storage (see also Schimel et al. 1990). The problem of predicting ecosystem processes is an active research field. Schimel and others (1991) summarize current activities. They stress the need for ecosystem modeling based on a small set of critical environmental variables linked to the biotic component of ecosystems. Work is urgently needed on the development of ecosystem models that include better information about the biotic component of the ecosystem and of ecosystem response to land-use practices.

The conclusion of this analysis is that useful theoretical frameworks for predicting ecosystem properties such as species diversity and primary production over large areas are limited. This scarcity of useful theory probably extends to most properties derived from individual ecosystem parts. Previous work shows that empirical relations can be derived from survey data. These relations can be useful for ecosystem management.

## CONCLUSIONS

Ecological theory that applies to ecosystem management is incomplete and does not evenly cover all topics of interest. Most theory focuses on succession concepts (Connell and Slatyer 1977) and biogeochemical cycles (Schimel et al. 1991). Landscape evaluation and ecosystem characterization, however, depend on pattern recognition and environmental correlations at various scales (Levin 1992). For ecosystem managers, the characterization of species and community patterns, habitat factors, biotic-abiotic relations, and ecosystem properties is of immediate concern. Accordingly, this paper has focused on some aspects of these areas.

Following are practical guidelines for landscape evaluation and ecosystem characterization that are likely to result in increased accuracy of results:

- Use the community concept to characterize ecosystems. Do not expect communities to be constant over large areas in their species composition and response to management.

- Use the continuum concept and its associated idea of an environmental space to characterize the range of variability in the response of species to environmental gradients within each ecosystem unit.

- Recognize that determining natural landscape units (stratifying indirect factors) is different from determining bioenvironments (stratifying the ecological factors to which species respond directly). Models should be developed eventually to go from one to the other.

- Recognize that the utility of mapping the environment depends on clearly stated biotic-abiotic relations. Develop testable ecological relations.

- Recognize that predictive models of species and community response to the environment may require developing complex curvilinear responses that differ with each case.

- Recognize that landscape surveys usually consider only spatial variability. Temporal variability and its interaction with spatial variability need to be investigated as well.

- Recognize that the problem of spatial and temporal representation of ecosystem processes in simulation models is still under investigation. Work closely with ecosystem modelers to link results from survey data with the appropriate ecosystem model.

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# Multiscale Organization of Landscape Heterogeneity

**M.G. Turner, R.H. Gardner, R.V. O'Neill, and S.M. Pearson**

## ABSTRACT

Environmental heterogeneity is hierarchical and is controlled by different processes at different spatial and temporal scales. Recent studies have demonstrated the existence of pattern within nested discrete scales on natural landscapes. A disturbance that disrupts this structure could have far-reaching ecological consequences; however, natural disturbance-recovery regimes often create and maintain spatial and temporal heterogeneity in landscapes. A broad framework for the description of landscapes that separates the spatial from the temporal scales of disturbance and recovery can be used to predict the resultant dynamics of a landscape. This framework permits the prediction of disturbance conditions that lead to qualitatively different landscape dynamics and demonstrates the scale-dependent nature of landscape equilibrium.

Results from numerous studies suggest that landscape connectivity is important to many ecological processes. Connectivity can change rapidly when landscape heterogeneity is altered, thereby, indicating the existence of critical thresholds. Critical thresholds in habitat abundance and connectivity can be identified for a variety of organisms, but the values of these thresholds differ with both the landscape pattern and the scale at which an organism can use the landscape. It is most difficult to predict the consequences of altered landscape patterns at intermediate levels of habitat abundance because of complex interactions between pattern and scale or resource utilization by different organisms. Suggestions for maintaining landscape heterogeneity at multiple scales are presented.

## INTRODUCTION

Describing environmental heterogeneity is challenging because heterogeneity occurs at various spatial and temporal scales and is controlled by a diverse set of processes. In this paper, we discuss the hierarchical nature of environmental heterogeneity, the implications of scale-dependence for disturbance dynamics, and the consequences of landscape patterns. We also propose general concepts for land management based on the implications of the multiscale organization of landscapes.

### Multiscale Heterogeneity in Landscapes

Environmental heterogeneity is hierarchical (Allen and Starr 1982, O'Neill et al. 1986, Urban et al. 1987 and is controlled by different processes at different spatial and temporal scales (Delcourt et al. 1983). The spatial distribution of life zones on a continent, for example, is controlled by climatic factors such as precipitation and temperature (Raunkaier 1934, Whittaker 1956). Within a life zone, however, the vegetation present at a particular location varies with soil type and topography; for example, landscapes in the southern Appalachian mountains are dominated by deciduous forest, but different species assemblages are characteristic of different topographic positions. Within a given soil type and topographic condition, tree density, stand age structure, and species composition also may vary due to disturbance history.

Recent studies have tested this hierarchical paradigm and demonstrated the existence of discrete scales of pattern on the landscape. O'Neill and others (1991), following a suggestion of Levin and Buttel (1986), examined six grassland and forested landscapes. By graphing an estimation of variance against spatial extent of the sample, Levin and Buttel demonstrated that a muftiscale structure existed on four of the landscapes. O'Neill and others (1992) used spatial analysis of transect data to demonstrate three to five distinct scales of pattern on three landscapes. Later efforts confirmed this result for four additional landscapes (O'Neill et al., unpublished). Hierarchical patterning in resources could affect consumer communities. An indirect demonstration of multiple scales was published by Holling (1992). He reasoned that if resources showed distinct scales, then the size of consumer home ranges would depend on these resource scales. Following McNab (1963), Holling suggested that discontinuities in the statistical distribution of home-range sizes would appear as clusters of body sizes in vertebrates. By examining existing data sets, he was able to establish the hypothesized clustering of body sizes. Holling also provides an extensive discussion of the endogenous and exogenous processes that generate these spatial scales.

Although the terms patch, matrix, and corridor commonly are used in landscape ecology, a rigid interpretation of these terms can impede our understanding of multiscale heterogeneity (Turner et al., in press). These terms are most useful when there is high contrast between patch and matrix (e.g., agricultural fields in a forested region) and this contrast is ecologically meaningful. It is difficult, however, to define patches in a landscape without being arbitrary. Through an organism-based perspective, patches have been defined in an ecological context as a discontinuity in an ecological variable affecting an organism (Wiens 1986). Analyzing landscape heterogeneity at the scale of an organism, especially a nonvertebrate, can reveal strikingly different environmental patterns and gradients than those apparent to humans (e.g., Buechner 1989, Wiens 1989, Wiens and Milne 1989). For instance, landscape connectivity (i.e., the degree to which sites are contiguous) will be perceived differently by an ant and an eagle.

Muftiscale patterning is the result of interacting physical and biological phenomena. Landscape heterogeneity often is produced and maintained by ecosystem disturbance and recovery dynamics. The resulting patterns have consequences for several ecological processes at the landscape scale. Recognizing these patterns of heterogeneity, as well as their causes and consequences, is necessary for developing management plans consistent with preserving the ecological integrity of landscapes.

## Disturbances and Hierarchies: The Implications of Scale Dependence

Because a landscape appears to be organized as a hierarchy of discrete spatial scales of pattern, it seems likely that any disturbance disrupting this structure could have far reaching ecological consequences. Such a disturbance might disrupt the scale of pattern in the spatial distribution of resources and could eliminate an entire component of the consumer community that depends on the scale of resource distribution affected. Thus, activities such as clearcutting or urbanization can substantively alter the natural hierarchical structure of a landscape.

Natural disturbance-recovery regimes often create and maintain spatial and temporal heterogeneity in landscapes. Natural disturbances often exhibit characteristic scales in time and space. Turner and others (in press) developed a broad framework for the description of landscapes that separates the spatial and temporal scales of disturbance, thereby allowing time and space to be considered separately. Four major factors characterizing the scale dynamics of landscapes are considered: (1) disturbance frequency, as indicated by the interval between successive disturbances (e.g., Baker 1989a, 1989b, Romme 1982); (2) rate of recovery from disturbance, as indicated by the length of time required for a disturbed site to recover (e.g., Pickett and White 1985); (3) the size or spatial extent of disturbance events (e.g., Baker 1989a, 1989b, Bormann and Likens 1979, Romme 1982; Shugart and West 1981); and (4) the size or spatial extent of the landscape (e.g., Baker 1989a and 1989b, Shugart and West 1981). These factors are then reduced to two key parameters representing time and space to describe potential disturbance dynamics.

The temporal parameter (T) is defined by the ratio of the disturbance interval (the time between successive disturbances) to the recovery time (the time required for a disturbed site to achieve recovery to a “mature” stage). Defining the temporal parameter as a ratio permits evaluation of three qualitatively different states, regardless of the type or time scale of the disturbance. These states are (1) the disturbance interval is longer than the recovery time ($T > 1$), so the system can recover before being disturbed again; (2) the disturbance interval and recovery time are equal ($T = 1$); and (3) the disturbance interval is shorter than the recovery time ($T < 1$), so the system is disturbed again before it fully recovers.

The spatial parameter (S) is defined by the ratio of the size of the disturbance to the size of the landscape. There are two qualitatively different states of importance here, again regardless of the type of disturbance: disturbances that are large relative to the size of the landscape, and disturbances that are small relative to the extent of the landscape. As defined in this paper, the parameter S can range from 0 to 1. Landscape dynamics cannot be predicted if the size of the disturbance exceeds the spatial extent of the landscape because the landscape is too small to characterize the effect and recovery from disturbance.

The use of ratios in both parameters permits the comparison of landscapes across a range of spatial and temporal scales. We use the parameters to describe a landscape state-space in which the temporal parameter is placed on the Y axis, and the spatial parameter is displayed on the X-axis (fig. 1).

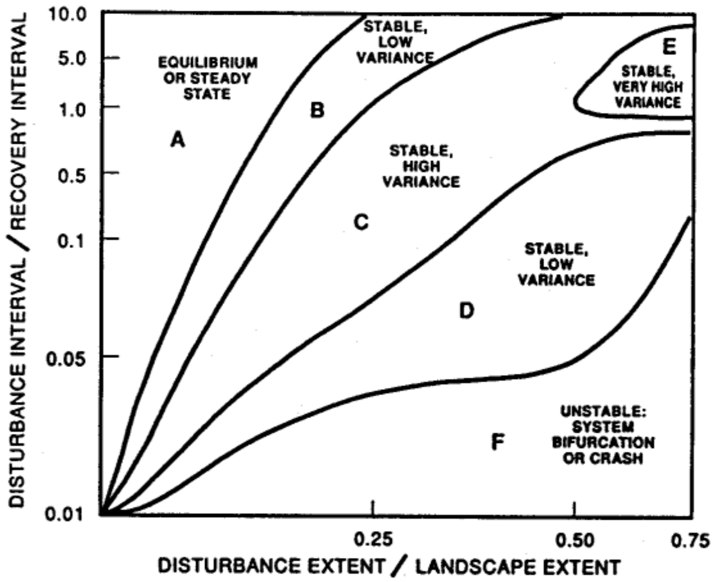


**Figure 1--State-space diagram of temporal and spatial parameters that Illustrate regions with qualitatively different landscape dynamics (from Turner et al., In press).**

A simple simulation model was developed to explore the implications of various combinations of S and T. Results indicate (fig. 1) that where disturbance interval is long relative to recovery time, and a small proportion of the landscape is affected, the system is stable and exhibits low variance over time (e.g., northeastern hardwood forests). These systems are traditionally considered to be in "equilibrium". Where disturbance interval is comparable to recovery interval and a large proportion of the landscape is affected, the system is stable but exhibits large variance (e.g., subalpine forests in Yellowstone Park). Where disturbance interval is much shorter than recovery time, and a large proportion of the landscape is affected, the system may become unstable and shift into a different trajectory (e.g., arid ecosystems with altered fire regimes). This framework permits the prediction of disturbance conditions that lead to qualitatively different landscape dynamics and demonstrates the scale-dependent nature of landscape equilibrium.

Scale-dependent disturbance dynamics have several important implications for land management. First, there is no spatial extent that can guarantee landscape equilibrium. Increasing spatial extent should, however, decrease the probability of a dramatic shift in landscape dynamics due to a rare disturbance event. Second, if the temporal or spatial scale of disturbance regimes are altered sufficiently (e.g., by climate change or land management), dramatic changes in landscape patterns are likely. Past climatic changes of small magnitude have caused significant changes in fire regimes in forested landscapes (Clark 1988, Hemstrom and Franklin 1982). Global warming may result in an increase in the frequency of dry years and, hence, an increase in the size or frequency of fire (Flannigan and Harrington 1988, Romme and Turner 1991, Sandenburgh et al. 1987. One could explore the implications of changes in a disturbance regime by locating the current position of a landscape in figure 1, then plotting a potential position within the state-space under a new disturbance regime. In this manner, the potential for a qualitative shift in landscape dynamics (e.g., from equilibrium to stable with high variance) could be identified. A landscape might, however, sustain a substantial change in disturbance regime, but remain within the same region of dynamics. Third, results of our model demonstrate the scale-dependent nature of landscape equilibrium. Conclusions regarding the apparent stability of a landscape are appropriate only for a specked spatial and temporal scale. Failure to recognize scale dependence can lead to sharply different interpretations about the same dynamics.

## Consequences of Landscape Patterns

Spatial patterns in the landscape may influence a variety of ecological phenomena (Turner 1989) such as the distribution and persistence of populations (Fahrig and Paloheimo 1988, Van Dorp and Opdam 1987), the horizontal flow of materials such as sediment or nutrients (Kesner and Meentemeyer 1989, Peterjohn and Correll 1984), the spread of disturbance (Franklin and Forman 1987, Romme and Knight 1982, Turner 1987, Turner et al. 1989), or net primary production (Sale et al. 1988). Heterogeneity in the landscape can increase gamma diversity by increasing the number of different habitats available. Excessive levels of heterogeneity, however, can result in the loss of species sensitive to habitat fragmentation. Heterogeneity, therefore, must be considered in the content and scale of a particular process or organism. For example, spatial heterogeneity as measured by variation in the type and phenology of food sources could provide a varied, nutritious diet for bears, but increasing spatial heterogeneity by adding unsuitable habitats, such as roads, would not enhance the bear population. In general, the risk of losing biodiversity and disrupting ecological function is greatly increased when natural patterns of heterogeneity are altered.

Results from numerous studies suggest that threshold of connectivity is important to the dynamics of many ecological processes including spread of disturbances (O'Neill et al. 1992, Turner et al. 1989), utilization of resources (O'Neill et al. 1988), and the movement and dispersal of organisms (Gardner et al. 1989, 1991). Landscape connectivity depends, however, on the ability of organisms or processes to move across the landscape. A plant with wind-dispersed seeds is more likely to colonize a small apparently disconnected cluster of habitats than is a heavyseeded plant that lacks a mechanism for long-range dispersal. Similarly, a river or highway might be a barrier to movement for a mouse, but a bird or deer might regularly cross such obstacles.

Critical thresholds in habitat abundance and connectivity can be identified for many organisms, but the values of these thresholds will differ with both the landscape pattern and the scale at which an organism can use the landscape (Pearson et al., in press). A series of simulation experiments conducted with hierarchically generated landscape patterns suggest that when suitable habitat or resources are abundant (e.g., > 80 percent of the landscape), neither landscape-level heterogeneity nor resource utilization scales are important; however, when suitable habitat is less abundant on a landscape, patterning and resource utilization scales become increasingly important. Simulation results suggest that fine-scale fragmentation of habitat poses a greater risk to landscape connectivity than the same percentage reduction of habitat distributed in a more coarse pattern. These results also suggest that the greatest opportunities for improving land management occur at low or intermediate levels of habitat abundance. It is most difficult to predict the consequences of altered landscape patterns at intermediate levels because of complex interactions between pattern and resource utilization scale.

## CONCLUSIONS

The recognition of hierarchical structure in landscapes, the effects of disturbances at different spatial and temporal scales, and the scale-dependent effects of heterogeneity requires new perspectives on land management. The following suggestions, originally geared toward maintaining biodiversity in managed landscapes (Pearson, et al., in press), should be useful for maintaining the integrity of landscapes across multiple spatial and temporal scales:

- View the landscape as a whole and use landscape-level indices to measure pattern at multiple scales. Do not focus solely on single, simple concepts like patches and corridors, and recognize that these concepts are scale-dependent.

- Match exploitative or disruptive activities to the natural patterns of heterogeneity. Do not disrupt natural processes such as fire or flooding that create and maintain heterogeneity. Attempt to maintain natural levels of heterogeneity in space and time.

- Maintain connectivity in the landscape by keeping the amount of native habitat in a landscape above potential thresholds of connectivity or by imposing coarse-scale structure on the landscape, or both.

- Be aware of the potential importance of crossing a critical threshold. Small changes in habitat abundance and pattern can suddenly fragment an otherwise well-connected landscape at some (but not all) resource utilization scales. Similarly, small changes in the spatial or temporal scale of disturbance or recovery dynamics can qualitatively change the overall stability of a landscape.

Coarse-grained patterning may have a less deleterious effect on organisms than fine-grained patterning because habitat connectivity can be maintained with less habitat if the habitat has more continuous acreage.

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# Natural Variability--Implications for Ecosystem Management

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## ABSTRACT

Information on the historical variability of ecosystem conditions and the natural disturbance regimes that influence such variability is increasingly used in design of ecosystem management systems. The rationale for this approach is, in part, that species have adapted to habitat and disturbance conditions of previous millennia, and increased deviation from those conditions is likely to result in increased risk of species loss and other undesireable ecological change. Use of information on natural variability is challenged by (1) limits in our ability to interpret past ecosystem variability, (2) effects of environmental conditions (e.g., climate change, exotic species, and engineered structures) on ecosystems that move those systems outside the range of historical variability, and (3) limited public participation in formulating ecosystem management and using information on natural variability. Despite these difficulties, the concept of natural variability is finding important application in ecosystem assessment and design. Broad-scale, cursory analysis of ecosystem dynamics complements intensive analysis in areas of 10,000 to 100,000 acres. Extensive analysis gives a broad geographic context, and intensive analysis provides detailed knowledge of a longer record including low to moderate severity events.

## INTRODUCTION

This paper examines the use of information on natural variability of ecosystems as a guide for ecosystem management (Overbay 1992, Society of American Foresters 1993) in Pacific Northwest forest landscapes. We use the term “natural variability” to refer to the composition, structure, and dynamics of ecosystems before the influence of European settlers. In this sense, natural variability is synonymous with other terms used in referencing natural variability in baseline conditions, such as “historical”, “pristine”, “prehistoric”, “prewestern technological man” (Kilgore 1987), and “primeval.” Natural variability can be characterized by: the range of ecosystem conditions (also referred to as “states”; sensu Brooks and Grant 1992a, 1992b), such as the extent of particular seral classes of vegetation, and by the disturbance regime (defined in terms of frequency, spatial arrangement, and severity of disturbances) that produced such conditions.

This paper adopts the perspective that managing an ecosystem within its range of natural variability is an appropriate path to maintaining diverse, resilient, productive, and healthy systems. It is also the most scientifically defensible way to meet society’s objective of sustaining habitat to maintain viable populations of native species, as stipulated in the National Forest Management Act, Endangered Species Act, and associated regulations. Our discussion addresses management of Federal forest lands as well as other lands where sustaining native species and ecosystem productivity are important objectives. Although we draw on many examples from terrestrial ecological research, we advocate a landscape perspective spanning upland areas as well as stream and riparian networks. Natural variability can be characterized and applied at spatial scales ranging from individual forest stands to landscapes or watersheds covering thousands to millions of acres.

A key premise of ecosystem management (based on natural variability) is that native species have adapted to and, in part, evolved with the natural disturbance events of the Holocene (past 10,000 year) environment. Accordingly, the potential for survival of native species is reduced if their environment is pushed outside the range of its natural variability. This rationale derives from recent developments in conservation biology and other fields of science and from judicial interpretations of efforts to apply ecological principles (sensu Craig 1987) to management of natural resources. Numerous ecological studies emphasize the close dependence of species on disturbance regimes. For example, Karr and Freemark (1985, p. 167) argue that “disturbance regimes . . . must be protected to preserve associated genetic (Frankel and Soule 1981), population (Franklin 1980), and assemblage (Karr 1982a, 1982b, and Kushlan 1979) dynamics.” Species loss and ecosystem change have been observed in areas where “natural” disturbance regimes and habitats have been substantially altered. Examples of such undesirable ecosystem change may include the decline in forest health in eastern Oregon, the buildup of fuels in areas where fire has been suppressed, forest regeneration failures (ferry et al. 1989), and apparent decline in habitat capability resulting in the actual or potential listing of species such as the northern spotted owl Strix occidentalis caurina , the marbled murrelet (Brachyramphus marmoratus), and many stocks of salmon as threatened or endangered (Johnson et al. 1991, Nehlson et al. 1991, and Thomas et al. 1990).

The use of natural variability as a reference point in ecosystem management is not an attempt to turn managed landscapes into wilderness areas or return them to any single pre-existing condition. Rather, the intent is to meet ecological objectives by bringing the range of existing conditions in a landscape within the natural range. Returning major portions of the Pacific Northwest landscape to the state existing before European settlement has been proposed, but has been criticized as unworkable on several points. First, critics say, too much has changed, including invasion of exotic species and construction of roads and other engineered structures. Second, selecting conditions at any particular date as the reference may be biologically arbitrary, because ecosystems have changed dramatically on a broad range of time scales. Third, elements of society seeking maximum emphasis on commodity production contend that the wilderness state is socially arbitrary and even irresponsible, given a growing human population and its demands for resources. Ecosystem management must balance these perspectives against ecological objectives best achieved by maintaining an ecosystem in its natural range of variability.

We propose that an understanding of natural variability provides the basis for designing management prescriptions as well as the reference points for evaluating ecosystem management. The use of natural variability in ecosystem management is part of an effort to find a new management paradigm to replace intensive plantation forestry on Federal lands (i.e., to shift from an old to a new forestry; see Shepard 1993). The success of intensive plantation forestry has been evaluated by measuring rates of production of wood fiber, but an analogous measure has not yet been established for ecosystem management. Measures of successful ecosystem management might include maintaining viable populations of native species and desirable levels of productivity and nutrient export. These measures may have limited usefulness, however, because of natural and management-induced variability in ecosystems and our limited ability to predict ecosystem behavior. Also, attempts to identify thresholds in the ecosystem (such as minimum viable population size) probably will not produce workable measures of the success of ecosystem management. Instead, we propose that natural variability be used as part of a broad strategy incorporating ecological principles into all aspects of management.

In this paper, we discuss approaches to characterizing natural variability and the rationale for using that information in ecosystem management. We also address three issues that affect the use of information on natural variability as a basis for ecosystem management: (1) our limited abilities to interpret past ecosystem variability; (2) the degree to which present and future environmental conditions (e.g., climate, exotic species, and engineered structures) may fall outside the range of historical natural variability and the effects of this deviation from natural variability; and (3) the extent to which the range of natural variability differs from ecosystem conditions desired by society. Finally, we briefly comment on two examples of ecosystem management based on information about the range of natural variability. This discussion focuses primarily on wildfire disturbance in uplands because of the importance of fire in producing the forest vegetation mosaic in the Pacific Northwest, and the large body of literature on fire history and patterns. Other processes (e.g., geomorphic, biotic, and wind) and other parts of landscapes (i.e., stream and riparian networks) also deserve attention.

## Significance of Natural Variability

The combination of traditional ecological research and the recent emergence of conservation biology and landscape ecology provides significant scientific background for the use of natural variability in designing ecosystem management. A long history of ecological studies of species-disturbance relations reveals many examples of the close dependence of species on disturbances. Notable examples in Pacific Northwest forest landscapes include: (1) the strong association of the northern spotted owl with old-growth forest habitat produced by successive episodes of disturbances such as wildfire and windthrow (Forsman 1980, Forsman et al. 1984); 2) the effects of disturbances such as dam construction or landslides on endangered salmon populations (Craig 1987); and (3) the effect of disturbance on individual life-history stages (e.g., fire needed for seed germination) (Noble and Slatyer 1980). These and other findings indicate the tight coupling of species with environmental variability.

Research in conservation biology has identified the need to incorporate information on natural variability into land management, but managers are just now getting around to doing it. Early conservation biology research focused on observing the life history of single species and developing empirical population dynamics models to predict future population change (Soule and Wilcox 1980). Population dynamics models of the spotted owl (described in Murphy and Noon 1992) and salmon (Trotter et al. 1992), for example, have attempted to answer the question, “What is the minimum effective population size required to maintain that population?” Researchers recognize that endangered populations may respond to two major sources of threat: (1) systematic human pressures such as forest harvest, alteration of streamflow regimes, and interruption of fish passage by dam construction, and (2) natural variability produced by stochastic phenomena in the environment (Salwasser 1986, Shaffer 1981, Soule 1983). Initially, three types of stochastic phenomena of natural and management origin were recognized: loss of genetic diversity; demographic stochasticity; and environmental variability, both temporal and spatial (Murphy and Noon 1992, Salwasser 1986). Early population dynamics models for endangered owls focused on genetic diversity and demographic stochasticity but ignored disturbance and other spatial phenomena, such as habitat fragmentation (see Salwasser 1986). Subsequent modeling of spotted owls has attempted to incorporate the effects of habitat fragmentation from logging on life history and dispersal (Lamberson et al. 1992, Murphy and Noon 1992). No analogous spatial analytic models have been developed for salmon. Characterization of the range of natural variability in Pacific Northwest forest ecosystems would be a major contribution to efforts to incorporate the effects of natural environmental variability and disturbance regimes into population dynamics models for owls, fish, and other species (Glenn and Collins 1993).

Concurrent progress has been made in landscape ecology research, which emphasizes the characterization of spatial and temporal aspects of environmental uncertainty and disturbance regimes. For example, Swanson and others (1988) describe how landforms affect both the long-term features of a landscape, such as distributions of plant associations or stream channel characteristics, and the spatial patterns of transient processes such as fire and landslides. Analysis and modeling of disturbance patterns have revealed important differences between natural and managed systems and the difficulty of “managing” a natural system (e.g., Baker 1992). Geographic information system technology and mapping from remotely sensed data can facilitate the characterization of natural variability, both the spatial distribution and location of vegetation classes and stream types, and the frequency, spatial arrangement, and severity of fire patches, landslides, insect outbreaks, and other episodic processes.

In summary, recent developments in science have provided the rationale for using natural variability information in ecosystem management strategies, and also provided the technology to characterize natural variability in real landscapes. More research and applied work are needed to give deterministic and quantitative form to those strategies.

## Approaches to Characterizing Natural Ecosystem Variability

Two approaches can be taken to characterize the natural variability of ecosystems. Both approaches are based on the assumption that landscapes are composed of definable patches of distinct “states” (Brooks and Grant 1992a, 1992b) and that patches move from one state to another as a result of vegetation succession and/or discrete disturbance processes. The first approach to characterizing natural variability builds on the concept of “natural states” of patches by emphasizing the measurement of the spatial extent of a set of patches belonging to a particular class (such as seral stages of vegetation or streambank stability classes). This approach can also include description of patches in terms of their size and shape distributions. The second approach builds on the concept of “disturbance regimes,” defined in terms of the patch size distribution, frequency, and severity of a sequence of disturbance events (such as fire, insect outbreaks, or landslides). Disturbances cause patches to move from one state to another and may rearrange boundaries between patches. A focus on disturbance events and disturbance regimes emphasizes processes of ecosystem change rather than system states.

### Characterizing Natural States

Characterizing natural variability based on some range of natural states was developed by the Sustaining Ecological Systems approach in the Northern Rockies (USDA Forest Service 1992) and has been applied in National Forests of the Blue Mountain in eastern Oregon (Caraher et al. 1992), eastern Washington (Shlisky 1993), and western Montana (O’Hara et al. 1993 and Hann et al. 1993b). This approach involves characterizing the range of natural states for a period, such as the mid-1800s to the mid-1900s, as a simple range displaying maximum and minimum values (fig. 1A). This characterization could be expanded to a probability distribution, that is, the probability of observing a particular state in a random sample of the landscape over an extended period (fig. 1B). The probability distribution provides more information than the simple range but may be more difficult to compile. An illustration of the range of natural states permits comparison of present or proposed conditions to the range of natural variability.

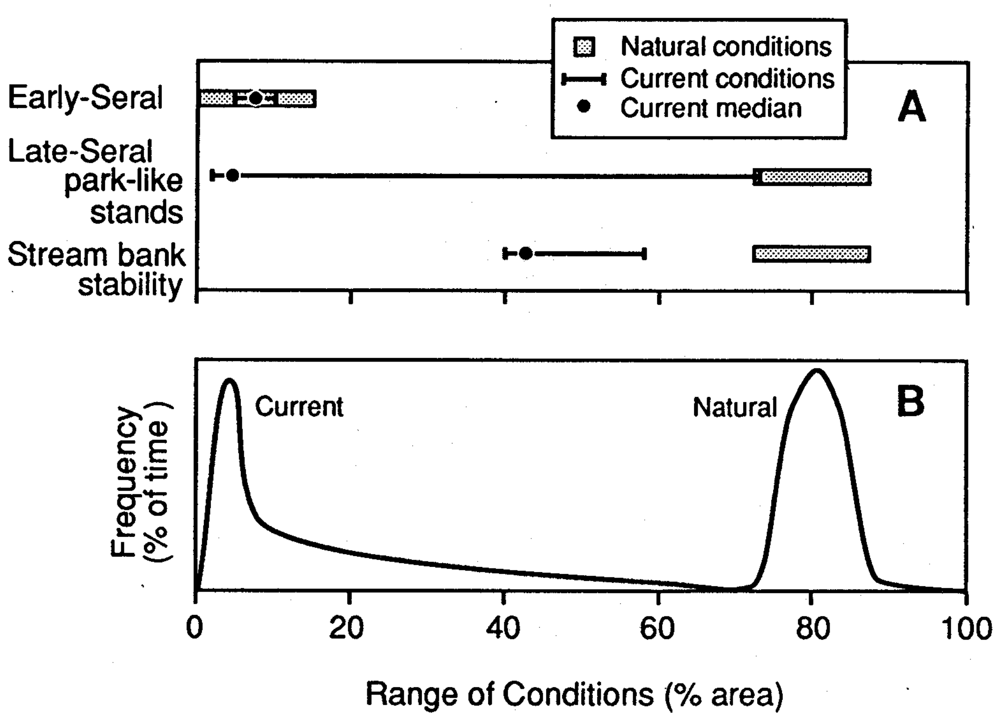


**Figure 1-Example display of range (A) and distribution (B) of ecosystem conditions. (A) Range of conditions for natural and current conditions are adapted from Silvies River area In Caraher and others (1992, p. 22); (B) Shows hypothetical distribution of ecosystem conditions of late seral, park-like stands for the same area.**

The range of natural conditions is interpreted from study of ecosystem history. A quick method for using this approach is to compile expert opinion (Caraher et al. 1992). Wildfire disturbance regimes can be quantitatively characterized by projecting backwards in time based on the timing and arrangement of fires which apparently produced the distribution of forest age classes in a landscape, both before and since fire suppression. Detailed reconstructions of history based on tree-ring analysis provide a fuller understanding of the actual age and extent of seral classes in a landscape. Detailed reconstructions are essential to determine the frequency of low- and moderate-severity disturbances which are important in maintaining certain stand conditions.

**Characterizing Disturbance Regimes**

The disturbance regime of a particular process may be described by the size of patches created, frequency, severity (e.g., percentage of live canopy cover retained), and other descriptors (White and Pickett 1985). A pictorial representation of these three variables (fig. 2) facilitates comparison among natural disturbance regimes and various management systems, which also can be considered as disturbance regimes. In figure 2, we depict a system of dispersed 40-acre clearcuts with a narrow range of disturbance characteristics that falls outside the range of the natural disturbance regime. Although such actions create vegetation patches whose size and frequency may fall within the wide, natural range of conditions maintained by wildfire, some natural successional states are not maintained. For example, the combined effects of dispersed, 40-acre clearcuts on a rotation of less than 100 years eliminates late-seral forest conditions.

Furthermore, clearcuts remove standing dead trees that provide many ecological functions, so the severity of clearcut disturbance exceeds the severity of wildfire disturbance.

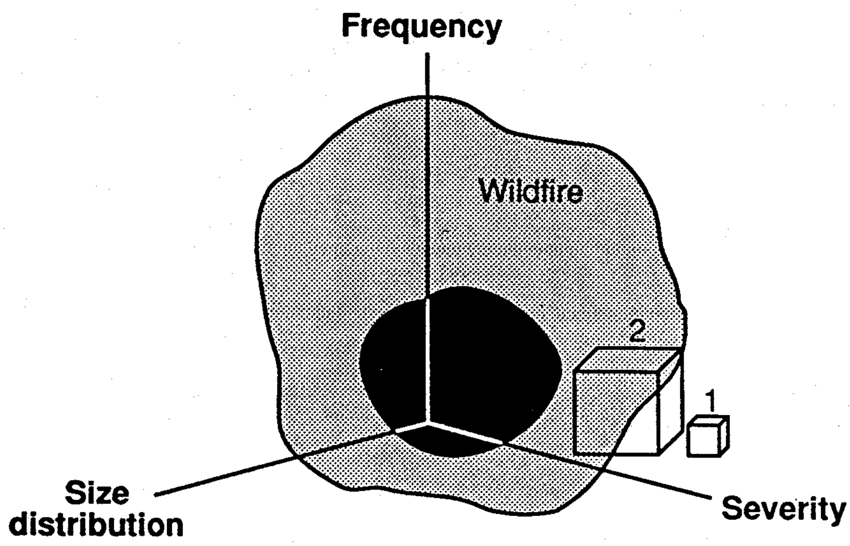


**Figure 2--Hypothetical representation of a natural disturbance regime with the large, irregular "cloud" showing a probability distribution of wildfire events interpreted from dendrochronologic observations. Box 1 represents a management system of dispersed clearcuts with broadcast burning. Box 2 represents the disturbance regime resulting from interaction of the design of the managed landscape (Box 1) and natural disturbance processes that could not be suppressed (e.g., windthrow at stand edges), accidents during management activities (e.g., escaped slash fires), and other factors. The black area represents a possible range of conditions under a form of ecosystem management.**

Applying the disturbance regime approach is complicated by the fact that natural disturbance processes continue in a managed landscape, albeit in a modified form, and that those natural disturbances commonly interact with landscape and stand structures modified by management actions (Franklin and Forman 1987). The desired disturbance regime, incorporating both natural and management processes, may appear something like the dark area of figure 2--within the range of natural variability but not fully occupying it, for reasons discussed below in the section entitled "Natural Variability and Desired Conditions." Future landscape structures are likely to be some difficult-to-predict hybrid created by the interaction of management actions and natural disturbance processes.

The range of natural disturbance regimes can be interpreted from field study of disturbance processes. Field methods for characterizing disturbance regimes have dealt mainly with wildfire and have been based primarily on dendrochronological and paleoecological methods, especially pollen analysis. Dendrochronologic and archival data have been used to produce historical maps of the extent of specific fires and to compute fire occurrence statistics. Fire disturbance regimes may be interpreted from mapped patterns of past fires or from sample plot data stratified by topographic position (especially aspect) or vegetation type (Agee et al. 1990, Morrison and Swanson 1990, Barrett and Arno 1991). Landslide disturbance regimes may be interpreted similarly from quantitative landslide hazard maps. For the most part, these approaches have been used to map and interpret histories of specific events. It is more useful for ecosystem management to synthesize this information in maps of disturbance regimes, using mapping units of disturbance frequency and severity (Hann et al. 1993a).

**Comparison of Natural States with Disturbance-Regime Approaches**

Each of the approaches discussed above has particular benefits and disadvantages. Many people are comfortable thinking in terms of natural conditions ("states"), such as vegetation classes or habitat types, and find it difficult to think in terms of disturbance regimes. Much of the modeling of ecosystem change has emphasized successional stages or classes and has treated disturbance in a simple, cursory fashion. Desired conditions defined in USDA Forest Service planning documents are typically described by the extent of conditions expressed as types of habitat or system outputs. The range of natural states can be described for many different types and properties of ecosystems and can be mapped in a spatially explicit fashion; thus, natural conditions have been widely used to characterize natural variability.

On the other hand, a disturbance-regime perspective, with its focus on processes, offers valuable insights for understanding ecosystem dynamics and this information can be used to design ecosystem management. Disturbance-regime information for a site can be used to design silvicultural treatments that may affect the ecosystem in a fashion similar to the natural disturbance regime. Dendrochronologic and paleoecological analysis to reconstruct disturbance history typically produces a longer record than can be obtained from using current age classes to characterize the natural range of states. A good characterization of disturbance regime requires examination of as long a record as possible from as large an area as possible, and thus a disturbance regime approach may be based on better information than an approach based on present distribution of forest age classes. Ideally, a combination of assessments of the range of historical natural conditions and disturbance regime would be most useful because the two approaches provide complementary results.

## Issues in Using Natural Variability as a Basis for Ecosystem Management

Technical and social issues arise in attempts to use concepts of natural variability as a basis for designing ecosystem management. These issues include (1) the limits of our abilities to interpret past ecosystem variability; (2) the interaction of management with natural variability, including possible effects of climate change, invasion of exotic species, and the presence of engineered structures; and (3) the relation between natural variability and society's view of desired conditions for ecosystems. The following discussion addresses each of these issues.

### Limits to Interpreting Historical Ecosystem Variability

Interpretations of historical ecosystem variability are limited by (1) temporal and spatial factors including the length of the historical record in relation to fire frequency, (2) the length, type, and magnitude of influence by both native and European humans, and (3) the frequency and severity of disturbance processes. The following discussion of these limiting factors focuses on fire, because it is generally a dominant and relatively well documented disturbance profess in many forest ecosystems of the region. The optimal conditions for interpreting natural disturbance regime are a long, pre-European settlement record with a high-frequency, low-severity disturbance regime, which permits sampling numerous events per site with dendrochronologic methods. Unfavorable conditions are a short record dominated by the European-influence period or sites with potentially long records, but with a dominantly catastrophic (such as stand-replacement fire) regime, which results in few disturbances recorded per site.

Where topography can be used as a proxy for fire pattern, landforms can be used to map the long-term wildfire disturbance regime. Large-scale landforms do not change significantly on the time scale of interest (Holocene) and do exert a persistent influence on vegetation, fuel, microclimate, and wind patterns that, in turn, influence fire behavior and frequency. Therefore, analysis of the disturbance regime of an area should include testing the strength of topographic and associated factors (e.g., soil and microclimate) on disturbance patterns and examining the potential for mapping disturbance regimes based on topographic features. The strength of association between topographic features and disturbance regimes is expected to vary with topographic complexity. In the low relief landscape of northern Michigan, for example, Frelich and Lorimer (1991) observed no significant geographic variation in disturbance regime, but Barrett and Arno (1991) found a strong influence of topography on disturbance regime in a steep, mountainous area of the northern Rockies. Where topography has a strong influence on fire patterns, it is possible to extend the interpretation of the wildfire regime well beyond the period of dendrochronologic records.

Paleoecological techniques can also be used to extend the length of record for disturbance regime characterizations. Examination of pollen records of vegetation change (e.g., ratios of early to late seral dominant species) and charcoal horizons can indicate the frequency of fire for a sampling point (lake or bog) for many thousands of years (Clark 1988); however, these point-in-space records are severely limited for making interpretations~of the areal extent, severity, and proximity of fire events.

Interpretation of fire history is hindered by spatial limits in areas of very large disturbances. Individual fires in the Pacific Northwest, for example, have exceeded 100,000 acres (Morris 1934), and there are many logistical problems in sampling such large areas. In addition, an area for sampling big events may extend over varied environments and disturbance regimes, so the analysis area may need to be geographically subdivided to accurately characterize the disturbance regimes.

In many areas of fire and wind disturbance, it is difficult to reconstruct the severity of past events which is critical information in designing ecosystem management. Density of trees surviving a disturbance is difficult to sample for events predating the most recent disturbances. Attempts to map severity of recent events have been limited to mapping burned areas by classes of severity interpreted from aerial photographs (e.g., Morrison and Swanson 1990), distinguishing regeneration (sites with tree origin dates only) from scarring events (Morrison and Swanson 1990, Teensma 1987), and examining distributions of tree age classes within a single study site (Frelich and Lorimer 1991). A knowledge of mortality characteristics by species, age class, and forest type is needed as a reference point to design silvicultural prescriptions that fall within the range of desired natural variability. For example, Morrison and Swanson (1990) point out that, if all recorded fires in a central Cascade Range study area had been stand-replacement fires, very little old-growth habitat would have existed over the past 500 years. The observed frequency of fires of low to moderate severity, however, suggest that such fires may have sustained extensive areas of old growth through multiple disturbances and through many centuries, and possibly through millennia.

An important question in defining natural ecosystem variability is how to treat burning practices by Native Americans. In many areas, intentional and unintentional burning by Native Americans probably occurred over a sufficiently long period (perhaps thousands of years) that effects were thoroughly incorporated in the ecosystem. Since European settlement, fire regimes have been altered to longer frequencies and sometimes greater magnitude, and are commonly accompanied by grazing, logging, or other practices that in some areas have caused the system to deviate markedly from pre-European settlement conditions. Furthermore, the period of Native American influence may have extended over much of the Holocene, so there may be no period when the existing complement of dominant species existed in the area without Native Americans present.

In summary, characterizations of natural variability have limited accuracy and completeness. Nevertheless, experience shows that reliable records of history of the natural systems can be developed and applied in a manner that is useful and even essential for ecosystem management.

**Interaction of Management with Natural Variability**

A disturbance regime that mimics the range of natural variability may interact with present and future environmental conditions to trigger ecosystem responses far outside the range of natural conditions. These changes may arise from the effects of exotic species, engineered structures, or climate change.

Roads are the most salient example of engineered structures in forested landscapes of the Pacific Northwest. Dams, armored streambanks, and other engineering works are dominant modifiers of river networks. Roads serve as conduits for dispersal of exotic species that can affect native plant communities through disease, insect attack, competition, and other processes. Roads also may function as sediment sources, extensions of the stream network that may affect peak flow generation, and sites for initiation of landslides that propagate downstream through drainage networks.

The sharp edges created by clearcutting against mature or older forest are an exotic, biotic structure in intensively managed landscapes. Wildfire and windthrow commonly create edges in natural landscapes, but these edges are more buffered by effects of residual green trees and abundant standing dead trees than are clearcut edges. Clearcut edges are particularly vulnerable to windthrow and other processes that operate preferentially at stand edges (Franklin and Forman 1987). Edges may increase the likelihood and extent of windthrow, perhaps even in areas where this process was relatively uncommon under natural conditions.

Exotic species have invaded most human-occupied landscapes in the world as a result of land use and use of transportation systems. Consequently, even if a former disturbance regime is repeated in the future, the presence of exotic species may alter the progression of landscape patches from one state to another and may produce dramatic, unexpected changes in the structure and composition of natural communities (Hobbs and Huenneke 1992).

Disturbance regimes are not static. Records of past variation in natural ecosystem conditions and disturbance regimes, for example, have been interpreted to be the results of climate change (Clark 1988, Brubaker 1991). Dramatic changes in plant and animal community composition, geographic distributions of species, and disturbance regimes (Clark 1988) can be expected as a result of climate changes in the range of variation experienced in the past millennium, or of the magnitude postulated in response to increased greenhouse gases in the atmosphere. Biological response to disturbance and successional development may be greatly altered by climate change. Therefore, climate change may greatly modify the implications of using knowledge of the range of natural states and disturbance regimes to design tomorrow's ecosystems.

These difficult issues of exotic species, engineered structures, and climate change do not negate, however, the importance of natural variability in designing systems for ecosystem management. Economic and ecological tradeoffs of engineered structures are being debated in many arenas, in part based on reference to the distribution of natural conditions, such as effects of dams on natural streamflow regimes and dependent biota. Exotic species are a growing concern in many ecosystems, regardless of management objectives and land use practices. Reference to natural variability and conditions may provide insights for developing new strategies to deal with exotic species. Reference to ecosystem response to past climate change is being used to interpret possible responses to projected climate change. We propose that an ecosystem managed for diversity, based on an understanding of natural variability is less likely to react catastrophically to dramatic climate change than is a simplified ecosystem. This hypothesis should be explored through field and modeling studies.

### Natural Variability and Desired Conditions

The range of natural variability of ecosystems and landscapes is likely to differ in some important respects from the conditions desired by society for many lands, perhaps even wilderness. In virtually all landscapes, a balance will be struck between natural processes and societal demands, reflected in part in the land use designation.

Acceptable management activities may deviate from natural variability where unchecked natural disturbances would create conditions undesirable to society. For example, very widespread disturbance events, such as wildfire covering more than 100,000 acres, have undesirable short-term effects on wildlife habitat, watershed conditions, and recreational values. Therefore, landscapes under ecosystem management may fall in the shaded area of figure 2, not occupying the full range of the natural disturbance regime for numerous reasons. Important questions remain, however, concerning how such effects of dampening of the disturbance regime might affect ecology and evolutionary biology in both terrestrial and aquatic systems.

One compromise between ecological and social considerations is to treat various functions of a given process differently. Fire, for example, may be necessary to sustain species at individual sites through its effects on regeneration, while also historically having shaped landscape patterns. In managed landscapes, fire may be used to play the former role, but management activities may replace fire as a determinant of landscape patterns.

Lands with a mix of ownerships and management objectives create added difficulty in balancing societal demands and maintenance of ecological processes. Uncoordinated activities between ownerships create great difficulty in managing landscapes. One owner may undertake management activities in less appropriate areas to compensate for actions by another owner. In the future, the benefits of cooperation rather than regulation may induce multi-ownership ecosystem management (Daniels et al. 1993, Lippke and Oliver 1993).

Ecosystem management, based on natural variability, is consistent with productive uses of the Pacific Northwest landscape, such as timber harvest and fishing, although near-term costs may increase. The socially acceptable balance between ecological and commodity objectives will be determined by the public. At present, the concept of managing from an understanding of natural variability has not been a subject of public discussion; however, it must be broadly discussed because the public and elected officials will determine the viability of this policy. Regardless of the outcome, an appreciation of natural variability is essential to making informed decisions.

## Examples of Use of Natural Variability

The following case studies illustrate two approaches to applying natural variability to ecosystem management. The approaches differ in their spatial scale, their focus on natural states versus disturbance regimes, and their use of expert judgment versus quantitative or historical analyses. Coarse-scale assessment, such as the Blue Mountain Assessment case study, can be used for planning, for prioritizing areas for management activities, and for developing broad-scale conservation and watershed management strategies. Fine-scale analysis, exemplified by the Augusta Project case study, is more useful for site-specific prescriptions. Together they provide complementary approaches to using information on natural variability for ecosystem management.

### Blue Mountain Assessment

The Blue Mountain Assessment (Caraher et al. 1992) provides an example of a broad-scale assessment of the range of natural conditions or states. Individual analysis areas cover drainage basins of many hundreds of square miles each. This assessment provides a basis for future management of forests in eastern Oregon and Washington. Extensive areas of trees in these forests are either dead or dying, apparently as a result of drought, insects, pathogens, and past management.

Natural conditions considered in the analysis include early and late seral stages of selected forest types, extent of pine types of low vigor (hence high susceptibility to insect damage), fuel loads, and stream and riparian conditions. The ranges of natural states and present condition of each of these variables are estimated based on “professional judgment and local knowledge” (Caraher et al. 1992, p.4). Results are reported as a simple range and median of natural conditions (fig. 1A). The present condition of many ecosystem variables is outside the range of natural variability. This information is used as a basis for proposing restoration practices in forest and river ecosystems by illustrating which ecosystem components are most removed from the range of natural conditions. High priority restoration efforts target these ecosystem components, especially those which are most critical in terms of societal objectives.

This “extensive” analysis approach is most useful for assessing large areas quickly within a common framework. Such a brief analysis, however, relies on the expertise of a few individuals, lacks rigorous, quantitative analyses of natural variability, and has limited depth of historical perspective.

### Augusta Project, Willamette National Forest

The Augusta Project provides an example of a small-scale assessment of both the range of natural conditions and the range of natural disturbance regimes. The project area covers 19,000 acres (about 30 square miles) in the Willamette National Forest and has been directed by the Blue River Ranger District and Cascade Center for Ecosystem Management. This exploratory project has emphasized ecological assessment and has had a strong research focus. Public participation in the project is in the early stages. Land use designations for portions of the project area include wilderness, a wild and scenic river corridor, and a limited extent of roadless area along some ridges. The following paragraphs describe the characterization of natural variability and the use of this information for ecosystem management for a large area of forest in the center of the project area.

Work for this project began with assessment of the wildfire history since 1400 AD, based on tree-ring counts of fire scars and tree origin dates. Maps of fire episodes (each episode representing a single fire or multiple fires over a period of several years) were interpreted and compiled from aerial photographs and from the distribution of sample sites at which fires were recorded. The maps showing fire episodes were used to map the long-term fire disturbance regime, using topographic controls to help extend the record. For example, areas with even-aged Douglas-fir overstory and little evidence of burning in the past 500 years and located in steep-walled, north-facing valleys, were considered to have had a long-rotation (greater than 400 years), stand-replacement natural fire regime. Dry, south-facing slopes, that have experienced numerous fires in the past few centuries and today exhibit a fine-scale mosaic of forest patches, were considered to have experienced frequent (less than 100-year rotation), low- to moderate-severity fire.

This interpretation of disturbance regime was used to devise a stand and landscape management system using blocks of land ranging in size from fifty to hundreds of acres. Larger blocks were generally used in areas of longer hillslopes and larger fire patches. Block boundaries generally extend from stream to stream to distribute the effects of cutting over several watersheds. Individual cutting units may be considerably smaller than the block size, but all the acreage within an individual block would usually be cut within a period of several decades. Silvicultural treatments and cutting intervals vary among blocks, based in part on the interpretation of the natural disturbance regime. For example, cutting intervals are shorter and more selective cutting may be used on areas with natural fire regimes of frequent, low- to moderate-intensity fires. Block-cutting sequences and resulting landscape patterns were scheduled over a 400-year period based on the proportions of the landscape desired for each sera) class. Landscape pattern objectives are, thereby, transformed into a set of landscape block trajectories used to guide stand management objectives.

This ecosystem management system is not intended to strictly mimic the natural disturbance regime, but follows it in important respects. The intent is to retain a distribution of vegetation seral classes (hence, wildlife habitats) within the range of natural variability interpreted from the fire history. This design also recognizes that natural disturbances will continue to interact with managed blocks of land. It retains stand conditions in topographic positions in the landscape where they occurred naturally, so that they may have a higher probability of being sustained in the face of natural disturbances.

This management system differs from the natural disturbance regime by omitting the wide temporal and spatial fluctuations in disturbance observed in the area over the past 500 years (fig. 3). Some individual fire episodes burned over half of the area, but management prescriptions affecting such large areas are considered to be undesirable for maintaining watersheds and other resources.

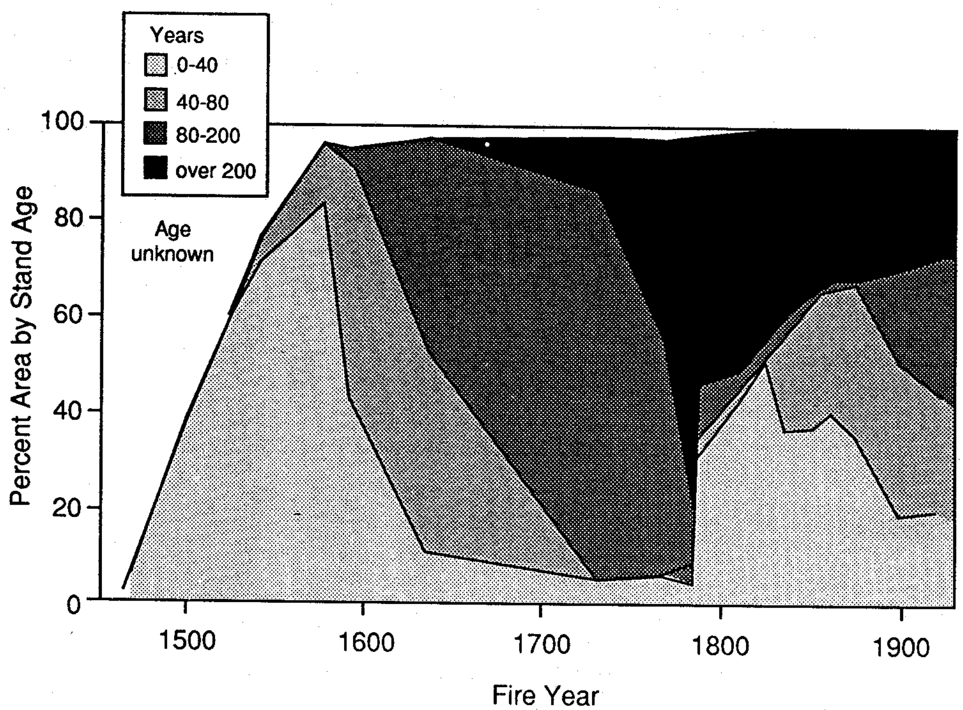


**Figure 3--Percentage area in various stand age classes for the non-wilderness portion of the Augusta study area (15,650 acres), Willamette National Forest, based on dendrochronologic reconstruction of wildfire history. Note that the assumption that burned areas experienced stand replacement fires results in underestimating older age classes and neglect of mixed-aged stands.**

**Complementary Nature of Intensive and Extensive Analysis**

The intensive Augusta Project and extensive Blue Mountain Assessment are complementary approaches. The intensive approach provides information on disturbance history and reveals some limitations of information and analytical techniques. This information can be used as a basis for designing more simplified, less data-intensive analysis procedures suitable for larger areas. Extensive analysis of large areas also provides a context for sites of intensive analysis. The extensive approach to characterizing and applying natural variability should be supplemented by a network of sites where the same concepts are tested and refined on a site-specific basis. For the Augusta Project area, uniform prescription of stand and landscape management derived from an extensive analysis would constitute an unnatural, probably unworkable, management system that would not meet biological or social objectives, because it ignores the large range of geographical variability within the project area.

## CONCLUSIONS--IMPLICATIONS FOR MANAGEMENT

Science provides a strong biological rationale for managing ecosystems within their natural range of variability to sustain native species and maintain ecosystem productivity. Information on the range of natural ecosystem conditions and disturbance regimes provides essential ingredients for designing sustainable ecosystem management. Many of today's contentious issues in natural resource management of Federal lands arise in part from actions causing deviation from natural ecological conditions.

Natural variability can be characterized using two complementary approaches: one focusing on the range of ecosystem states and the other concerning the disturbance regime. The analysis can be extensive, quickly assessing a large area based on judgments of experts, or intensive, requiring quantitative evaluations with historical reconstructions of past disturbances. Information on natural variability can be applied in planning, assessment for restoration practices, and design of regional-, landscape-, and stand-level ecosystem management practices.

The long-term effectiveness of this approach to sustainable ecosystem management can be tested only over many decades of research and adaptive management. Although this discussion has focused on fire disturbance in upland areas, more work is needed to characterize natural variability for managing stream and riparian networks. These latter areas experience geomorphic disturbances characteristic of stream systems as well as forest disturbances, such as fire and windthrow. For example, roads and clearcuts substantially increase the frequency of debris slides and debris flows over only 1 percent or so of upslope areas, but these processes may affect a substantial fraction of the length of first- through third-order stream channels and associated riparian zones (Benda and Dunne 1987, Swanson and Dyrness 1975, Swanson and Lienkaemper 1978). Stream-riparian networks commonly experience more extensive disturbance (under natural conditions) than adjacent upland areas in steep landscapes prone to mass movement. In some landscapes, management may be accentuating this contrast.

Reference to natural ecosystem conditions does not provide specific, quantitative direction for ecosystem management. Rather, this approach makes management planning and decisions more challenging because the range of management decisions is much broader than a few years ago when management issues focused on the question of preservation versus intensive plantation forestry. Ecosystem management in the context of the range of natural variability and disturbance regimes requires balancing social and ecological values. The use of natural variability defines a range within which a compromise between social and ecological values will have to be struck.

The public has a critical role in formulating ecosystem management. Public participation is important if concepts of natural variability are to be used in developing and implementing ecosystem management, but currently there is no forum for public participation. The natural variability of ecosystems and the inevitability and roles of disturbances are important concepts that all participants must understand, if they are to contribute to management decisions.

## ACKNOWLEDGMENTS

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# Design and Use of Ecological Mapping Units

**R.G. Bailey, M.E. Jensen, D.T. Cleland, and P.S. Bourgeron**

## ABSTRACT

This paper presents an overview of the theory, design, and use of ecological mapping units. Such mapping units delineate similar biophysical environments for land evaluation and planning and may be defined at various hierarchical scales depending on management needs. The criteria commonly used in ecological map unit design (e.g., climate, landform, geology, potential plant communities) do not change substantially after management activities. Consequently, these criteria provide a template on which data concerning the existing condition of the land (e.g., vegetation stand maps, wildlife surveys) may be overlaid to improve assessments of ecosystem health. Ecological units provide a consistent basis for predicting what the land could be; other resource maps describe its current status. Characterizations of historical variablity, predictions of plant succession pathways, descriptions of natural disturbance regimes, and estimates of potential productivity are commonly stratified by ecological unit types. Accordingly, ecological units are critical to all planning and analysis efforts for ecosystem management.

## INTRODUCTION

Ecosystem management includes the use of renewable resources (e.g., wood, forage) in a sustainable manner. Accordingly, land use planning requires prediction of ecosystem response after different kinds and intensities of management practices. One of the first tasks of ecosystem management should be to delineate and describe land units that behave in a similar manner given their potential ecosystem composition, structure, and function (Jensen et al. 1991). Such delineations represent ecosystems with similar response potential and resource production capabilities and are called “ecological units” by the USDA Forest Service (USDA Forest Service 1990). These delineations are similar to land units (Zonneveld 1979), biogeoclimatic ecosystems (Meidinger and Pojar 1991), and land systems (Christian and Stewart 1968, Wertz and Arnold 1972), in that they represent tracts of land that are ecologically homogeneous at a given scale of analysis.

Assessments of ecosystem health and condition commonly require use of two types of maps: maps that delineate areas with similar potential for management based on more permanent landscape components, such as climate, soils, geology, and landform (i.e., ecological units), and maps that delineate the existing status of landscape components that are readily influenced by management practices or display high temporal variability (e.g., existing vegetation).

Both types of maps are required if the health of a given ecosystem is to be assessed in land use planning. Ecological unit maps commonly are used to describe how the landscape could look or function under natural processes as well as under different management scenarios. The second type of map describes how the landscape currently looks. Overlaying these two types of maps helps describe landscape health; that is, what is the landscape currently like contrasted against what it could (or should) be, given management objectives?

This paper describes some of the basic theoretical and design considerations relevant to ecological map unit construction. It also presents examples of how such maps are used in ecosystem management.

## Spatial Hierarchies

Ecosystems are three-dimensional segments of the earth where life and environment interact (Rowe 1980). They may be defined at any scale and can be conceptualized as occurring in a nested, geographic arrangement, with smaller ecosystems contained within larger ones (Allen and Starr 1982, O’Neill et al. 1986). The hierarchy of these systems is organized in descending orders of scale by various driving variables (e.g., climate, landform) that influence biological function. Patterns and processes at any given scale operate within the context of higher scales of a spatial hierarchy. For example, coarser scale influences such as regional climatic regimes affect embedded ecosystems such as riparian habitats. Additionally, properties of smaller ecosystems emerge in the context of the surrounding, larger system (e.g., plant community pattern is commonly recognized within broad landscape stratifications).

The linkages that exist among different ecosystems must be described to address planning issues that transcend National Forest and Regional boundaries (e.g., air pollution, anadromous fisheries, biodiversity). National Forests must be considered in the context of "global" ecology and economy, if ecosystem management is to succeed. Accordingly, descriptions of how geographically related ecosystems are linked to form larger systems are required. Biophysical ecosystem delineations (i.e., ecological units) used to indicate similar potential for land management planning must be defined at different hierarchical levels.

## Scales and Boundary Criteria

Schemes for recognizing different scales of ecological units have been proposed and implemented in many countries (e.g., Meidinger and Pojar 1991, Wertz and Arnold 1972, Zonneveld 1979). The system proposed by Miller (1978) recognizes ecosystem linkages at three scales of perception and is useful in illustrating the nature of most systems. At the smallest scale (microscale), ecosystems are homogeneous sites commonly recognized by foresters and range scientists. Such sites can be delineated at scales ranging from 1:10K to 1:80K. At the mesoscale, linked sites create a landscape mosaic that looks like a patchwork commonly mapped at a scale of 1:25K to 1:1 M. At macroscales, mosaics are connected to form larger systems, which are called ecoregions. The mapping scale of ecoregions is commonly 1:3M.

A fundamental question facing all ecological land mappers is, how are the boundaries of different size systems determined? To screen the effects of disturbance or succession, such boundaries should be based on semipermanent landscape components important in differentiating ecosystems at various scales (e.g., landform). This basis allows recognition of an ecological unit regardless of present land use or existing vegetation. To show linkages between systems, and establish a hierarchy, boundaries should ideally be based on attributes common to all scales.

## Controlling Factors and Scale

The logic and criteria for establishing ecosystem boundaries of different sizes have been presented by Bailey (1983, 1985, 1987, 1988b), following concepts advanced by Rowe (1980), Miller (1978), Crowley (1967), Isachenko (1973), Leser (1976) and Forman and Godron (1986). The following discussion is a brief summary of ideas developed by those authors.

The operation of ecosystems is controlled primarily by climatic regime (i.e., diurnal and seasonal fluxes of energy and moisture). Climate regime, in turn, is modified by the structural characteristics of an ecosystem (i.e., its land surface form). Consequently, ecosystems at all scales respond to climatic factors which may be modified by different ecosystem features at different scales. For example, latitude, continentality and elevation exert primary control on regional climate; however, landform, topography, and vegetation modify regional climatic factors to produce local climatic conditions. An understanding of how various environmental factors influence climate across different scales is required before optimum boundary criteria can be derived for ecological unit maps.

## Macroscale Maps: Ecoregions

At the macroscale, ecosystem patterns are controlled primarily by latitude (irregular solar energy), distance from the sea (continentality or oceanic influences), or elevation. Macroclimatic units (i.e., the climate that lies just beyond the local modifying irregularities of landform and vegetation) are delineated at this level and are similar to the broad climatic region maps of Koppen (1931), Troll (1964) or Walter et al. (1975). Such maps outline eco-climatic zones with repeatable patterns of major ecosystem types. These maps are important sources of information to climatologists and can be used to help determine ecosystem boundaries at the regional scale.

Each ecoclimatic zone is clearly defined by a particular type of climatic regime and (with few exceptions) corresponds to zonal soil types and climatic climax vegetation. These zones are also indicative of those major ecosystems that biogeographers have traditionally recognized as biomes (Whittaker 1975). Therefore, two series of ecoclimatic units are recognized in mapping: lowlands and highlands. Highlands are considered to be azonal members of the lowland zone in which they occur. Highland settings differ climatically from the zone from which they rise and must be considered separately in most mapping efforts.

Direct mapping of ecoclimatic units is difficult because meteorological stations are sparsely distributed in many areas and data are unavailable. Consequently, biological indication (Kuchler and Zonneveld 1988) is commonly employed to predict climatic boundaries in broad-scale ecological unit mapping. For example, the composition and distribution of vegetation was used by Koppen (1931) in his search for important climatic boundaries, and vegetation is a major criterion in the ecosystem region maps of Bailey (1980, 1983, 1989) and Walter and Box (1976).

Climatic differences useful in recognizing ecological units at the macroscale are indicated by vegetation in several ways (Damman 1979): (1) changes in forest stand structure, dominant life forms, and topography of organic deposits; (2) changes in dominant species and in the toposequence of plant communities; and (3) displacement of plant communities, changes in the chronosequence of a habitat, and minor changes in the species composition of comparable plant communities. Kuchler (1974) and Van der Maarel (1976) provide other examples of climatic biological indication.

## Mesoscale Maps: Landscape Mosaics

Macroclimate accounts for the largest share of systematic environmental variation at the macroscale or regional level. At the mesoscale level, however, broad ecoclimatic zones are modified primarily by geology and topographic (landform) features (fig. 1). For example, solar energy will be received and processed differently by a field of sand dunes, lacustrine plains, or upland hummocky moraines.

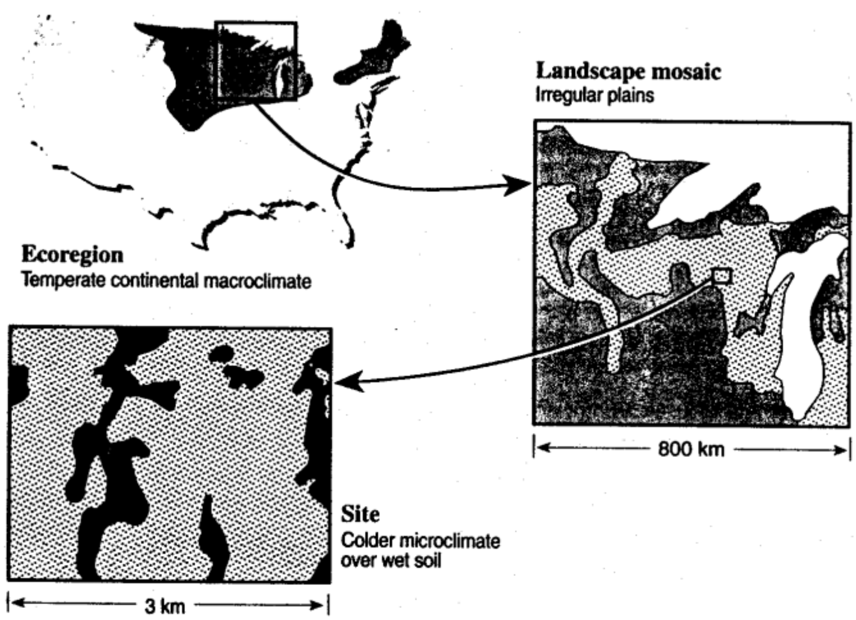


**Figure 1--An example of ecosystem maps at different scales.**

Landforms (with their geologic substrate, surface shape, and relief) influence the variability of ecological factors such as water availability and exposure to radiant solar energy. Through varied height and slope of the ground surface, landforms interact with climate and directly influence hydrologic and soil-forming processes. Consequently, the primary correlate of vegetation and soil patterns at the mesoscale is landform because it controls the intensities of key factors important to plants and to the soils that develop within them (Hack and Goodlet 1960, Swanson et al. 1988).

Landforms may be described at various scales of mapping and are central to several approaches to classifying forest land (e.g., Barnes et al. 1982). On a continental level (within the same macroclimate), broad-scale landform patterns commonly exist that modify zonal climatic units. The landform classification of Hammond (1954), who classified land surface forms by existing surface geometry, is useful in determining the boundaries of landforms for regional climatic modification.

According to its physiographic nature, a landform unit may be visualized as being constructed of different ecosystem types. For example, a delta has ecosystem types that differ from those found on a moraine landscape next to it. Within a landscape, such ecosystem types are arranged in specific patterns that commonly follow a toposequence of site types (Major 1951).

## Microscale Maps: Sites

The identification of ecological units at coarser scales is facilitated primarily by macroclimate, geology, and broad-scale landform patterns; however, at the microscale, such delineations follow differences in microclimate as influenced by topographic position and soil factors. Within a landform, differences in slope and aspect commonly exist which act to modify the macroclimatic regime. Accordingly, the following topoclimate classes may be identified on most landforms (Thomthwaite 1954): normal, hotter than normal, and colder than normal. Ecological units derived from these classes are referred to as site classes by Hills (1952).

Site classes may be further modified by identification of soil moisture regimes which influence plant community distribution (e.g., very dry, dry, moist and wet sites). Deviations from the normal topoclimate and soil moisture regime occur in various combinations within a region and are called azonal and intrazonal site types by Hills (1952).

In Hills' scheme, zonal site types are microscale units that reflect the normal topoclimate and soil moisture regime for a given area. Azonal site types are zonal in a neighboring zone, but are confined to an extrazonal environment in a given zone. They are hotter, colder, wetter, or drier than the reference zonal site. Intrazonal site types occur in exceptional situations within a zone and commonly reflect environments with extreme soil types that support different climax vegetation associations than the theoretical climatic climax association for a region (Drury and Nisbet 1973).

## Construction and Validation of Mapping Units

Ecological mapping units may be developed through a combination of individual ecosystem component maps, or through simultaneous synthesis of a combination of components (Bailey 1988a). The former approach is commonly used when separate functional inventories exist for an area (e.g., soil map, potential vegetation map, geology map) and an integration of these ecosystem component themes is desired. Geographic information systems software commonly is used to generate ecological unit maps when the integrity of individual resource maps need to be maintained for specific interpretation needs. Synthesis of individual map themes through a geographic information system is commonly done in the development of detailed (large scale) ecological unit maps. Limitations associated with this approach are described by Zonneveld (1989) as follows: the land unit (i.e., ecological unit) as a system is not a mere compilation of independent components; and the boundaries of separately surveyed land attribute mapping units rarely coincide because of orientation errors, classification errors, real classification differences, and no correlation among the land attributes.

Development of ecological units by simultaneous synthesis of ecosystem components minimizes the boundary problem by avoiding the first three problems listed above. The fourth problem (no correlation among land attributes) can be detected during a survey. Map unit design is then adjusted to coincide with the landscape attributes that best define the status of an ecological unit for mapping purposes. The primary advantages of the simultaneous integration approach are that the ecological units delineated reflect observable ecosystems on the ground; communication is improved among the different disciplines involved in map unit design (commonly a soil scientist, geologist and plant ecologist work together in this effort); and the unit costs associated with map development are far less than those associated with generating and combining individual resource maps.

Simultaneous integration is used most commonly in the development of broad-level (small-scale), ecological unit maps. An interdisciplinary team approach to map unit design should always be used when this method is selected.

Different levels of classification may be used to describe ecosystem components of the landscape. Broad-level classifications should be used to describe ecosystem component composition of coarse-scale ecological units (table 1), because if detailed classifications are used in coarse-scale mapping, too many taxa need to be described, which greatly complicates both the design and analysis of mapping units. The level of classification used to describe composition of an ecological unit is a direct function of the types of interpretations that need to be made. Finer levels of classifications (which are used to describe detailed ecological map unit delineations) allow for more detailed interpretations; however, costs associated with more detailed mapping are exponentially higher. For this reason, coarser-scale mapping units (and classifications) may be required for certain analysis efforts because of budget or time constraints. Coarse-scale units are also required for certain types of planning that cover large analysis areas (e.g., Forest planning, Regional planning).

**Table 1--Examples of Hierarchical Classification Systems Used In Describing Ecological Unit Composition**

| Ecological Unit Scale | Soils (a) | Potential Vegetation (b) |
|---|---|---|
| Macro | Order (Mollisol)<br>Suborder (Boroll) | Class (forest)<br>Subclass (coniferous forest)<br>Formation (Temperate Mesophytic forests) |
| Meso | Great Groups (Cryoboroll)<br>Subgroups (Lithic Cryoboroll) | Series (grand fir) |
| Micro | Family (clayey, Lithic Cryoboroll<br>Phase of family (eroded phase) | Plant association (grand fir/ginger)<br>Ecological site (sandy substrate phase) |

**(a) Taxa presented follow Soil Taxonomy (USDA-Soil Conservation Service 1975)**
**(b) Taxa presented follow Driscoll and others (1984).**

Developing an ecological unit map requires a map unit ID legend which describes the differentiating criteria used in map unit delineation (e.g., landform, topography, climate zonation), and a map unit description which addresses the composition and relations among the different ecosystem components included in the map unit (e.g., soils, potential vegetation, geology). Experienced mappers never initiate a survey without some idea of what they will find. Predictive skills are important for a mapper because it is often impossible to visit every map unit delineation within a survey area, so a mapper must extrapolate knowledge gained from field transects or traverses in one area to unsampled areas. Extrapolations are based on the relation between the taxa being described (e.g., potential vegetation types, soil types), and other more readily observable landscape features such as elevation, aspect, and geologic material (Bourgeron et al. 1993). The landscape features that correlate with the different types of taxa described in mapping are the differentiating criteria used in map unit design. The sum of these relations is documented in the map unit description which the mapper is continuously testing and revising during the survey. This description provides a structured means for communication between mappers as well as a hypotheses for statistical testing to validate relations between map unit taxa and coarser-level landscape features.

The differentiating criteria used in ecological map unit design must always be checked to ensure that the boundaries they produce have ecological significance (Rowe 1980). For example, a climatic map that delineates such key factors as temperature and precipitation is not necessarily an ecological map until its boundaries are shown to, correspond to sign'if'icant biological boundaries. Likewise, maps of landform, vegetation, and soils are not necessarily ecological maps unless they co-vary with one another.

Ecological unit maps should be thoroughly tested and modified (if necessary) before they are used in environmental analysis (Bailey 1984). Such mapping units are commonly hypothesized to circumscribe a population of sites with similar characteristics (e.g., potential vegetation or soil patterns). If data on site characteristics are assembled and evaluated statistically, the validity of the map can then be objectively evaluated. Gradient-oriented sampling and generalized linear analysis models are especially, useful for validating ecological map units (Margules and Austin 1991, Bourgeron et al. 1993).

## Ecological Unit Mapping Examples

Ecosystem mapping involves the use of multiple environmental factors in map unit design (Spies and Barnes 1985). The idea of mapping mulitfactor ecosystems is not new; however, interest in this approach has increased significantly in the last 50 years (Küchler and Zonneveld 1988). The following discussion provides a brief overview of some of the systems commonly used for ecological unit mapping.

The “CSIRO-Land System Maps” of Australia and New Guinea emphasize the use of landform (geomorphology), soils, and potential vegetation for map unit delineation and classification (Zonneveld 1979). Such maps are used extensively in land use planning and evaluation. The Northern, Intermountain and Eastern Regions of the Forest Service have historically used the Land Systems Inventory Method (Wertz and Arnold 1972) for ecological map unit construction, which is similar to the CSIRO approach, The theoretical basis for land system inventory is that landforms, patterns of soils, and climax plant communities are all products of the interaction of climatic forces with the geologic structure of the surface of the earth (USDA Forest Service 1976). This system integrates the sciences of geomorphology, soil science, hydrology, and plant ecology to classify, map, and describe lands for land management planning. This system (table 2) recognizes climatic ecoregions as described by Bailey (1980, 1983) at coarser scales. Climatic and geologic properties of the land are emphasized at the province, section; and subsection levels. At the lower levels of the system hierarchy (i.e., landtype association, landtype, landtype phase, and site), landforms, soils, and potential plant communities are primarily used to differentiate terrestrial ecosystem units. Valley bottom setting, stream type, and fishery habitat components are commonly used to delineate riverine systems at the landtype, landtype phase, and site levels of mapping.

**Table 2--Examples of Primary Design Criteria Used in Land Systems Inventory**

| Mapping level | Typical size (scale) | Primary design criteria | Associated characteristics |
|---|---|---|---|
| Domain | 100,000 square miles (1:30,000,000) | Climatic zone or group | Repeatable patterns of vegetation classes or subclasses and soil orders or suborders |
| Division | 50,000 square miles (1:15,000,000) | Climatic type (Koppen 1931) | Repeatable patterns of vegetation subclasses or formations and soil suborders or great groups |
| Province | 5,000 square miles (1:3,000,000) | Hammond's (1964) land surface form, plant climax formation patterns | Repeatable patterns of vegetation formations, and soil great groups |
| Section | 1,000 square miles (1:1,000,000) | Climax plant series patterns following Küchler (1964) | Repeatable patterns of vegetation series and soil subgroups or great groups |

| Mapping level | Typical size (scale) | Primary design criteria | Associated characteristics |
|---|---|---|---|
| Subsection | 100 square miles (1:500,000) | Geologic (e.g., lithology structure), physiographic (e.g., glaciated mountain slopes) and state-wide climatic zones | Repeatable patterns of vegetation series and soil subgroups |
| Landtype Association | 10 square miles (1:250,000) | Physiographic and geologic criteria (e.g., fluvial dissected, granitic, mountain breaklands) | Repeatable patterns of plant association groups and soil subgroups |
| Landtype | 1 square mile (1:63,000) | Physiographic criteria (e.g., landform, shape, elevation, range, drainage, aspect, dissection characteristics) | Repeatable patterns of plant associations, soil families, and stream types |
| Landtype Phase | 0.1 square miles (1:24,000) | Topographic criteria (e.g., percent slope, position, aspect), plant association soil family, stream type) | Repeatable patterns of soil series, ecological sites, and fishery habitat components |
| Site | 0.01 square miles (1:15,840) | Ecological site, phase of soil family, fishery habitat components (e.g., pools) | |

The biogeoclimatic ecosystem classification of the British Columbia Ministry of Forests (Pojar et al. 1987) uses a hierarchical scheme for ecosystem description with three levels of integration: regional, local, and chronological. At the regional level, vegetation-soil relations are used to infer the regional climate into zonal classifications that define broad-scale biogeoclimatic mapping units. At the local level, ecosystems are classified into vegetation and site units by using vegetation and soils information. At the chronological level, ecosystems are organized into site-specific chronosequences of vegetation according to site history and successional status.

## Applications

There are several applications for ecological map units including those that improve our ability to bring data together in a meaningful way for planning, management, and conservation of ecosystems. Some examples are described below.

### Assessments of Ecosystem Condition

The components commonly used to develop ecological map units (e.g., climate, landform, geology, soil) do not change substantially after most management activities; consequently, such maps may be used to consistently describe similar biophysical environments. Given this fact, ecological units provide a basic template for interpretation of data that commonly display change after management treatment (e.g., existing vegetation, cobble embeddedness, animal abundance). These types of data describe the “existing condition” of the landscape and are commonly overlaid on appropriate ecological unit maps to determine the condition or health of a given area. The effects of management practices on the landscape are most efficiently described by contrasting the “existing condition” of an area with other managed or unmanaged areas that occur on the same ecological unit. The natural variability among sites is minimized by this process; consequently, the difference in the observed condition may be correlated to the different types of treatment imposed.

**Temporal Variability Characterization**

Ecological map units (or their ecological classification components) also provide a basic template for interpreting the temporal variability expressions of a landscape. For example, the vegetation present on a landscape is not static. Instead, it changes with time because of plant succession and disturbance regime processes. Predicting the response of a plant community to disturbance is facilitated by identifying the ecological unit on which it occurs (Arno et al. 1986). The types of plant communities that may occur on an ecological unit can be described and their successional pathways illustrated through use of the “cone model” of plant succession (Huschle and Hironata 1980). This model uses ecological units to describe the basic biophysical environments within which plant succession occurs. Developing successional pathway predictions by ecological units provides a powerful tool for assessing the effects of management practices on vegetation which, in turn, influences the value of the land for multiple-use management (Jensen et al. 1991).

**Environmental Analysis**

The classification and inventory of ecological units also provides basic information for natural resource planning and management. When combined with information on existing conditions and process, ecological units may be used in resource assessments, environmental analyses, establishment of desired future conditions, and monitoring of natural resources.

The hierarchical framework used in most ecological unit designs is useful for conducting multiscaled assessments of resource conditions. For example, in assessing population viability of mobile species like bears or wolves, conditions at both broad-landscape and site levels are important. Across the landscape, there may be reproducing populations in high-quality habitats that are sources for populations in other areas. In other locations, there may be less suitable habitats where mortality exceeds reproduction. In the latter case, populations are dependent on immigration from source populations to maintain existing numbers.

Within a landscape, the quality of local habitats depends on many factors including ecosystem diversity, vegetative composition and structure, and land use (e.g., road densities, fragmentation). Ecological units may be used to assess ecosystem diversity and vegetative potentials (e.g., seasonal forage). Ecological units also may be used to identify potential for manipulating vegetation to improve habitat and to develop habitat suitability models (Glenn and Collins 1993). Multiscale analyses can detect landscape and local ecosystem relations. For example, population viability can be analyzed at both broad landscape and local levels, and marginal habitats can be expanded, linked or otherwise improved by using information on existing conditions and ecological unit potentials.

Another application of this perspective is in the linkage of terrestrial and aquatic systems. Because of the interdependence of geographical elements, aquatic systems are linked or integrated with surrounding terrestrial systems through the processes of runoff and migration of chemical elements. By delineating areas with similar watershed conditions in terms of terrestrial site characteristics, the embedded freshwater aquatic systems are thereby delineated. Aquatic systems delineated in this indirect way have many characteristics in common, including hydrology and biota (Frissell et al. 1986, Minshall 1993). Overlays of hierarchical watershed boundaries on ecological mapping units are useful to most watershed analysis efforts.

Ecological units can also be related to past, present, and future conditions. Past conditions serve as a model of functioning ecosystems and provide insight into natural processes. It is unreasonable, for example, to attempt to “restore” systems like oak savannas or old-growth forests in areas where they did not occur naturally. Moreover, natural disturbance regimes (e.g., flooding) are often beyond human control. The impact of management decisions on disturbance regimes and landscape configuration can be analyzed in a hierarchical fashion (Milne 1993, Turner et al. 1993). Ecological units are useful in understanding landscape patterns and processes and in devising desired condition scenarios for land use planning which can be attained and perpetuated. Accordingly, desired conditions can be portrayed at several spatial scales. Conflicting resource uses (e.g., remote recreational experiences versus developed motorized recreation; habitat management for area-sensitive species versus edge species) can be minimized by considering the effects of projects at several scales of analysis (Brenner and Jordan 1991, Milne 1993).

### Monitoring

Monitoring the effects of management requires baseline information on the natural and existing conditions of ecosystems at different spatial scales. Effects induced by management are departures from these baselines. Landscape, community and species-level biological diversity, forest productivity, water quality, and other concerns can best be approached by establishing baselines for ecological units and then monitoring changes.

Ecological unit hierarchies enable land managers to identify repeatable geographic patterns in ecosystems. Thus, resource managers are in a position to design efficient sampling networks for inventory and monitoring (Minshall 1993). Because ecosystems commonly recur in predictable patterns within a region, representative sample sites may be described and used to characterize analogous (unsampled) sites. This stratified sampling approach greatly reduces the cost and time of inventorying (Bourgeron et al. 1993) and monitoring (Bailey 1991).

## CONCLUSIONS

Ecosystems exist at all spatial scales, from the global ecosphere to local sites. They are defined by associations of ecological factors such as climate, geology, landform, soil, water, plants, and animals (USDA 1990). Although the association of all factors is important in understanding ecosystems, each factor is not equally important in defining such systems at all spatial scales. Accordingly, the primary challenge of ecological classification and inventory is to distinguish natural associations of ecological factors at different spatial scales. Additionally, the differentiating criteria for ecological map unit design must reflect those factors that exert primary control on the levels of organization contained within the scale of ecosystem being described.

The ideas presented in this paper suggest that ecosystems may be recognized by differences in climatic regime. The basic idea is that climate, as a source of energy and moisture, acts as the primary control on ecosystem distribution. Climatic effects on ecosystems change with scale. Broad-scale macroecosystems, or ecoregions, may be described as areas of homogeneous macroclimate. At the mesoscale and microscale, landforms are important critera for recognizing smaller ecosystem divisions because they influence localized climatic regimes. Therefore, boundary criteria for ecological mapping units may be determined by climate as modified by landform. This approach offers a logical basis for delineation on both large- and small-scale ecological mapping units.

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# SECTION 3 - SAMPLING DESIGN AND DATA ANALYSIS

## Summary of Sampling Design and Data Analysis Papers

This section addresses basic features of survey design, data collection, and data analysis, as they relate to ecosystem evaluation and characterization at regional and landscape scales. Optimum sampling designs can only be derived from explicitly stated objectives; consequently, we avoid specific recommendations of “what to measure and how to measure it” in these papers. Instead, the papers of this section synthesize appropriate sampling theory and provide general discussions of sampling, applications. This information is useful to the development of sampling designs for specific inventory and analysis objectives. The first paper of this section (Bourgeron et al.) describes general concepts of landscape survey design, which constitutes the initial observational phase of ecosystem characterization and land evaluation. These authors stress that the scale and purpose of a survey need to be explicitly stated before an optimum sampling design may be derived. Depending on the purpose of the survey, different types of existing data (remote sensing imagery, herbarium records, and plot data) may be appropriate for analysis. Additionally, such data can be efficiently used to identify gaps in knowledge for an area, in terms of geographic and ecological space.

Sampling techniques commonly used in the description of ecological patterns and rare-element detection are reviewed by Bourgeron and others, who conclude that stratified random sampling along environmental gradients is more efficient than traditional random sampling or systematic sampling designs. Stratified random sampling designs are particularly appropriate to situations where biotic and abiotic interactions are poorly understood (most wildland reconnaissance surveys) because they greatly increase the probability that biotic and abiotic rarities will be sampled. These authors describe a two-stage modification of a stratified random sampling design (i.e., GRADSECT) that is very effective in the recovery of ecological patterns. They also discuss efficient methods for data interpolation in wildland surveys and suggest that generalized linear models are particularly appropriate to sampling design optimization, data analysis, and data interpolation.

The second paper of this section (Milne) discusses analysis methods useful in describing the pattern of landscapes. A brief overview of techniques used in landscape correlation analysis (standard parametric and nonparametric statistics and multivariate data reduction methods) is provided. Milne suggests that because correlation analyses provide a characterization of the potential interactions in nature, they may be useful in determining the effects of management activities on ecosystems and landscapes (i.e., correlation analyses provide a simplification of nature that assists in our understanding of an otherwise complex system).

Milne also demonstrates that the characterization of ecosystems and landscapes at multiple spatial scales enables land managers to partition the causes of system change according to the scale at which relevant ecological forces operate. Miles demonstrates this by providing examples of resource analysis at various spatial scales. His analyses indicate that (1) resource density (e.g., basal area and community cover) varies with the scale at which it is assessed; (2) there is a statistical regularity to density variation (i.e., fractal dimension) that allows this variation to be characterized for interrelating densities at several scales; (3) species operate at different scales; consequently, they perceive different densities within the same landscape; and (4) the locations of dense versus sparse resources used by species differ with the scale at which species perceive density. Milne uses an example of tree harvesting strategies to demonstrate how these principles might be used in resource management. He also shows how fractal geometry may be used to define the domains of scale over which particular processes and species operate. Domain detection is important to land management because it permits manipulations to be performed at scales that will not interfere with valuable processes operating at other scales (e.g., stand harvest system designs that do not adversely effect watershed hydrologic function).

The third paper of this section (Glenn and Collins) reviews the appropriateness of various models used in predicting species distribution and abundance. They suggest that the abundance and distribution of species are space and time-scale dependent. This fact is important to the development of management plans for long-term persistence (sustainability) of the biotic component of ecosystems. Glenn and Collins also suggest that most of the models currently used by land management agencies for predicting species abundance and distribution (e.g., habitat suitability models) do not consider the temporal and spatial variability of species interactions; consequently, their appropriateness to ecosystem management is questionable. The authors suggest that metapopulation-based models offer promise to land managers, and they describe how landscape issues of community structure may be analyzed in terms of spatial patterns of corridors and patchiness.

The fourth paper of this section (Minshall) addresses the sampling and analysis of stream-riparian ecosystems in a hierarchical framework of spatial and temporal scales. Minshall states that many landscape disturbances are spatially heterogeneous and initiate sequences of temporal recovery events that profoundly influence the composition and structure of stream ecosystems. Historical characterizations and analyses of such systems have often ignored this fact; instead, stream ecosystems have been viewed as spatially homogeneous and temporally static. Minshall suggests that a more realistic approach to stream and riparian ecosystems is achieved through a landscape perspective, which incorporates greater spatial and temporal considerations into 'basin-wide studies of stream ecosystems. He indicates that a principal operational paradigm for the study of stream ecosystems is facilitated by the 'River Continuum concept,' which assumes that important ecosystem processes (and their resultant patterns) change in a predictable fashion with stream size (order). He also states that the importance of different factors (climate, land use and mass wasting) in influencing stream ecosystems changes with the scale of system description. Accordingly, land managers must be careful in selecting the spatial scale at which a stream ecosystem is studied or managed.

Minshall also reviews previous approaches to the assessment of habitat and biotic conditions in stream-riparian ecosystems. He suggests that a hierarchical approach is required for: the classification of stream-riparian ecosystems, the identification of appropriate reference areas for condition assessment, and the selection of monitoring variables and the level of information obtained in surveys.

Jurgensen and others provide the fifth paper of this section, which describes risk assessment methods commonly used in land management. These authors provide a summary of the different techniques used in assessing soil, wildlife, fire, and insect-disease conditions and risk.

The last paper of this section (Jensen et al.) describes a multiresource database and analysis system (ECODATA) for describing and evaluating ecosystems. The study of landscapes for ecosystem management involves the description of biotic and abiotic variability over various spatial scales. Accordingly, standardized ecosystem attributes must be described for sampled sites ff ecosystem process-pattern relationships are to be developed. Consistent database structures are also required if analysis software is to be efficiently used in landscape evaluation. The ECODATA database and analysis system of Jensen and others is useful to ecosystem management because its design facilitates consistent collection, storage, and interpretation of basic ecosystem information. Such database and analysis systems are required if land managers are to efficiently use most of the data analysis methods described in this section.

# General Sampling Design Considerations For Landscape Evaluation

**P.S. Bourgeron, H.C. Humphries, and M.E. Jensen**

## ABSTRACT

This paper presents sampling design procedures applicable to evaluating landscapes and characterizing ecosystems at regional scales. The discussion emphasizes a sampling design that provides primary descriptions of the biotic and abiotic variability over entire regions, as well as descriptions of the environmental variables and processes that control species, community, and ecosystem distributions. Aspects of sampling design reviewed include: formulation of the purpose of survey, use of existing data, replicated versus randomized sampling, choice of stratifying variables, selection of sampling sites, incorporation of local variables, cost-effectiveness of the survey, and efficiency in recovering ecological patterns.

## INTRODUCTION

Landscape survey design is the initial observational phase of landscape evaluation and ecosystem characterization. Data collection determines to a large degree the accuracy and precision of all analyses performed thereafter, such as pattern recognition. Ecologists tend to emphasize data analysis techniques over sampling design procedures. This emphasis is unfortunate because the quality of the interpretation depends on the quality of the data, both in terms of the thoroughness of coverage of the area sampled and in terms of the type of information collected at each sample site. Accordingly, the following three questions need to be addressed in survey design: What are the general data requirements of regional surveys for landscape evaluation and ecosystem management? What standard ecological sampling procedures are appropriate for regional ecological surveys? and What is the trade-off between statistical theory, logistical practice, sampling efficiency, and the use of data for interpolation and extrapolation?

Landscape evaluation and ecosystem characterization usually take place in extremely large areas, and are aimed at characterizing the entire range of variability of the biotic and abiotic components of ecosystems (e.g., plant communities in relation to environmental gradients). All segments of the range of variability should be described; that is, both common and rare elements. Such inventories differ in purpose from standard sampling designs, which are formulated to determine unbiased estimates of the mean of variables over entire populations (Austin and Heyligers 1991). Another condition of standard sampling design is the complete enumeration of the units from which a sample can be selected; however, such knowledge of the sampling frame does not exist for unknown ecosystems. This paper describes all aspects of the sampling design process used to answer the three questions listed above, including (1) the purpose of the survey and use of existing data, (2) the need for representative sampling based on environmental variables, (3) the choice of location and number of samples, (4) the detection of rarities, and (5) interpolation and extrapolation from samples.

### Purpose of Survey and Use of Existing Data

The first step in conducting a regional survey is to clearly formulate the purpose of the project (Green 1979). For example, is the primary purpose of a survey to characterize only the forested ecosystems of a region or all ecosystems? If the survey is aimed at forests only, is it restricted to ecosystems of a certain age (e.g., old growth) or does it incorporate all successional stages? Does it focus on all taxa (plants and animals) or on a subset (e.g., trees of a specific genus or granivorous birds)? The sampling design can meet the purpose of the survey only if the objective is explicitly defined (Gauch 1982, Austin 1987, 1991a).

The second step is to determine what kinds of data are already available and whether they are useful and cost effective for a specific purpose even if biases exist (Austin 1991b). Maps of vegetation, soils, geology and climate often exist and can be used initially as surrogates for field data. Problems can arise if data layers have different scales, adding to the uncertainty and error rate inherent in each map. This approach, however, has been used successfully by various investigators. For example, in Australia maps of land systems (Christian and Stewart 1953) were used to build a database to describe the range of natural environments in the study region. This database was later used to construct a stratified sampling scheme to describe the range of natural variability contained in various areas in the region (Pressey and Nicholls 1991).

Remote sensing imagery is readily available over most of North America. Use of remotely sensed data may be a cost effective approach for broad-scale mapping of land cover. There are significant challenges, however, in producing maps from remote sensing data that pertain to the scale of measurement of the data (Davis et al. 1991). There has been a recent trend to acquire ground and remote sensing data over a range of scales (Davis et al. 1991, Running et al. 1989, Sellers et al. 1988). Davis and others (1991) concluded that some of the problems in the use of remote sensing data relate to understanding the spatial scale dependence of processes and patterns. This search for multiscaled patterns and processes (Bourgeron and Jensen 1993, Levin 1992, Milne 1993, Minshall 1993, and Turner et al. 1993), in turn, requires the collection of ground data at the appropriate intensity and frequency.

Herbarium and museum records are another source of data. Databases have been developed that compile this kind of information. The work of Heritage Programs and The Nature Conservancy (Jenkins 1985) with Biological Conservation Database (BCD) software provides an example. Herbarium and museum records for specific plant and animal species, plant communities, and ecosystems (identification, location, ownership, and any relevant biological and management information) are collected and entered into the BCD for each state. Although the information in the database is descriptive, cost effective uses can be made of the data for many purposes. Another example of the use of existing data is the work of the U.S. Fish and Wildlife Service Gap Analysis Project (Scott et al. 1987) or GAP. State GAP programs combine species distribution information with vegetation maps. Museum records for selected animal species are combined with vegetation maps to determine species-habitat relations. This exploratory analysis defines geographic areas with suitable habitats for selected species. Heritage's BCD and GAP databases are often used together to optimize use of existing data. Useful analysis of the data may be sufficient without new field surveys. For example, information in the BCD can be used to meet the goals of threatened and endangered species programs, and GAP analyses can be used for the preliminary assessment of the diversity of selected species in large areas. Furthermore, new survey costs can be minimized because old records may suggest historically suitable areas for species or communities that have not been visited in decades. Repeat visits to areas with historical information have yielded important data on changes in grassland and forest ecosystems in the western United States (Gruell et al. 1982, Phillips 1963).

A final source of existing data is plot data that are commonly collected in routine vegetation surveys. In eastern Oregon and Washington, such surveys include habitat typing projects (e.g., Daubenmire 1970, Johnson and Clausnitzer 1991, Johnson and Simon 1987). If these data are incorporated into a standardized database, a very cost effective tool for landscape evaluation and ecosystem characterization can be created. For example, all previous survey data of the USDA Forest Service, Northern Region, have been incorporated into ECODATA, an integrated data management and analysis system (Jensen et al. 1993). These data have been used for preliminary landscape evaluation and ecosystem management (Hann et al. 1993, O'Hara et al. 1993). One limitation of using existing survey data is that the same biotic or abiotic information is rarely collected in each survey; however, if a minimum list of attributes common to all surveys is established, data can be used for a variety of purposes (Austin 1991b, Jensen et al. 1993). Jensen and others (1991) illustrate the use of such data in evaluating cumulative environmental effects. Other examples of the regional use of similar databases for land management decisions in Australia is provided by the work of Austin and others (1983, 1984, 1990) and Margules and others (1987).

Such databases can also be used to determine which areas or which parts of environmental gradients have not yet been sampled. If existing data are systematically reexamined, duplication of survey effort can be avoided and maximum use of resources can be made. Disadvantages (cost of establishing a database, limitation to a minimum list of attributes at the regional scale) are offset by potential cost reduction and increased effectiveness of new surveys, and by providing an initial regional overview of the information.

## Need and Rationale for Representative Sampling Based on Environmental Variables

The primary goal of regional surveys is to characterize as many of the ecological patterns as possible. This purpose, the recovery of ecological patterns, is not necessarily met by commonly used statistical sampling procedures. Sampling theory emphasizes randomization to provide the probability structure for statistical analysis or to give credibility to the statistical model used (Gillison and Brewer 1985). Gillison and Brewer (1985) argue that randomization procedures may be counterproductive to the intent of ecological surveys, especially where natural pattern is known to be non-random.

In ecological surveys, two aspects of pattern recognition should be considered: the delineation of the pattern itself (e.g., a specific forested ecosystem), and the frequency and distribution of patches of the pattern (i.e., spatial distribution, number and size of stands of the forested ecosystem) (Godron and Forman 1983, Gillison and Brewer 1985). In landscapes, patch frequency and distribution vary as a scale-sensitive function of environmental complexity and of the resolution of the ecological classifications used to characterize the pattern (Gillison and Brewer 1985). Landscape configuration variability should be analyzed in terms of the driving variables (the abiotic factors) controlling the biotic component of the ecosystem (Bourgeron et al. 1993b). In standard sampling design, each spatial point in the landscape is given an equal probability of being sampled. Random placement of sample sites will not accurately reflect the full range of variability of the biotic and abiotic components of ecosystems at regional scales unless the sampling intensity is very high (Gauch 1982, Orloci 1978, Pielou 1974).

To alleviate the shortcomings of standard random sampling, stratified sampling schemes have been used to provide statistical validity and accuracy in the recovery of patterns. Stratified sampling divides a study area into compartments and locates samples randomly within compartments. This approach has been used successfully over large heterogeneous areas with mostly unknown patterns. For example, a nested stratified random sampling design according to landforms and ecoregions was used in southern Yukon, Canada, to characterize vegetation pattern and its underlying environmental gradients (Orloci and Stanek 1979). The results of this study indicate that the selected stratifying variables accounted for a large part of the regional variation in vegetation.

Orloci and Stanek’s (1979) sampling design is similar to a methodology known as gradient directed transect (gradsect) sampling. Gradsect sampling is a variant of stratified random sampling schemes. This approach, first described by Gillison and Brewer (1985), is based on characterizing the distribution of patterns along environmental gradients. The gradsect sampling design (Austin and Heyligers 1989, Gillison and Brewer 1985), is intended to provide a description of the full range of biotic variability in a region by sampling along the full range of environmental variability present. Transects that contain the strongest environmental gradients in a region are selected to optimize the amount of information gained in proportion to the time and effort spent during a survey (Austin and Heyligers 1989). In addition, sampling sites are deliberately located to minimize travel time. This method has been shown statistically to capture more information than standard sampling designs (Gillison and Brewer 1985).

Heiman (1983) and Austin and Heyligers (1989, 1991) have expanded the gradsect methodology to include different levels of environmental stratification within each gradsect. Their modified gradsect procedure utilizes a two-stage sampling design: first, gradsects are chosen, and then adequate environmental stratification and replication are performed within gradsects.

## Sampling Design Procedures

The two-stage gradsect sampling design has been used to describe the rain forests of southern New South Wales, Australia, (Heiman 1983) and a mixture of eucalypt and rain forests in northern New South Wales (Austin and Heyligers 1989, 1991). In the U.S., an opportunity to use gradsect methodology and to evaluate its efficiency in recovering ecological patterns arose as result of a request from The Nature Conservancy to survey the Gray Ranch in southern New Mexico. The goals of this study were to characterize the vegetation patterns and their associated floristic variability in relation to the range of environmental variability within the ranch and to use this information to assess the conservation value of the area (Bourgeron et al. 1993a, Engelking et al. 1993). The

primary constraint to sampling was that resources permitted only 2 weeks of fieldwork by two crews of two surveyors each to characterize an area of 130 000 ha.

Stratification variables for sampling (and their classes) were chosen based on information from previous studies, the decisions of a group of experienced ecologists and soil scientists, and the availability of suitable maps. This approach was taken because of the lack of spatially referenced databases containing defined attribute data for the survey area. The dominant landscape level variables considered to influence the distribution of species and plant communities at the Gray Ranch included: geology as an indicator of the nutrient regime, elevation as an indicator of precipitation and temperature, and soil type as an indicator of water availability. These variables were grouped into classes (Bourgeron et al. 1993a, Engelking et al. 1993) and arranged in a factorial design for use in sampling design stratification. Each factorial combination (elevation X soil X geology) was considered to represent a physical environment. Class intervals were chosen to produce maps (1:100,000 scale) for each variable. These maps were overlaid to produce a map of the physical environments suitable for visual assessment of the environmental gradients.

Access roads were considered when positions for gradsects were established. Two main gradsects were chosen for sampling that contained as many of the physical environments as possible. After comparing the physical environments in the gradsects with those occurring in the Gray Ranch as a whole, two short additional gradsects were chosen specifically to capture those environmental combinations which did not fall within the two main gradsects. Some physical environments of very restricted extent were not represented in the gradsects, but given the time constraints, they were not targeted for sampling. The gradsects sampled included 49 of the 55 physical environments identified for the study area.

Geographical replication of similar environments allows the capture of biological variation arising from criteria other than those used to define the gradsects. Geographical replication was included twice in the sampling design. First, the overlap in the environmental envelopes of the gradsects provides a degree of geographical replication of similar environments. Second, the main gradsects were each divided into three segments to further enhance geographic replication.

Explicit sampling procedures were used to locate sample sites along gradsects. First, the total number of sample sites that could be visited during the 2-week period was estimated to be 100 sites. This number was increased to 120 to provide for flexibility during field work. Second, researchers decided to sample physical environments according to their representation at the Gray Ranch. A grid with 0.4-km spacing was generated and overlaid on the map of the physical environments. Each grid point was assigned to one of the environments. From the number of grid points intercepted by each combination, a percentage (number of grid points for a given environment divided by total number of grid points) was calculated. These percentages provided estimates of the spatial extent of each physical environment and were used to calculate the number of samples per combination. For example, the combination of alluvium substrate by elevation class 1250-1500 m by argid soil suborder occurred on eleven percent of the Gray Ranch and should have received eleven percent of the total number of samples. Based on this rule, however, the environments representing less than 1 percent of the Gray Ranch would not have received a sample. Researchers decided to sample all physical environments present in the gradsects in order to cover as much of the range of environmental variability as possible. Therefore, one sample was assigned to each infrequent environment. This kind of decision can be made by investigators to adjust the sampling scheme to meet the needs of the study.

All physical environments in the two short gradsects were infrequent (less than 1 percent of the total area) and were each allocated one sample (total = 11). Choice of the sample locations was simplified because often these combinations were found in only one location. Accordingly, 109 sites remained for sampling along the two main gradsects. Each main gradsect was allocated samples in proportion to is surface area. Within a gradsect, each segment was allocated one-third of the number of samples. The number of samples assigned to each physical environment in a segment was in proportion to is representation in the segment. The number of grid points intercepted by each environment within the segment was used to calculate the percent representation, and an appropriate number of samples was assigned.

Actual sample location within each segment was chosen randomly for a given physical environment. Bias due to accessibility was made explicit by taking into account the location of access roads. To decrease the effect of intense grazing and disturbance on vegetation patterns, however, the design was constrained so that plots were at least 0.8 km from a road, water tank, or windmill, except for a few sites which were deemed to be of high quality. The design further allowed alternative sample sites to be selected within the same physical environment when field survey showed that the sites were too inaccessible (time constraint) or too severely affected by cattle or human activity.

At the local scale, variables other than those chosen for the sampling design may have a strong influence on vegetation composition (e.g., slope or aspect). Consequently, a further stratification based on the response of the vegetation to the environment was imposed. Within each sample, 20 x 20 m plots were located in each physiognomic type found (e.g., forest, shrubland, grassland), or in dominance types. Finally, at each sample site the classes of the stratifying variables were verified (soil and geology) to compensate for errors due to the various map scales used in the initial sampling design. If an error was found, changes were made in the design to stay on track with the required number of samples for each physical environment.

After 2 weeks of vegetation survey, 97 plots were sampled at the Gray Ranch. These plots occupied 36 of the 49 physical environments identified in the gradsects used in sampling. The 13 environments not visited were infrequent and scheduled to have only one sample taken. These environments were often found exclusively in hard-to-access areas requiring long hikes, which would have taken too much time to sample given the 2-week survey period. All unsampled physical environments at the Gray Ranch were prioritized for survey during the next field season.

## Sampling Efficiency

The relative efficiency (i.e., the ability to recover ecological pattern) of gradsect sampling compared to other sampling techniques has been analyzed (Austin and Adomeit 1991, Gillison and Brewer 1985). An existing vegetation map derived from high-density point sampling of a 424-$km^2$ area was used as the test base for comparing gradsects to random transects (Gillison and Brewer 1985). Gradsects were found to be 27 percent shorter, on average, than random transects and logistically better located. Gradsects also were an average 21 percent richer in vegetation types than random transects of the same length.

A simulation study was conducted to evaluate three traditional statistical sampling methods and gradsect sampling (Austin and Adomeit 1991). The simulated data set used for these tests was based on an actual landscape, with realistic biotic-environment relations (Belbin and Austin 1991). Austin and Adomeit (1991) provided “informed guesses” of sampling costs for each method. Among the three statistical sampling methods compared, random sampling recovered more species than systematic sampling or transect sampling at intermediate cost. Transect sampling was found to have the highest costs and detected the fewest species. Gradsect sampling methods were compared to random sampling. Rules for gradsect sampling within a gridcell were included in the test. These rules included resampling within a gridcell, and choice of samples from topographic units. Gradsects detected more species than random sampling and some gradsect sampling rules achieved this at lower cost.

These results show that gradsects are generally more efficient than traditional statistical techniques in recovering the greatest amount of ecological pattern per sampling effort. The gradsect method allows placement of samples in logistically more accessible areas than traditional statistical techniques such as systematic or random sampling, and thus improves cost-effectiveness. Costs can also be reduced and effectiveness maximized when stratifying variables are carefully chosen using existing information (Austin and Adomeit 1991).

## The Detection of Rarities

The problem of detecting rare ecological elements is an important topic in ecosystem management for three reasons. First, rarity is a criterion widely used in designing conservation strategies. The detection of rarity during landscape surveys is desirable because it assists land managers in meeting their legal requirements (e.g., maintaining an accurate threatened and endangered species list, designing representative research natural areas). Second, for ecosystem management, biotic-abiotic relations need to be characterized to provide land managers with the ability to predict the response of an ecosystem to various management scenarios (Bourgeron and Jensen 1993). When rare ecological elements of a landscape are not identified, land units supporting them may be assigned to other known elements. This misidentification of biotic-abiotic relation may lead to erroneous predictions of ecosystem response at the landscape level. Third, landscape configurations change over time. Ecosystems that are restricted today may become more extensive in the future. In the context of ecosystem management, it is crucial to identify all segments of the range of biotic and abiotic variability, whether abundant or rare. Only then can ecosystem management consider all aspects of natural variability at landscape and regional levels.

Gillison and Brewer (1985) argue that gradsect design is more efficient than random and systematic design in recovering rare elements of an ecological pattern. They reason that since gradsect sampling recovers significantly more patterns at finer scales of distribution, the likelihood of locating rarities should increase. Two other reasons make gradsects or any variant of the methodology more likely to locate rarities. First, because physical environments have to be defined and mapped, the less common environmental combinations are clearly identified and located (Austin and Heyligers 1989, 1991, Bourgeron et al. 1993a, Engelking et al: 1993). If there is any biotic rarity associated with such environments, it will be surveyed as well. Second, gradsect methodology requires intensive study of the range of environmental variability in an area, and this leads to a more thorough examination in the field. Hence the probability of locating rare ecological elements is increased.

## Interpolation and Extrapolation From Samples

Ecological inventories in most large areas are unlikely to be complete because costs are prohibitive. Therefore, it will often be necessary to use models to interpolate or extrapolate survey results to areas that have not been sampled. Predictive statistical models such as Generalized Linear Models (McCullagh and Nelder 1989) are powerful and flexible tools for generating quantitative relations between species or communities and environmental factors (driving variables) using location-specific data sets (Nicholls 1989). Such models can be used to predict distributions of species or communities in unsampled areas if the environmental characteristics of an area are known or can be derived from maps or simulation models. A method for obtaining predictions of regional distributions of species or communities consists of generating relevant environmental data at each intersection of a regularly spaced grid overlaid on a map of the area. Models are then used to produce predictions of biotic abundance at each grid intersection.

Generalized linear models are a class of statistical models that include linear regression and analysis of variance models as special cases (McCullagh and Nelder 1989). The usual assumptions for classical linear regression models, that is, constancy of variance and normal distribution of errors, may not be met in many kinds of biological data. Generalized linear models can be constructed for a variety of distributions in addition to the normal distribution. Biotic data in the form of counts or abundance classes can be modeled using a Poisson or multinomial distribution. A binomial distribution can be used for developing predictive models of presence-absence data. There is also flexibility in the choice of environmental attributes used as explanatory variables. Once the type of model has been chosen based on biotic response variable characteristics, a generalized linear model is constructed by fitting environmental variables. If a large number of variables are available, it may be desirable to use a selection procedure to choose a subset of the variables to be included in the model. The goal of generalized linear modeling is to develop a model that combines a parsimonious set of explanatory variables with a good fit to the data. After the model has been fitted, regression diagnostics can be applied to further evaluate the adequacy of the model. McCullagh and Nelder (1989) offer further information on the use of generalized linear models.

Complete coverage of all combinations of environmental factors in the data set used to generate modeled relations is crucial to the successful use of generalized linear models to predict biotic distributions in unsampled areas. The reliability of spatial prediction is increased when the frequency of observations is evenly distributed across the environmental space (Nicholls 1989). Survey design should include explicit consideration of how well the full range of environmental factors is sampled when the use of statistical models for prediction of unsampled areas is anticipated. The gradsect approach (with its purpose of covering the whole range of biotic variability over the range of environmental variability encountered) is appropriate for generating required data. Logistical constraints may make perfect coverage unattainable; however, it should be attempted in survey design when possible. Evenness of coverage is very important to most survey efforts because it determines the extent to which resultant predictions are an interpolation within the range of collected data rather than an extrapolation outside the sampled range (Margules and Stein 1989).

The environmental coverage found in an undesigned data set and is implications for predictive ability are presented in figure 1. In this example, data were collected on forested plots throughout the southwestern United States. Species abundance was measured for all species on a plot along with environmental characteristics (Muldavin et al. 1990). A simulation model, MTCLIM (Hungerford et al. 1989), was used to derive climate attributes for each plot. A generalized linear model analysis of the 883 plots sampled in this study indicated that maximum July temperature and first quarter (January to March) precipitation were significant environmental factors that determined presence or absence of Gambel oak *Quercus gambelli* (Engelking et al. 1993). An examination of the distribution of the plots in the environmental space described by maximum July temperature and January to March precipitation, also showed that coverage was least complete at the extremes of both environmental variables. Consequently, predictions from models developed using these data would have less reliability at the limits of the temperature and precipitation distributions than nearer their centers. Additional sampling of plots would be most profitable in areas where coverage is sparse in this study.

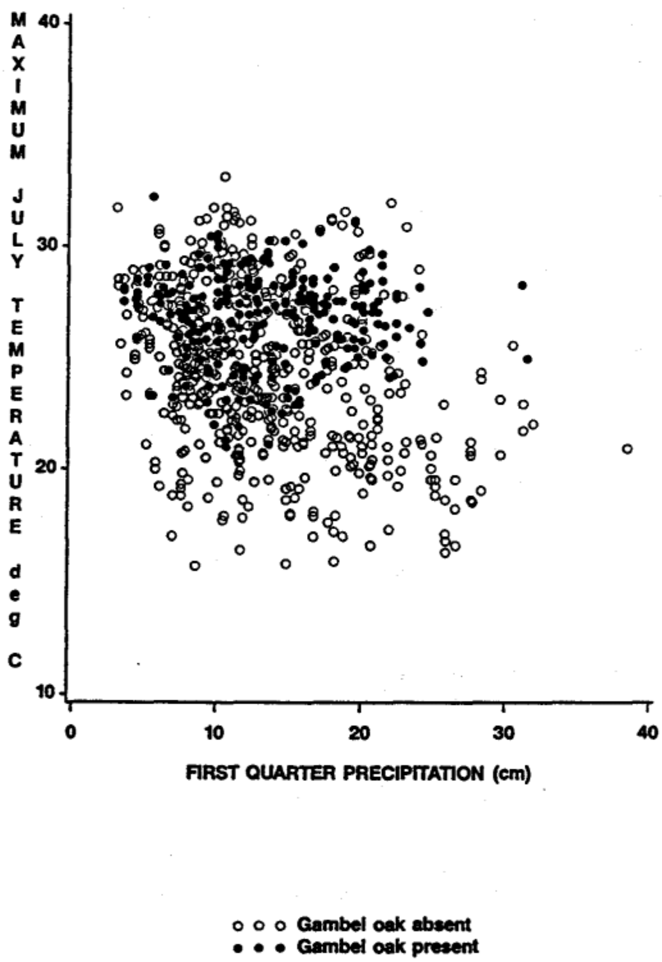


**Figure 1-- Distribution of southwestern forested plots with and without Gambel oak in the environmental space determined by the climatic attributes of maximum July temperature and first quarter (January to March) precipitation.**

## CONCLUSIONS

A major difficulty of landscape evaluation and ecosystem characterization is the simultaneous use of classifications of the biotic and abiotic components of ecosystems. For example, matching plant community and soil classifications is not always an easy task. One reason for this difficulty is that samples of ecosystem components are usually widely dispersed over a study area (Gillison and Brewer 1985), because sampling intensity for one discipline is not usually adequate for another. Use of gradsects can reduce this problem by varying the intensity of sampling according to the steepness of the environmental gradients. The development of sampling methods that allow biotic and abiotic data to be recorded simultaneously in the context of broad quantifiable gradients will improve characterizations of biotic-abiotic relations in ecosystems.

Gillison and Brewer (1985) state that the general principles of sampling along environmental gradients are scale-free; however, the driving variables that influence biotic pattern often change according to the scale of the analysis selected. For example, at continental scales, the climatic attributes that lead to the characterization of climatic regions (Mitchell 1976) are likely to be among the driving variables controlling biotic variability (Neilson et al. 1989). At a finer within-region scale, broad-scale climatic attributes are less important and geology and landforms commonly become the driving variables (Bailey et al. 1993). At increasingly finer scales, other variables (e.g., aspect, slope position, etc.) account for most of the variability in ecological patterns. Therefore, matching the biotic and abiotic components of ecosystems for explicit sampling design is a function of the scale and purpose of the survey. Identification of specific sampling procedures (number and location of samples) is linked to the purpose and scale of the survey, which requires that the scale-dependent driving variables controlling biotic variability be described.

The following are general practical guidelines for surveyors involved in landscape evaluation and ecosystem characterization:

- Define the scale and the purpose of the survey as clearly as possible.

- Review existing data and use them where possible for analysis or as templates for designing new surveys. Recognize that the gain realized by establishing databases for existing data may very well offset the cost.

- Consider the potential of two-stage sampling along environmental gradients (gradsect sampling) as a cost-effective and efficient survey design.

- Choose an appropriate level of technology. Although computer technology such as geographic information systems is faster and allows greater precision than manual methods, the survey principles described in this paper can be applied using maps, pens, and paper.

- Determine the most important environmental variables for selecting gradsects. Match the scale of the variables with the scale and purpose of the survey.

- Determine the appropriate environmental and geographical stratification within gradsects. If necessary, identify finer-scale variables that seem to account for biotic variability within gradsects.

- Make careful decisions about sampling infrequent environmental combinations and the amount of replication within common combinations in light of available resources and the purpose of the survey.

- As appropriate, assess the representativeness of the data collected and test the efficiency of the design.

- Use suitable models such as generalized linear models to complement the survey results. Use results from those models to review the need for additional sampling.

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# Pattern Analysis for Landscape Evaluation and Characterization

**B.T. Milne**

## ABSTRACT

Manipulating cover types and resources within landscapes requires a spatial perspective because of the inherent flows between locations. Recent ecological perspectives that incorporate spatial and temporal scale as surrogates for processes that affect systems are illustrated in three ways. First, species may integrate information about the landscape at different scales depending on home-range size and other ecological and behavioral traits. By analyzing landscape maps at various scales, the concentrations of cover types used by species that operate at different scales can be visualized. Second, a study of the spacing of eagle nests along the coast of Admiralty Island illustrates the use of fractal geometry to estimate habitat availability at many scales. Third, a fractal thinning procedure is introduced which could be used to remove trees and to control the density of remaining trees to provide habitat for species that operate at a particular scale. Interestingly, the a priori choice of thinning scale determines which trees are removed and results in different patterns of remaining trees depending on scale. These three examples illustrate strategies for landscape management that incorporate multiscale sensitivity to resource density and species perceptions of the landscape.

## INTRODUCTION

For management purposes, landscapes are defined as kilometers-wide regions with repeating patterns of various elements (Forman and Godron 1986, Naveh and Lieberman 1984) such as forest patches, agricultural fields, roads, and rivers. The issues of habitat fragmentation, land use change, and resource sustainability in the landscape pose unique challenges to management. The management challenge stems from, among other factors, two quintessential ecological properties: manipulation of one part of the ecosystem may affect other parts, as when nutrients applied to agricultural fields leak into waterways; and landscape elements tend to be clustered so that some types of ecosystems or habitats occur next to other types more often than expected by random chance (e.g., riparian vegetation along streams). Although the familiar riparian example may seem trivial, it represents the prevalence of correlated structure within landscapes. In general, landscapes are ecologically distinct because the flows of resources, organisms, pollutants, and sediment (Cooper et al. 1987) through space are controlled to some extent by the connections between points on the landscape (Turner 1989). Thus, manipulation or management of one landscape element may affect some elements more than others because of the arrangement and spatial context of environments.

Resource management and environmental problems can be effectively addressed at various spatial scales because some environmental factors influence ecological interactions at some scales but not at others (Meentemeyer 1989). Thus, spatial scale can be used as a surrogate for better knowledge about relevant controlling factors (see Johnson et al. 1992). Characterizing ecosystems and landscapes at multiple spatial scales enables managers, ecologists, and designers to partition the causes of system change according to the scale at which the relevant ecological forces operate (Urban et al. 1987, Wiens 1989). Management decisions can be made once the ecological implications of the location, juxtaposition, and flow of resources, pollutants, and species are known (Risser et al. 1984).

The main goal of this paper is to provide examples of resource analysis at various scales and thereby illustrate: (1) resource density may vary with the scale at which it is assessed, (2) there is a statistical regularity to the variation of density that allows it to be characterized for interrelating densities at several scales, (3) species operate at different scales and thereby perceive different densities within the same landscape, and (4) the locations of dense versus sparse resources used by species differ with the scale at which species perceive density. An application to tree harvesting is provided to show how these insights might be used in resource management.

The second goal of this paper is to provide an introduction to the literature of correlation analysis in the broadest sense, both for nonspatial and spatial applications. The quantitative approaches described here enable the landscape to be envisioned as an entity with unique properties, much like the properties which Lamont Cole (1954) identified as unique to populations (e.g., intrinsic growth rate), survivorship schedules, and first age of reproduction. This approach contrasts with more common-sense perspectives in which the landscape is viewed as a collection of parts, each with individualistic properties pertaining, for example, to patch area, perimeter length, and distance to neighbors. The key to applying a landscape perspective is to treat large areas as entities with intrinsic properties.

## Implications of Scale

The long-term perspective of paleoecologists reinforces the understanding of ecologists that ecosystems are affected by forces that operate across an enormous range of scales (Delcourt et al. 1982, Prertice et al. 1991). For example, Woodward (1987) discusses the variation in climate caused by everything from glacial cycles to minute-by-minute variation in sunflecks. Fortunately, pervasive ecological factors exhibit a positive correlation between temporal variation and spatial extent (Delcourt and Delcxiun 1987). Disturbances such as tree falls in forests are small and of short duration compared to the broader, and longer lasting, effects of forest fires (e.g., Shugart and West 1981, Urban et al. 1987). Thus, to understand ecosystem behavior at a particular point requires information at many scales (Rykiel et al. 1988).

A specific example comes from a recent characterization of grazing. Senft and others (1987) show that animals make choices about food at many scales. At the scale of individual plants, grazers must select between adjacent plants based on palatability. Grazers must also choose among clumps of plants, among communities of plants within a landscape, and among landscapes within a region. The spatial distribution of forage plants at each scale is controlled by competition, soil conditions, and land use history at the levels of clump, community, and landscape, respectively. Management of rangeland benefits from a multiscale perspective.

## Correlation Within Landscapes

In the broadest sense, correlation is the nonrandom association between two or more variables. Ecologists have developed and adopted techniques to assess correlations among species, correlations between species responses and environmental factors (Davis and Goetz 1990, ter Braak 1986, 1988), and the correlations of resources or species through space and time (Allen et al.1977). Correlation can be measured in many ways, including the standard parametric and nonparametric coefficients of Pearson and Spearman (Zar 1984), or by multivariate data reduction approaches such as cluster analysis and ordination (James and McCulloch 1990, Minchin 1987, Pielou 1984, ter Braak 1983, Whittaker 1967), or by methods designed to represent nonrandom structure at many scales simultaneously (Mandelbrot 1982, Milne 1991a, 1992a, Palmer 1988, Voss 1988).

Correlations summarize the main patterns rather than portraying all the details, many of which are probably irrelevant (Gauch 1982). Correlations also characterize the potential interactions in nature (e.g., Bergeron 1991). Although correlations are strictly descriptive and do not confirm any causal relations between the measured species or resources, the body of evidence shows that natural systems and landscapes are not random (Gardner et al. 1987), and therefore, the correlations among landscape components are potentially useful for determining the extent to which management applications may ripple through ecosystems and landscapes. Likewise, such analyses may reveal how the proximity of incompatible neighboring habitats or environments may stymie management efforts. Correlations provide a simplification of nature for better understanding of otherwise complex systems. Several introductions and reviews of the various techniques for correlation analysis are available. Among others, techniques for analysis of multivariate relations include Austin (1985), Beals (1984), Green (1979), Gauch (1982), Ludwig and Reynolds (1988; several errors are found in the first edition, but it provides software and a valuable overview), Pielou (1984), and ter Braak and Prentice (1988). Excellent recent overviews and texts about spatial analysis are provided by Cressie (1991) and Rossi et al. (1992).

## Associations Among Environments Vary with Scale

A major organizing concept in landscape ecology is that patterns and correlations vary with the scale at which landscape measurements are made (Mandelbrot 1982, Meenterneyer and Box 1987). Analysis at different scales, as shown below, may reveal different relations between variables (Milne 1991b, Noy-Moir and Anderson 1973). As an analogy, we know that a masterpiece painting can be evaluated in many ways. Aesthetic perceptions of paintings depend on the pigments used by the artist to alter the distribution of light from the visible part of the spectrum (400-700 nm). A technical art historian may use shorter wavelength x-rays or ultraviolet radiation to see very different aspects of the painting and thereby determine the age, authenticity, origin of materials, and ontogeny of the image. Thus, by manipulating the wavelengths of light, observers can learn very different things about the painting. Similarly, by purposely manipulating the scale at which measurements are made, landscape ecologists can tune observations to reflect various processes that operate in the landscape (Johnson et al. 1992) or improve sampling strategies to evaluate the effectiveness of management practices.

From the standpoint of organisms, there is a rich set of cover types or habitats (sensu Whittaker et al. 1973) throughout the environment. If one imagines a landscape populated by overlapping animal home ranges or seed shadows of plants, it is apparent that each of these units intersects a unique set of environments. One home range of a bird may straddle a river and the surrounding riparian area, whereas another may encompass a recently bumed area

Species responses to environment also depend on the area from which the environment is evaluated. A major principle in ecology is that body size dictates many important ecological and behavioral aspects (Peters 1983). For example, Harestad and Bunnell (1979) indicate that home range size of mammalian herbivores generally increases with body mass as:

$$\text{Area} = 0.032\ \text{Mass}^{0.998}$$

where

Area = home range area (km$^2$),
Mass = body mass (kg).

Thus, within a given landscape, small rodents (80 g) may sample the environment within 0.0025 km$^2$ areas whereas ungulates, such as antelope, may sample home ranges over many square kilometers.

As an exercise in visualizing the different perceptual scales of animals living in a common landscape, an analysis was made of a digitized aerial photograph of the pinon juniper woodlands and surrounding grasslands in the Sandia Park quadrangle of New Mexico (fig. 1A). The image, representing woodlands in dark tones, grasslands in lighter tones, and rock outcrops and dense grassland in medium tones, was clustered into three discrete cover types by using an automated cluster-analysis method. The resulting cover-type map (fig. 1B) shows how different vegetation types come together in different admixtures. Given a cover-type map, it is possible to ask how a particular arrangement of cover types might appear to animals that integrate spatial information at different scales. Presumably, species with small home ranges perceive different relative densities of vegetation types than species that integrate information at broader scales.

To make such a comparison, the density of each cover type surrounding each cell on the cover-type map was measured by centering a window of 3 x 3 cells (about 14 m$^2$) on each cell. The number of cells of each cover type was counted and stored at the cell location. Given the three cover types, each cell on the map then contained three numbers corresponding to the densities of the three cover types near the cell. A second cluster analysis was made with the measurements from the 3 x 3 cell windows (fig. 1C). The process was repeated for measurements from 9 x 9 windows (127 m$^2$, fig. 1D). In both cases, the total number of cover classes was held at three to allow comparisons with the original cover-type map (fig. 1B).

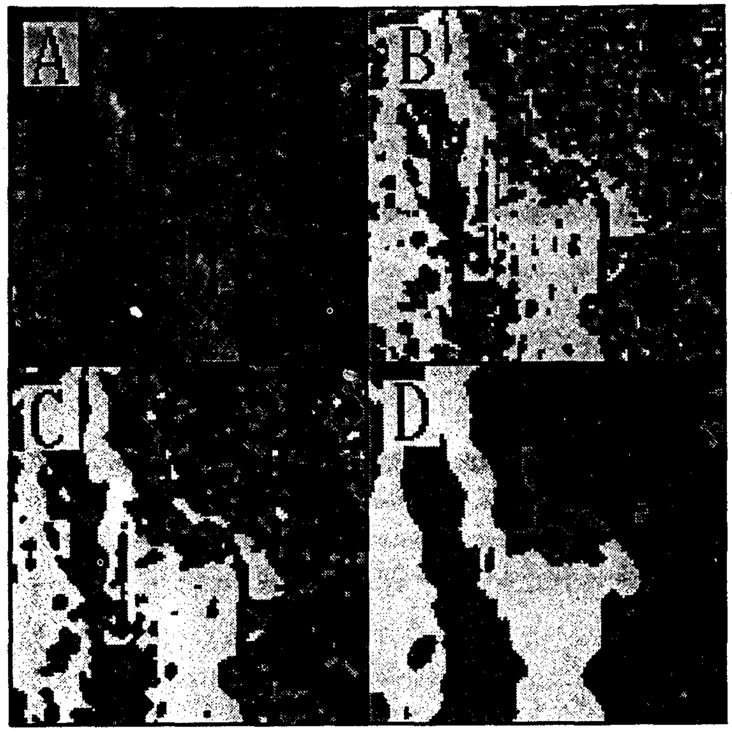


**Figure 1—Changes in the association of cover types with the scale at which habitat density is measured: (A) Aerial photograph of woodlands (dark) and grasslands (light or medium tones); (B) initial cluster analysis of the photograph, showing three distinct cover classes including woodland (black), rock outcrops or tall grass (medium) and short grass (light); (C) reclassification based on densities of cover classes in 3 x 3 cell neighborhoods; and (D) reclassification based on cover-type densities within 9 x 9 cell neighborhoods.**

The most striking feature of the classifications based on the 3 x 3 and 9 x 9 cell neighborhoods is a successive smoothing of the landscape pattern. Basically, the larger neighborhood sizes indicate the dominant cover type in each neighborhood, essentially reducing the prevalence of less dense cover types. This result reflects the general tendency of small sampling areas to produce high intersample variance (Levin 1992, Meentemeyer and Box 1987), thereby suggesting that small species perceive greater variation in habitat from place to place than do large organisms.

A second feature of the results is that a small specialist on any one of the cover types will tend to encounter a more fragmented distribution of the preferred habitat (Milne 1992a). Moreover, because a given cell from the 3 x 3 cell neighborhood analysis may be of one class, but the same location in the 9 x 9 cell analysis may be of another (cf. fig. 1C and D), it is conceivable that species that integrate information at different scales, but nonetheless have the same environmental preferences (e.g., both prefer woodland), will find suitable densities of habitat at different locations within the same landscape (Milne 1992a): Thus body size and the associated variation in home-range size (among other features, Milne et al. 1992) may enable species to coexist.

These insights are based on the study of correlations among habitats measured at different scales. The method for studying correlations is standard in the ecological literature (see James and McCulloch 1990), but the purposeful manipulation of the scale at which measurements are made is more recent (Ver Hoef and Glenn-Lewin 1989, but see Noy-Meir and Anderson 1973).

## Characterization of Patterns at Many Scales

A manager may wonder how a fire or an alien species may propagate through a landscape (Rykiel et al. 1988) or the extent to which management practices affect species and habitats other than those targeted by the management action. For example, an economical plan to eradicate a pest, such as a rust fungus, may entail determining the minimal amount of thinning of a host species population to break the life cycle of the parasite. The critical density is equivalent to a critical interhost spacing distance, or scale, above which hosts are of such low density as to be useless to the pest. Measurements of the landscape must be made at multiple scales to answer these questions.

This section introduces the use of fractal geometry as a tool for manipulating the scale of observation and for delineating the domains (Wiens 1989) of scales over which particular processes arid species operate (Milne et al. 1989, Milne et al. 1992, Peters 1983). The detection of domains could be important in management because domains may enable manipulations to be performed at scales that will not interfere with valuable processes operating at other scales.

## Tools to Characterize Scale Dependence In Landscapes

An operational description of scale includes four components: extent, grain, window, and lag (Milne 1991a, 1992a). Extent is the spatial or temporal breadth of a set of observations (O'Neill et al. 1986), and grain is the smallest resolvable element on the landscape. The grain may be the size of a study plot, the cell size in a digital satellite image, or the smallest mapping unit in photointerpretation. Although landscapes may be represented by high-resolution aerial photographs with photographic emulsions capable of detecting centimeter-long objects, the transfer of the photographs to computerized image analysis systems or geographic information systems often reduces the spatial resolution of the image, thereby resulting in a larger grain size. The precise consequences on subsequent analyses of the coarse-grained, digital representation of the original image may be unknown, although exact assessments of the coarse-graining procedure may be controlled in some cases (e.g., Gould and Tobochnik 1988, Milne 1992b). Grain size and extent are major limits on the ecological information represented by maps.

For purposes of habitat or resource analysis, an analyst may purposely manipulate the size of windows (i.e., analysis scale) within which quantities (e.g., tree density) are evaluated. Here, the length of a window will be represented by L while the extent, or width of the landscape, will be represented by E. Thus, the resolution of landscape data is E / L The smallest possible window size equals the width of a grain. The ability to manipulate window length provides a powerful method to view the landscape at multiple scales, as described below. In geostatistical analysis (Rossi et al. 1992) and in Hurst analysis of fractal time series or profile data (Feder 1988), measurements within windows may be compared among windows which are separated by fixed distances called lag distances. The spatial pattern is then characterized by systematically varying the distance between windows and accumulating a statistic based on comparisons among windows at each of several lags. Lagged analyses generally reveal a critical distance below which measurements are correlated and above which measurements are uncorrelated (fig. 2, Cressie 1991, Davis 1986, Palmer 1988, Robertson 1987). Separation of sample units (e.g., plots, quadrats, transects) at distances greater than the critical distance ensures that the units are indeed independent replicates (see Hurlben 1984), thereby preserving the ability of statistical tests to make accurate inferences (i.e., independence preserves the Type I error rate). Other approaches have been developed for problems involving the analysis of variance for autocorrelated samples (Bhatti et al. 1991, Legendre et al. 1990). Bhatti and others (1991) developed an analysis in which autocorrelation among samples is incorporated into the statistical design to control for spatial correlations among plots or sample units. The adjustments resulted in significant effects that otherwise would have been masked by autocorrelation. An alternative approach of Legengdre and others (1990) is to repeatedly randomize measurements in inherently correlated samples and thereby control for correlations among samples.

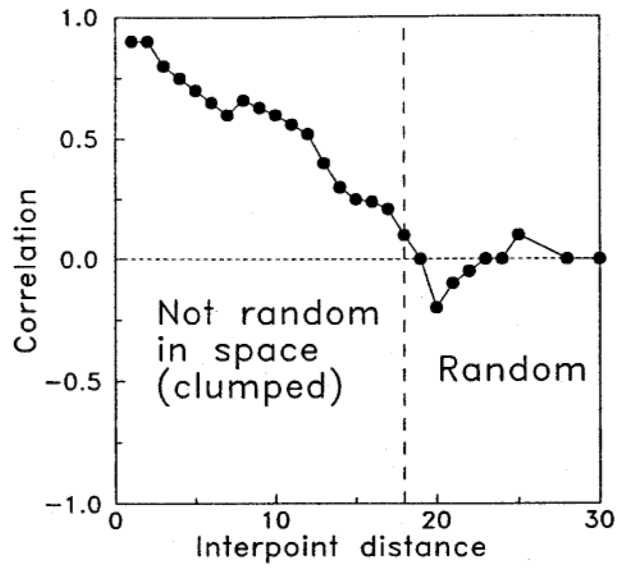


**Figure 2-Theoretical relation between lag distance and correlation. A critical lag distance (vertical dashed line) may be found below which landscape patterns are correlated and above which measured quantities are randomly associated.**

## Fractal Geometry

Fractal geometry (Mandelbrot 1982) has gripped the scientific community because of its usefulness in many kinds of systems (Barnsley 1988, Milne 1991c). Fractal geometry is useful for describing and modeling natural patterns such as terrain, clouds, stream networks, river discharge, archipelagos, resource density, and movements of animals (De Cola 1989, Feder 1988, Gupta and Waymire 1989, Krummel et al. 1987, Lovejoy 1982, Milne 1991a). Fractal patterns share one common feature: magnification of a small part of a fractal pattern reveals a pattern that is very similar to the whole. A simple example is provided by a wadded-up piece of paper. If one tears about one-quarter of a page off of a whole page and then crumples both pieces separately, the two wads will exhibit similar patterns of folds, lumps, and crevices, despite the difference in absolute size.

Quantitatively, fractal patterns are said to obey power laws between some quantity that is measured (e.g., biomass) and the length scale at which it is measured (Tel 1988). Length scale is the length of a sample plot, or the radius of a circle, within which the quantity is measured. The fractal power law is:

$$Q = kL^D$$

where

Q is a measured quantity, such as biomass,
k is a constant,
L is the length scale,
and D is a fractal dimension describing the complexty of the pattern

The power law indicates that the quantity varies as a function of the length L of the plot within which the measurement is made, raised to an exponent. For aerial measurements such as plant cover, the exponent D is expected to equal 2 if the cover is spread uniformly throughout a study area. It may seem peculiar to ecologists, but in fractal geometry, both random and evenly dispersed patterns have a dimension of 2 because they both increase as $L^2$. The fragmentation or clumping of many natural patterns, however, may reveal a particular range of length scales over which cover increases at a slower rate than $L^2$ and is therefore fractal (Orbach, 1986) if the power law is valid over two or more orders of magnitude.

There are many ecological and environmental consequences of fractal geometry. First, the power law illustrates that the apparent amount of a resource may depend on the scale at which it is measured (Goodchild 1980).

Mandelbrot (1982) demonstrates this consequence by asking, “How long is the coast of Britain?” At first one might expect an absolute answer, say 8642 km. If the coastline is measured at successively higher magnification (as when using maps of successively larger scales, e.g., 1:250,000; 1:24.,000; 1:1000), however, one finds an inordinate increase in the estimated coastline length because of the discovery of ever more nooks and crannies. Thus, one must answer Mandelbrot’s question by saying, “Coastline length depends on the scale at which it is measured.” Many “substrates” that are subject to pollution are probably fractal (e.g., the Alaskan coastline, which was contaminated in the Exxon Valdez spill). Fractal geometry has profound implications for the estimation of contaminant concentration in sediments, the abundance of animal habitat, the density of organisms living on fractal substrates, and diffusion (Johnson et al. 1992).

For example, fractal geometry can be used to estimate the number of eagle nests along a coastline. Robards and Hodges (1976) observed that bald eagles nest on the coast of Admiralty Island, Alaska (fig 3, inset). The fractal geometry of the coast is measured by estimating the number of times a caliper can be placed along the coast for several caliper gap sizes (fig. 3). The roughness of the coastline at all scales is described by the slope of the relation between the logarithm of the estimated coastline length and the logarithm of the caliper gap width (fig. 3). The slope of the relation is equal to 1 - D, where D is the fractal dimension of the coastline (Mandelbrot 1982), that is, D is close to 1 for smooth coastlines and D approaches 2 for very rough coastlines. Even though D for coastlines only varies between 1 and 2, a small difference in D corresponds to a large difference in coastline length because D is an exponent. Also, the power law is used to describe the pattern because it is necessary to know L to estimate coastline length at each scale.

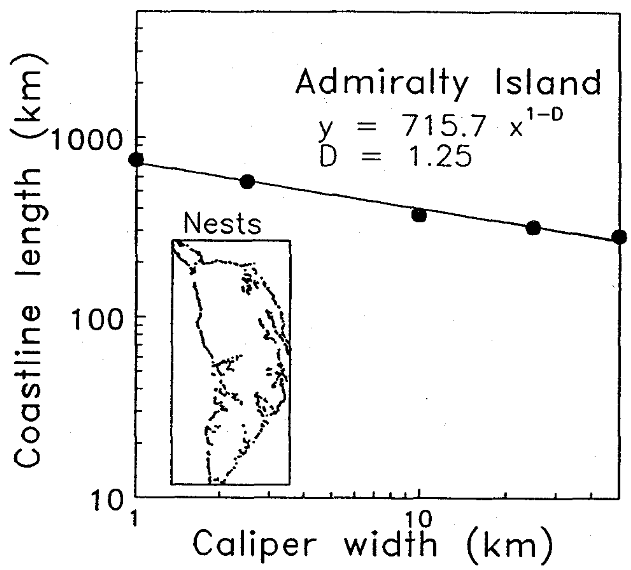


**Figure 3--Fractal geometry of Admiralty Island. Estimated coastline length decreases with Increasing caliper width or decreasing resolution. The inset shows a map of eagle nests tracing the coastline.**

Thus, the number of nests along the coast should depend on the average distance between nests (i.e., territorially controlled length scale) and the fractal dimension of the coastline. Given an inter-nest distance of 0.782 km (Robards and Hodges 1976), the coastline length from the perspective of the eagles is $715.7(0.782)^{-0.25} = 761.1$ km. Thus, one would expect a total of 761.1 / 0.782 = 973 nests. Robards and Hodges (1976) observed 893 nests. The discrepancy suggests either an error in their estimate of coastline length (which they probably measured at some arbitrary scale and not at the scale at which eagle nests are spaced) or perhaps a smaller observed population than expected because of some ecological limitation on size of the eagle population. If one assumes that the fractal characterization of the coastline applies to barnacles (which are ~0.00002 km long) then as many as 535,110,000 barnacles could form a chain around the island. One can barely imagine the number of molecules of crude oil that could fit along such a coastline. Extrapolations below the grain size of the original data can only be done under the assumption that the fractal scaling law is valid. It would be possible to validate the law for Admiralty Island by analyzing the coastline from high-resolution aerial photographs with grain sizes (i.e., minimum values of L) equal to that of the length of the organism of interest.

Second, “length scale” may be thought of as the “scale at which organisms operate.” Species differ greatly in the scale at which they operate due to differences in body size, home range area, and movement rates (Peters 1983). Thus, there is potentially a tremendous disparity in the pattern and concentrations of a resource or habitat available to two species which operate at different scales in the same landscape (Milne 1992a). Milne and others (1992) simulated the effects of fractal landscape structure on different species of animals living within an 810-ha landscape. Forage biomass within 30-meter-wide grains was estimated from remotely sensed data, and the simulated animals were allowed to forage within movable home ranges during three winters. The model relied on empirical allometric relations to control all the ecological, behavioral, and physiological traits of the species. Thus, all species behaved according to the same rules which were scaled according to body mass. There were major differences in the time at which the animal populations of the different species starved simply as a function of body size and home range area because forage density varied with scale. The study suggests that fractal characterizations of landscapes and resource density could be incorporated into demographic models to measure the starvation risk each species would face.

Estimates of resource (or pollutant) abundance depend on the scale at which resources are perceived either by cartographers or by nonhuman species. For example, imagine a landscape with woodland cover that increases with scale as $M(L) = 0.5\ L^{1.4}$, where M(L) is the average number of square meters of woodland surrounding any particular grain. If a cartographer maps a particular woodland cover type with a grain size of L = 300m, then the average number of square meters of woodland surrounding any particular grain is $0.5(300)^{1.4} = 1468$. If small rodents use the woodland within home ranges that are L = 30 m, however, they will see an average of 58 $m^2$ of woodland within their home range. Without the fractal relation, one might use a Euclidean transformation of the cartographic estimate which did not account for the fractal scale dependence. Specifically, if the map at a 300-m resolution showed 1468 $m^2$ of woodland, then a Euclidean estimate of cover at the 30-m scale would be $1468(30/300)^2 = 14.68\ m^2$ which is 14.68/58 = 25 percent of the area actually available. The discrepancy comes from the failure of the Euclidean approach to account for the clumping of woodland.

Fractal geometry is a tool for characterizing resource abundance at various scales, for cartographically rendering spatial distributions so that the perceptions of other species are acknowledged, and for explaining how mixtures of species coexist in a given landscape. The mechanisms of species coexistence are important as explanations of the origin and maintenance of biodiversity, one of the major concerns of conservationists, ecologists, and politicians.

## The Diversity of Landscape Models

Beginning with Mandelbrot (1982) and followed by the reviews by Burrough (1986), Milne (1991a), Stanley (1986), and Sugihara and May (1990) it has become clear that there is a wealth of fractal models for use in characterizing ecological patterns. Although a complete discussion is impossible here, there are several general approaches for use in ecology and resource management. For static landscape patterns composed of patches there are three general approaches to fractal modeling: “box counting,” perimeter-area measures, and correlation dimension approaches based on Mandelbrot measures (Voss 1988).

In the box-counting method, a grid of cells larger than the grain size is placed on a map of some cover type that occupies less than 100 percent of the area and the number of occupied grid cells is counted. Then, successively coarser grids are placed and the counts are repeated. The number of occupied cells will decrease as the length scale (L) increases. It is useful to refer to this dimension as $D_b$ to indicate that it was obtained by the box-counting method. This method is fairly simple but somewhat sensitive to the placement of the grid, particularly for large grid cells (e.g., E / 3). Whether large gaps within the fractal pattern fall in, or out, of large grid cells depends on the placement of the grid. Best results are obtained by repeatedly shifting the grid until a minimal number of cells are occupied at each grid mesh size. The problem is minor for small grid sizes.

In the perimeter-area approach (Krummel et al. 1987, Lovejoy 1982) the logarithm of patch perimeter length is regressed against the logarithm of patch area to obtain a slope (slope = $D_P/2$) that indicates how rapidly perimeter increases with patch area; smooth patch edges yield a dimension $D_P$ (where p indicates the perimeter dimension) close to 1.0 and rough edges have dimensions that approach 2.0. Estimates of dimensions from this approach are extremely sensitive to several procedural nuances. First, patches touching the edge of the map should not be included because the perimeter is truncated by the map edge. Second, raster digital maps of patches cause severe distortion of the patch shape for small patches so care must be taken to limit estimates of dimensions to large patches (Milne 1991a). Together, these limitations reduce the number of usable patches to just a few large patches that are completely enclosed by the extent of the map.

In general, there is no null hypothesis to describe the expected perimeter-area fractal dimension for a given map. The approach of Krummel and others (1987), however, can be extended to provide a null hypothesis that the dimension is constant for all size patches. Krummel and others (1987) observed 2000 forest patches and ranked them from small to large. Then they regressed the logarithm of perimeter versus the logarithm of area for the smallest 200 patches and plotted the resulting slope (i.e., $D_p$) as a function of the mean patch area for the group of 200 patches. They repeated the analysis for successive groups of larger patches and plotted $D_p$ as a function of mean patch area. A transition between smooth, small patches and rough, large patches was found that probably reflected a transition between the scales at which agricultural practices control patch shape and the broader scales at which geomorphic features prevent farmers from maintaining straight patch edges. An appropriate null model for the analysis would be formed by randomly selecting 200 patches from among the 2000 and then computing the dimension for the subset. Repeated resampling in this fashion would produce a distribution of dimensions for comparison with the dimensions observed for subsets from the rank order. If the observed dimensions fell outside of the confidence interval generated by the randomization process, then the observed dimensions would indicate that some particular process was operating at those scales.

By far, the most versatile approach to estimating dimensions and characterizing the scale dependence of pattern is based on the Mandelbrot measures (Voss 1988) which effectively measure the local density of cells on binary images at various scales (Milne 1992a). In essence, the method involves passing windows of different lengths over binary images in which cells are either “occupied” by a cover class of interest or “unoccupied.” By convention, the windows are centered on occupied cells and the number of occupied cells is then counted. This placement method is used because it provides a direct measure of the association within a class of cells. After completing counts at a window size L, the mean number of occupied cells is computed and the logarithm of the mean values are then regressed versus the logarithm of L to obtain a slope, which is the fractal dimension of the occupied cells. Given that digital analyses require windows of an odd length (e.g., 3, 5, 7) the windows can only be centered on occupied cells which are greater than (L-1)/2 cells from the edge of the map. Consequently, larger windows miss more occupied cells near the edge, so the mean densities are based on fewer windows. Thus, a premium should be placed on using large maps for which L is much greater than the grain size but is no more than one-third the extent. The versatility of the Mandelbrot measure approach stems from its ability to detect disturbance within fractal landscapes (Milne 1992b) and to provide visualizations of resource density at various scales (Milne 1992a). An example of this approach is provided next.

## Forest Thinning

In this example, a forest is evaluated for harvesting. Tree density is evaluated at two scales (i.e., window sizes) and 25 percent of the trees are removed. The locations of tree removal, however, depend on the scale at which density is evaluated, so evaluation of density within small windows leads to the removal of different trees than when density is measured in larger windows. This difference has potential consequences for species that use the remaining trees as habitat, with the consequences depending on both the scale at which trees were removed and on the scale at which the organisms evaluate habitat availability.

The fractal thinning procedure begins with an image of trees in the Sandia Park quadrangle, New Mexico (fig. 4A) where each cell in the image represents a 1.25-meter-wide area. The gray-scale image was divided into areas with trees (dark) and without trees (light, fig. 4B). Then, tree density was evaluated at a scale of 8.75 m and 43.8 m by centering windows on cells occupied by trees and counting the number of occupied cells within the window.

Thus, a map of tree density was obtained (fig. 4C) by using windows of 8.75 m to count the number of occupied cells in figure 4B. The most dense locations were identified (fig. 4D) and 25 percent of the trees were eliminated (thinned) from the most dense areas at the scale of 8.75 m. Thinning produced a new map (fig. 4E). The procedure was repeated at the 43.8-meter scale to produce a density map (fig. 4F), a map of removed trees (fig. 4G), and the final forest cover-type distribution map (fig. 4H).

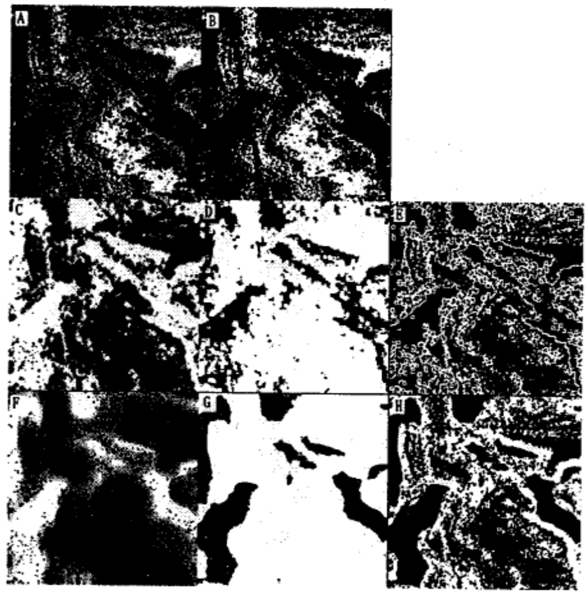


**Figure 4--Visualization of forest thinning at two scales: (A) Digital image of woodlands showing trees in dark tones, (B) classification of photograph to provide a map of trees (black); (C) tree density measured within 7 x 7 cell windows is proportional to brightness, (D) candidate tree cells for thinning 25 percent of the total tree cover based on the most dense 7 x 7 cell windows (black), (E) remaining trees after thinning (white) at the 7 x 7 scale, (F) tree density measured within 35 x 35 cell windows is proportional to brightness, (G) candidate tree cells for thinning 25 percent of the total tree cover based on the most dense 35 x 35 cell windows (black), and (H) remaining trees after thinning (white) at the 35 x 35 scale.**

The fractal approach to tree thinning reveals three main features. First, the relative density of trees within different sized windows varies throughout the landscape. Locally dense places at one scale may not be dense at another scale (fig. 4C and F). Scale-dependent density has implications for the flow of parasites and fire, both of which are processes with inherent limitations in the rate at which spread occurs. The neighborhoods of potential host plants for a parasite can be modeled by a large window size to represent the relatively large dispersal distance provided by birds. In contrast, the neighborhoods of potential competitors for light or water could be modeled with a smaller window to represent the limited distances over which trees interact. Second, exactly which trees are removed depends on the measurement scale even though the total number of trees removed may be constant. There are important tradeoffs in resource extraction costs associated with this approach, as it is many times more difficult to remove trees that have been selected with a small window size because they are less clustered in space. Third, the pattern of remaining trees differs among thinning scales (fig. 4E and H) which has implications for the effective densities of habitat that would be perceived by species of various sizes living in the thinned landscape.

The fractal thinning approach contrasts with traditional methods of clearcutting in several ways. First, the fractal method uses the existing pattern of tree cover to determine which places will be cut. Given the natural variation in tree density with slope, aspect, and elevation, the fractal method tends to preserve the natural pattern. Thus, the visual properties of the forest tend to be preserved. Second, the fractal method creates gaps of different sizes in the forest, especially at small scales, which may reduce wind damage at the edges of cut areas (see Franklin and

Forman 1987). Most of the microenvironments created by thinning may help in the reestablishment of trees and simultaneously reduce erosion and soil temperature compared to traditional clearcuts.

## CONCLUSIONS

The perspectives outlined in this chapter emphasize the connections between one place on the landscape and another. The assumption in these analyses is that processes and species interact with the environment at various scales, which can correspond to dispersal distances, home range areas, or flow rates. In attempts to represent the connections, analyses are made at several scales to reveal differences in the patterns and correlations among habitats as they may interact with resource management decisions.

## ACKNOWLEDGMENTS

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# Species Richness and Scale: The Application of Distribution and Abundance Models

**S.M. Glenn and S.L Collins**

## ABSTRACT

Metapopulation models of species distribution and abundance may apply to many ecostystems and organisms. These models assume that natural communities are dynamic and that species do not necessarily occupy all the available suitable habitats. We describe how to test several metapopulation models to allow land managers to determine if the models apply to their systems. These models may, be used to predict community structure and dynamics. Current models of wildlife and habitat relations do not adequately represent species that act as metapopulations and do not consider the spatial structure of landscapes. Therefore, several recommendations are made to managers for applying both wildlife habitat models and metapopulation models.

## INTRODUCTION

There is growing evidence that metapopulation models of species distribution and abundance may apply to many ecosystems and organisms. These models assume that natural communities are dynamic and are structured with some type of spatial patchiness. Given that these assumptions are true, land managers should address processes driving these models. If the models hold in a specific ecosystem, they may be used as predictors of community structure and dynamics. The primary objective of this paper is to review some existing theoretical models of community structure, discuss testing of these models, and re-examine current wildlife habitat models.

Much research in community ecology has focused on patterns in species richness as they relate to disturbances and species interactions, such as competition (Connell 1978, Pickett and White 1985). The processes directly responsible for diversity at a site are local immigration and extinction. Immigration is the arrival of a new species at a site, and extinction is the disappearance of a species from a site. Study of such processes is difficult and has rarely been attempted, as seen in island biogeography studies (see McCoy 1982). Detection of a new immigrant at a site requires not only the ability to locate the species in an inventory, but also documentation from prior inventories that the species was not present earlier. Local extinction is often harder to detect because the species may not be obvious. These problems hamper attempts to quantify changes in species richness and to determine if there are predictable patterns in these changes. The problems are confounded when defining the presence of viable populations. For example, it may not be important to determine if a viable population of a robust sunflower has immigrated when studying patterns of species richness in herbaceous plants among 1 square-meter plots. Simply the presence of the sunflower may have ecological consequences. Managers of an old-growth stand in a National Forest may be concerned, however, with maintaining a viable population of a rare species that is dependent on that vegetation type. They may also be concerned with immigration of new species to the stand, if these species are dominant, exotic species taking advantage of open space, or rare species migrating to a previously unoccupied location. Therefore, the focus of monitoring the health of a natural area is more than just the total number of species. An understanding of how communities are structured may help predict possible consequences of management activities.

The problems in assessing species immigration and extinction and in predicting diversity have led some researchers to use mathematical models to paraphrase nature. Given the number of possible interactions of species and the complexity of natural communities, there have been many simplifying models proposed to help understand communities. In this paper, examples of theoretical models proposed to describe communities are presented. The implications and testing of such models also are discussed. Many models currently used in studies of wildlife-habitat relations are empirically derived and are applicable to only a narrow set of sites. The theoretical models presented here are difficult to test but tend to be more general in application. It is necessary to test the assumptions and mechanisms of these models to determine if the models are applicable to management. Each model is concerned with the analysis of patterns of species distribution across a region and species abundance at sites in a region. Dominant species are defined as species found in high abundance at a site, such as big bluestem in tallgrass prairie, or racoons in campgrounds. Many models assume that dominant species are often widely distributed. There are some species, however, that are not locally abundant at any site but have wide distributions, such as many large predators. Some rare species may be locally abundant but have very restricted distributions. Therefore, it is important to understand the assumptions of the models before using them to drive management plans. The species of interest for a specific project may not be the species best represented by any of the models.

## Distribution and Abundance Models

Species distribution is the geographical area over which a species can be found. Abundance is the population size or number of individuals at a location. Distribution and abundance are quantifiable, especially for dominant species, and predictions can be made regarding patterns in these variables. We will consider four multispecies models that predict how species are distributed across a region or landscape. These theoretical models have different assumptions about processes that control species distribution and abundance. Explicit descriptions of these assumptions are not always available but are implicit in the model structure. These four models are given as examples of available models (see Gilpin and Hanski 1991).

Levins (1969) developed a model based on local immigration and extinction dynamics of populations. In this model, immigration of a new population to a site is proportional to the number of sites already occupied by a species. Extinction is random and unrelated to the number of sites occupied. This model predicts that most species will be distributed at an intermediate number of sites in a region. The other species will occupy somewhere between all sites and no sites in a region (fig. 1A).

The core-satellite model developed by Hanski (1982a) is a modification of Levins' model in which extinction is related to the number of sites occupied. Hanski's model also assumes that the species distributed across all sites will be abundant at those sites (abundance is correlated with distribution). Hanski's model predicts that some species will be found at very few sites while other species will be found at all sites (fig. 1B). The infrequent species are termed satellite' species and the frequent species are 'core' species.

The models of both Levins and Hanski are metapopulation models that assume subpopulations are distributed across the region and are connected by dispersed individuals (Hanski and Gilpin 1991). There is no strong environmental gradient or heterogeneity, and all sites are able to support all subpopulations. The models are dynamic; that is, species are more or less frequent in samples depending on random fluctuations in immigration and extinction. In Hanski's model, species can switch between core and satellite categories, being very common one year and then rare the next year. Hanski later notes that core-satellite switching is rare. Detection of switching, however, may depend on the length of time the community is observed.

Brown (1984) developed a model to predict regional patterns of species distributions based on the assumption that a species would be most abundant at the center of is range and would decrease at the edges of its range because of limiting environmental conditions. Species ranges overlap, with different species having peak abundance at different locations. Such species distributions may be dynamic. When these distributions are sampled over a region, most species will be found at very few sites, and there will be few species found at all sites because few species are found in optimal conditions at all sites (fig. 1C).

Kolasa (1989) proposed a hierarchical regional model of species distribution and abundance. Kolasa assumed that communities could be divided into species that partitioned the environment into fine or coarse habitats at different scales. Some species would be widely distributed across all environments and would be abundant everywhere. Other species may perceive the environment differently, such that habitat is divided into smaller patches for which each species is specialized. These limited species distributions were nested in the distributions of the dominant species and these species were not abundant at any given site. Further, subdivisions may occur with additional levels of subdominant species. The result of sampling a region is that there will be several species found at all sites, another mode of species found in few sites, and so on down through the levels of subdominant species (fig. 1D).

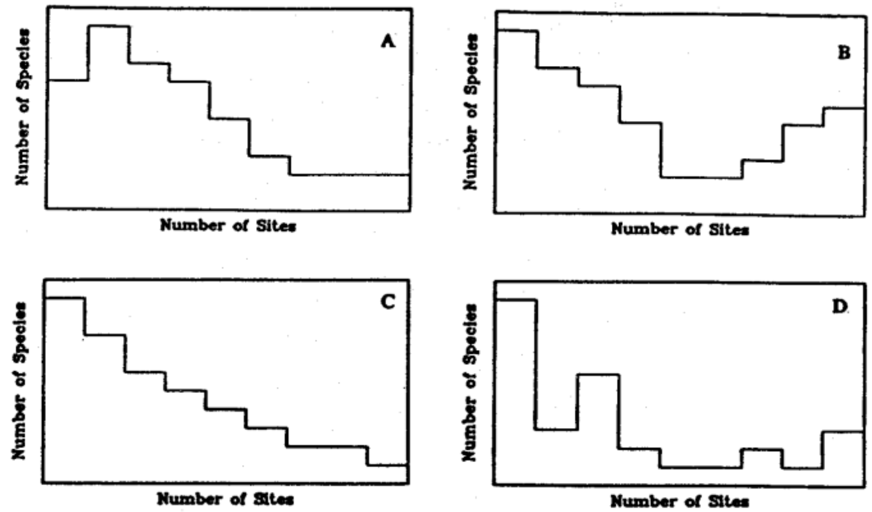


**Figure 1-Number of species found In different numbers of sites based on four models (see text): A) Levins (1969), B) Hanski (1982a), C) Brown (1984) and D) Kolasa (1989).**

## Model Testing and Scale Issues

The most common application of distribution-abundance models is to graph the number of species found with different frequencies to determine if patterns are consistent with the models (Collins and Glenn 1990, Gaston and Lawton 1990, Gotelli and Simberioff 1987, Hanski 1982a, 1982b). A unimodal distribution is predicted by both Levins' (1969) and Brown's (1984) models, with Brown's models specifying that the mode should be at few sites. A bimodal distribution (core-satellite) is predicted by Hanski's model. Kolasa's (1989) model predicts several modes depending on how many levels are found in the hierarchy. Given any sampling protocol, species presence-absence data are used to determine species frequency (number of sites where a species is found). The number of species with a given frequency (or frequency class) are graphed and compared to model predictions. Results of these comparisons have differed and may indicate that some taxa are not well represented by metapopulation models.

In analyzing patterns in species distributions, it is important to address the appropriate scale of the model. Although all these models are regional models, the region is not explicitly defined. Some regions may be defined spatially, other regions may depend on the distribution and dispersal abilities of the metapopulations. Given that the definition of an appropriate scale is not clear, it may be necessary to examine the models at different spatial scales. For example, data on vascular plant species were collected from several prairie sites at the Konza Prairie Research Natural Area in Kansas (Collins and Glenn 1990). The number of sites in which each species was found was used to create a frequency diagram to determine if the predictions were consistent with Hanski's model. The frequency diagrams differed with spatial scale, with predictions from some scales agreeing with Hanski's predictions more or less than at other scales (Collins and Glenn 1990). Different factors may be operating at different scales, thereby making one model more appropriate at a given scale and less appropriate at a larger or smaller scale (Gaston and Lawton 1990).

Model validation is complicated by different models often making the same predictions. The models of both Levins and Brown are not distinguishable if most species are found at few sites; the single mode of Levins' model may fall at zero as in Brown's prediction. Therefore, validating a model does not necessarily mean that the mechanisms or processes hypothesized in the model are true.

The assumptions of any model need to be tested. For example, if abundance is measured as the number of individuals, Hanski's model assumes that this value will be positively correlated with the number of sites species occupy (fig. 2). Abundance often is not directly measured, however, because it may be difficult to distinguish individuals. For example, a measure of canopy cover or total biomass may be a more appropriate measure of abundance in a clonal grass, where it is difficult to define where an individual starts and stops. Although abundance has been measured in several ways, a positive relation between distribution and abundance has been found in several studies (Collins and Glenn 1991, Gotelli and Simberloff 1987, Hanski 1982a). Hanski's model also incorporates assumptions about immigration and extinction rates. In taligrass prairie plant communities, immigration was defined as the appearance of aboveground living biomass of a species in a plot that did not contain the species during the previous sampling period (Collins and Glenn 1991). Local extinction was assumed when no aboveground living biomass was found for a species that had been in the plot during the last sampling period (Collins and Glenn 1991). Rapid dynamics consistent with the assumptions of Hanski's model were found in prairie plant communities at large spatial scales (square kilometers) (Collins and Glenn 1991).

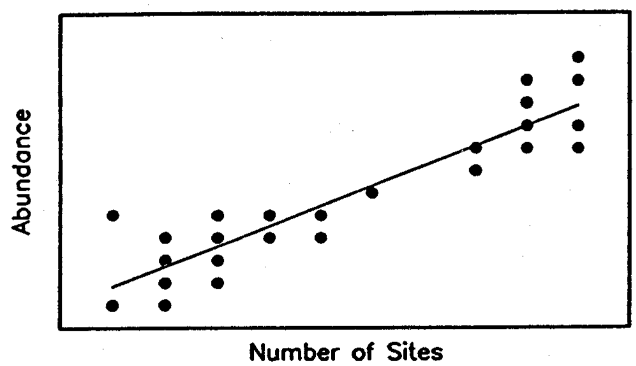


**Figure 2-Relation between abundance of a species at a site and the number of sites the species occupies as assumed by Hanski's (1982b) core and satellite species model.**

The spatial dynamics of metapopulation models imply that suitable habitat patches may not be filled to capacity at any one time (Harrison 1991). Rapid temporal dynamics have been found for some taxa (Braithwaite 1991, Glenn and Collins 1992); however, species switching (from becoming infrequent one year to being found at all sites another year) implies that these "empty" habitats may simply be a sampling artifact. Therefore, species distributions and abundances should be monitored over several years to determine if these spatial dynamics occur. Hierarchy theory predicts that species dynamics will be rapid at fine scales of analysis but constrained by factors operating at larger scales (Allen and Starr 1982). The scale of sampling affects the patterns observed; consequently, constraints on species distributions and abundance may be more easily identified at some scales than others. A hierarchical sampling strategy, therefore, may be useful to determine the scale most applicable to management needs.

## Landscape Issues

The consequences of fragmented habitat are important issues in conservation biology. Additionally, changes in landscape configuration have become a major focus in landscape ecology (Forman and Godron 1986). Accordingly, most current research concerns of landscape ecology are related to many issues tackled in metapopulation models, that is, survival of species and communities, fragmented environments, patch definition, and corridors (Forman and Godron 1986). The study of fragmentation in any context involves several components, such as patch size, heterogeneity, surrounding habitat composition, and edge effects (Rolstad 1991). The concept of metapopulations operating in a fragmented environment has warranted attention of the conservation community (Hanski and Gilpin 1991). A species in a newly fragmented habitat, however, may lack sufficient dispersal ability to act as a metapopulation (Hanski and Gilpin 1991). The role of the "rescue effect" is explicit in some metapopulation models, where a small population may be saved from local extinction by repeated immigration of individuals (Brown and Kodric-Brown 1977, Hanski 1985). If a species is unable to disperse across newly cleared land, small isolated populations may have only a limited probability of survival.

Metapopulation models may be applied in cases where a land system is being designed to encourage migration between metapopulations. The following items should be considered in such land management designs: spatial patterns of corridors and patches are important in determining population persistence (Fahrig 1990); there may be general response trends of metapopulations to corridors dependent on the populations dispersal abilities; dispersal may occur early in the reproductive season in some species occupying early successional habitats (Hansson 1991); habitat specialists may be very sensitive to boundaries with altered habitats (Hansson 1991); and it may be possible to predict genetic consequences of metapopulation structure (Gilpin 1991).

## Fish and Wildlife Habitat Models

Habitat suitability models are empirical models used in the United States to predict species distributions (U.S. Fish and Wildlife Service 1981). Such models tend to be both species- and site-specific and assume that some aspect of a population can be predicted from a measure of habitat (Salwasser 1984). In construction of these models, an abstract variable is derived from combining several suitability indices. These indices are scaled indicators of the species' expected performance at a site with a given value of a measured environmental variable. The environmental variables measured are assumed to be the major controls of presence, abundance, or distribution of the species. Because of the specific and empirical, nature of these models, they are best viewed as verifiable predictions of where a species is likely to survive (i.e., they do not describe causal functions) (Schamberger et al. 1982). As in other community models, the processes that control species distributions should be tested to support use of these models. Several reviews of the usefulness and limitations of these models (Cooperrider 1986, Morrison et al. 1992, Van Home and Wiens 1991) suggest that the initial choice of variables to be included in such models often limits the usefulness of their predictions. This observation is based on the fact that distribution of species may be controlled by many interacting factors and may not be adequately modeled by using two or three arbitrarily chosen variables. Important habitat features, such as edge effects, species interactions, and landscape pattern are rarely incorporated into such models as the controlling environmental variables. Other limitations of habitat suitability models involve the implementation of the model, and the inability to fully test model output (Van Horne and Wiens 1991). These models often produce a synthetic index that cannot be compared to a field measurement to gauge the accuracy of the index.

Several other models have been proposed for use because of limitations in the habitat suitability models; however, no particular model has been widely adopted. The following is a brief description of some of the proposed models. Simple models assume that species presence is correlated with habitat and can be predicted from a habitat map covering the species' geographic range (Ruggiero et al. 1988, Scott et al.1993). Habitat capability models often predict additional attributes, such as population density or biomass, by using methods similar to habitat suitability models. Multispecies models are being developed and are more likely to be successful if all species are controlled by strong environmental factors (Bain and Robinson 1988, Van Home and Wiens 1991). A probabilistic approach, such as found in the Bayesian and pattern recognition models used to determine habitat suitability, may be more appropriate for modeling spatially and temporally dynamic species. In these models, the population depends on the probabilities of various environmental factors occurring, and such models may be integrated with vegetation response models Kirkman et al. 1984, Moeur 1984). In this way, predictions can be made that take into account successional changes in vegetation. In some cases, spatial effects, such as minimal required area, are specified (Davis and Detain 1984). Spatial context is, however, rarely considered.

Habitat evaluation procedures incorporate the quantities of different habitats in an area in addition to the habitat suitability index of each site to rank areas for potential protection or mitigation. These procedures explicitly incorporate some landscape factors that affect habitat diversity. Satellite imagery and geographic information systems are being used to make this process more cost-efficient (Wakely and O'Neil 1988). The models need to be spatially explicit for taxa sensitive to a patchy environment. For example, in Australia the position of monsoon forest patches in relation to sedgelands was critical in determining water buffalo abundance and effects on fauna (Braithwaite 1991). Top-down models have been proposed for systems too complex to model all species individually (Van Home and Wiens 1991). Only relevant details pertaining to distribution of a group of species are incorporated in these models, and there usually is ho underlying mechanism hypothesized. Geographic information systems are useful for this type of modeling in situations where species are constrained by spatial patterns of the environment.

## CONCLUSIONS

Metapopulation models may apply to some species of management concern that are not well represented by current wildlife habitat models. The following recommendations should be considered by managers interested in modeling species distribution and abundance:

- Testing of existing models should be carried out at different spatial scales, and should include testing model assumptions and mechanisms, not solely model predictions. Current models have not been adequately tested to distinguish among them and to determine which are most appropriate at which scales and for what taxa. This step is needed before it can be assumed that metapopulation dynamics are important or unimportant considerations in management decisions.

- Spatial-temporal variability in populations should be measured and used in model development instead of assuming that all available habitat should be occupied as assumed in habitat suitability models.

- Field crews should note the position and surroundings of a patch of habitat in a landscape. This may be as simple as noting surrounding land cover type(s) and land use(s).

- New Wildlife habitat models should be developed that incorporate aspects of spatial structure. Models that predict optimal spatial patterning of habitats are needed. Spatial processes may be easier to model with top-down models; however, these models should be tailored to fit specific land management planning needs.

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# Stream-Riparian Ecosystems: Rationale and Methods for Basin-Level Assessments of Management Effects

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## ABSTRACT

Stream-riparian ecosystems are directly responsive to land management practices. This paper addresses the sampling and analysis of stream-riparian ecosystems in a hierarchical framework of spatial and temporal scales. The paper also establishes recommendations for a protocol to assess the effects of land management practices on aquatic and adjacent terrestrial (riparian) conditions extending to basin-wide and other landscape levels.

## INTRODUCTION

Of all aquatic habitats, streams show the greatest and most intensive interaction with terrestrial forest lands. Streams are products of their catchments (Hynes 1975), and particularly in forested areas, their environmental conditions and biotic communities are strongly influenced by the nature and state of the surrounding lands within a catchment or basin. The adjacent streamside (riparian) environment is the principal interface between the terrestrial uplands and streams. Because of the integral relation between stream and riparian environments, the two are best considered together and are often regarded as a single ecosystem (Cummins et al. 1989, Gregory et al. 1991, Minshall 1988).

The critical ecological role of stream-riparian environments establishes their importance to resource managers. These environments are, perhaps, the most diverse and productive areas within a watershed; consequently, they also have the highest potential for conflict on the use of timber, grazing lands, recreation, water, and wildlife resources (Thomas et al. 1979).

This paper addresses the essential features of stream-riparian ecosystems from the perspective of resource management. The key parameters in these ecosystems are identified, and the critical role of the riparian environment as an interface between the uplands and the stream is described. Spatial and temporal dimensions of stream-riparian environments also are reviewed to identify the appropriate focus for studying and monitoring management effects and to provide a basis for applying a landscape approach to their analysis. Because many resource management activities, particularly those on the land, result in displacement of stream-riparian ecosystem conditions away from those found before European settlement, the changes in these systems are considered both generically and with specific reference to major land management activities.

Human disturbances on these systems often have occurred so far in the past and have been so gradual or persistant that resource managers typically do not recognize their occurrence. Recovery, if it occurs, may require decades to centuries. Consequently, measures of predisturbance conditions are required for comparison. This requirement may be met by using conceptual models and reference sites. When predisturbance conditions are not known, or closely paired reference and disturbed watersheds are not available, space-for-time substitutions and subsequent inferences can also be useful. The concept of biological integrity and its quantification have paved the way for the approach and procedures recommended below; the procedures are intended for use on forest and rangelands at various scales of evaluation. The ideas developed in this paper need to be tested and refined through application in these settings.

### Key Ecosystem Parameters in Stream-Riparian Ecosystems

Stream ecosystems are organizational units of interacting physical, chemical, and biological entities, and possess structural and functional attributes within a variety of spatial and temporal dimensions. The biotic community of a stream, consisting of all the organisms (biota) in a given area, interacts with the physical and chemical environment to produce a flow of energy and cycles of materials leading to discernible structural (e.g., biotic diversity) and functional (e.g., trophic) organizations. In natural stream ecosystems, biotic components adapted to each other and to their abiotic environment over periods of evolutionary time.

The principal abiotic factors in stream ecosystems are flow, substratum, light, temperature, and dissolved chemicals. Strong unidirectional flow and the consequent continuous and relatively rapid renewal of water distinguish streams from other aquatic environments. Flow implies movement (current), turbulence, erosiveness, and transport of organic and inorganic loads in solution or suspension. Water renewal implies a continual replenishment of nutrients and gases and removal of waste products. Current acts as an energy subsidy in transporting materials and organisms and decomposing matter, including plant litter (Hall 1972). Turbulence results in mixing and in the virtual elimination of thermal and chemical vertical stratification in all but the deepest and slowest rivers. Important components of flow are velocity (speed of water movement), volume (discharge), and annual pattern (regime).

The principal biotic components of streams are (1) primary producers- attached algae (periphyton) and rooted vascular hydrophytes; (2) terrestrial-plant litter (allochthonous detritus)-leaves, twigs, bark, and so on; (3) consumers-invertebrates, especially mollusks, crustaceans, and insects (ranging from microscopic to macroscopic), fish, and a few amphibians, reptiles, birds, and mammals; and (4) decomposers-bacteria and fungi. Within an ecosystem, important measures of structural organization are species composition, richness, and dominance; measures of functional organization are food web composition -and complexity (Cohen 1989, Pimm 1982) in both stream and riparian components. In addition, functional feeding group structure, and the relative importance of photosynthesis to community respiration and the ratio of grazers to consumers of terrestrial-plant fitter, commonly are evaluated in streams. Key ecosystem processes in streams are community metabolism (primary production and community respiration), processing of terrestrial-plant litter, organic matter transport (including invertebrate drift) and storage, and nutrient spiraling.

The riparian zone generally is regarded to encompass the streambank and flood-plain vegetation but also should include any vegetation outside the flood plain likely to enter the stream by gravity (recruitable debris). This broadened operational boundary is especially important in steep-sided valleys and in forests of tall trees, where the fallen materials may directly enter the stream. The riparian environment forms a transition zone between the stream and the adjacent uplands, and together the two habitats constitute a connective corridor that integrates conditions within and among streams of different size. The riparian interface may be a sharp boundary (edge) or a gradual transition (ecotone) between the two, depending on the sharpness of the environmental gradients, and will affect the ease with which the riparian area is delimited. The sharpness of the environmental gradient is itself an amalgam of several factors including climate, topography, landform, and geological control (constraint) (Gregory et al. 1991). Although often visualized as “ribbons of continuity,” riparian ecosystems frequently are encountered as heterogeneous patches and clusters or as isolated islands (Gregory et al. 1991), especially in areas of intensive land use or in more arid regions.

Riparian habitats are especially important as refugia during periods of environmental stress, such as annual drought or rapid shifts in long-term climate patterns, because of the improved climatic conditions they provide to many species along river valleys (Gregory et al. 1991, Minshall 1992). As a result of the favorable conditions and heterogeneous nature of riparian habitats, riparian plant communities exhibit a high degree of structural and compositional diversity (Gregory et al. 1991). In addition to the environmental factors that riparian habitats have in common with streams (e.g., light, temperature, nutrients), other abiotic factors are specific to riparian environments, including soil type and depth, and moisture availability (e.g., proximity to the water and extent, frequency, and duration of flooding) (Minshall et al. 1989). The primary producers in riparian habitats include woody terrestrial plants (especially shrubs), sedges, and grasses instead of algae and flaccid vascular plants common to streams. The vertebrate consumers of riparian habitats are mainly birds and mammals, whereas fish are the vertebrate consumers in streams. Key features used in describing riparian ecosystems include the width of the riparian zone; the age composition, structural composition, cover, and overhang of terrestrial vegetation; and the amount, size, and distribution of woody debris (e.g., Burton et al. 1992, Cowley 1992, Plaits et al. 1987). In addition, bank features, including stability and degree of undercutting, often are used to describe riparian ecosystems.

## Role of the Riparian Environment

Riparian areas perform a number of important functions with respect to streams including physical filtration of water (Cooper et al. 1987); bank stabilization; water storage and recharge of subsurface aquifers; nutrient retention, transformation, and release (Cooper and Gilliam 1987, Green and Kaufman 1989, Lowrance et al. 1984a, 1984b); provision of organic matter to aquatic consumers (Cummins et al. 1989, Minshall 1967); regulation of light and thermal conditions in streams; and provision of corridors for the dispersal of plants and animals (Gregory et al. 1991).

The riparian environment strongly influences the microclimate, physical structure, and food resources of the stream (Gregory et al.1991). This influence is driven largely by the makeup and density of the vegetation, which in turn is strongly influenced by soil, water, temperature, and light conditions. Terrestrial woody debris provides physical habitat, modes streamflow and channel conditions, and retains organic matter of smaller sizes. Woody material, which enters streams from the adjacent banks, serves to retain particulate organic matter and inorganic sediment in the stream. Retention has long been identified as a critical component in stream channels. Vannote and others (1980) considered debris dams, filter feeding by invertebrates, and other retention elements to be important to ecosystem stability.

Physical structures are well-documented retention mechanisms for enhancing energy flow and nutrient cycling. The potential of the riparian zone for retention of dissolved solutes is, however, largely unexplored (Triska et al. 1989a, 1989b, 1990). Large woody riparian debris also control channel morphology, physical characteristics of aquatic habitats, and biological activity of aquatic consumers (Swanson et al. 1982). Streamside vegetation is a major source of energy and nutrients for instream communities. Coarse particulate organic matter, which enters streams from the adjacent land in the form of leaves, twigs, and so on, plays an especially important role in the trophic dynamics of flowing water (Cummins et al. 1989, Minshall 1967, Vannote et al. 1980). The extent to which the surface of a stream is shaded is largely determined by the adjacent riparian vegetation. The availability of light regulates the occurrence and growth of algae and higher aquatic plants. Shading by riparian vegetation moderates the thermal regime of stream communities by providing cooler temperatures, which benefit most aquatic life (Swanson et al. 1982). Removing riparian vegetation can increase water temperatures and alter quantities of dissolved oxygen, and numbers of invertebrates and salmonids.

## Ecological Patterns in Space and Time In Riverine Landscapes

The boundaries of ecosystems have spatial-temporal dimensions (Bourgeron and Jensen 1993, Levin 1992, O'Neill et al. 1986) but this perspective has only recently been applied to stream ecosystems. Yet, the perception of flowing water ecosystems is profoundly influenced by the particular frame of reference used. This section addresses some important principles for the management of streams from a landscape perspective.

### Spatial Dimensions

The air-stream interface has long presented a barrier to the study and delineation of stream ecosystems. Early perspectives were framed at the scale of a square foot or some comparable and rather narrowly defined dimension (fig. 1). Even when the perspective was larger, such as a habitat or reach within the stream, the sampling unit often remained the same size and effectively influenced the resulting perspective. The use of narrowly defined sampling units in stream ecosystem research has led to some serious practical and interpretational problems. For example, even now, it is common to publish results in the context of entire river basin responses even though the original data are from only a few points representing a limited number and variety of stream sizes in a large-order river system. Such extrapolation of data and analyses beyond sampling boundaries requires extreme caution. Perceptions of stream ecosystem boundaries have recently expanded outward from the point and transect orientations in all three spatial dimensions: longitudinally along the stream, laterally from the wetted channel, and vertically into the streambed/water column interface and groundwater realms (Minshall 1988).

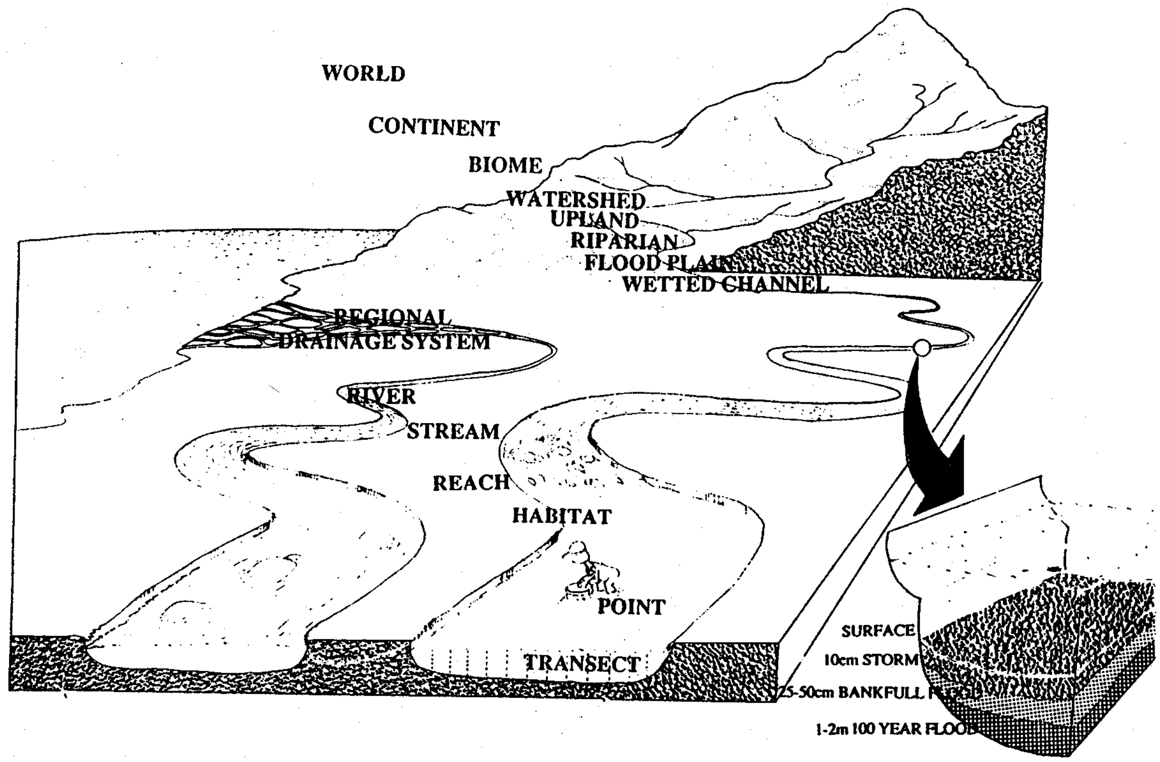


**Figure 1--Spatial boundaries of flowing water ecosystems (from Minshall 1988). Inset illustrates how boundaries below the surface of the streambed may differ with flow, but the depth dimension also will differ from place to place (e.g., pools versus riffles).**

The principal operational paradigm for the study of stream ecosystems in the longitudinal dimension is the river continuum concept (Minshall et al. 1985, Vannote et al. 1980). A basic tenet of this concept is that important ecosystem process-functions change in predictable fashion with stream size (order) but that within a size class, many similarities exist.

Stream habitats frequently are spatially heterogeneous. Stream ecologists have not dealt effectively with heterogeneous distributions at any of the spatial scales, treating them mainly as a bother when applying parametric statistics. Interpatch variability in a stream habitat type, however, should be viewed as-information rather than as statistical noise to be overcome with excessive sample size (Pringle et al. 1988, Resh and Rosenberg 1989). Likewise, some stream ecologists (e.g., Townsend 1989) have had difficulty reconciling the longitudinal conceptual framework with the patchy framework, perhaps because they have viewed the two as a dichotomy rather than an interaction.

Patchiness is a level of complexity to be superimposed on the existing spatial (and temporal) framework (Pringle et al. 1988). Flowing-water ecosystems are patchy at any given site but also are highly spatially oriented, exhibiting a predictable downstream longitudinal succession of communities (Fisher 1990). Patch dynamics is considered to be nonequilibrium by some (Townsend 1989), .at least when viewed from a restricted temporal perspective. From the viewpoint of patch dynamics, streams are best viewed as complex landscapes (pools-riffles, small versus large streams, channel-wetland-upland transition) with flows across their boundaries (Naiman et al. 1988). The regional landscape pattern of patches is best viewed as a major consequence in determining the attributes of specific streams (Karr 1991). In many situations, multiple scales of inquiry may be required because different patterns and insights emerge at different scales of inspection and analysis (Milne 1993).

Many attributes of stream ecosystems are patchy because their structure is nonlinear or nonuniform (Pringle et al. 1988). Patchiness may be found even within a single riffle (as documented by Needham and Usinger (1956) in Prosser Creek, CA), throughout a reach, across an entire catchment, or over a complex region.

Patchiness at the regional level was created by the 1988 fires in Yellowstone National Park (Minshall et al. 1989). That set of fires ranged over five major stream systems, varied in intensity from cool ground fires to roaring crown fires, and left totally burned to partially burned or unburned areas. Streams from first through sixth order in size were affected and the patchwork pattern produced by the fire was superimposed on two distinct climatic zones and two types of geology. Thus, as noted by Pringle et al. (1988), examination of patch dynamics can improve the utility and predictive power of unifying concepts in stream ecology, such as the river continuum concept, through focus on organism and process-specific building blocks of flowing-water systems.

The principal operational model for the study of stream ecosystems in the lateral dimension is the stream-riparian linkage. Sometimes viewed as a distinct boundary and other times as a gradual transition, the size and distinctiveness of the riparian border depend on the steepness of the environmental gradients encountered between open water and uplands. In the vertical dimension, the streambed/water column interface is analogous to the riparian zone as a region of integration-in this instance, between surface and ground waters.

As the conceptual horizons of stream ecologists expanded longitudinally, laterally, and vertically, it became clear that at different scales of resolution different classes of operational factors are important (fig. 2) (see also Milne 1993). Therefore, one must be careful to identify the spatial scale at which a stream ecosystem is studied or managed. From the perspective of landscape ecology, factors such as regional climate, land use patterns, and many of the factors listed in the lower right of figure 2 are especially important.

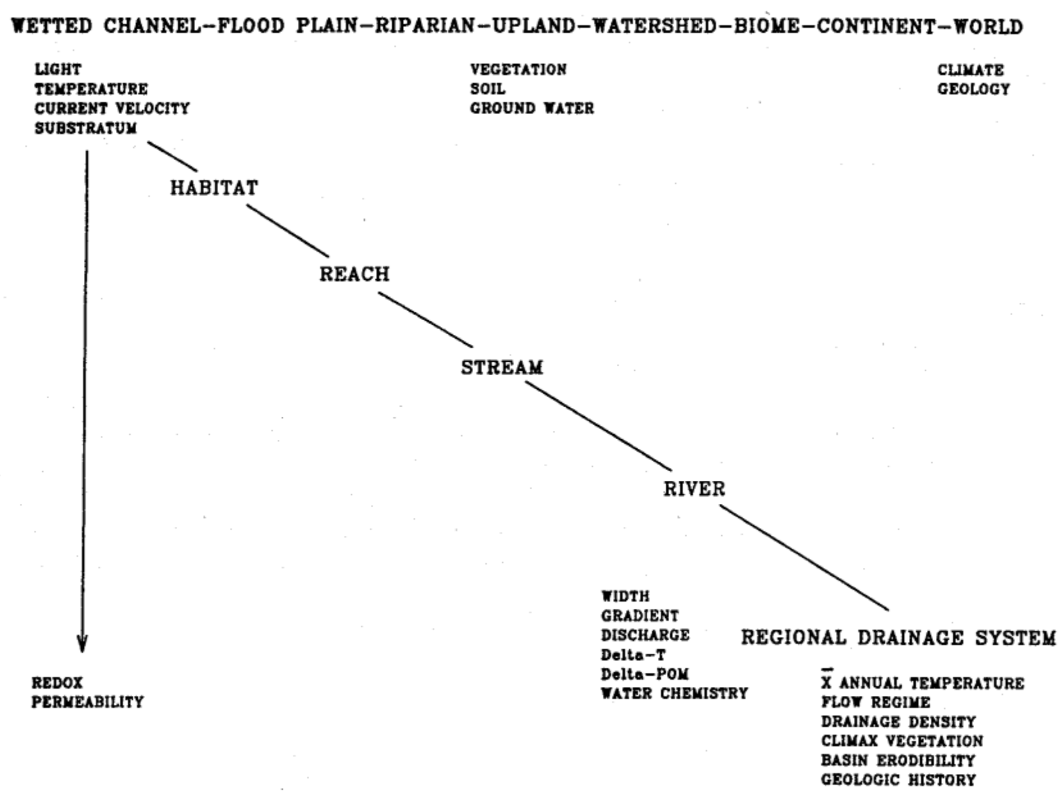


**Figure 2--Illustration of how the principal factors responsible for the structure and behavior of stream ecosystems may differ with scale of resolution (from Minshall 1988).**

### Temporal Patterns

Temporal changes in stream ecosystem structure and function usually occur in response to a disturbance, whether natural or human-caused (Fisher 1990). Major causes of temporal responses in streams of the western United States are logging, grazing, fire, and rain on snow events. These factors after recovery and conditions of stream ecosystems. Ecologists believe succession, or “patterned recovery over time,” following a disturbance (defined broadly as change in average environmental conditions) occurs in flowing-water ecosystems (see recent reviews by Fisher 1990 and Mackay 1992). Because of the restricted number of potential colonists, their biology, and the environmental conditions found in a particular area or region, the process of succession generally is repeatable and predictable. Succession in streams spans a broad range of temporal scales from a few days or weeks to many years (Minshall 1988). Recovery time is directly related to the spatial scale and intensity of the disturbance. Distinct seasonal differences in the ability of stream communities to respond to disturbance also exist (Minshall 1988, Minshall et al. 1985).

Terrestrial vegetation, especially in forests, displays some predictability in recovery patterns. Although actual pathways may differ, a limited array of endpoints usually is involved in forest succession (e.g., Crane and Fischer 1986). Because of the strong interaction between the stream and its valley (Hynes 1975), it is possible to postulate the long-term recovery pattern for stream ecosystems aftera watershed or larger scale disturbance, such as wildfire (Minshall et al. 1988).

The impact of a disturbance and the time required for recovery are functions of size, frequency, and intensity of disturbance events and the resilience of the stream-riparian system (e.g., Fisher 1990). Effects of disturbance are superimposed on and confounded by daily, seasonal, and longer term schedules of abiotic factors that also produce change in streams, although not successional change (Fisher 1990). The effect of a disturbance of a given spatial scale may be heterogeneous. For example, pools and the streambed/water column interface show higher resistance to flooding than do riffles and runs (Fisher 1990). Indeed, Townsend (1989) views patch dynamics of streams largely in the temporal dimension. He argues that the organization of stream communities is largely determined by temporal phenomena induced by disturbance. Temporal heterogeneity or variation contributes to coexistence and diversity (Townsend 1989), an idea implicit in the river continuum concept (Vannote et al. 1980).

### Space and Time

O’Neill and others (1986) point out that ecosystems must be defined simultaneously in terms of space and time and that ecological dynamics occur over a broad spectrum of space-time scales. Current views of stream ecosystems explicitly link space (in the longitudinal direction) and time, through the river continuum and nutrient spiraling concepts, including the aspects of the physical habitat templet and spatial-temporal heterogeneity (Fisher 1990, Minshall 1988, Poff and Ward 1990). Stream ecosystem responses to disturbance occur at scales ranging from millimeters and minutes to hundreds of kilometers and millions of years (fig. 3). Small-scale disturbance events recur with relatively high frequency, whereas larger scale disturbance events occur progressively less often. At each scale, some aspects of the system seem constant and others dynamic. The most important scales generally available to contemporary stream ecologists for analysis fall within the range of individual rocks (or small patches) to wildfires (fig. 3). But, as described later, it is especially at the scales of stream reaches to biogeoclimatic regions, and of years to hundreds of years, that are of particular importance to resource managers.

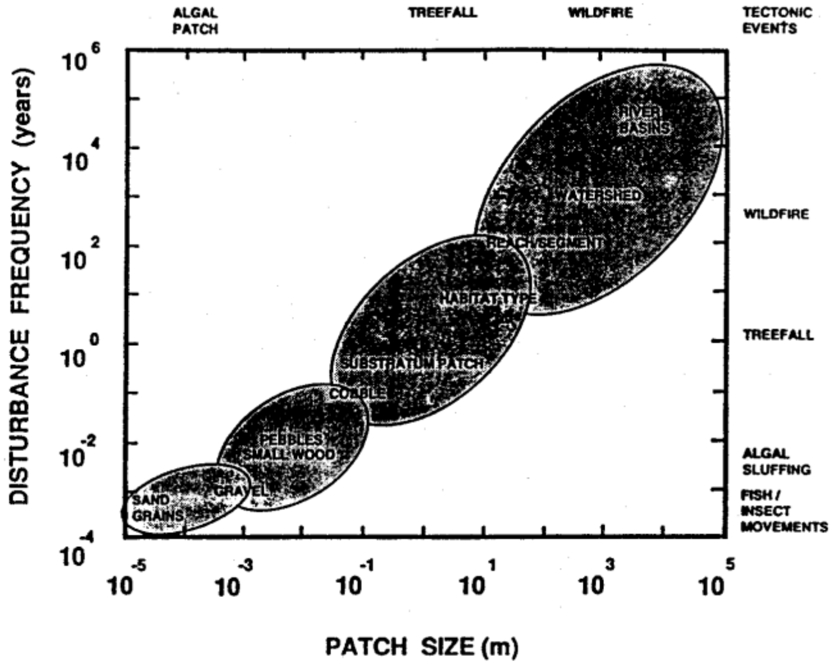


**Figure 3--Time-space continuum for stream ecosystems (based on values given in Minshall 1988, tables 3 and 4).**

Natural geomorphic processes, such as mud slides, debris torrents, and floods, may be viewed as disturbances from an ecosystem perspective (Swanson et al. 1988). Comparable responses may result secondarily from fire, clearcutting, overgrazing, and insect infestations. Geomorphic surfaces create physical patterns in space and time. Fluvial processes in unconstrained channels are frequent and widespread. These processes and surfaces may be organized hierarchically because the force required to modify geomorphic surfaces at different spatial scales is directly linked to the recurrence intervals of floods or other geological events of similar magnitude (Gregory et al. 1991). It is generally accepted that bankfull discharges, with recurrence intervals of 1.5 to 2.2 years, are responsible for the maintenance of channel form and bed structure. Aquatic organisms are assumed to be adapted to this magnitude and frequency of change in the physical habitat (Minshall 1988; Poff and Ward 1989, 1990). Consequently, it is the high-magnitude, low-frequency event that would be most likely to affect changes in stream-riparian ecosystems.

Natural fires also tend to have predictable recurrence intervals and operate at a variety of spatial scales (e.g., Knight and Wallace 1989, Minshall et al. 1989, Romme and Despain 1989). Even within the perimeter of a fire, different catchments and portions within a catchment may burn at different intensities and to different degrees, thus imposing different recovery trajectories on the spatial mosaic (Minshall et al. 1989). The spatial-temporal interactions resulting from the disturbance of stream-riparian ecosystems are complex and difficult to present conceptually. One possibility is shown in figure 4.

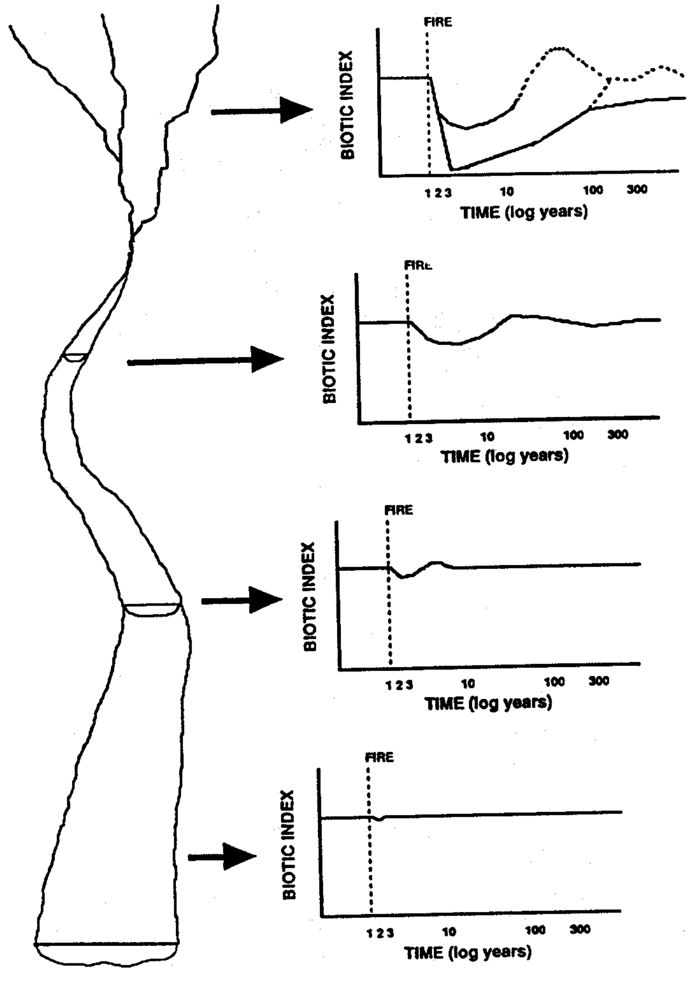


**Figure 4--Postulated temporal response of stream communities, at different locations along a river drainage net, to a disbursed (heterogeneous) landscape-scale disturbance.**

## Disturbance and Hierarchical Scales in Stream-Riparian Ecosystems

Most land and water development and management alters or disrupts stream-riparian ecosystems. When these changes after ecological conditions, they constitute a disturbance (Minshall 1988, White and Pickett 1985), which may have short-term (ecological) and long-term (evolutionary) implications (Poff 1992). A disturbance can be defined quantitatively by its intensity, duration, frequency of recurrence, and predictability; all are characteristics of the disturbance event (Poff 1992, Resh et al. 1988). These characteristics determine the level of organization that responds to a disturbance (O'Neill et al. 1986) and the type and magnitude of the response.

Stream-riparian ecosystems experience a host of disturbances, which have characteristic scales (Fisher 1990). These scales form a hierarchy ranging in size from landscapes (e.g., Yellowstone and Glacier National Parks fires in 1988; clearcutting of checkerboard tracts of corporate forest lands in northern Idaho and western Montana), to whole catchments (e.g., pesticide application to control insect outbreaks, heavy metal and toxic gas releases from smelters), stream reaches (e.g., grazing, road and bridge construction), small patches the size of an animal's footprint (e.g., wading by livestock or fishermen), and individual rocks. A disturbance at one scale (e.g., a rock) may not constitute a disturbance at another scale (e.g., a stream segment). Therefore, it is important that scale be specified for both the characteristics of the disturbance and the ecological response (Poff 1992). In addition, disturbances of stream-riparian ecosystems may differ in type, such as chemical versus physical, toxicant versus nutrient, or pesticide versus acid, which may have different effects.

The history of stream ecology is replete with classifications of waterways (see Hawkes 1975, Naiman et al. 1992), most of which have failed to provide substantial insight into the operation or management of stream ecosystems. The relatively recent application of hierarchical theory to stream habitat classification, as pioneered by Charles Warren (1979), his colleagues (Lotspeich 1980), and students (Frissel et al. 1986), offers promise for inventorying, monitoring, and managing stream resources. Naiman and others (1992) recently reviewed the principles of river classification in respect to conservation. Habitat classification schemes, popular among fisheries workers (e.g., McCain et al. 1990) and useful for identifying local determinants of fish biomass, often are too narrowly focused and subjective for widespread application to broadscale stream-riparian management issues.

A hierarchical framework has been suggested for the investigation of pattern and process in flowing-water ecosystems from the perspective of landscape ecology and patch dynamics (Pringle et al. 1988). This framework complements the unifying theories of stream ecosystems emphasizing longitudinal linkages (Minshall et al. 1985; Newbold et al. 1981, 1982; Vannote et al. 1980) and the watershed view of stream habitats (Frissell et al. 1986). The patch dynamics perspective considers additional characteristics of patches, such as size distribution, density, juxtaposition, diversity, duration, and mechanisms affecting formation (Pringle et al. 1988). A hierarchical classification of riparian ecosystems has been developed for the Great Basin (Minshall et al. 1989). The approach consists of sequential delineation according to hydrologic unit, geomorphic valley form, water regime, physiognomy of the community, community type, and other descriptors. This system could apply equally well to other parts of the West (see also Bailey et al. 1993).

Disturbances of a given frequency often are associated with a particular spatial scale in the natural landscape (O'Neill et al. 1986). Small forest fires occur frequently but in restricted areas. Fires occurring across larger areas have much longer recurrence times. In general, the longer the recurrence interval of a disturbance, the larger the spatial scale and the higher the organizational level of the system that must be considered (O'Neill et al. 1986); however, other variables such as recovery time and intervals between disturbance also need to be considered (Turner et al. 1993). These relations for stream-riparian ecosystems are shown in figure 3; specific examples are given by Minshall (1988), Pringle and others (1988), Fisher (1990), and Mackay (1992).

## Major Alterations Caused By Land Management Activities to Stream-Riparian Ecosystems

Most lard use and subsequent land management activities either directly or indirectly disturb stream-riparian ecosystems. Depending on the spatial-temporal scale on which a stream-riparian ecosystem is viewed (figs. 1 and 3), a disturbance event may appear to be relatively insignificant and mainly background noise, the main "signal" in the formation and maintenance of ecological communities, or a dominating restructuring event involving major geomorphological and evolutionary processes. One would surmise that any event for which the influence extended beyond evolutionary time scales or regional spatial scales (e.g., major geologic or climatic events) would lead to the demise of stream-riparian ecosystems as we know them. Therefore, from the viewpoint of resource management, as well as most ecological investigations, disturbances that are main signals or dominant restructuring events are of greatest interest and usually entail substantial portions of the landscape. Even intrusions that seem to be small-scale when viewed in isolation, may have significantly larger-scale implications when their cumulative effects across space or time are recognized. Examples include low-head hydroelectric development on a fifth-order tributary at 50 or more sites and the effect of poorly regulated grazing across an extended portion of a catchment. In these and comparable cases, a single or a few small-scale effects may be undetectable or relatively restricted but, when taken together, may result in severe degradation of stream-riparian conditions.

Major human-caused disturbances that affect stream-riparian ecosystems in the western United States include livestock grazing, forestry and logging practices, mining, beaver introduction and removal, urban usage such as domestic utilization and sewage discharge, agricultural practices (sediments, nutrients, toxicants, dewatering, etc.), and fish management practices such as poisoning unwanted species, or the introduction of exotic species (Resh et al. 1988). Pesticides applied to forest and agricultural lands often reach water ways. Mining and smelting operations release heavy metals and other poisonous substances via surface, subsurface, and aerial pathways. Other important human-caused influences involve dam building, diking, channelization, and removal of woody debris (for navigation, flow "enhancement," flood control, fish passage, etc.) (Power et al. 1988). Not only may each one of these activities be important to the integrity of stream-riparian ecosystems, but the effects may be cumulative among types (Sidle 1990).

Some large-scale disturbances that affect stream-riparian ecosystems are rapid and dramatic. In the West, examples include massive deforestation, urbanization, development of crop and pasture lands, forest fire, large blowdowns of trees, and plant disease or insect outbreaks. Other disturbances occur gradually or over extended periods of time, or both, and, hence, often are not recognized as problems until the situation becomes extremely difficult or impossible to reverse. These disturbances include some types of logging and mining, livestock grazing, fire suppression, irrigation, and, potentially, global climate change (Firth and Fisher 1992). Rapid or gradual, disturbances of stream-riparian ecosystems may result in changes in water temperature or runoff, channel straightening, scouring and sedimentation, loss of physical habitat, alteration of food base, and waterlogging or drying of riparian soils. Global climate change could profoundly alter stream-riparian ecosystems through is effect on terrestrial vegetation, thermal and hydrologic regimes, nutrient cycles, and so on (Firth and Fisher 1992).

The topics discussed above represent some of the major areas of concern to land managers charged with protecting or improving conditions for stream-riparian ecosystems. Additional factors of direct concern to those managers include protection of threatened and endangered species, maintenance of biodiversity and ecosystem function, and development of productive capacity (such as desired condition). Collectively, these areas of concern also represent the immediate major challenges for the State and Federal agencies entrusted with land-water use. Application of the principles of ecosystem management at the landscape level is essential for finding solutions to these management problems.

## Substitution of Space for Time in Understanding the Long-Term Effects on and Recovery of Stream-Riparian Ecosystems From Land Management Activities

Long-term studies of the recovery of stream ecosystems to disturbances larger than individual rocks or small stretches of streams are rare (Mackay 1992). This lack of information, from the standpoint of resource management, is at least partly due to management agencies being most responsive to crises and lacking the incentive and means to sustain extended research or monitoring programs. Whatever the reason, this lack of knowledge means that rates of recovery from medium- to large-scale disturbances, or even whether such recovery occurs, are largely unknown. Under these circumstances, management responses are hampered and may even be inappropriate or counter-productive.

One way to gain information on recovery processes relatively quickly is to simultaneously examine a series of sites known to have been exposed to the same type of disturbance at different times. In effect, the differences in space are equivalent to differences in time, and inferences may be made as to what is likely to occur at a single site over time. The procedure has a number of shortcomings and pitfalls and is no substitute for carefully designed, long-term research or monitoring, particularly where explanatory mechanisms are being sought (Pickett 1989). It does, however, provide the means for rapidly acquiring insights into the recovery process and possible outcomes, and for partially circumventing current institutional constraints. Care must be taken in these applications to select study sites that have comparable underlying biophysical conditions, as identified by standard ecological mapping units, or physical environments (Bailey et al. 1993, Bourgeron et al. 1993, Frissell et al. 1986).

## Previous Approaches to the Assessment of Habitat and Biotic Conditions in Stream-Riparian Ecosystems

Early attempts to assess conditions in stream-riparian ecosystems consisted of developing lists of organisms and descriptions of structural elements in comparison to conditions found in adjacent, unimpacted areas. This procedure generally focused on selected subcomponents of the aquatic community and water quality factors and was performed in restricted areas of a stream in response to known or suspected cases of water pollution (Mackenthun 1969). Interest in water quality, renewed and broadened in the 1960s and 1970s and was accompanied by several landmark pieces of Federal legislation that, among other things, included the requirement to protect and restore the biological integrity of aquatic environments (Karr 1991). Cairns (1977) considered the concept of biological integrity and its quantification from the standpoint of assessing conditios in stream ecosystems. He recognized the need to treat the concept more broadly as “ecological integrity” to encompass the ecological structure and function characteristic of a locale. Karr and Dudley (1981) argued that “integrity” encompasses all factors affecting the ecosystem, and they defined ‘biological integrity’ as the ability to support and maintain “a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of they region.”

Recent legislation and advances in applied stream ecology have led to efforts to directly assess biotic integrity (Karr et al. 1986, Plafkin et al. 1989). Application of these procedures on a forest-wide, basin-wide or larger scale have been facilitated by efforts to document environmental variability across landscapes (Omernik 1987) and throughout basins (Vannote et al. 1980). Appropriate regional and stream-size adjustments of water quality standards (EPA 1983) and measures thus have been developed. Although the scope has broadened to spatial aspects, the bioassessment of stream-riparian conditions still focuses on structural features of a few subcomponents of the biotic community. Functional and ecosystem-level features, such as decomposition, community metabolism, and nutrient cycling, are rarely if ever examined directly (e.g., MacDonald et al. 1991, Plafkin et al. 1989, Platts et al. 1983, 1987).

Most assessments of habitat and biotic conditions are at the scale of stream segments or smaller units (some notable exceptions are several recent state-initiated surveys, for example, Ohio EPA 1988) and at time scales of a few years or less. Biotic measures have focused primarily on specific biotic subcomponents (fish, macro-invenebrates, algae) of ecosystem structure (richness, abundance, biomass, and various indices) and function (e.g., functional feeding-group composition or a “shredder” index) to the exclusion of process-related measures (e.g., metabolism or nutrient spiraling). Recognition is increasing, however (Cairns 1977, O’Neill et al. 1986, among others) that both structural and functional attributes of ecosystems must be measured because the two operate independently and are necessary to adequately describe an ecosystem state or condition. Habitat measures have been mainly subjective, limited in the scope of factors considered, and oriented mainly toward fish. As with water quality, characterization of habitat conditions alone cannot adequately describe or predict community status or ecosystem condition.

At present, the most widely accepted procedures for assessing biotic conditions in streams at the landscape scale (e.g., county, district, state, region) use indices of biotic integrity for fish and macroinvertebrates and are codified in a set of rapid bioassessment protocols (RBP) (Plafkin et al. 1989). These protocols are designed for the efficient and economical acquisition of information on biological indicators of ecological conditions, and are especially well-suited for preliminary surreys and routine monitoring. The rapid bioassessment protocol approach is hierarchical in that there are several tiers of intensity, with the expectation that additional tiers will be added when a problem is identified or additional detail is required.

The RBPs consist of five separate protocols (I-III for macroinvertebrates and IV-V for fish). RBP I and IV are used as screening or reconnaissance techniques. Benthic RBPs II and III and fish RBP V are progressively more rigorous and intended to provide more objective and reproducible evaluations than I and IV. This procedure is being extensively tested and refined by various State and Federal agencies due to the Federal requirement that biological criteria be incorporated into water quality standards over the next few years.

## Need for a Landscape-Scale Approach to the Assessment of Ecosystem Conditions

Landscape patterns influence many ecological phenomena. Many features of streams and their corridors and many aspects of land use that affect stream-riparian ecosystems operate at regional, river basin, and other levels associated with and affected by these patterns and scales of influence. The occurrence of natural geographic variation in the ecological features of undisturbed aquatic systems must be recognized in any effort to assess ecosystem responses at the landscape scale (Hughes et al. 1986, Karr 1991). Attributes such as slope, aspect, and geology tend to make large sites diverse and heterogeneous, even within a single landform. Superimposed on this may be spatial-temporal patterns of disturbances, both natural and human-induced, resulting in a mosaic of patches of different age, size, and composition (White and Pickett 1985).

Geology and climate are regional factors that influence the characteristics of a river basin or watershed ecosystem (Minshall et al. 1985) and, thus, act at the scale of the landscape (Hughes and Larsen 1988, Omernik 1987).

Variations in geology within a region affect the relative erosiveness of the parent material in a drainage basin and, consequently, watershed topography, chemical load, bed composition, and so on. Climate affects the type and density of terrestrial vegetation, and the effect of climate on the stream is, in turn, intimately bound to the pattern of vegetation. Precipitation and vegetation interact to affect runoff and erosion and, hence, sediment yield and organic-matter loading. Climatic variability affects the biota through its influence on the seasonal patterns of flooding, or in a larger sense, on the hydrology of the watershed (Poff and Ward 1989). These factors influence the evolution of stream biotas and, following watershed modifications by humans, the persistence of species and communities in flowing waters (Karr 1991). Although partially determined by climate and geology, distinctive landforms characteristic of an area, such as mountains or plains, result from additional, largely independent factors (Skinner and Porter 1987) and hence are best considered separately when characterizing landscapes for ecological purposes.

Climate, geology, landform, and potential vegetation are commonly used criteria in ecological map unit design (Bailey et al. 1993). They provide a useful templet for evaluating conditions at various scales (e.g., county, forest, or forest region), inventorying natural resources, establishing reference states against which existing conditions can be evaluated, and formulating and predicting desired conditions.

Landscape is the scale of many forest-wide and region-wide land uses and associated management practices (e.g., logging, mining, livestock grazing) which affect stream-riparian ecosystems. It also is the scale of many larger-scale phenomena (drought, acid rain, forest and range fires, disease and pest outbreaks). Landscape-scale events may affect stream-riparian ecosystems at various lower scales of resolution because of the hierarchical nature of these systems (Frissell et al. 1986, Pringle et al. 1988). Thus, there is no single correct scale for the study of stream-riparian or any other ecosystems (Bourgeron and Jensen 1993, Levin 1992, O'Neill et al. 1986); rather, the appropriate scale depends on the question or management problem being addressed. Also, the particular environmental factors of importance and the interpretation of measurements taken on stream-riparian ecosystems differ with scale (Minshall 1988, O'Neill et al. 1986).

Ecosystem spatial boundaries must be correlated with the temporal framework appropriate for a particular disturbance (O'Neill et al. 1986). At the landscape scale, the life histories of the organisms involved are often long, and disturbances at this scale may affect genetic diversity (Robinson et al. 1992, Sweeney et al. 1992) and are evolutionary in scope (Poff 1992). In this context, genetic diversity is driven by large events and life-cycle turnover times. Infrequent, high magnitude events, for example, are expected to set in motion long-term recovery sequences of 50 to 300 years or permanently alter the possible endpoints for such "recovery" (e.g., Minshall et al. 1989). These points have practical implications for defining reference conditions and for evaluating degree of impact. Characterization of historical variability should include these low-frequency events as part of the background against which the aquatic flora and fauna of a particular location have evolved. The location of a given stream-riparian site on the recovery trajectory must be known, to properly evaluate is usefulness as a reference condition, or to determine its deviation from the norm when subjected to impact. Finally, risk assessments (Jurgensen et al. 1993) should incorporate an understanding of the magnitude and frequency of disturbance processes.

Numerous features of pristine stream-riparian ecosystems change progressively throughout a river basin and require a landscape perspective for proper interpretation (Vannote et al. 1980). For example, the influence of riparian vegetation, the annual amount of terrestrial leaf litter in the channel, the availability of dissolved organic matter, and the modal size of particulate organic matter all decrease with distance from the headwaters of a stream system. The relative contributions of photosynthesis, community metabolism and the composition of functional feeding groups also change gradually and in a predictable fashion along the river continuum. The effects of disturbances also differ along a river system, with the effects of some disturbances (especially if widely dispersed) becoming dissipated with increasing stream size, and other effects acting cumulatively.

The watershed is an appropriate landscape unit for examining stream-riparian ecosystem responses to disturbances on the order of years to decades (fig. 3; see also O'Neill et al. 1986). For events occurring at intervals of 100 to 10,600 years, such as wildfire, the focus becomes the entire forest, which iself may cover numerous catchments. For example, the fires in the greater yellowstone area ecosystem in 1988 encompassed both sides of the Continental Divide and included many tributary watersheds and mainstem riparian portions of the Yellowstone, Clark's Fork, Shoshone, Snake, Madison, and Gallatin River Basins (Minshall et al. 1989).

## Recommended Approach and Procedures

Development of procedures for the delineation, sampling, and analysis of stream-riparian ecosystems, particularly at the basin or landscape scale, is still in its infancy and is likely to see considerable evolution within the next decade. But there are compelling reasons (ranging from regional threats to water quality and biodiversity to major natural "catastrophes" such as Mount St. Helens and the recent fires in Yellowstone and Glacier National Parks) to inititate characterization programs immediately. The broad outlines of such programs exist now and future refinements will serve mainly to provide fuller understanding, better precision, or greater predictive power to the process.

The recommended approach is hierarchical in the classification of stream-riparian ecosystems (and in the selection of their appropriate reference conditions), and in the selection of factors to be measured and the amount of information obtained. A hierarchical approach lends itself to the study of ecosystems, because it resolves the dilemma between simplifying the apparent complexity of and accounting for observable behaviors (O'Neill et al. 1986). Finally, it provides means for stratified sampling in the development of ecologically sound and cost-effective environmental assessment and monitoring programs (Bourgeron and Jensen 1993).

### Classification of Stream-Riparian Habitats

Frissell and others (1986) present a spatially nested hierarchical framework for classifying stream systems that lends iself to resource management at the landscape level. Their approach is widely applicable because it includes the idea that different variables are important at the different time and space scales at which a system is viewed. Frissell and others set the stream system, at the scale of entire watersheds, as the upper element in their classification scheme. At successively lower scales, their stream "systems" consist of stream segment, reach, pool-riffle, and microhabitat subsystems. For application to multiple watersheds distributed across complex, heterogeneous landscapes (such as those often encountered by resource managers at the scales of individual forests and water districts in the western U.S.), their scheme has been extended upward one scale to include biogeoclimatic regions (table 1). Operationally, this is accomplished by distinguishing between "regional" versus "local" aspects of climate, geology, and terrestrial vegetation. Proper classification at the regional scale requires long-term records of atmospheric temperature, precipitation, and stream discharge and knowledge of presettlement, terrestrial vegetation distributions (Bailey et al. 1993, Swanson et al. 1993). Environmental data may be available from weather and stream-gauging stations in a region (e.g., Finklin 1988, Mosko et al. 1990) but frequently such data will be lacking. The development or reestablishment of a representative network of instrumented recording stations is required. In the past, shortsighted, cost-cutting practices have resulted in the elimination of many such stations. Terrestrial vegetation records may be obtained from sources such as Franklin and Dyrness (1973), Hall (1973), and Steele et al. (1981). The revised system for stream habitat classification also has been modified to include the aspects of flow and thermal regimes and substratum heterogeneity, whose importance to stream ecosystems is widely recognized (e.g., Poff and Ward 1990).

Initial stratification of sites by biogeoclimatic regions can be done with the ecoregional scheme of Omernik (1987) or Bailey and others (1993) (also see Gallant et al.1989). Including flow regime, by using the procedure of Poff and Ward (1989), further refines the biogeoclimatic aspects and makes the characterization more directly related to flow as a major environmental driver of stream-riparian ecosystems. As used here, a regime is regarded as incorporating the characteristics of intensity, frequency, duration, and predictability; predictability, in-turn, consists of the attributes of constancy (temporal uniformity) and contingency (temporal variability, but in an ordered sequence) (Poff 1992, Poff and Ward 1989, Resh et al. 1988, Vannote and Sweeney 1980). In addition, other aspects of variability (e.g., heterogeneity), pattern (e.g., mean interval between events), and intermittency have been used (Poff and Ward 1989, Vannote and Sweeney 1980).

The scale of stream systems (table 1) reflects the more local conditions of climate, geology, topography, and plant cover contained within individual watersheds. For the Pacific Northwest, the map of Omernik and Gallant (1986) is helpful, but finer resolution eventually may be needed. Because of the smaller spatial scale, shorter term climatic and discharge records may be adequate or possibly can be established by correlation of recent measurements for the watershed with longer term records elsewhere in the region. Incorporation of thermal regime (Vannote and Sweeney 1980) at this scale allows further separation of streams that may have similar

**Table 1--Hierarchical Classification of Stream Riparian Habitats (after Frissell et al. 1986)**

| | | ------------ Boundaries ------------ | | | |
|---|---|---|---|---|---|
| Stream habitat (linear spatial scale) | Defining measures | Longitudinal | Lateral | Application | Source of information |
| Biogeoclimatic region ($10^5$ m) | Regional climate<br>Regional geology<br>Regional topography<br>Regional terrestrial vegetation<br>Flow regime | | | Region; State; Forest; District | Topographic maps (15')<br>Geologic maps (15')<br>Landsat photos<br>Annual discharge records |
| Stream system ($10^3$ - $10^4$ m) | Local climate<br>Local geology<br>Local topography<br>Local terrestrial vegetation<br>Thermal regime | Drainage divides, and seacoast, or catchment area | Drainage divides bedrock faults, joints controlling ridge valley development | Basin-wide surveys; Cumulative impacts; Integration of sites within watersheds | Topographic maps (7.5')<br>Geologic maps<br>Vegetation maps<br>Aerial photos<br>Annual temperature recorc |
| Segment system ($10^2$ - $10^3$m) | Tributary junctions<br>Major geologic discontinuities | Tributary junctions major falls; bedrock lithologic or strucutural discontinuities | Valley sideslopes or bedrock out-crops controlling lateral migration | Paired watersheds<br>Segment classes (e.g. uplands vs lowlands) | Topographic maps (7.5')<br>Ground reconnaisance<br>Low level aerial photos |
| Reach system ($10^1$ - $10^2$m) | Channel slope<br>Valley form<br>Bed material<br>Riparian vegetation | Slope breaks: structures capable of withstanding < 50-year flood | Local sideslopes or erosion-resistant banks; 50-year floodplain margins | Local effects; grazing allotments; dredging | Ground survey/mapping |
| Pool/riffle system ($10^0$ - $10^1$m) | Bed form and material<br>Origin<br>Persistence<br>Mean depth and velocity | Water surface and bed profile slope breaks; location of genetic structures | Mean annual flood channel; midchannel bars; other flow-splitting obstructions | Aquatic habitat inventories; fisheries censuses | Ground survey/mapping |
| Microhabitat system ($10^1$ - $10^0$m) | Surface particle size;underlying particle size; water depth; velocity; overhead cover (type) | Zones differing substrate type; size arrangement | Same as longitudinal | Characteriz-ation of local spatial heterogeneity and effects (e.g., wading by fishermen) | Direct measurement |

external and biogeoclimatic controls but differ in their thermal environments because of different combinations of ground and surface water or different orientations to the sun.

Stream segment systems are designated on the basis of stream orders (Strahler 1957) or links (Shreve 1966) and major geologic discontinuities. Incorporation of thermal regime, as recommended by Vannote and others (1980) and Poff and Ward (1990), permits stratification by catchment-scale differences due to aspect and water source. Although Frissell and others (1986) provide criteria for distinguishing reach system classes, the procedure developed by Rosgen (1985) (and since modified in a 1989 table available from him) may prove more widely applicable. Substratum characteristics are expected to be important at this scale (Frissell et al. 1986), including particle size heterogeneity (Minshall 1984, Poff and Ward 1990) and woody debris accumulations. Also at the scale of stream reach, the use of valley form (Minshall et al. 1989, Rosgen 1985) in place of side-slope gradient is suggested as better for characterizing features likely to be important to riparian and stream dynamics.

General survey procedures and detailed analytical measurements are available for determining habitat characteristics at the stream reach scale, depending on the particular need. Designations of stream habitats at the scale of pool-riffle systems have been refined, since Frissell and others (1986) published their ideas, to include more than 20 types (McCain et al. 1990). Although developed strictly from the perspective of fish distribution and habitat preference, the categories also should prove adequate for describing conditions for other aquatic resource values at this scale. Microhabitat systems are appropriate for addressing fine-grained heterogeneity, such as the distribution of benthic macroinvertebrates (Minshall 1984) and efficiency of nutrient cycling within microbial communities (Pringle et al. 1988). This scale most likely will provide too fine of a resolution for most management issues.

Replication of reference sites across a region is recommended to incorporate natural geographic variability into the assessment approach (Hughes et al. 1986, Karr 1991). As noted above, natural variability also may be expected along the length of a river basin, which will require stratification of reference sites by stream size. Use of reference-condition sites provides a control during the study or monitoring period and also gives an index of recovery or desired condition (in the sense used by the USDA Forest Service). Paired comparisons with the reference site will indicate changes in the managed site across time, relative to natural changes from climate or infrequent natural events, such as a 100-year flood, encountered in both the managed and reference sites. The reference site also establishes a baseline against which the affected site can be evaluated to quantify its present condition. The reference site provides the data to separate the effect of treatment from the variability shared by both systems. Use of replicated reference sites are recommended to provide stronger statistical evidence of cause and effect (Burton et al. 1991).

**Selection of Factors to Measure**

The evaluation should involve measures of stream-riparian ecosystem structure and function and include measurements of principal physical (see Platts et al. 1983, 1987 for procedures) and chemical qualities (see APHA 1989, Stednik 1991 for methods). In addition to the routinely employed analyses of structural organization, a complete assessment of stream ecosystem health ultimately must include measures of functional organization and ecosystem behavior (i.e., process rates and system function). Where possible, seasonal differences should be documented and annual or greater temporal sampling should be synchronized by using an energy-input measure such as accumulated degree-days for the year. Established (standard) procedures should be used, where possible, to permit rapid sampling and to ensure comparability among studies and technical personnel. The procedures should be sufficiently robust to be applicable over a wide range of situations.

In general, it is important that the procedures be cost effective and that the results be easily interpretable and available in a timely manner. Otherwise, it is unlikely that the approach will be adopted by resource managers. This goal must neither be viewed strictly in the short-term nor used as an excuse for incomplete or inadequate assessment, or for the failure to develop additional procedures. At the same time, practical constraints dictate that key indicators of stream-riparian ecosystem health be employed whenever possible, rather than comprehensive measurement of the myriad of internal “details” or mechanistic explanations of cause and effect (i.e., a holistic rather than a reductionist approach is recommended).

The use of multiple measures is recommended because it is unlikely that any one measure will have sufficient sensitivity to be useful in all circumstances (Karr 1991). For the same reason, the values for each measure should be kept separate (to maintain its information value), as opposed to summing the values to produce a single index value. It is recognized, however, that simplified single ecosystem ‘scores’ may be required at times for communication with nontechnical users.

A nested series of procedures arranged hierarchically is recommended to progressively increase the information available for management decisions and permit adjustments for specific types of problems. A hierarchical arrangement ensures that a basic set of comparable measurements will be made in all instances and also permits further tailoring of the program for specific needs and available resources. The scale of the question or application (see table 1), type of problem (e.g., nutrients vs toxic metals), use to which the information will be put (e.g., a local management question versus a full-scale legal battle), and other factors, will determine the particular scale of analysis needed. Because the more descriptive aspects of physical habitat structure are incorporated in the selection and delineation of an analysis scale (table 1), their measurement is presumed and is addressed only secondarily.

The recommended measurement approach for stream-riparian ecosystems is given in table 2. The approach is applicable to all habitat scales (table 1) by appropriate adjustments in measurement location and frequency. The procedures are organized into four stages. Each subsequent stage is assumed to incorporate the measures of the previous stage, unless otherwise noted. Resh and others (1988) identify nine structural and functional components of the stream biota for examining the effects of disturbance: standing crop biomass, transport-drift, primary production, secondary production, taxonomic richness, trophic-functional diversity, pattern of life history tactics, size spectra, and biotic interactions. Of these, the first six particularly lend themselves to routine application to resource management questions. Two of the components (biomass and trophic diversity) are measured in stage 2 of the recommended approach and the other four components are measured in stage 3.

**Table 2--Hierarchical Sequencing of Stream Environmental Conditions Suitable For Application at the Stream Segment Scale and Lower (Excluding Habitat Features Addressed In Table 1)[1]**

| Stage 1 | Measurement per Feature | Purpose |
|---|---|---|
| Environmental factors: | | |
| Temperature | 24-h maximum and minimum during warmest month of year | Estimate of annual maximum and diel change ($\Delta T$) |
| Discharge | Summer baseflow | Characterization of stream size; permit calculation of fluxes |
| Substratum | Mean and coefficient of variation of x axis for ≥100 randomly selected particles | Mean particle size distribution and heterogeneity |
| Alkalinity<br>Hardness<br>pH<br>Specific conductance<br>Turbidity | Grab samples analyzed using standard methods | General water quality |
| Biotic Factors: | | |
| Macroinvertebrates | Rapid bioassessment protocol III | Biotic condition indicators<br>Community structure indices |
| Fish community structure | Rapid bioassessment protocol V | Biotic condition indicator<br>Community structure indices |

| Stage 2 | Measurement per Feature | Purpose |
|---|---|---|
| **Environmental factors:** | | |
| Solar radiation | Percent incoming PAR reaching stream surface at 9, 12, 3, and 6 on a clear day in summer | Relative density and shading by vegetative and topographic features |
| Temperature | Seasonal 30-d thermograph records | Improved characterization of thermal regime and heat budgets |
| Discharge | Seasonal instantaneous (5 random times each) measurements or 30-d stage height records | Improved characterization of flow regime |
| Substratum | Embeddedness and stability | Estimate of suitability of streambed for fish (egg) and invertebrate survival |
| Calcium<br>Magnesium<br>Nitrate-nitrogen<br>Phosphorus (ortho)<br>Sulfate | Filtered sample<br>Colormetric field procedure | Delineation of main cations<br>Principal plant nutrients<br>Further delineation of main anions |
| **Biotic factors:** | | |
| Algae | Periphyton chlorophyll and biomass | Quantification of an important food source and biotic indicator |
| Benthic organic matter | Total | Quantification of an important food source |
| Invertebrates | Total density<br>Total biomass | Improved indicators<br>Estimates of 2° consumer production |
| | Analysis by functional feeding groups | Definition of trophic organization (in combination with similar data for fish) |

| Stage 3 | Measurement per Feature | Purpose |
|---|---|---|
| **Environmental Factors:** | | |
| Solar radiation | Stream surface, std., depth and bottom PAR seasonaly on clear days | Estimate of solar input |
| Temperature | Annual thermograph records | Improved information content |
| Discharge | Annual hydrograph records | Improved information content |
| Current velocity & depth | Measured at random locations throughout study area. Determine mean and current velocity | Characterization of stream habitat suitability; determination of hydraulic stress |
| Ammonia-nitrogen | Laboratory analysis of filtered samples | Further detail regarding nitrogen dynamics |
| Nutrient flux (N,P) | Concentration X discharge (with concentration determinations upgraded to laboratory quality) | Measure of resource availability (Fisher 1990) |

| Stage 3 | Measurement per Feature | Purpose |
|---|---|---|
| Biotic Factors: Algae | Diatom community metrics | Biotic condition indicator |
| Benthic organic matter | Partitioned into coarse and fine sizes and main sources | Refined food resource analysis |
| Transported organic matter/invertebrate drift | Same as for benthic organic matter | Estimate of exported organic matter and food available for filter feeders and fish |
| Leaf packs | Processing rates | Estimate of decomposition by microbial and invertebrate "detritivores" |
| 1° Production/community respiration | Activity rates of colonized trays of native substrata measured in recirculating chambers | Index of community P/R rates |
| Nutrient uptake | Response to standard nutrient additions (e.g. "pot" releases). Uptake rates in recirc. chambers | Plant nutrient-growth status |
| **Stage 4** | **Measurement per Feature** | **Purpose** |
| Environmental Factors: Solar radiation | Annual solar radiation | Determine solar radiation regime and energy input |
| Biotic Factors: Ecosystem production/ respiration | Total-system metabolism using diel up/down or sum of individual component/compartment values | Measure of ecosystem behavior, productivity, and trophic state |
| Nutrient spiraling | Turnover length and time. Index of retentiveness | Measure of ecosystem behavior and utilization/ retention efficiencies |
| Secondary production | Monthly measurements of invertebrate standing crops | Measure of impacts on fish-food producing capability of streams |

[1]Arranged in order of increasing detail, with each subsequent stage intended to be cumulative unless noted otherwise.

Stream-riparian ecosystem assessments that use the Rapid Bioassessment Procedures of the Environmental Protection Agency (Plafkin et al. 1989) for both habitat and biotic (macroinvertebrates, fish) components (see also MacDonald et al. 1991) currently emphasize the Stage 1 procedures. The combination of RBP III and V procedures are recommended for routine use in the ecosystem assessments addressed in this document. The RBP III protocol should be modified to involve the analysis of 300 or more specimens and use of 250 um-mesh Surber net or comparable quantitative sampling device as standard procedures. Implicit in this stage is a basic evaluation of physical habitat (table 1: Plafkin et al. 1989 in combination with Petersen 1992) and diagnostic water quality conditions.

Stage 2 provides a more complete assessment of environmental conditions and an analysis of the food resources available to the heterotrophs. Thermograph records are helpful for identifying and quantifying important aspects of the thermal regime. These records are equally important for quantifying thermal budgets (e.g., cumulative degree-days), which are used to explain aquatic invertebrate and litter-processing responses (see also stage 3) (Cummins et al. 1989). The amount of sift and sand and the relative stability generally are regarded as critical measures of substratum conditions. A number of methods have been proposed previously, but all have their conceptual and practical limitations. Assessments of embeddedness and of riffle armor stability are still used widely and the information they provide generally is regarded as useful. The benthic invertebrate analysis is expanded to include total density (abundance per unit area), biomass (which requires accounting for all organisms in a sample), and partitioning of the results by functional feeding group. At this stage, habitat features should be quantified by using procedures such as those described by MacDonald and others (1991) and Platts and others (1983, 1987). A standard quantified protocol for habitat analysis comparable to the subjective protocols presented by Plafkin and others (1989) and Petersen (1992) is yet to be developed.

Stages 3 and 4 supply additional environmental details and address some of the most important aspects of stream ecosystem function: decomposition rates, energy metabolism, and nutrient cycling. These two stages differ primarily in the detail involved. Stages 3 and 4 are more difficult to implement than the preceding stages but are required for a thorough assessment of ecosystem dynamics. At the level of stage 4, consideration should be given to the development of vegetation analysis focused on litter input (see Cummins et al. 1989) as a means of quantifying important riparian influences not addressed in the earlier stages. Such an analysis takes into consideration the types of vegetation in terms of processing rates (e.g., “fast” versus “slow”), the timing and rate of delivery to the stream, and the size of the tallest plants relative to the width of the channel.

## CONCLUSION

Stream-riparian ecosystems are intimately linked to the conditions of their catchments; consequently, they are directly responsive to land use and management practices. Stream-riparian environments and biota also are inseparably linked in an interacting and intergrading fashion along a river system. This observation requires that watersheds be viewed as single riverine ecosystems comprised of individual habitats, reaches, and segments (e.g., tributaries) that are integrally bound together, in terms of structure, function, and responsiveness to change, along a river continuum. Failure to recognize these interdependencies, has led to serious effects on these ecosystems since the European settlement. These effects have resulted in the general degradation of stream-riparian ecosystems at the scale of individual land management actions (e.g., a timber sale or a grazing allotment) to entire river basins. Such effects have routinely gone unnoticed or been ignored by the land managing agencies.

Many landscape disturbances, both natural and human-caused, are spatially heterogeneous and set in motion a sequence of recovery events that profoundly influence the nature of stream ecosystems. Ecologists and resource managers have tended to view stream ecosystems as spatially homogeneous and temporally static or fixed in time. This perception is an artifact of the strong influence that parametric statistics and equilibrium concepts have exerted on the field of ecology in general. A more realistic and productive approach from a landscape perspective is to view riverine ecosystems as spatial-temporally heterogeneous. If correct, this view suggests the need to incorporate greater spatial and temporal diversity into so-called basin-wide studies of stream ecosystems. The objectives and methods of landscape ecology provide the means for accomplishing this.

The river continuum and landscape perspectives of stream-riparian ecosystems suggest the need for hierarchically based classification and assessment systems that will provide resource managers with adequate information at appropriate temporal and spatial scales to evaluate (in the present) and avoid (in the future) detrimental effects resulting from piece-meal and short-sighted management decisions. The information obtained from the recommended classification and assessment schemes is readily incorporated into geographic information systems and can greatly aid resource managers in obtaining a holistic view of stream-riparian conditions within a district, forest, or region. The resulting expanded perspective will allow more accurate determination of the opportunities and constraints associated with individual and cumulative events and actions, whether natural or human-caused.

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# Risk Assessment Methodologies in the Interior West

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## ABSTRACT

Risk assessment has been and will continue to be an important component of land management planning in the western United States. Broader applications of ecosystem management in this region, however, will require a shift from more local, economic, stand-oriented assessments to ones that encompass landscape scale consideration. Traditional, empirically based risk assessments are rapidly being replaced by more complex holistic and probabilistic approaches that incorporate an understanding of ecosystem processes into risk model development. This paper presents an overview of risk assessment methodologies for four important ecosystem risk factors in western conifer forests: insects and diseases, fire, soil erosion, and wildlife populations. The advantages and disadvantages of current risk assessment techniques are reviewed, and suggestions concerning possible improvements to such methods are provided.

## INTRODUCTION

The Yellowstone Fire of 1988 and the continuing controversy on the current viability and long-term survival of the northern spotted owl in old-growth forests of the Pacific Northwest have highlighted the importance and the problems of using risk assessment in ecosystem management. Risk rating systems to assess the susceptibility of forest stands to insect and disease attack or fire have been an integral part of forest management for decades. Many of these traditional risk assessment strategies have stressed possible timber loss or other resource allocation changes (e.g., Reed and Errico 1987, Teeter and Dyer 1986); however, with the current emphasis on landscape or ecosystem management, risk assessment has taken on a much broader orientation (Byler and Zimmer-Grove 1991).

Understanding the capabilities and goals of forest management is essential in developing and using risk assessment technology. As an indicator of stand or ecosystem response to management activities, risk assessment has an important role in adaptive management strategies and procedures (Everett et al. 1993, Rolling 1978). A risk rating system will likely go unused, unless the risk assessment system is compatible with the management regime (Redden 1981).

Risk assessment can be defined as estimating the scientific and management uncertainties associated with forest management (Marcot 1992). The information used in risk assessment has various degrees of reliability and applicability. Scientific information obtained from field studies is statistically based, applicable to specific situations, and highly credible. When such information is not available, however, ecological principles and models are often used to extrapolate information to new locations or situations (Marcot 1992). Less reliable, but extremely important in forest management is professional judgment, which relies on personal knowledge of local forest ecosystems. Professional judgment may be useful for many purposes, but it also can be subjective and biased, depending on the training and experience of the observer.

Monitoring of biotic and abiotic attributes of stands or ecosystems to determine if management actions recommended by risk assessment produce expected responses is an essential part of risk management. Such information is used to validate current assessment models and serves as the basis for developing new models. In this paper, we present an overview of risk assessment methodologies for four important ecosystem risk factors in conifer forests of the western United States: insects and diseases, fire, soil erosion, and wildlife populations. Emphasis is placed on describing the advantages and disadvantages of risk assessments currently being used or developed, and on the possible improvements needed to reflect changes in forest management practices. The scientific names of all insects, pathogens, and trees referred to in this paper are listed in table 1.

**Table 1--List of Common and Scientific Names**

| Common Name | Scientific Name |
|---|---|
| Annosum root disease | Heterobasidion annosum (Fr.) Bref. |
| Armillaria root disease | Armillaria ostoyae (Romag.) Herink |
| Asian gypsy moth | Lymantria dispar Linnaeus |
| Balsam woolly adelgid | Adelges piceae Ratzeburg |
| Douglas-fir | Pseudotsuga menziesii (Mirb.) Franco |
| Douglas-fir tussock moth | Orgyia pseudotsugata (McDunnough) |
| Douglas-fir beetle | Dendroctonus pseudotsugae Hopkins |
| Douglas-fir dwarf mistletoe | Arceuthobium douglasii Engelm. |
| Fir engraver | Scolytus ventralis LeConte |
| Grand fir | Abies grandis (Dougl. ex D. Don) Lindl. |
| Laminated root rot | Phellinus weirii (Murr.) Gilb. |
| Larch casebearer | Coleophora laricella Hübner |
| Lodgepole pine dwarf mistletoe | Arceuthobium americanum Nutt. ex Engelm. |
| Lodgepole pine | Pinus contorta var. latifolia Dougl. ex Loud |
| Mountain hemlock | Tsuga mertensiana (Bong.) Carr. |
| Mountain pine beetle | Dendroctonus ponderosae Hopkins |
| Ponderosa pine | Pinus ponderosa Dougl. ex Laws |
| Western spruce budworm | Choristoneura occidentalis Freeman |
| Western larch | Larix occidentalis Nutt. |
| Western larch dwarf mistletoe | Arceuthobium laricis (Piper) St. John |
| Western dwarf mistletoe | Arceuthobium campylopodum Engelm. |
| Western pine beetle | Dendroctonus brevicomis LeConte |
| White fir | Abies concolor (Gord. & Glend.) Lindl. ex Hildebr. |
| White pine blister rust | Cronartium ribicola Fischer ex Rabh. |

## Insects and Diseases

Most insects and pathogens affecting forested ecosystems in the interior West are native to western North America. The most notable exceptions are the white pine blister rust, Asian gypsy moth, larch casebearer, balsam woolly adelgid, and several species of lesser importance. Native tree, insect, and pathogen populations have coevolved a mutual tolerance. Nowhere in the western United States is the long-term viability of a coniferous species threatened by a native pathogen or insect. These organisms are an integral part of these forest ecosystems. The extent of their influence on vegetation is a function of their biotic and abiotic environments, the disturbance history of ecosystems, and the abundance, arrangement, vigor, and degree of evolved tolerance of their hosts. In contrast, newly introduced pathogens and insects often rapidly overcome hosts that are similar to those in their natural range because there is no coevolved tolerance.

Entomologists and pathologists have developed, over the last few decades, various hazard and risk rating systems in an attempt to evaluate and possibly control insect outbreaks and pathogen epidemics. The intention of such work has been to develop accurate predictive models that would enable resource managers to intervene and prevent undesirable outcomes. Forest entomologists in the western United States make a distinction between hazard rating and risk rating systems. Hazard rating systems classify trees and stands according to their susceptibility to insect infestation (Molt 1963) using tree, stand, site, climate, and other environmental or edaphic characteristics (Waters 1985). Hazard rating attempts to determine how conducive vegetation is to infestation by a particular insect. Risk rating systems evaluate stand vulnerability to damage or risk of injury. Risk rating systems integrate vegetation susceptibility (hazard), with an evaluation of injury that will likely occur when a damaging insect population is present in a stand, or in nearby stands.

**Previous and Current Techniques.**

Hazard assessment techniques vary in their scope and sophistication. Most hazard rating systems for western forest insects, principally conifer defoliators and bark beetles, were designed to evaluate the susceptibility of trees or stands to infestation by a single insect species. These methods have enabled resource managers to prioritize with some success, susceptible stands for control treatments. Hazard rating systems, however, have not been developed for most major western forest diseases (dwarf mistletoes and root diseases), although rating systems have been used in localized situations (Byler et al. 1990, McDonald et al. 1987, McDonald 1990, Williams and Marsden 1982).

At one end of the spectrum, climatological variables have been used to predict outbreak frequencies of western spruce budworm (Kemp 1983, 1985) over large areas of the western United States. Heller and Kessler (1985) used aerial photo-interpreted data to rate budworm-susceptible stands in localized areas. Stoszek and Mika (1984, 1985) studied the Douglas-fir tussock moth and western spruce budworm in Idaho and developed multiple regression equations to predict defoliation hazard from categorical data. In contrast, Wulf and Carlson (1985), and Carlson and others (1985) devised a stand susceptibility rating system for the western spruce budworm in the Northern Rocky Mountains using indices of stand composition, density, height-class structure, vigor, maturity, site climate, regional climate, and continuity of surrounding host types. The concepts developed in this model may be applicable to other geographic areas of the interior West.

Numerous hazard rating systems have been devised for assessing the susceptibility of lodgepole pine stands to the mountain pine beetle (Amman and Anhold 1989, Safranyik et al. 1974, 1975, Shore et al. 1989, Stuart 1984). Crookston and others (1977) provided a simple hazard rating scheme by mapping areas of historical beetle infestation. Amman and others (1977) developed a stand hazard rating system using elevation-latitude, mean stand diameter at breast height (DBH), and mean stand age. Mahoney (1978) used periodic growth ratio (PGR) to rate stand susceptibility to beetle infestation. A stand hazard rating (SHR) was calculated by Schenk and others (1980) using a crown competition factor (CCF) and the proportion of stand basal area in lodgepole pine. Berryman (1978) developed a method of stand hazard rating that used stand resistance to beetle attack, (the ratio of PGR/SHR), with the percentage of the lodgepole pine basal area with phloem thicker than 0. 1 inch (a measure of a stands ability to support beetle populations through successful brood development). Stuart (1984) found that quadratic mean diameter and the number of growth rings in the outer 1 cm of the bole, best predicted outbreaks of mountain pine beetle in lodgepole pine stands of south-central Oregon. Similarly, Mitchell (1987) expressed hazard to lodgepole pine stands in central Oregon as a function of the number of trees per acre larger than 9 inches DBH, and a stand vigor rating (Mitchell et al. 1983a, 1983b). Waring and Pitman (1980) ranked stand susceptibility to mountain pine beetle using an index of stand growth efficiency.

Other hazard rating systems for mountain pine beetle have been developed that were based on climate (Safranyik 1978); tree age, diameter, and climatic zone (Safranyik et al. 1975, Shrimpton 1973); habitat types, tree diameter, and elevation (McGregor 1978, Roe and Amman 1970); phloem thickness and tree diameter (Cole 1978, Cole and Cahill 1976); phloem thickness, tree vigor, and beetle population dynamics (Berryman 1982); and site climate, tree diameter, tree age, phloem thickness, crown competition factor, and periodic growth ratios (Cole and Amman 1980). Shore and others (1989) and Stuart (1984) evaluated a number of these hazard rating systems and found that none of them satisfactorily predicted beetle infestation. As Paine and others (1984) concluded, the risk of beetle damage is a function of stand susceptibility and the proximity of host stands to damaging populations of beetles.

Much less has been done on hazard assessment of mountain pine beetle on ponderosa pine. Miller and Keen (1960) provided age and crown vigor characteristics of individual ponderosa pine trees susceptible to western pine beetle attack, but methods for identifying susceptible stands were not developed.

**Hazard Variables Used.**

Many different tree, site, and stand characteristics of interior West forests can influence the probability of insect outbreaks or pathogen epidemics; the degree of hazard, and hazard variables are strongly influenced by geography. Factors that influence the magnitude of insect or disease effects, such as long- or short-term climate flux, or other important biotic and abiotic factors that internally and externally regulate pathogen and insect populations will not be discussed. These factors are not well understood, are highly variable by locality, cannot be presently mapped, or cannot be integrated with other factors. The following discussion summarizes some of the better understood variables influencing insect and disease hazard assessment.

Defoliators--The most influential conifer defoliators of the interior West are the western spruce budworm, and the Douglas-fir tussock moth. Defoliation affects both stands and landscapes by increasing tree mortality, tree growth loss, topkilling, reduced tree cover or cover quality, altering species composition and stand structure, increasing fire hazard, and increasing susceptibility to damage by other agents (bark beetles, root diseases, dwarf mistletoes, and drought). There has been no apparent change in western spruce budworm outbreak frequency in response to forest management practices, but outbreak extent, duration, and severity have increased (Anderson et al. 1987, Carlson et al. 1983, 1985; Fellin et al. 1984), and these factors provide the basis for budworm hazard assessment. Budworm outbreak extent, duration, and severity are dependent on the amount, structure, quality, and spatial distribution of available host types, and conducive environmental conditions for all budworm life stages. Large continuous areas of late-seral and mature Douglas-fir, grand fir, and white fir stands, with multiple canopy layers are most susceptible to defoliation by the western spruce budworm.

The extent and severity of Douglas-fir tussock moth outbreaks are dependent upon many of the same variables as the budworm, although tussock moth outbreaks tend to be cyclic, and their duration is more often regulated by parasites, predators, and virus epizootics (Torgersen and Dahlsten 1978, Mason and Luck 1978, Stoszek and Mika, 1978). Stand and landscape susceptibility for both the western spruce budworm and the Douglas-fir tussock moth are linked to the composition and arrangement of stands according to site quality, host abundance, host vigor, age and density, canopy structure, and continuity of host types (Carlson et al. 1985).

Bark Beetles--The principal bark beetles of the interior West are the Douglas-fir beetle, western pine beetle, mountain pine beetle, and fir engraver. Stand and landscape effects resulting from bark beetle attack are tree mortality, strip attacks, topkilling, reduced tree cover or cover quality, altered species composition and stand structure, and increased fire hazard. Bark beetles initially attack stressed, windthrown, injured, or weakened trees. Once beetles have occupied such trees, they typically move into nearby healthy trees. Stressed or low vigor ponderosa pine and lodgepole pine stands are most frequently associated with over-crowding, poor site quality, and advanced stand age. In late-sera) mature stands of the Douglas-fir, grand fir, or white fir habitat series, stressed trees are abundant as a result of root diseases, drought, dwarf mistletoes, and defoliators. Landscape susceptibility to bark beetle attack is linked with stand composition and arrangement according to site quality, host abundance, host age and size, density of stands, the incidence and severity of dwarf mistletoes and root diseases, windthrow abundance, drought susceptibility, and defoliation severity (Hadfield et al. 1986, Miller and Keen 1960, Mitchell 1987, Mitchell and Martin 1980, Mitchell et al. 1983, Mitchell and Preisler 1991, Sartwell 1971, Sartwell and Stevens 1975, Scott 1991, Wright et al. 1984).

Dwarf Mistletoes--Dwarf mistletoes are parasitic seed plants that infect both vigorous and non-vigorous hosts via ballistically discharged seeds. Dwarf mistletoe species are highly host specialized (Hawksworth and Wiens 1972). The most influential dwarf mistletoes in forests of the interior West are those of western larch, Douglas-fir, lodgepole pine, and ponderosa pine. Because of their wide distribution, dwarf mistletoes together are responsible for the greatest tree growth and mortality effects in western conifers. Bolsinger (1978) reported that at least 47 percent of the western larch, 42 percent of the Douglas-fir and lodgepole pine, and 26 percent of the ponderosa pine in eastern Oregon and Washington were infected with dwarf mistletoes.

The effects of mistletoe infection in stands and landscapes are tree mortality, tree growth loss, modified crown structure, altered species composition and canopy structure, increased fire hazard, and increased susceptibility of infected trees to other mortality factors (bark beetles, drought, defoliators, and root diseases). Dwarf mistletoe damage is dependent upon the presence of mistletoes in host stands. Actual inventory data on the distribution and extent of dwarf mistletoes are needed for accurate hazard assessment. Without this information, only the presence of susceptible hosts in susceptible arrangements can be assessed. Stand and landscape susceptibility to dwarf mistletoes are linked with the composition and arrangement of stands according to site quality, host abundance, canopy structure, and age (Hawksworth and Johnson 1989, Knutson and Tinnin 1980, Parmeter 1978, Strand and Roth 1976).

Root Diseases--The principal root diseases of the interior West are laminated root rot, Armillaria root disease, and S- and P-group annosum root diseases. Collectively, laminated root rot, Armillaria root disease, and S-group annosum root disease greatly affect the Douglas-fir, grand fir, and white fir habitat series, particularly in mid- and late-seral stands. Laminated root rot aggressively kills Douglas-fir, grand fir, white fir, and mountain hemlock. Other coniferous species are moderately susceptible, tolerant, or resistant to damage (Hadfield et al. 1986), and may develop extensive butt decay at maturity. Armillaria root disease primarily affects Douglas-fir, grand fir, and white fir, although all other coniferous species may be attacked. Stressed, weakened, or injured conifers of any species are especially susceptible to this root disease. There are two known types of annosum root disease: the S-group is host specific to true firs, hemlocks, and spruces; the P-group is host specific to pine species, primarily attacking ponderosa pine on the driest sites.

The effects of root disease on stands and landscapes are tree mortality and growth loss, altered species composition and canopy structure, reduced tree cover or cover quality, increased fire hazard, and increased susceptibility of root diseased-trees to bark beetles and drought. Root disease mortality produces small gaps in the forest canopy that favor the release or regeneration of shade-tolerant species, many of which are highly susceptible to root disease. Actual inventory data on the distribution and extent of root diseases is needed for accurate hazard assessment. Lacking these data, only the presence of susceptible hosts in susceptible arrangements can be assessed. Stand and landscape susceptibility to root diseases are linked with the composition and arrangement of stands according to site quality, successional stage, host abundance, canopy structure, and the presence of other mortality factors (bark beetles, drought, and defoliators).

Root disease damage is dependent upon the presence of viable root disease inoculum in susceptible stands. Root pathogens survive several decades in root systems of infected trees, snags, and stumps (Hadfield et al. 1986). This is the inoculum for continued transmission of disease to both vigorous and non-vigorous hosts. Root diseases spread from diseased trees or stumps to healthy trees by mycelial extension, root to root contact, or in the case of Armillaria root disease, by rhizomorphs. Stands with high inoculum potential may exhibit little disease incidence when tolerant or resistant hosts are in abundance. Mortality from root diseases is greatest in stands dominated by susceptible hosts.

**Advantages and Disadvantages**

Insect and disease hazard rating techniques designed for a particular geographic area have limited applicability to other areas without recalibration and validation. Climate, vegetation, and site conditions are highly variable throughout the interior West, and different variables are influential or limiting, depending upon location. Hazard rating systems have been developed over a short span of decades of relatively stable climate. Periods of changing or differing climatic regime would necessitate recalibrating or redesigning most existing systems. Many of the variables used in hazard assessment techniques involve simple, standard measurements. The use of nonstandard variables would require training and practice to learn new measurement techniques and interpretation skills.

Hazard rating techniques were designed to classify stand susceptibility but few accurately predict damage. Some of these rating techniques are time intensive and expensive for the narrowness of their application. Most hazard rating techniques assess the susceptibility of vegetation to a particular insect or pathogen, yet changes in factors that influence the susceptibility of vegetation tend to influence suites of pathogens and insects that often interact with each other. Hazard rating techniques that assess susceptibility to multiple, interacting agents are needed. Existing hazard rating systems are based upon ecological data, theory, and to some extent, local expert knowledge. Systems that use expert knowledge or opinion may be biased by the experience of the developers.

Insect hazard assessments, when conducted, should be considered as preliminary evidence of hazard pending field verification or cross-checking with field examination records. Once hazard conditions are accurately classified and verified, the occurrence probability (uncertainty) of infestations should be estimated. Occurrence probability is a function of the proximity of susceptible conditions to potentially damaging insect populations. Existing stand hazard rating systems tend to inaccurately predict outbreaks in specific stands because the proximity of susceptible vegetation to potentially damaging insect populations is not considered (Paine et al. 1984, Shore et al. 1989). Occurrence probabilities can be developed from empirical data from prior outbreaks. For a given area and point in time, the risk of a potential insect outbreak can be estimated by quantifying the distribution and abundance of host types, surveying the distribution and impact severity of the insect, and projecting insect population dynamics and damage. Subsequent simulation (for example, Monte Carlo) can aid in applying appropriate probabilities to the projected outcomes. Occurrence probabilities that are based on guesswork and assumptions probably interject additional uncertainty, and they are of limited value.

**Improvements Needed**

Hazard rating schemes for forest pathogens, like those developed for forest insects, have been little used by pathologists, and perhaps with good reason. With the exception of S- and P-group annosum root diseases, which colonize new areas by infecting freshly cut stumps, most root disease is found in areas that have had a long history of root disease (Dickman and Cook 1989, Kile et al. 1991, Shaw and Roth 1976). While limited spread of *Phellinus weirii* and *Armillaria ostoyae* to new areas via wind-blown spores is assumed, new centers of disease are infrequently found. Most stands and landscapes with susceptible vegetation and having no Armillaria root disease or laminated root rot inoculum present, will likely remain free of these root diseases for a very long time. The same is true with dwarf mistletoes. Although mistletoe infection centers enlarge, and birds and small mammals do passively vector viable seeds to hosts some distance away, areas with dwarf mistletoe today will mostly be the same a century or two from now.

Hazard classification schemes are needed that classify the root disease and mistletoe susceptibility of plant associations or habitat types (Hessburg and Flanagan 1991, 1992). Plant association guides should include a discussion of the pathogens and insects ordinarily associated with each plant association, and the population trends and effects of each agent on each successive developmental phase or seral stage. Vegetation susceptibility (hazard) and extent of damage (risk) are a function of current and evolving vegetation conditions.

Because dwarf mistletoes and root diseases tend to occupy stands and landscapes for a long time, it would be highly advantageous to inventory their current distribution and severity. This would give resource management personnel the ability to compare the effects of alternative management strategies on root disease and dwarf mistletoe distribution and their effects on stand growth and development. Dwarf mistletoe and root disease hazards and risks (with exception for S- and P-group annosum root diseases) should be assessed within the current distribution of each pathogen. Risk assessments for these forest diseases elsewhere would have little relevance.

The Stand PROGNOSIS Model (Stage 1973) with its many regional variants, can be used to model growth and development of coniferous stands throughout the interior West using ordinary stand examination or inventory data (Wykoff et al. 1982). Root disease (Stage et al. 1990) and dwarf mistletoe (Hawksworth et al. 1992) modeling extensions are available with each PROGNOSIS variant, but they are little used. When running these PROGNOSIS model extensions, model users must include disease information with the sample tree attribute data collected from stand examinations or inventories. The models simulate change in disease distribution and damage for alternative stand development histories. The Parallel Processing Extension of the PROGNOSIS Model (Crookston and Stage 1991) is available for all regional variants that are based on version 6.1 of the Prognosis model. This extension allows simultaneous simulation of 1000 or more stands for periods of up to 400 years. This modeling extension makes possible landscape-scale simulation of root disease and dwarf mistletoe effects. Using the PROGNOSIS modeling system, projections of root disease and mistletoe hazard and risk can be made for unique stands and landscapes throughout the interior West, and occurrence probabilities can be estimated from simulations. To be useful in management planning and decision-making, risk assessment methods must deal with uncertainty.

Annosum root disease is unique among the major forest pathogens because long distance spread by spores is commonplace, and new centers of disease often arise when freshly cut stumps are infected by spores. A recent prototype annosum root disease model was developed (Eav and Adams 1992) for experimental use with a restricted set of regional PROGNOSIS Model variants. This addition to the PROGNOSIS modeling system will ultimately enable users to project annosum root disease risks under alternative stand development histories anywhere that hosts are growing, or can be grown. Population dynamics and/or damage models for the western spruce budworm (Crookston et al. 1990, Sheehan et al. 1987, 1989; Stage 1973), the Douglas-fir tussock moth (Monserud and Crookston 1982), the Douglas-fir beetle (Marsden et al., in press), and the mountain pine beetle (Cameron et al. 1990, Cole and McGregor 1983, Crookston et al. 1978) are also part of the PROGNOSIS modeling system.

The PROGNOSIS modeling system provides an improved capability to forecast insect and disease effects on forest stands and landscapes; however, operational use of the system by project planning teams is a goal yet to be realized. Existing model extensions are continuously improved and streamlined but not at the pace required. Greater investment in this modeling system is needed to validate and refine existing extensions, make the modeling system easier to use, and build new extensions for other insects and pathogens. Also needed is a multiple insect and pathogen model that allows users to simultaneously simulate the population dynamics and effects of several insects and pathogens operating in the same stands or landscapes. At present, no such analysis is possible but future development has been proposed (Eav and Adams 1992).

## Fire

Since the turn of the century, land use and the climate of fire-adapted ecosystems in eastern Oregon and Washington have changed, perhaps irreversibly. Successful fire suppression over the past 80 years has interrupted several fire cycles, contributing to a shift in species composition and a build-up of fuels. Douglas-fir and true firs have assumed dominance over ponderosa pine and western larch, by competing with them for limited water and nutrients. Douglas-fir and true firs have encroached on meadow and range-land areas. This human-altered ecosystem has proven to be susceptible to insect infestation, disease epidemics, and catastrophic wildfire, and is less adaptable to climate change. As a result, many scientists, land managers, law makers, and local residents have concluded that the forests of eastern Oregon and Washington are becoming increasingly unhealthy.

### Previous and Current Techniques

Fire is a natural disturbance process in nearly all western forest types. Because fire is often viewed as a destructive force on many forest sites, most previous and current risk assessment efforts have focused on the risk of fire occurrence, and little work has been done to assess the risks associated with long-term fire suppression. For example, a stand-scale risk rating system for stand replacement fire is being developed for the Boise National Forest (Reinhardt, in progress). This risk rating system is designed to use data available from USDA Forest Service stand examination databases, and reflects the risk that a stand on a given site will be replaced as a result of a fire. If this information is examined with historical fire occurrence data, it can be used to assess the risk of fire to a forest ecosystem.

This risk rating system is derived by using fire behavior fuel models (Anderson 1982), slope, and an assumed wildfire and weather scenario to model surface fire intensity (Rothermel 1983). Average probability of death of overstory trees is then calculated by using surface fire intensity, species, diameter, and height as inputs (Ryan and Reinhardt 1988). Three adjustments to the simple model can be made to improve risk ratings: (1) if risk ratings for insects and disease are high and would likely result in a short-term fuel increase, a different fuel model is used; (2) if ladder fuels are present (the stand is layered), risk of stand replacement fire will be greater as torching or crowning add to damage from surface fire; and (3) if deep duff or heavy loadings of large woody fuels are present, there is an increased risk of tree death due to bole and root damage. Currently, the scale of this risk rating system is a single stand and assesses only the risk to the timber resource. The output is a quantitative estimate of probable death of the stand overstory.

### Risk Factors Used

Risk factors used in this model are fuels described in terms of the fire behavior fuel model--duff depth, presence of ladder fuels, and loading of large woody fuel, slope, and stand characteristics including species and diameter of overstory trees.

### Advantages and Disadvantages

This risk assessment method is simple and based on well-documented models; the output is easy to interpret, and information required is easily obtained. There are, however, many aspects of fire-associated risk not addressed by this method. Omissions from this model include risk that an ignition will develop into a large fire (this risk can be related to historic weather records and fuels), risk of reduction in long-term site productivity due to fire (this risk is not yet well quantified, but site productivity can be affected by soil heating), risk to resources such as watersheds (Tiedemann et al. 1979) or residential areas (Arno and Brown 1989, Simmerman and Fischer 1990); and risk of declines in forest health, reductions in biodiversity, and loss of seral species associated with fire exclusion (Brown and Arno 1991).

**Improvements Needed**

A risk analysis currently being developed for eastern Oregon and Washington ecosystem health assessment is designed to consider a larger spatial and temporal scale, and to assess risk to ecological processes rather than to the timber resource. This new system may provide a considerable improvement to current fire risk and air quality tradeoff assessment methods.

In this analysis, historical (1932-1959) and current (1981-1992) aerial photographs were interpreted for live vegetation characteristics for polygons within sample watersheds of the Pend Oreille, Methow, Wenatchee, Yakima, Grande Ronde, and Deschutes river basins in eastern Oregon and Washington. The characteristics were matched to the closest situation represented in one of several fuel and fire behavior photo series publications (Fischer 1981, Maxwell and Ward 1976, 1980) by developing a key based on vegetation composition and structure. Of the fuel and fire behavior photo series available, 36 photos were selected or stylized to represent the range of fuel conditions within the six river basins. These photos were applied to fuel complexes representing nonforested conditions, forested in natural conditions, and conditions after post-logging, thinning, and other management activities. For each fuel and fire behavior photo series, information on fuel loadings by size class, spread rate, flame length, and resistance to suppression was available to develop the fuel loading and fire behavior database. Fuel loading and fuel moisture content were entered into the CONSUME model (Ottmar et al. 1993) to estimate fuel consumption. Fuel consumption was multiplied by an appropriate emission factor (Hardy and Teesdale 1991, 1992) to determine emissions produced for wildfires and prescribed fires. Historical and current emission production rates were compared.

The sampling unit used within a river basin was the watershed. All polygons within a watershed were combined to obtain a mean value for the fire and smoke-related attributes. Mean values for the watersheds were obtained for the following variables: forest fuels (tons/acre), fuel consumption (tons/acre) in prescribed and wildfire scenarios, fire rate of spread (chains/hour), flame length (feet), fire resistance to suppression (chains/person-hour) for wildfires, smoke emission factors of particulate matter 10 micrometers in diameter or less (PM 10) (pounds/ton of fuel consumed) and smoke production of PM 10 (pounds/acre) for prescribed fires and wildfires.

Once the watershed mean value for fire and smoke-related attributes were obtained, the procedure to calculate the historical and current means for the river basin followed the procedure described by Lehmkuhl and others (1993). Lemkuhl and others also described the statistical analysis used to assess the change across time of fire behavior and smoke production attributes. These results can be used to estimate the extent by which historical changes in vegetation may be altering fire regimes of ecosystems in eastern Oregon and Washington.

## Soil Erosion

Erosion is a geomorphic process that is a natural component of any forest ecosystem. Two broad groups of erosion processes occur on forest hillslopes: surface erosion and mass erosion. Natural and humancaused disturbances can accelerate both types of erosion processes. Wildfire is by far the most common cause of accelerated erosion from natural disturbances (Megahan et al. 1981), whereas the effects of insect attack and windstorms usually are negligible. Human activities affecting hillslope erosion include forest management activities (cutting, skidding, and site preparation including prescribed burning), road construction (both access and skid roads) requiring cut and fill operations, and grazing.

Surface erosion is defined as the movement of individual soil particles by wind, water, or gravity. Four different types of surface erosion processes are dominant on forest lands in the western United States:

sheet, ravel, rill, and gully. Sheet erosion is caused by the effect of raindrops and sometimes by uniform overland flow. Sheet erosion can occur on any slope where mineral soils are exposed. Ravel, sometimes called dry ravel or dry creep, occurs during nonrain periods when mineral soils on steep slopes are exposed and move downslope due to gravity. Rills consist of downcutting caused by channelized overland flow of water. Gullies are simply large rills (greater than 30 cm deep) caused by large volumes of concentrated overland water flow.

Mass erosion (landslides in this context) is defined as the movement of many soil particles in a single mass, primarily under the influence of gravity. Mass erosion occurs when shear stresses within a slope exceed shear strength. Unlike surface erosion, which progresses from the surface downward, mass erosion usually includes the entire soil mantle and often part of the underlying parent material or bedrock as well. The most common kinds of mass erosion include debris failures, slumps, and earthflows. Debris failures, which include debris slides and flows, are the rapid sliding of surface soil over the underlying bedrock or parent material. Debris failures occur most often on steep slopes (greater than 60 percent) and usually occur in slope depressions that serve as water accumulation zones. Slumps and earthflows tend to be deep-seated with the failure surface usually well beneath the soil. Slumps are manifest as a well-defined rotational failure with a distinct shear plane. Earthflows are characterized by a long-term deformation of material and can occur on gentle slopes.

**Previous and Current Techniques**

Soil erosion risk assessment is a process of extrapolating information about erosion from reference sites to other sites having similar characteristics. Information about erosion obtained at reference sites may range from simple observations that erosion is occurring under certain circumstances to detailed measurements of soil loss. Knowledge of erosion processes and observational experience are critical to all stages of the erosion risk-assessment process. Selection of a methodology for assessing hillslope erosion risk depends on the type of erosion and the nature of the disturbance on the slope. The desired information obtained from the assessment is also an important consideration. For example, a relative ranking of erosion risk for alternative management strategies may be suitable for broad-scale land management planning but inadequate for assessing long-term cumulative effects (NCASI 1992).

Extrapolation of erosion information to other locations is based on an assessment of the factors affecting erosion at the new site. These factors are assessed by using aerial photographs, soil and vegetation surveys, rainfall and geologic maps, and field inventories. Erosion risk assessments are made either as a relative risk rating or a prediction of actual amounts of soil loss (Megahan and King 1985).

Risk of surface erosion by water and wind can be evaluated by direct measurement using a variety of techniques such as catchments, erosion bridges and pins, and rainfall simulators (Dunne 1977, Mutchler et al. 1988). These techniques are commonly used to monitor the results of land management practices. Erosion data also are used to develop empirical relations to relate site characteristics to erosion potential. Such relations can then be used to predict erosion at other locations with similar site characteristics prior to implementing land management practices. Although desirable, it is often not possible to develop an erosion database for the area where risk assessment is needed. Instead, erosion models developed at other locations are used to estimate erosion for the area of concern. Such models may be empirical, linking erosion occurrence and amounts to site characteristics, or more physically based models that simulate actual erosion processes. In either instance, the model selected must meet the objectives of the erosion risk assessment, and it must be appropriate for the location. Determining this fact often requires collecting erosion data for validation purposes.

The universal soil loss equation is the most widely used empirical method to predict surface erosion (Wischmeier and Smith 1978). This equation was originally designed to predict average annual sheet and rill erosion on agricultural lands. A modified soil loss equation was developed to adapt the universal soil loss equation to forest lands (Warrington et al. 1980) and subsequent work adapted the cover factor for application on forest lands (Dissmeyer and Foster 1981). The revised universal soil loss equation, a recent update of the universal soil loss equation, corrects many of the deficiencies of both the universal and modified universal soil loss equations (Renard 1991). Another empirical model was developed by Cline and others (1981) to provide estimates of average annual sediment production from watersheds, based on estimates of average surface and mass erosion. Several versions of this sediment yield model have been developed for specific geographic areas in the western United States. The water erosion prediction project is developing a process-oriented surface erosion model for forest lands (Foster, 1987). Probably the most common method for delineating surface erosion hazards on forest lands uses erosion hazard ratings based on stratification of important variables identified in the universal soil loss equation, or other empirical equations applied to specific site conditions on the ground. Such ratings commonly are used to define relative erosion hazards rather than to quantify erosion rates.

Risk assessments of mass erosion usually are based on landslide surveys and local experience relating landslide occurrence to terrain and physical features such as geology, landform, and hydrologic characteristics. Information from remote sensing, maps, and field reconnaissance is used to analyze risk factors and to make inferences about probable landslide occurrence in other areas with similar site characteristics (Swanston et a1.1980). The use of slope stability models developed by geotechnical engineers also can be used to assess landslide risk. Many of these models exist for all types of landslides, but only those developed for debris avalanches are practical for widespread application on forest lands. Hammond and others (1992) provide a method for defining debris avalanche risk on a probabilistic basis.

**Risk Factors Used**

Examples of risk factors used to predict surface erosion using the universal soil loss equation and its derivatives include rainfall intensity and duration, soil erodibility (often inferred using texture, structure, and organic matter content), slope length and gradient, vegetative and soil cover characteristics, type of soil disturbance, and erosion control practices. Risk factors used in the sediment yield prediction method developed by Cline and others (1981) are bedrock weathering, soil texture, and landform characteristics.

The relative importance of factors used for assessing mass erosion risks differ by location. These risk factors can include landform features (slope gradient and shape, terrain origin), soil characteristics (soil depth, drainage properties, cohesion, and the internal angle of friction), bedrock lithology and structure (rock type and degree of weathering, attitude of bedding planes, the degree of jointing and faulting), vegetative cover (tree volume and rooting characteristics), and the potential for large volumes of water from rain and snowmelt.

**Advantages and Disadvantages**

Direct measurements of surface erosion generally are inexpensive and easy to install and use; however, a commitment of time and money is required to monitor the large variations in soil erosion over time and space. Catchment studies provide a measure of net soil loss from the catchment area, but data obtained may be influenced by catchment size. Erosion bridges and pins only provide a measure of change in soil depth at the measurement point, thereby making the results difficult to quantify and interpret (Dunne 1977, Mutchler et al. 1988). Models and equations can be used to estimate erosion amounts and to provide an evaluation of the erosional consequences of alternative management practices. Prediction modeling of surface erosion by using the derivatives of the universal soil loss equation can be a valid tool, but limitations of the models need to be recognized (Warrington et al. 1980). The water erosion prediction project technology promises improvements for some of the universal soil loss equation limitations, but it will require considerably more site-specific information. Many models calculate average erosion across an area, but in reality, most of the erosion is contributed by critical source areas and is not uniform (Megahan and King 1985).

Mass erosion is a complex process, and the relations among the factors that influence stability are not always clear. Geologic conditions are often complex and groundwater conditions are seldom reliably known. Risk evaluation techniques, such as the procedure described by Swanston and others (1980), rely on the ability of the interpreter to accurately recognize and analyze risk factors. The reliability and accuracy of the resultant evaluation can differ depending on the interpreter’s knowledge and the availability of reliable risk factor data. Mass erosion models can work well when detailed data are available, but the techniques require considerable geotechnical expertise and data collection is usually costly. The application of such models over broad areas, consequently, is impractical. The alternative is to define erosion hazards on a probabilistic basis, as is done in the level 1 stability analysis developed by Hammond and others (1992). This model introduces the concept of uncertainty to soil erosion assessment, and may make some types of land management decisions more difficult.

### Improvements Needed

New assessment techniques and models are being proposed and tested as our knowledge of risk factors and their relations improve. In addition to these improved assessment tools, better inventories of risk factors are needed along with the ability to manipulate and analyze these data. Progress is continuing in database design and analysis. Use of geographic information systems is improving spatial analysis capabilities and accuracy. The ability to analyze data for use at multiple scales that range from regional to site-specific is also increasing.

## Wildlife

### Previous and Current Techniques

Risk assessments for wildlife management differ along gradients of taxonomic scope and procedural complexity. Broad biodiversity risk assessments consider many different species and use relatively simple procedures, whereas assessments of risk to single species usually employ more complex procedures. Single-species assessments have narrow taxonomic scope, and differ in complexity from simple deterministic to more complex probabilistic rank estimates. Probabilities define the amount of uncertainty associated with unpredictable biological or management activities. Complexity of both biodiversity and single-species assessments also can differ from a generalized consideration of risk under current conditions to an analysis of several management alternatives.

Biodiversity assessment methods have developed from early wildlife habitat relations programs (e.g., Brown 1985, Lehmkuhl and Patton 1984, Patton 1978, Thomas 1979). Wildlife habitat relations databases include information on wildlife use of plant communities, and often contain ordinal versatility and other indices of habitat use. This information is used mostly by managers in project-scale analyses to assess how manipulation of vegetation might change wildlife communities. In a few instances, quantitative data, rather than subjective rankings, have been used to link historical and projected forest type acreage with vertebrate populations to estimate trends in the abundance of vertebrates and functional species groups (Raphael 1988, Raphael et al. 1988).

A second type of biodiversity risk assessment is used in setting priorities for state-wide or global conservation efforts (Burke and Humphrey 1987, Mace and Lande 1991, Millsap et al. 1990, Niemi 1982, Ranjit-Daniels et al. 1991), or to assess the effects of large-scale habitat loss and fragmentation (Hansen and Urban 1992, Johnson et al. 1991, Lehmkuhl and Ruggiero 1991). These types of assessments use wildlife habitat relations information to develop ordinal ratings of risk for many species, but differ from habitat-oriented models by considering species life history, population structure, and the environment within a population viability framework (Gilpin and Soule 1986).

Risk assessments of individual species have narrow taxonomic scope and are typically more complex. Models based on habitat relations are designed to rapidly assess risk to single species for use in project-scale analyses. These models include habitat suitability index models (Schamberger et al.1982), habitat evaluation procedures (Flood et al. 1977, U.S. Fish and Wildlife Service 1980), and Bayesian pattern recognition models (Grubb 1988). Morrison and others (1992) give a complete review of these models. Population parameters usually are not considered in these procedures.

Population and environmental variables are considered in more complex single-species assessments, which are generically termed population viability analyses (Boyce 1992). These assessments model the effects of demographic, genetic, or environmental variability on population stability to examine how expected time to extinction changes with the environment, population structure, or behavior (Royce 1992, Gilpin and Soule 1986, Shaffer 1981). An important innovation of this risk assessment method that distinguishes it from others is the consideration of uncertainty due to unknown or unpredictable events. Uncertainty is incorporated by modeling variation in population parameters and estimating probabilities of extinction over specified periods of time, instead of using a single estimate for an unspecified time as many wildlife habitat relations methods do. Viability analyses have been done for the northern spotted owl (Strix occidentalis) (Lande 1988), grizzly bears (Ursus arctos) (Shaffer 1983), checkerspot butterflies (Nymphalidae Euphydrys) (Murphy et al. 1990), louseworts (Pedicularis) (Menges 1990), and primates (Kinnaird and O'Brien 1991). Population viability analyses have been used mostly in a generalized sense rather than specifically to assess risk from alternative management.

The extension of population viability analyses to estimate animal risk under several management alternatives has been an important innovation for assessing and managing risk (Marcot 1986, Salwasser et al. 1984). Various population viability analyses have been used to rank risk from management alternatives for the northern spotted owl (Johnson et al. 1991, Marcot and Holthausen 1987, Thomas et al. 1990, USDA 1988) and other species associated with late-successional forests (Johnson et al. 1991). Burke and others (1991) and Soule (1989) have done similar risk analyses for other vertebrates.

The highly structured, quantitative decision analysis method is another form of risk assessment that incorporates probabilities of biological and management uncertainty to calculate expected values of management alternatives (Maguire et al. 1987, Raiffa 1968). Decision-tree risk analysis is the most commonly used procedure and has been done for grizzly bear (Maguire and Servheen 1992), black-footed ferrets (Mustela nigripes) (Maguire et al. 1988), tigers (Panthera tigris) (Maguire and Lacy 1990), and Sumatran rhinoceros (Dicerorhinus sumatrensis) (Maguire et al. 1987). Other methods of decision analysis, such as expected value of perfect and sample information, and Bayesian statistics may be useful for wildlife risk analysis (Marcot 1992). Expert system computer models are increasingly being viewed as an important tool to analyze risk in relation to specific resource management problems (Marcot 1992, McNay et al. 1987).

**Risk Factors Used**

Wildlife risk assessments for land management purposes typically stress the loss and fragmentation of habitats and how life history and population structure enable species to persist in spite of habitat fragmentation (Lehmkuhl and Ruggiero 1991). Factors used in wildlife risk analysis describe the interactions among species life history, population structure, and their abiotic and biotic environments. Key risk factors for wildlife populations are population structure, movements, and social behavior. Forest management normally changes stand or ecosystem environmental conditions, usually by reducing the amount or quality of habitat area, but it also can modify competitive and predator-prey interactions. Risk factors associated with environmental change include changes in vegetation composition and pattern; attributes of species population structure, habitat selection, and movements; and social behaviors that influence persistence in changing environments.

Gilpin and Soule (1986) summarize the life history, population, and environmental components of population viability analyses for wildlife risk assessment. Life history factors include animal morphology, physiology, behavior, movements, and habitat selection. Important attributes of population structure include the frequency of occurrence, size, interaction of population subunits across the landscape, age and size structure, sex ratio, growth rate, and growth rate variance within and among subpopulations. Environmental components of population viability include not only habitat quantity, but habitat quality in terms of resource density, abundance of interacting species, and patterns of disturbance (duration, frequency, severity, and spatial scale).

Identification of wildlife risk factors in multiresource ecosystem risk analyses presents problems different from species-oriented approaches because of analysis scale (Nash 1991). The ecosystem and its individual components first must be defined. Wildlife components may include multiple species with different habitat requirements and population structures (e.g., Johnson et al. 1991), which have different responses to ecosystem stress and desired conditions. Nevertheless, the same population viability analyses framework can be used to identify risk factors for individual species or functional groups of species.

**Advantages and Disadvantages**

Simple methods of biodiversity assessment and wildlife habitat relations models that produce ordinal ratings or indices of species risk have the advantages of being broadly applicable, rapid to use, and able to incorporate local expert knowledge where research data are few. Simple ranking techniques usually are easily understood by managers in other disciplines and can be applied to estimate risk for many species over large areas. The generality of the procedure, however, brings with it disadvantages. Information based on expert opinion may be biased by the personal experience of the contributors. Important life history or population information that is critical for some species may be absent from the analysis. Such generalized procedures might not incorporate risk variation associated with the variability of species populations, environmental conditions, or uncertainty in species response. Such data are important for application to local management conditions, such as stands and parts of watersheds. Management alternatives and their costs may not be explicitly stated, so application is necessarily at a strategic planning scale.

More detailed risk analyses, usually associated with population viability analyses of one or a few species, use more life history and population information, consider uncertainty, and often examine effects of management alternatives (e.g., Johnson et al. 1991, Marcot and Holthausen 1987, Thomas et al. 1990). Uncertainty in management and species responses to risk factors are specifically stated as a range of probabilities. These risk analyses require more information and analysis time, which usually limit the number of species that can be considered. Management, social, and economic costs usually are not explicitly included in the calculation of risk scores, but are considered in separate adjunct analyses.

Highly structured risk assessment procedures using decision analysis have the advantage of explicitly stating all assumptions and possible management alternatives in a standard format (Maguire et al. 1987, 1988; Marcot 1986, 1987). Management actions and species responses are given as probabilities with a defined range of expected values and variation. Different types, sources, and units of management and ecological information can be incorporated to facilitate management decisions. There are several disadvantages, however, when using decision analysis. Probabilities often are difficult to accurately assess, and assumptions to estimate probabilities may inject too much additional uncertainty. Small changes in probabilities may greatly affect the results of the analysis. All management alternatives may not be foreseen, thus biasing the management decision. Defining specific management objectives for individual species and quantifying habitat conditions from biological, social, and political viewpoints may be difficult. Extrapolation of small scale analysis (e.g., a forest stand) to larger scales (e.g., watersheds) may become unwieldy (Marcot 1987, Nash 1991).

### Improvements Needed

Improvements in wildlife risk assessments can be made in several areas. Better information is needed on species population structure and their range of variability over space and time, habitat selection, and effects of management on population structure and habitat selection. This information can be gained through additional research and also from improved ecosystem monitoring within the framework of adaptive management. Just as important is the need to better estimate the economic and social values of wildlife and the costs incurred by different management practices (Nash 1991, Starfield and Herr 1991). Administrative goals and objectives for forest management must be made compatible with wildlife and ecosystem conservation if conflicts among special interests over the management of public lands are to be resolved. For example, sustained timber yield may be incompatible with maintaining well-distributed vertebrate populations. Essential to both wildlife and ecosystem management is defining essential ecosystem components and linking processes, and the endpoints or desired conditions of these ecosystems (Nash 1991).

## CONCLUSIONS

Risk assessment has been and will continue to be an important component of land management planning in the western United States. Broader applications of ecological management in this region will require a shift from local economic, and stand-oriented assessments to landscape-scale considerations. Empirically based risk assessments are giving way to more complex holistic approaches, which incorporate a better understanding of ecosystem processes into risk model development. Improved risk models will incorporate assessments of various ecosystem risk factors to produce an integrated stand or landscape assessment. Such work is already being done in some fire models where the risk of insect attack and disease is used to project changes in fuel loadings; however, much more work needs to be done. Ecological management is a dynamic and evolving concept; accordingly, risk assessment methodologies will evolve with it.

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# ECODATA--A Multiresource Database and Analysis System for Ecosystem Description and Evaluation

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## ABSTRACT

ECODATA is a term used to describe a set of relational databases and analysis programs developed for environmental assessment and ecosystem analysis. This system contains a series of mufti-intensity sampling methods and databases that facilitate consistent and efficient descriptions of various ecosystem components (e.g., vegetation, soil, streams, wildlife, and topography) at the site or plot scale. Ecosystem analysis programs (ECOPAC) are also included which access such data and produce standard reports, statistical summaries, and resource value interpretations for plots or plot groupings (i.e., classifications). Output from such programs may be used to describe different types of classification (e.g., existing vegetation, potential vegetation, soil, and stream types) in various classification databases (ECOCLASS). These databases are linked to a series of polygon or map unit sampling methods and databases which are used to describe digitized map themes in environmental assessment efforts. The hierarchical design of this system accommodates rapid updating of map theme attributes based on site-scale inventory data. Consequently, ECODATA is a powerful tool for implementing ecosystem management.

## INTRODUCTION

Evaluation of landscapes for ecosystem management involves the description of biotic and abiotic variability over various spatial scales. Additionally, environmental processes that control patterns of species, community, and ecosystem distribution must be identified (Margules and Austin 1991). Accordingly, standard ecosystem attributes of sampled sites must be described to facilitate identification of ecosystem process-pattern relations at any scale of analysis (Bourgeron et al. 1993). Consistent database structures are also required if analysis software is to be efficiently used in landscape evaluation (Margules and Austin 1991).

An obstacle to implementing ecosystem management is the lack of standard systems for ecosystem characterization and analysis (RISC 1983). For example, vegetation and environmental data from most ecological studies or inventories are rarely collected in a similar manner because sampling methods are seldom compatible (McClure et al. 1979, Pfister and Arno 1980). Such data are of limited use in ecosystem management because they cannot be shared across studies and, therefore, cannot be used in broad-scale regional analyses (Landau 1980, O'Brien and Van Hooser 1983). Nonstandard data collection also inhibits coordinated analysis of resource information between disciplines (Chalk et al. 1984, Hann 1989, Bastedo and Theberge 1983). This lack of standard data collection commonly results in increased inventory costs because repeated visits to sampling sites are often required for multidisciplinary analysis (Landau 1980, Cost 1978).

In 1985, the Ecology Staff Unit of the Northern Region of the USDA Forest Service recognized the need to develop a standard set of sampling procedures for the inventory and monitoring of vegetation that would be appropriate to all resource functions (Keane et al. 1990). The sampling methods and databases developed by this group were termed "ECODATA" and have expanded to include other resource information such as stand stream data (Jensen et al. 1992).

After standard sampling methods and databases were developed for ecosystem description, it then became feasible to develop analysis software that could access such data for efficient, consistent resource interpretations. The analysis system that uses the various plot-scale databases of ECODATA is called "ECOPAC" and has been continually updated since 1985 to facilitate improved ecosystem analysis. This paper presents a brief description of the ECODATA sampling, database, and analysis system (Jensen et al. 1992).

## System Overview

The ECODATA database and analysis system is designed to facilitate consistent collection, storage, and interpretation of basic ecosystem information. The following hierarchical structure was used in system design:

1. basic plot-scale inventory data are summarized by appropriate ECOPAC analysis programs and results are written to various ECOCLASS databases (fig. 1)

2. the classification information contained within ECOCLASS is linked to a set of Polygon Sam plin Databases. These databases describe the percentage of composition of various taxa (e.g., soil, vegetation, and stream types) within uniquely labeled polygons (e.g., watershed and vegetation stands) or ecological mapping units (Bailey et al. 1993), and

3. polygon attribute information is further processed by various LANDPAC software programs for spatial analysis of ecosystem information in environmental assessment efforts.

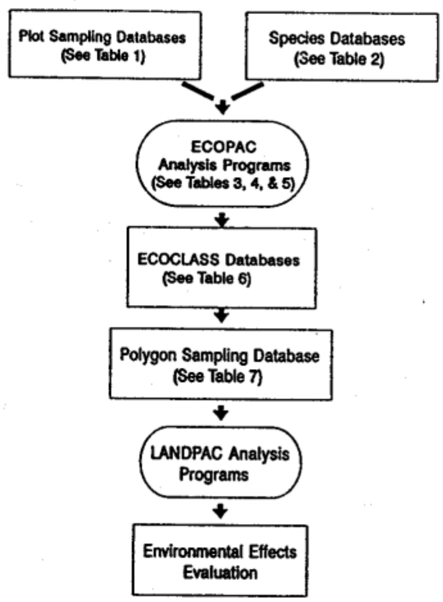


**Figure 1--Overview of the ECODATA database and analysis system.**

The flow path depicted in figure 1 represents a progressive series of data reduction steps, whereby (1) basic, measured site data are summarized into classification taxa, (2) such taxa are then used to describe polygon components, (3) polygon components are further summarized to yield average resource values for map themes, (4) map themes may then be digitized and spatial analysis performed in a geographic information system environment, and (5) spatial indices (e.g., pattern analysis) and resource prediction system software (e.g., sediment models) may then be used to further interpret map themes for various types of environmental assessment. The discussion that follows provides a brief description of components in the ECODATA system.

## Plot Sampling Methods and Databases

The primary sampling unit of ECODATA is a macroplot. For terrestrial systems, the macroplot sampling entity is commonly a relatively uniform plant community and its environment within a map unit or geographic area. The sampling entity should occur on one potential vegetation setting (e.g., one aspect, slope, soil, and environment) and possess similar vegetation structure and composition. For riverine systems, the sampling entity described by a macroplot in reconnaissance riparian surveys is usually a representative reach of a stream segment (i.e., a segment with similar gradient, substrate, bankfull width and depth), and a fisheries habitat type (e.g., pool and riffle) in intensive survey efforts (Jensen and Manning 1991).

Macroplots can be of any size or shape; however, a 1/10-acre (400-$m^2$) circular plot is commonly used in terrestrial inventory work. Such plots are often located subjectively in a representative portion of a stand without preconceived bias (Mueller-Dombois and Ellenburg 1974). These plots may also be located by random or stratified-random sampling designs based on inventory or monitoring objectives (Bourgeron et al. 1993).

Several sampling methods and databases currently exist in the ECODATA system for plot-scale descriptions of ecosystem components (fig. 2, table 1). Additional sampling methods are continually added to the system based on user needs. The sampling methods presented in table 1 were designed for basic ecosystem component description by common survey methods. For example, foliar cover of plant species may be described by rapid visual estimation (i.e., plant composition), or by intensive replicated sampling of microplots (i.e., cover frequency) or line transects (i.e., line intercept) dependent on study needs. The flexible design of the ECODATA plot-sampling methods allows the user to select appropriate methods for sampling based on purpose of sampling, accuracy and precision requirements, cost, and personnel experience. Accordingly, these methods may be used to accommodate most integrated inventory and monitoring needs.

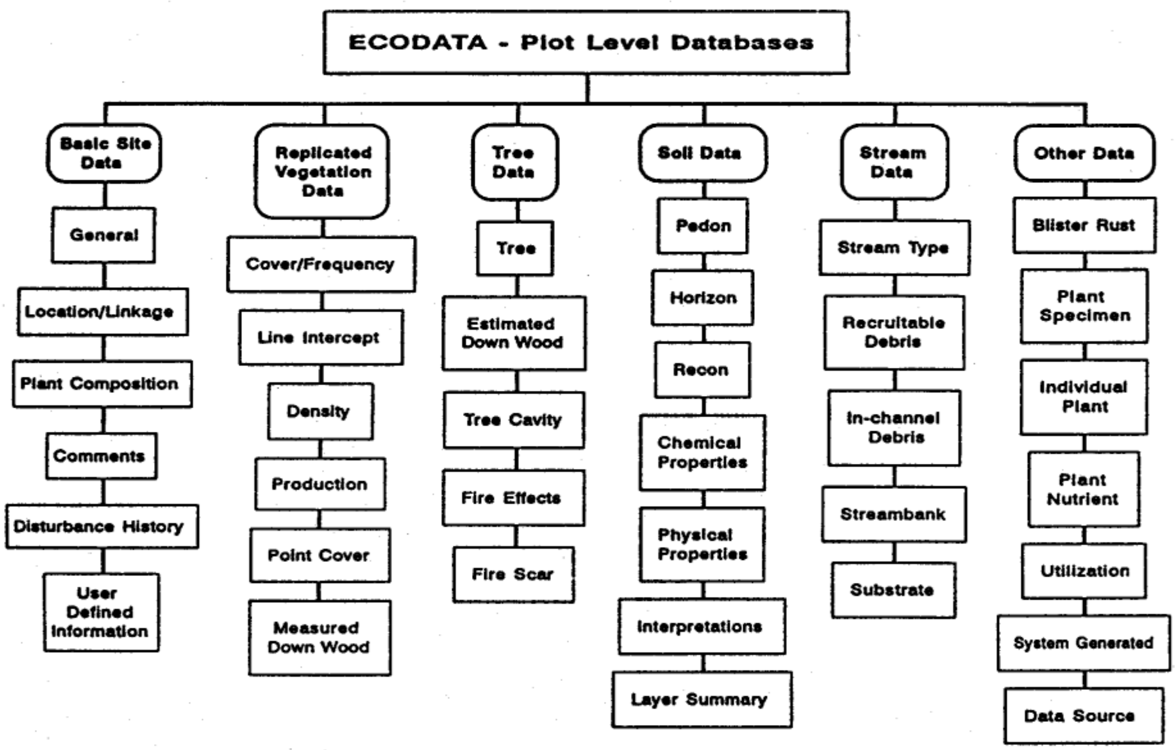


**Figure 2--Diagram of ECODATA plot scale databases stratified by six general types of sampling needs. Each box corresponds to a sampling methodology, field form, and relational database.**

## Table 1--ECODATA Plot-Sampling Methods and Databases

| Type of data | Sampling method | Description |
|---|---|---|
| Basic site data | Sample | Describes the types of samples collected at a plot. |
| | General | Broad vegetation and environmental site descriptors (e.g., elevation, aspect, tree cover, ground cover, fuel loading). Always used to describe a terrestrial plot. |
| | Location and Linkage | Required information for locating and linking a plot to other databases (e.g., UTM coordinates, spectral values, watershed ID, stand ID). |
| | Plant Composition | Visual estimates of plant cover, height, and synecological information for plant species on a macroplot. |
| | Comments | User-specific comments concerning a plot. |
| | Disturbance History | Information concerning the disturbance or treatment history of a plot. |
| | User Information | Generic sampling method that accommodates user-specified optional data fields. |
| Replicated vegetation data | Cover and Frequency | Foliar cover and nested rooted frequency data are recorded by microplots within a macroplot sampling unit. |
| | Line Intercept | Interception of foliar cover is recorded by species along line transects. |
| | Density | Density (individuals per unit area) is recorded by item (e.g., plant species, deer pellets) within belt transects or circular microplots. |
| | Production | Quantitative biomass estimates of individual species (or groups of species) are facilitated by clipping and weighing plant matter within a series of microplots. |
| | Point Cover | Ground cover (or plant species cover) are recorded along systematically located points on line transects. |
| | Measured Down Wood | Fuel loadings and decay classes of down woody material are measured along line transects. |
| Tree data | Tree | Basic tree attributes are recorded by species (e.g., incremental growth, crown ratio, disease status). |
| | Estimated Down Wood | Visual estimates of fuel loadings and down wood biomass are recorded. |
| | Tree Cavity | Information concerning tree cavity number, size, and shape are recorded by tree species. |
| | Fire Effects | The fire effects present on a macroplot are recorded (e.g., duff consumption, soil oxidation, scorch height). |
| | Fire Scar | The presence and timing between fire events are recorded by tree fire scar measurements. |
| Soil data | Pedon | General soil pedon features are recorded at a macroplot (e.g., depth to water table, permeability, soil name). |
| | Horizon | Soil properties are recorded by horizon layer (e.g., color, texture, coarse-fragment content). |
| | Recon | Reduced list of pedon and horizon features are recorded for reconnaissance survey efforts. |
| | Chemical* | Soil chemical properties are recorded by horizon layer. |
| | Physical* | Soil physical properties (e.g., bulk density, texture) are recorded by horizon layer. |

Table 1--ECODATA Plot-Sampling Methods and Databases (continued)

| Type of data | Sampling method | Description |
|---|---|---|
| | Interps* | Soil pedon interpretations are recorded (e.g., water-holding capacity, suitability ratings). |
| | Layer Summary | Layer summaries are created from soil horizon data and stored by user-specified layer types (e.g., surface layer, rooting layer). |
| Stream data | Stream Type | Data required for classifying a stream are recorded (e.g., bankfull width and depth, gradient, sinuosity). |
| | Recruitable Debris | Numbers of recruitable stream debris are recorded by size class and species. |
| | In-Channel Debris | Numbers of in-channel debris are recorded by size class and species. |
| | Streambank | Visual estimates of streambank cover, stability, and textural characteristics are recorded. |
| | Substrate | Visual estimates of stream substrate cover are recorded. |
| Other data | Blister Rust* | Various indices of blister rust are recorded by tree species. |
| | Plant Specimen* | Data concerning plant specimens are recorded by species. |
| | Individual Plant* | Data concerning individual plants are recorded for intensive monitoring purposes |
| | Plant Nutrient* | Plant nutrient data (e.g., digestible nitrogen content) are recorded by species or groupings of species. |
| | Utilization* | Various types of biomass utilization indices are recorded by plant species. |
| | System Generated | Computer-generated data for a plot from various ECOPAC analysis programs (e.g., SITCLIM, FORAGE). |
| | Data Source | Identifies the sources of data for various fields of the plot level databases. |

* Indicates databases that are being developed and are not available for distribution at this time.

The sample and general sampling methods (table 1) are always completed for a terrestrial ECODATA macroplot. These methods contain the minimum amount of information needed for most ecosystem description and analysis purposes and also provides linkages to the other plot-sampling methods. The remaining plot-sampling methods are used to collect more detailed macroplot information depending on survey objectives. Each sampling method has its own field form, coding instructions, and database.

The PROM Utility (Data General 1984) was used to create most of the entry screens for the ECODATA plot level databases which are stored in INFOS II flat files on the mainframe Data General System (Data General 1985). These databases are currently being reformatted to the ORACLE database management system (ORACLE 1989) for future development of a “PC” version of ECODATA in cooperation with The Nature Conservancy and the Intermountain and Pacific Northwest Research Stations. Other Forest Service Regions are also involved in this effort.

## Species Databases

The plot-sampling methods of ECODATA do not provide all the information required for ecosystem analysis by the ECOPAC software programs. Accordingly, a series of support databases exist in the ECODATA system that contain descriptive information about plants and animals that are accessed by various ECOPAC analysis programs. These databases (table 2) contain information about the plants and animals of the northern Rocky Mountains and can be used outside the ECOPAC environment. For example, the Autecology database provides detailed autecological information for specific plant species that can be queried by botanists and ecologists in their work. A generic structure was used in the development of these INFOS II databases; consequently, they can be used in areas outside the northern Rockies by adding or deleting records as appropriate to local analysis needs.

**Table 2--ECODATA Species Databases**

| Database name | Description |
|---|---|
| Flora | Complete species list of all vascular plant species in Montana, Idaho, North Dakota and South Dakota. Contains general distribution information and scientific and common names. This database is accessed by the ERRCHK, FORAGE, and REPORT programs of ECOPAC. |
| Sensitive Plant | List of sensitive, threatened, or endangered plant species along with administrative and ecological characteristics. |
| Autecology | Detailed, quantitative autecological description of selected plants in the Flora database. This information is accessed by the plant succession prediction models of ECOPAC (e.g., FORSUM, VEGSUM). |
| Forval | Forage value indexes are stored by plant species to denote their value as forage for various animal species by season of use. This database is accessed by the FORAGE program of ECOPAC. |
| Model Parameter | Parameters needed to model plant succession in the VEGSUM and FORSUM programs of ECOPAC are stored in this database. |
| Location | Location information concerning populations of selected species in the Sensitive Plant and Flora databases are stored in this database. |
| Weed* | Important autecological, control, and treatment information for weed species in Flora are stored in this database. |
| Rehab* | Reestablishment information concerning species response to various types of major disturbance are stored in this database. |
| Fauna | Complete species list of all mammals, reptiles, and amphibians in the nortern Rocky Moutains. Contains general distribution information and scientific and common names. This database is accessed by the WILDCOV and WILDHAB programs of ECOPAC. |
| Species Abundance | Predicted and measured abundance values for selected species of the Fauna database. |
| Habitat | General information of habitat usage for selected species of the Fauna database. |
| Structure Value* | Detailed information of habitat usage by structural stage and canopy closure for selected species of the Fauna database. |
| Cover Type* | Value of various plant community cover types to selected species of the Fauna database. |

* Indicates databases that are being developed and are not available for distribution at this time.

## ECOPAC Analysis Programs

ECOPAC (fig. 3) is a collection of computer programs and models that are directly linked to the ECODATA plot sampling databases for data input. This software package was developed to perform efficient, standard ecological analyses for various resource specialists (e.g., ecologists, soil scientists, range conservationists, and wildlife biologists). The analysis programs of ECOPAC contain current, literature-based, state-of-the-art, computational algorithms that are updated as new research results become available.

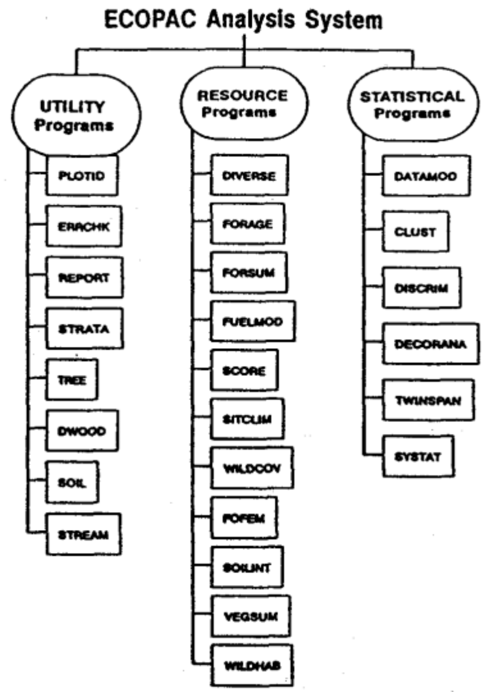


**Figure 3--Diagram of ECOPAC analysis software stratified by three basic types of program applications.**

Most of the ECOPAC programs were written in FORTRAN 77 language by the authors of this paper for execution on the Data General minicomputer and IBM-compatible microcomputers. Each program is run interactively by the user through a set of standard menu queries that specify details of program analysis and execution. Output may be printed directly to a line printer, stored in external disk files, or used to populate appropriate fields of the ECOCLASS database system.

ECOPAC is divided into three modules that contain programs of similar analysis purpose: UTILITY, RESOURCE, and STATISTICAL (fig. 3). The UTILITY module (table 3) is composed of programs that summarize plot-scale information and perform basic data analyses. Commonly used UTILITY applications include: report writing (REPORT), error checking (ERRCHIn, and plant community analysis (STRATA). The STRATA, SOIL, and STREAM programs of the UTILITY module may also be used to develop ecological classifications and populate ECOCLASS databases with appropriate information for future environmental analysis.

**Table 3--ECOPAC Utility Programs**

| Program name | Description | Common application | Database inputs |
|---|---|---|---|
| PLOTID | Creates macroplot files for input to all ECOPAC programs | Plot grouping | All databases |
| ERRCHK | Checks for logic, boundary, and syntax violations in data | Error scanning | All databases |
| REPORT | Prints summary reports and performs reduced statistical analysis | Report writing, monitoring, and analysis | All databases |
| STRATA | Vegetation and site classification package that computes summary tables and statistics for plot groupings | Vegetation classification, spectral imagery analysis, ecological classification | General, all vegetation cover databases |
| TREE | Checks for logic, boundary, and syntax violations, produces violations and summary reports of Tree Data | Error scanning, report writing | Tree |

## Table 3--ECOPAC Utility Programs (continued)

| Program name | Description | Common application | Database inputs |
|---|---|---|---|
| DWOOD | Checks for logic, boundary, and syntax violations; produces summary reports of measured down wood data | Error scanning, report writing | Measured down wood |
| SOIL* | Soil classification and horizon aggregation package that computes summary tables and statistics for soil pedon and horizon groupings | Soil classification, horizon summaries, edological classification | All soil databases |
| STREAM* | Stream type classification package that computes summary tables and statistics for stream reach groupings | Stream classification | All stream databases |

* Indicates programs that are being developed and are not available for distribution at this time.

The RESOURCE module of ECOPAC (table 4) contains programs that are frequently used in calculating the value of a plot (or group of plots) for a particular resource use. For example, the FORAGE program is commonly used in range analysis to calculate the forage value of a site or vegetation type for a particular animal species by seasons of use. The WILDCOV program may be used to determine the habitat suitability of different vegetation types through calculation of security and thermal cover indices. Vegetation succession models are also included in the RESOURCE module to facilitate resource response prediction (Keane 198. These models predict plant species coverages after various types of disturbance based on empirical (FORSUM) and expert systems (VEGSUM) techniques.

## Table 4--ECOPAC Resource Programs

| Program name | Description | Common application | Database inputs |
|---|---|---|---|
| DIVERSE | Computes diversity measures for plots or plot groupings | Diversity evaluation and monitoring | General, all vegetation cover databases |
| FORAGE | Computes the forage value of a plot (or group of plots) for animal species by season of use. Also calculates plant species composition by user-specified usage levels | Rangeland and wildlife life habitat evaluation | General, all vegetation cover databases |
| FORSUM | Predicts the cover of plant species after various types of disturbance | Wildlife, range, and forest planning | General, plant composition |
| FUELMOD | Summarizes fuel characteristics of a plot (or group of plots) and computes fire behavior for various weather scenarios | Fire and fuel planning | General, plant composition, down wood |
| SCORE | Rates floristic similarity of a plot to a user-specific community type (e.g., potential natural community) | Range and forest planning | General, all vegetation cover databases |
| SITCLIM | Predicts the climate of a plot using weather station input and various site variables | Ecological research, land use planning | General, plant composition |
| WILDCOV | Computes measures of wildlife cover (e.g., security, thermal) for a plot or group of plots | Wildlife habitat evaluation | General, all vegetation cover databases |
| FOFEM | Predicts the first-order fire effects after a natural or prescribed fire | Fire and fuel planning | General |
| SOILINT* | Calculates interpretations for a soil pedon or group of pedons (e.g., engineering properties, limitations) | Watershed planning | General, all soil databases |
| VEGSUM* | A vegetation succession model based on expert systems technology that predicts plant species abundance following disturbance | Land use planning | General, all soil databases |

**Table 4--ECOPAC Resource Programs (continued)**

| Program name | Description | Common application | Database inputs |
|---|---|---|---|
| WILDHAB* | Computes habitat capability and suitability for selected wildlife species | Wildlife habitat evaluation | General, all vegetation cover databases |

* Indicates programs that are being developed and are not available for distribution at this time.

The STATISTICAL module (table 5) contains a series of programs used in the quantitative .analysis of plot-scale data. Most of these programs are commonly used in ecological research and are of particular value to the development and testing of ecological classifications (e.g., TWINSPAN, DECORANA--Hill 1979a and 1979b). The DATAMOD program provides flexibility to the analysis system by reformatting data for different types of statistical software packages (e.g., SPSSX, SAS, CANACO).

**Table 5--ECOPAC Statistical Programs**

| Program name | Description | Common application | Database inputs |
|---|---|---|---|
| DATAMOD | Reformats plot data for analysis by statistical software packages (e.g., SAS, SPSSX) | Ecological research, predictive modeling | All databases |
| CLUST | Cluster (or group) plots based on site, soil, or vegetation data similarities | Ecological classification | General, soil databases |
| DISCRIM | Performs discriminant analysis of site, soil, and vegetation data | Classification validation, map extrapolation | General, soil databases |
| DECORANA | Performs detrended correspondence analysis of plot site and vegetation data | Ordination, vegetation classification | General, all vegetation cover databases |
| TWINSPAN | A two-way, indicator species analysis package that groups plots into similar types based on plant species richness and abundance | Vegetation classification | General, all vegetation cover |
| SYSTAT | A statistical analysis package that operates on the Data General System. Data run through the DATAMOD program can be analyzed by this software | Basic statistical analysis | All databases |

## ECOCLASS Databases

The ECOCLASS Database System (table 6) is a collection of relational database tables (ORACLE 1989) that store information for a user-specified classification of potential or existing vegetation, soil type, or stream type. The design employed in ECOCLASS development is generic to ensure that new types of ecosystem classification can be accommodated as they become available (e.g., lake types, and airshed types). ECOCLASS records (i.e., classifications) are primarily used to quantify taxonomic components and resource values of uniquely labeled polygons (e.g., vegetation stands) or map unit delineations (e.g., ecological units). This database system is directly linked to the Polygon Sampling Method databases of ECODATA (fig. 1).

**Table 6--ECOCLASS Databases**

| Database name | Description | Sources of data |
|---|---|---|
| INFO | Stores general information for a classification. This database is always completed for a classification record | User-entered, ERRCHK, STRATA, SOIL, & STREAM programs |
| INTERP- Continuous | Stores site and resource value interpretations for a classification based on continuous variable information | User-entered, all UTILITY and RESOURCE Programs |
| INTERP- Categorical | Stores site and resource value interpretations for a classification based on categorical variable information | User-entered, all UTILITY and RESOURCE Programs |
| VEG | Stores basic plant species information for a classification (e.g., constancy, average cover) | STRATA Program |
| TREE | Stores basic tree information for a classification by tree species and size class | TREE Program |
| TIME | Stores successional sequence information for a classification | User-entered, VEGSUM Program |
| PLOT | Stores the record ID's of ECODATA plots used in classification development | STRATA, SOIL, and STREAM Programs |
| COMMENT | Stores any user-specified comments concerning in classification | User-entered |

The eight databases which comprise ECOCLASS are usually populated from the STRATA, SOIL, or STREAM programs of ECOPAC, if plot-scale data were used in classification development. The user may, however, enter data directly into any of these databases if their classification was developed from information other than ECODATA plots (e.g., published literature).

## Polygon Sampling Methods and Databases

The sampling methods and databases used to describe uniquely labeled polygons (e.g., vegetation stands and watersheds), stream reaches, or ecological mapping units are presented in table 7. The Polygon Database System consists of relational ORACLE database tables that can easily be updated as new polygon variables are identified in ecosystem mapping (e.g., watershed morphometric features). These databases provide the basic attribute files for map theme descriptions, and are accessed by the LANDPAC analysis programs of ECODATA (fig. 1) for various types of environmental assessment.

**Table 7--Polygon Sampling Methods**

| Sampling method | Description |
|---|---|
| Polygon Description Data | This method is used in all polygon sampling efforts. It provides linkages between other polygon and plot-scale sampling methods. Additionally, it provides fields for special codes the user may need to create for specific inventory objectives. Information from this sampling method is stored in four databases: Linkage (identifies the polygon or map unit being described), Sample (lists the sampling methods used in polygon inventory), Plot-Linkage (lists the ECODATA plots used in sampling), and Local-Codes (stores user-specified data for customized inventory needs). |
| Polygon Comment and Disturbance Data | Comments and disturbance history for a polygon or taxon (e.g., soil type) are recorded by this method and stored in two databases (Comments and Disturbance). |
| Polygon Vegetation Composition Data | This sampling method is used to describe the potential and existing vegetation types of a vegetation stand, stream reach, or ecological map unit. Linkages are provided to the ECOCLASS database system. |
| Polygon Soil Composition Data | This sampling method is used to describe the soil composition of a vegetation stand, stream reach, of ecological map unit. It is commonly used in combination with the Vegetation Composition Sampling Method in describing the soil composition of a landscape. Linkages are provided to the ECOCLASS database system. |

Table 7--Polygon Sampling Methods (continued)

| Sampling method | Description |
|---|---|
| Stream Type Composition Data | This sampling method is used to describe the stream type composition of a watershed, stream reach, or ecological map unit. Linkages are provided to the ECOCLASS database system. |
| Polygon Taxa Location Data | This sampling method is used to describe taxon location within a polygon. It is most commonly used to identify the relation between soil and vegetation taxa and selected environmental variables (e.g., elevation range, geology, slope position). |
| Polygon Vegetation Stand Linkage Data | This method provides basic descriptive data for a vegetation stand and is commonly completed in the office. Linkages are provided that facilitate integration of range, timber, soil, hydrology, wildlife and fisheries information for a stand from polygon-scale databases. |
| Stream Reach Linkage Data | This method provides basic descriptive data for a stream reach. Linkages are provided that facilitate integration of other polygon-scale databases to a specific stream reach. |
| Ecological Unit Data* | This method provides a basic description of the environmental features that characterize an ecological unit (e.g., elevation range, geology, dissection). It is designed to store data for ecological unit description that are not contained in the taxonomic databases (e.g., ECOCLASS and Soil Composition). |
| Ecological Unit Interpretations* | This database stores resource interpretations for an ecological unit. Such interpretations may be derived through summary of appropriate taxa composition (e.g., weighted average of vegetation taxa forage values) or by direct user entry of non-taxa specific map unit interpretations (e.g., mass-wasting potential). |

* Indicates databases that are being developed and are not available for distribution at this time.

The Linkage and Sample databases of the Polygon Description method (table 7) are always used in sampling. These databases describe the type of polygon (Linkage) and the types of sampling (Sample) employed in polygon description, and provide linkages to other polygon databases. The remaining sampling methods and databases of the polygon system are used according to specific inventory needs. For example, Soil, Vegetation, and Stream Type Composition Sampling Methods can be used to describe the composition of taxa within polygons. Their databases, in turn, are linked to the ECOCLASS system which facilitates specific descriptions of classification components and their resource values.

Additions to the polygon sampling methods listed in table 7 are continually being made to accommodate other resource mapping needs (e.g., watershed and fisheries). Consequently, this portion of the ECODATA system will probably experience considerable growth in the next few years.

## LANDPAC Analysis Programs

LANDPAC is a collection of analysis programs and data interface commands used in the spatial analysis of digitized polygon themes. An operational version of LANDPAC is not available for distribution at this time; however, a prototype version will be released for beta-testing later in 1993. The components of LANDPAC that are currently being developed include linkages to various polygon-scale, resource prediction models (e.g., WATSED, habitat suitability models); user interface macros for efficient access of the polygon attribute tables; user interface macros for simplified execution of DRIS, SPATIAL, and ARCINFO geographic information system software; expert systems linkages, and incorporation of various FORTRAN programs for landscape pattern and spatial analysis.

## CONCLUSION

ECODATA is a database and analysis system useful for ecosystem management. At its inception eight years ago (Keane et al. 1990), ECODATA was designed primarily for integrated vegetation inventory and monitoring. It has grown, considerably in the last few years and is now a critical component of the environmental effects analysis process of the Northern Region (fig. 4). The application of ECODATA in environmental effects analysis has been reviewed by Jensen and others (1991) who emphasize that the first step in effects analysis is to ensure resource managers use common terms and databases when characterizing and analyzing the ecosystems they manage. Consequently, the primary objective in ECODATA development has been the creation of a platform for efficient and consistent environmental assessment.

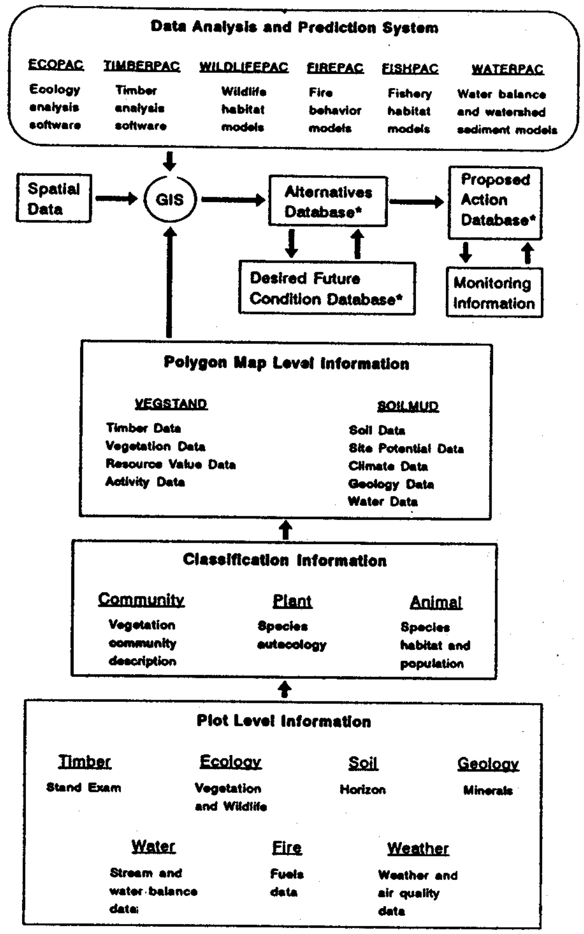


**Figure 4--List of some databases and analysts programs used by the USDA, Forest Service, Northern Region, in environmental effects analysis. Databases denoted by an asterisk are currently being developed.**

ECODATA is designed to be of use to not only the Forest Service but also to other government agencies, universities, research institutions, and environmental interest groups. Accordingly, representatives from some of these organizations have been actively involved in the development and testing of the ECODATA system (e.g., The Nature Conservancy, National Park Service, Bureau of Land Management, and various universities). About 40,000 ECODATA plots have been collected throughout the western United States by cooperative user groups and Forest Service personnel. Such standard data are critical to broad-scale landscape evaluations (Bourgeron et al. 1993).

Interest in the ECODATA database and analysis system has increased tremendously in the last few years and will undoubtedly continue as the philosophy of ecosystem management is fully implemented into the stewardship of public lands. Detailed descriptions (Jensen et al. 1992) and computer software for the ECODATA database and analysis system are available from the authors on request.

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## SECTION 4 - CASE STUDIES

### Summary of Case Study Papers

This section presents four land management planning case studies that use various aspects of the landscape ecology principles, sampling design, and data analysis methods described in .sections two and three of this document. These case studies are provided to demonstrate how landscape ecology concepts can be efficiently used in land evaluation. The techniques presented in these papers represent evolving approaches to ecosystem analysis that will likely change over time. Accordingly, the ideas presented in this section represent a starting point for ecosystem-based land evaluation, the foundation on which ecosystem management is implemented.

The first paper of this section (Engelking et al.) describes techniques for assessing the value of conservation areas at regional scales. These authors conducted assessments of representativeness (the degree to which the natural range of variability in a region is represented in a conservation area) in three different study areas (Gray Ranch Preserve in southwestern New Mexico, Yampa River Basin in northwestern Colorado, and the Greater Yellowstone Ecosystem). The following variables were used in describing the representativeness of each study area: climate and ecoregions, biological-physical environments, floristic elements, and vegetation patterns. The sampling designs (GRADSECT sampling) and analytical techniques presented in this paper are applicable to most conservation planning efforts that promote the maintenance of ecosystems and biodiversity.

Engelking and others state that the criteria land managers typically use in selecting areas of land for conservation emphasis (e.g., significant, high quality, and important) are too vague. The authors suggest that measurable criteria such as rarity, diversity, and representativeness are of much greater value to ecosystem management and should be increasingly used. These authors demonstrate how such criteria can efficiently be used in the evaluation and design of conservation areas.

The second paper of this section (O'Hara et al.) describes how landscape ecology theory may be incorporated into integrated resource analysis. Two case studies presented in this paper demonstrate extensive (Elkhorn planning unit of the Helena National Forest) and intensive (North Flints planning unit of the Deerlodge National Forest) approaches to integrated resource analysis. In the extensive method, broad-scale ecological mapping units (i.e., land type associations, see Bailey et al., this document) were used to describe areas with similar ecosystem processes (e.g., fire, erosion, and herbivory) and historical vegetation pattern. The historical vegetation cover type ranges identified by land type associations were contrasted against current cover type maps of the area to identify declining habitats (e.g., riparian areas and grasslands) and potential for their restoration. Such information was used to develop "desired future condition" strategies by land type association which were implemented through various projects under a single National Environmental Policy Act (NEPA) document. The wide range of activities prescribed through this single landscape-scale analysis provided improved efficiency over traditional analyses; however, more detailed analysis of vegetation pattern was necessary for project implementation.

The intensive-analysis approach described by these authors used historical stand-scale maps of vegetation pattern in analysis. Such maps were constructed for 1880 (the year of the last major stand replacement fire), 1950 (the period before major timber harvest), and 1991 (the current vegetation pattern as influenced by recent timber harvest and fire suppression). This information was used to develop desired future conditions for the analysis area, which emphasized maintenance of natural processes and vegetation patterns to the extent possible given the altered landscape. The disturbance patterns and patch sizes identified by historical vegetation map themes were very useful in quantifying realistic desired conditions for the analysis area. The detail employed in this analysis facilitated relatively precise spatial and temporal treatment scheduling; however, coarser-scale analyses may still be required to meet regional and forest conservation strategy goals.

The third paper of this section (Hann et al.) provides an example of how ecosystem characterization and evaluation techniques may be incorporated into forest plan implementation strategies. The authors present an intensive evaluation of ecosystem process (fire) and composition (current and historical vegetation patterns) for the Trail Creek Drainage of western Montana. Hann and others demonstrate how an understanding of the natural variability in ecosystem processes and composition is useful in selecting target stands for forest plan implementation. The authors also suggest how such information may be used in the design of projects to meet forest plan goals and objectives.

The final paper of this section (Shlisky) demonstrates how landscape ecology principles can efficiently be used in the development of restoration plans for altered landscapes. The Umatilla National Forest restoration project used a landscape analysis and design process to evaluate current ecological conditions of the North Fork John Day River Basin against natural variability ranges. Such information was used to identify watershed restoration project priorities. The analysis identified the following elements for the river basin: landscape structure and pattern (patch types, corridors, and matrix); landscape flow phenomenon (water, fire, and wildlife); ecosystem functions that maintain landscape flows (production and nutrient cycling); natural plant succession; and disturbance processes. This information was used to prioritise watersheds within the river basin for restoration. Departures of key landscape structures and functions from historical "natural" conditions were used in assigning restoration rankings. Restoration "tools" were then described for the study area, and their effects on accomplishing restoration objectives were contrasted.

The methods presented in this paper provide a framework for the development of ecologically based management objectives, which may be further refined by social, economic, and forest plan objectives. The author suggests that an understanding of ecological function and structure should be included at the forefront of planning and project prioritization, instead of being treated solely as a mitigation concern, as has been done in the past.

# Regional Conservation Strategies: Assessing the Value of Conservation Areas at Regional Scales

L.D. Engelking, H.C. Humphries, M.S. Reid, R.L. DeVelice, E.H. Muldavin, and P.S. Bourgeron

## ABSTRACT

Assessment of representativeness (the degree to which the natural range of variability in a region is represented in a conservation area) was conducted for three case studies as part of the landscape evaluation and ecosystem characterization process. The three study areas differed in site characteristics, existing information available, length of time for the study, and conservation goals. Determination of regional climate and ecoregion characteristics for each study area provided land managers with a regional context for assessing representativeness at the landscape scale. Ecologically relevant environmental variability of the study areas was characterized for two purposes: to provide a basis for gradient-oriented sampling designs, and to determine which environmental characteristics of a region are contained within particular conservation areas. A floristic analysis provided information for managers about general adaptations of species to climatic features. Quantification of vegetation patterns within and among communities and of biotic-abiotic relations provided a picture of the representation of regional vegetation patterns in an area and the environmental constraints under which the biota operate.

## INTRODUCTION

The long-term sustainability of regional and continental patterns of biotic elements (species and communities of plants and animals) and their environment must be ensured by using appropriate conservation strategies. Conservation areas include traditional core areas (e.g., private preserves, research natural areas, and wilderness areas) as well as private and public land managed for both conservation and natural resources. The selection of conservation areas and of appropriate management strategies is part of the general process of landscape evaluation and ecosystem characterization (Bourgeron et al. 1993a).

Conservation evaluation of areas requires both a rationale and data for implementing the rationale (Austin and Margules 1986). The rationale or set of conservation values includes rarity, diversity and representativeness. Biotic elements are considered rare if they occur in one or very few locations. Diversity refers to the number of biotic elements present. Representativeness means that a conservation area or system of areas should contain biota or biophysical characteristics, or both, that represent the range of natural variability found within a land class or region (Austin and Margules 1986). Assessing representativeness requires the same basic data as the process of landscape evaluation and ecosystem characterization for ecosystem management (Bourgeron et al. 1993b). Each step of representativeness assessment also requires the recognition of various temporal and spatial scales (Bourgeron and Jensen 1993, Turner et al. 1993).

Representativeness assessment uses the sampling strategies (Bourgeron et al. 1993b) and analytical tools and techniques (Milne 1993) of landscape evaluation and ecosystem characterization. In practice, however, gaps in biological and biophysical data may seriously restrict the application of ideal landscape evaluation and ecosystem characterization procedures, thereby limiting the short-term ability to make scientifically based decisions about the location of conservation areas and the formulation of management policies (McKenzie et al. 1989). Furthermore, specific projects may require that only part of the landscape evaluation process be conducted. The objectives, availability of data, and time and resource constraints all need to be considered when a specific project is undertaken.

The objectives of this paper are: (1) to present the specific use of various landscape evaluation and ecosystem characterization procedures to assess the conservation value of large areas for three projects; (2) to discuss results from explicitly designed field surveys; and (3) to discuss the use of numerical analysis, geographic information systems, and generalized linear modeling using data with varying detail and extent.

## The Case Studies

Results from the following three case studies are presented and discussed:

- Gray Ranch preserve--The Gray Ranch is a 130 000 ha working cattle ranch located in the boot heel of southwestern New Mexico. The Nature Conservancy's major goals for the Gray Ranch project were: to assess its representativeness (i.e., how much of the regional biological and environmental variability is contained in the Gray Ranch), and to provide information to be used as part of a preliminary management plan. The study had to be conducted within three months, including fieldwork. There was no georeferenced database available (climate, soils, geology, vegetation map, etc.).

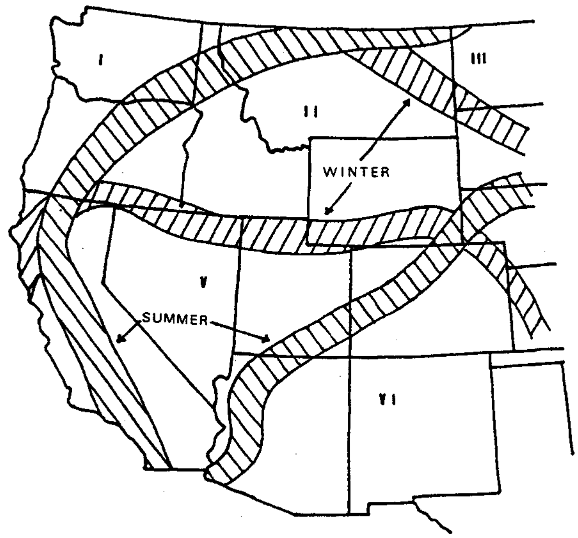
- Yampa River basin--The study basin, over 19 000 $km^2$, is part of the upper Colorado River basin and spans northwestern Colorado and a small area of south-central Wyoming. The major goal of the pilot project initiated by The Nature Conservancy was to assess the environmental variability in the Yampa River basin and the corresponding response of the riparian vegetation, and to use the resulting information as a baseline toward a basin-wide conservation strategy. The pilot project was to be conducted within one year, including fieldwork. Complete georeferenced databases were not available.

- The Greater Yellowstone ecosystem--The area includes the northwestern section of Wyoming and portions of south-central Montana and eastern Idaho. The goals in studying this area were to determine how adequately the Greater Yellowstone ecosystem encompasses the communities of the high-elevation spruce-fir forests distributed regionwide from Canada to the Southwestern United States, and to assess the representativeness of the bioenvironments of a research natural area system in a smaller area near the Greater Yellowstone ecosystem in the Blackfoot River region, using rule-based models and geographic information systems. Both projects were short term (2 weeks), were based on previous analytical work, and did not include de novo fieldwork. For the bioenvironment study, georeferenced databases were used.

Assessment of representativeness was conducted in the following areas: climate and ecoregions, bioenvironments-physical environments, floristics, and vegetation. When three or more of these five areas were investigated, the integration of results led to a final diagnosis of whether the goals of a project were achieved in a particular location.

## Climate and Ecoregions

The climate, involving such properties as atmospheric flow, photoperiod, temperature; and precipitation patterns, constrains the biotic elements of an ecosystem to those species able to tolerate the environmental variability within a given climatic region (Bailey et al. 1993, Neilson 1986). A first step in understanding climatic influences on ecosystems is the characterization of the regional climate in terms allowing the study of causal relations between climate and vegetation boundaries. Climatic regions provide land managers with a first approximation of broad-scale controls affecting ecosystems. Mitchell's (1976) climatic regionalization of the western United States provides divisions that appear to coincide with changes in vegetation patterns. Mitchell's six climatic regions are based on air mass boundaries (fig. 1) and are correlated with the distribution of biomes (Neilson 1986, Neilson et al. 1989).




Ecoregionalization is the process of delineating large land units that contain contiguous ecosystems assumed to be under the influence of the same higher order climate, elevation, soil conditions, etc. (Bailey and Hogg 1986, Bailey et al. 1993). These land units provide the regional context for evaluating the representativeness of conservation areas (Austin and Margules 1986). We used Bailey's (1980) map of the ecoregions of the United States in our analyses.

Gray Ranch--The Gray Ranch is contained in climatic region VI, defined by the summer monsoon boundary (Mitchell 1976), which separates the region from drier air overlying most of the interior west during summer (fig. 1). Most precipitation comes in convectional storms during the summer months and occasional rains in winter. This region differs from the others in its summer control of the climate.

The Mexican Highlands Shrub Steppe ecoregion (Bailey 1980) reflects the influence of a number of north-south oriented mountain ranges. The boundaries of the ecoregion were slightly altered to better reflect the actual distribution of the criteria used to define it. The mountain ranges alter regional climate through the effect of elevation, topography, and land masses. The ecoregion, located between the Sonoran desert on the west and the Chihuahuan desert on the east, contains high plains from 1200 to 2100 m and mountains up to 2750 m. The modified ecoregion covers more than 50 000 km$^2$.

The Gray Ranch is at the boundary between two precipitation regimes as characterized by a climate transect from Houston, Texas, to San Diego, California (Neilson 1987). The ecoregion is in a portion of the transect that is transitional in terms of the timing, frequency, and intensity of precipitation between conditions in the Sonoran desert to the west and in the Chihuahuan desert to the east (Neilson 1986, 1987).

Yampa River basin--The portion of the Yampa River basin in Colorado lies primarily within Mitchell's (1976) climatic region V (fig. 1). During summer, this climatic region is influenced by interior air in contrast to moist air to the south (see Gray Ranch discussion above). Winter is characterized by relatively infrequent intrusions of Pacific air compared to the area to the north of region V, where more westerly surface winds bring frequent Pacific air intrusions.

This general climate pattern is modified by the relatively high elevation of the study area. The study area lies within two highland ecoregions, the Wyoming Basin and the Rocky Mountain Forest ecoregions (Bailey 1980). The Wyoming Basin is characterized by plains interrupted by isolated divides and low mountains. The region is arid, with short hot summers and cold winters. The Rocky Mountain region is composed of rugged, glaciated mountains up to 4300 m in elevation. The climate is strongly influenced by the generally north-south orientation of the mountains and topographic influences on temperature and precipitation. Most precipitation falls in winter as snow.

Greater Yellowstone ecosystem--This ecosystem is in Mitchell's (1976) climatic region II (fig. 1). This region is affected by the frequent intrusion of Pacific air masses during winter but is under the influence of dry interior air during summer. The climate is a semiarid regime in which most precipitation falls in winter. Total precipitation is moderate but greater than on the plains to the west and east. The bases of the mountains receive 25 to 50 cm of precipitation, which increases to 100 cm at higher elevations. Climate is influenced by the prevailing westerly winds and the general north-south orientation of the mountain ranges, which result in east slopes that are much drier than west slopes. The Greater Yellowstone ecosystem is contained in the Rocky Mountain Forest ecoregion (described above) and encompasses the Douglas-fir Forest section of the ecoregion.

In conclusion, the climatic and ecoregional characterizations of the study areas allow the definition of areas with unique ecological sequences and landscape configurations. This scale provides a meaningful context for the analysis of community and ecosystem properties (Bourgeron et al. 1993a).

## Bioenvironments and Physical Environments

In assessing how representative an area is with respect to regional environmental variability, patterns in key environmental factors are established and their representation within conservation areas analyzed. Environmental factors used to define patterns of variability fall into one of three types (Bourgeron et al. 1993a): (1) indirect factors (e.g., landform, elevation, soil type, or geological substrate), (2) direct factors (e.g., temperature), or (3) resource gradients (e.g., available soil moisture or nitrogen). Combinations of environmental factors are termed bioenvironments when direct factors are used, and are termed physical environments when indirect factors are used. Indirect factors are used for two purposes: to develop ecological map units (Bailey et al. 1993); and as surrogate variables to estimate ecological regimes and develop maps of bioenvironments (Bourgeron et al. 1993a). Although in theory it is preferable to estimate ecological regimes from direct factors or resource gradients, in practice, the lack of available data, time, and resources make the use of any combination of the three types of factors necessary.

In addition to characterizing the environmental variability of the study areas, physical environments were used to drive a gradient-oriented transect (gradsect) sampling design (Austin and Heyligers 1989) for two of the areas, the Gray Ranch and the Yampa River. This sampling design provided a consistent, repeatable, quantitative, and cost-effective description of floristic variability in the area by including sampling along the full range of environmental variability. Standardized methods for collecting floristic and environmental data were used in implementing the gradsect sampling designs. Bourgeron and others (1993b, 1993c) describe the sampling design for the Gray Ranch.

Gray Ranch--Factors for describing environmental variability were based on past work conducted in the southwestern United States and chosen after consultation with a group of experts in vegetation, soil, geology, and hydrology. Because information on the spatial distribution of direct factors and resource gradients was not available, the following indirect factors were used: geologic substrate as an indicator of the nutrient regime; elevation as an indicator of precipitation and temperature; and soil suborder as an indicator of soil water availability. Definition of class intervals for each variable was constrained by the scale of available mapped information over the ecoregion, the need to keep the number of classes manageable, and the need to have ecologically meaningful classes. The resulting classes are shown in table 1.

**Table 1--Classes of Environmental Factors Used at the Gray Ranch**

| Geology | | Elevation | | Soil | |
|---|---|---|---|---|---|
| Class | Description | Class | Range (feet) | Class | Suborder |
| 1 | Quartz latite, andesite, and latite andesite | 3 | 3500-5000 | 1 | Orthids |
| 2 | Alluvium & bolson deposits | 4 | 5000-6500 | 2 | Argids |
| 3 | Basalt & basaltic andesite flows | 5 | 6500-8000 | 3 | Ustolls |
| 4 | Gila conglomerate | 6 | >8000 | 4 | Orthents |
| 5 | Quartz latite, rhyolite, and andesite | | | 5 | Fluvents |
| 6 | Intrusive rocks--grantitic, dioritic, rhyolitic, and andesitic | | | | |

To generate distributions of physical environments at the Gray Ranch and in the Mexican Highlands ecoregion, a regular grid was superimposed on map overlays of geology, elevation, and soil. The physical environment of each grid point was determined by identifying the combination of classes of each factor at that point. Different mapping scales were used for the Gray Ranch (1:100,000 with a grid spacing of 3422 m) and the ecoregion (1:500 000 with a grid spacing of 5501 m) because of the lack of regional map coverage at the 1:100 000 scale. Thirty of the 161 physical environments in the ecoregion were represented at the Gray Ranch. The twelve physical environments found both on the Gray Ranch and elsewhere in the ecoregion represented 40 percent of the total at the Gray Ranch and 7 percent of the total in the ecoregion. For these shared physical environments, no statistical difference in distribution among classes was found between the Gray Ranch and the ecoregion using a G-test (Sokal and Rohlf 1981). Consequently, the Gray Ranch can be considered a representative “snapshot” of the ecoregion for these physical environments. Eighteen physical environments were found only on the Gray Ranch. Although the land area of the Gray Ranch covers only 3 percent of the total surface area of the ecoregion, it encompassed 19 percent of the regional physical environments.

Yampa River basin--Environmental factors used to characterize the environmental heterogeneity of the Yampa River basin were chosen based on previous riparian work in the central Rocky Mountains and after consultation with a group of experts (Reid and Bourgeron 1993). In addition, a technical committee of land managers reviewed the decisions. The variables selected were elevation, geologic substrate, valley width-to-depth ratio as an indicator of the solar radiation regime, and drainage basin length, which is strongly correlated with stream discharge, as an indicator of the moisture regime. The gradsect sampling methodology was modified for this study to reflect the fact that perennial streams were treated as transects. No stream segment or geographic area was excluded from possible sampling. This approach maximized geographic replication and enabled the sampling team to visit most of the study area.

Values of each variable analyzed were divided into classes (table 2). Physical environments (e.g., elevation, geology, valley width-to-depth ratio, and drainage basin length) were mapped along perennial stream or river segments. The areal extent of each physical environment was estimated and the proportion that each environment occupied of the total drainage basin length of the Yampa River basin was determined. Numerical criteria were developed to allocate sample sites according to the proportional representation of the physical environments.

**Table 2--Classes of the Environmental Factors Used to Stratify the Yampa River Basin**

| Elevation | | Geology | | Vally W/D | | Stream discharge | |
|---|---|---|---|---|---|---|---|
| Class | Range (meters) | Class | Grouping | Class | Description | Class | Basin length (kilometers) |
| 1 | 1525-1830 | 1 | Sedimentary | 1 | Broad/shallow | 1 | 0-6.5 |
| 2 | 1830-2135 | 2 | Alluvium | 2 | Broad/deep | 2 | 6.5-16 |
| 3 | 2135-2440 | 3 | Igneous | 3 | Narrow/deep | 3 | 16-48 |
| 4 | 2440-2745 | 4 | Metamorphic | 4 | Vertical-walled | 4 | 48-145 |
| 5 | 2745-3050 | | | | | 5 | 145-435 |
| 6 | > 3050 | | | | | 6 | 435-1304 |
| | | | | | | 7 | > 1304 |

Once the number of sample sites per physical environment was determined, the actual location of each site was selected randomly. Sampling at individual sites allowed for capturing variation in environmental variables operating at a local scale (e.g., aspect or topography), as reflected by vegetation patterns. Plots were placed for sampling floristic (species cover) and environmental data (e.g., landform or distance to water) in each physiognomic vegetation type found at a site. A total of 80 sites were visited, with 111 individual plots collected over three months.

Greater Yellowstone ecosystem--The Blackfoot study (DeVelice et al. 1993) differs from other bioenvironmental work in that bioenvironmental aggregates were generated by using ecological knowledge rather than statistical properties (e.g., Mackey et al.1988). A rule-based model was developed based on two hypotheses: that species survival and growth area function of moisture, temperature, radiation, and nutrient regimes (Nix 1982); and that biotic patterns are mostly shaped by limiting factors (see Coughlan and Running 1989).

Soil moisture limitations are based on the fact that, in the area studied, most effective recharge of soil moisture results from snowmelt (Nemani and Running 1989, Weaver 1980). Estimates of temperature limitation are based on studies showing that species range limits are strongly influenced by temperature extremes and growing season length (Botkin et al. 1972, Woodward 1987). Soil limitations in soil fertility are based on nutrient supply indices for each soil type (Mackey et al. 1988, 1989).

Using Arc/Info geographical information systems software and a 1:250,000-scale digital elevation model, researchers recorded elevation, slope, and aspect at 15,500 points on an approximately 345-$m^2$ grid. For the same grid points, soil moisture holding capacity and soil fertility were derived from the 1:250,000 scale STATSGO database (STATSGO 1991). The rule-based model estimated the magnitude of soil water, temperature, and soil fertility limitations. Each point was then classified based on the model diagnosis. Bioenvironmental land units were derived by aggregating adjacent sample points belonging to the same class.

Maps of the bioenvironments were generated at about 12-ha resolution. The area occupied by each bioenvironment was determined for the total study area and for the system of research natural areas within it. The existing research natural area system was evaluated for how well the bioenvironmental diversity was represented. The salient conclusion of the study was that the research natural area system is primarily comprised of cold-limited bioenvironments. All other classes are poorly represented. The study provided land managers with an analysis of some of the critical ecological constraints under which local ecosystems function.

## Floristics

Floristic groups are conceived of as species assemblages with variable and often overlapping areal extent (McLaughlin 1986). Such areal types (Whittaker and Niering 1965) or floristic elements (McLaughlin 1986) classify species by patterns of geographic distributions. Analysis of areal extents or floristic elements provides information about the history and past ecological influences of an area, as well as migratory paths of species. This information is used to help land managers understand the scale of environmental processes that act on local biota. Floristic analysis is also used in representativeness assessment to determine whether an area includes all aspects of biodiversity patterns. This portion of representativeness assessment was conducted only at the Gray Ranch.

At the Gray Ranch, 305 species were found in the sampled plots. The areal extent of the species was characterized by using floristic subdivisions of McLaughlin (1986, 1989) and Whittaker and Niering (1965). Eleven categories were identified: Madrean, Chihuahuan, Sonoran, Rocky Mountain, Southwestern, Western, Plains, Temperate, Northern, Holarctic, and Latin American. Areal types were assigned to all species based on information in Kearney and Peebles (1960), Correll and Johnston (1970), Wagner (1977), and Whittaker and Niering's (1965) table. Although much floristic work has been done in the southwestern United States, phytogeographical categories are not standardized (McLaughlin 1986, 1989). Where there was consensus among the various sources, the determination of the previous authors was used. Otherwise the experience of two of the investigators (Moir and Muldavin) was used to make final decisions about species assignments to areal types.

The highest percentage of species found at the Gray Ranch belong to the Southwestern element (39 percent) and the second highest to the Madrean element (25 percent). As discussed above (Climate and Ecoregions), the Gray Ranch is at the boundary between two precipitation regimes. A majority of species found at the Gray Ranch (64 percent) either are distributed over an area where both precipitation regimes are encountered (Southwestern element) or are distributed in the boundary area itself (Madrean element). Species belonging to elements that coincide with only one of the two precipitation regimes are not well represented at the Gray Ranch (2 percent Sonoran element and 7 percent Chihuahuan element). Dependence on a particular precipitation regime appears to be a less successful strategy for plants in the area of the Gray Ranch. Floristic ties with floras of the southwest (including Chihuahuan, Madrean, Sonoran, Southwestern and Southern Rocky Mountains elements) are strong (77 percent of species at the Gray Ranch).

The floristic analysis indicates that plant species at the Gray Ranch must possess either wide tolerance in moisture requirements or the ability to occupy habitats that buffer them against the unpredictability of precipitation. Either characteristic would allow the distribution of a species across the two precipitation regimes. For example, the Fremont cottonwood Populus fremontii is found along streams which provide the species with a water source that is greater than and largely independent of the amount of precipitation at the site. Two yucca species Yucca elata and Y. bacata can photosynthesize throughout the year by tapping water that percolates to intermediate depths in coarse-textured soils. The yuccas' generalized root systems allow uptake of water whenever it is available, an adaptation to a highly unpredictable soil moisture environment.

## Vegetation

When assessing the representativeness of an area based on vegetation, three questions should be asked: How much of the regional patterns in vegetation occur in the area of interest? What parts of the gradients in intracommunity composition are present in the area? What parts of the environmental gradients that constrain each vegetation pattern are found in the area? Answering these questions requires knowledge of the pattern of distribution of communities and of the intracommunity compositional variation within a community's distribution.

In the three case studies, a draft of a regional hierarchical vegetation classification of the western United States was used that includes all available data on vegetation patterns (Bourgeron and Engelking 1993). This hierarchical system uses the UNESCO physiognomic classification (UNESCO 1973), adapted for the United States (Driscoll et al. 1984), at the highest levels and a floristic classification at the lowest levels.

The rationale for coupling physiognomy and floristic systems has been made by various authors (see discussion in Werger and Sprangers 1982) and is justified on evolutionary, ecological, and management grounds (e.g., Webb et al. 1969, Westhoff and Van der Maarel 1978, Werger and Sprangers 1982). The UNESCO framework (table 3) was chosen because it provides continental coverage, is for the most part internally consistent and ecologically meaningful, and may be modified during validation. Two floristic classification subdivisions, series and plant associations (sensu Westhoff and Van der Maarel 1978), were added under the UNESCO framework to characterize the biotic variability at landscape and site scales. Both classifications identify plant assemblages of any successional status by using floristic composition, physiognomy and habitat conditions (recurring combinations of environmental factors) at increasingly finer levels.

**Table 3--Numbers of Types within Levels of a Hierarchical Classification of Western U.S. Vegetation for the Gray Ranch and the Mexican Highlands Shrub Steppe Ecoregion**

| Level of classification | Number at Gray Ranch | Number in ecoregion |
|---|---|---|
| Formation class | 4 | 5 |
| Formation subclass | 8 | 9 |
| Formation group | 9 | 12 |
| Formation | 12 | 16 |
| Series | 26 | 71 |
| Plant associations | 45 | 215 |

Gray Ranch--Floristic and environmental information was collected at the Gray Ranch (Bourgeron et al. 1993c). Floristic data were classified using TWINSPAN (Hill 1979) and Ward's technique (Vllishart 1982). Results from the two techniques were compared to each other and to an initial field classification. As a result of using a stratified random sampling method based on physical environments (Bourgeron et al. 1993b, 1993c), many vegetation types were represented by only one or two plots. This representation is not~unexpected because the sampling design maximizes the placement of plots over the range of variability of the vegetation in response to the environment. The lack of replication of some types created difficulties, however, in identifying patterns through numerical techniques. Ordination of species and environmental factors was performed using canonical correspondence analysis (Ter Braak 1987). Ordination results were used to characterize the dynamics of the classification units and to draw the boundaries of the core and managed conservation areas.

Following correlation of the Gray Ranch classification to the existing regional classification framework, the representativeness of the Gray Ranch vegetation was assessed in terms of the types found in the ecoregion. As was the case for the assessment of physical environments, the Gray Ranch had a high percentage of vegetation patterns at all levels of the classification (table 3). At the finest level (plant associations), the Gray Ranch contained 21 percent of ecoregional diversity; however, it only occupied 3 percent of the total surface area. In addition, a high number of plant associations are documented only on the Gray Ranch.

Yampa River basin--Using the floristic and environmental variables collected, classification (using TWINSPAN and Ward's technique) and ordination (using canonical correspondence analysis) analyses were performed. The data set included high biotic and environmental variability as a direct consequence of the sampling design. A preliminary classification resulted in the definition of 38 riparian plant communities based on 102 plots. Five of the communities had not previously been described for the western United States. In addition, 13 communities defined in this study had not previously been described for Colorado, although they were known from Utah and southwestern Idaho (Padgett et al. 1989).

Ordination revealed complex patterns of variation in the riparian vegetation of the Yampa River basin. Overall, groups of species found at lower elevations and in marshy habitats at all elevations were separated from groups of species found in montane and subalpine forests (fig. 2a). The first axis of the ordination appeared to represent a complex environmental gradient most strongly influenced by elevation, but also by broader scale drainage basin characteristics such as drainage basin length, valley width, and drainage basin relief. The second axis was most strongly correlated with height above the water table. The second axis separated a group of plots representative of marshes dominated by sedge (Carex) species, with water tables at the surface from a group representative of low-elevation deciduous forests, dominated by boxelder maple (Acer negundo) or Fremont cottonwood, generally found on older alluvial terraces high above present-day water tables.

The data were stratified for further investigation of environmental correlations. For montane and subalpine willow communities, elevation and valley width-to-depth ratio were strongly correlated with the first axis of the ordination (fig. 2b). Willow communities in broad, shallow valleys at moderate elevations were clearly separated from those found at higher elevations in narrow, deep valleys. The second ordination axis was weakly correlated with north or west aspects. In this study, aspect is the downstream direction of the valley in the vicinity of a sample site. These correlations suggest that in narrower, deeper valleys at high elevations, north- or west-facing aspects become important variables that influence the distribution of plant species. This result is ecologically meaningful. North-facing valleys receive less solar radiation and therefore are cooler and retain snowfall longer in spring. West-facing valleys receive high amounts of solar radiation in summer and have the potential for locally warmer and drier conditions. It is likely that different species will be found in these local habitats, thereby resulting in different communities.

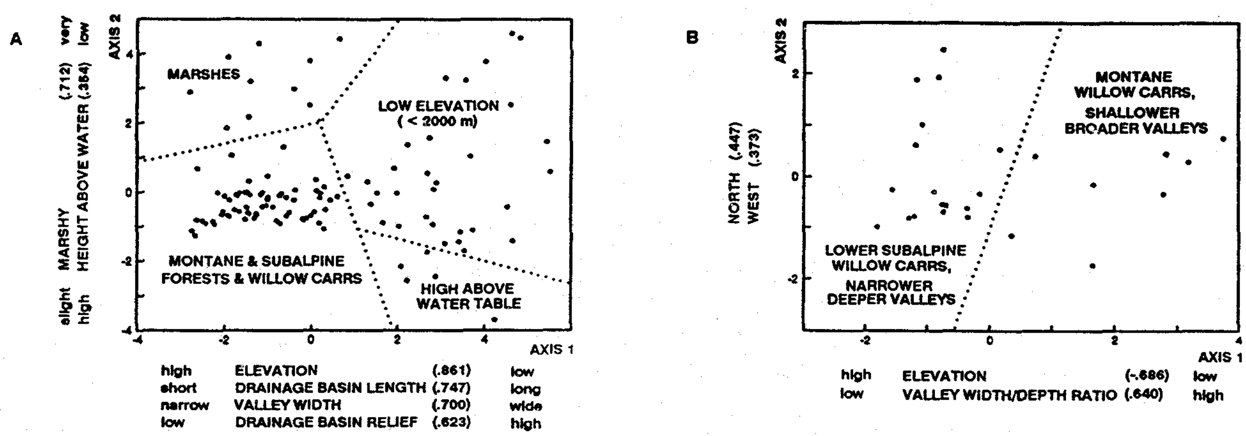



**Figure 2--Ordination of the Yampa River basin data using detrended canonical correspondence analysis of (A) all plots, and (B) upper montane and subalpine willow plots. Environmental variables with highest correlations with axes are shown. Values in parentheses are correlations.**

Uncommon plant species and communities were found in all segments of the strongest gradients influencing the patterns of riparian vegetation in this study. Therefore, a system of conservation areas established to protect all aspects of riparian vegetation should not focus on only one portion of a gradient. This study reveals that no single gradient is most important in explaining patterns in the distribution of riparian vegetation over the study region. For example, for willow-dominated communities, the influence of local valley aspect may be more important at high elevations whereas at moderate elevations, the shape of the valley (width-to-depth ratio) may be more important. Any system of representative conservation areas would need to represent these segments of environmental variability as well.

Greater Yellowstone ecosystem--A quantitative reanalysis of existing classifications led to the development of a regional classification of the Rocky Mountain high-elevation spruce-fir forests (Engelking et al. 1991). Numerical classification of data based on percentage of cover was conducted using Ward's technique. Ordination was performed with detrended correlation analysis (Hill 1979). Thirty high-elevation spruce-fir plant associations were identified over the region. Although their physiognomy is essentially constant (see Peet 1988), these forests exhibit regional and local patterns of change in species composition. A regional latitudinal gradient influencing the vegetation is correlated to a large extent with climatic regions (Mitchell 1976, Neilson 1986). Mesoscale climate effects reflecting orographic influences contribute to within-climatic-region landscape-induced variation in vegetation. A floristic latitudinal gradient reflects the influence of both local and regional processes.

Ten of the 30 plant associations occur in the Greater Yellowstone ecosystem. Three of these plant associations are widely distributed regionally and are characterized by the subalpine fir (Abies lasiocarpa) and grouse whortleberry (Vaccinium scoparium). Each plant association, however, differs in most other species present. The names of these regional plant associations have been abbreviated ABLA/VASC1, ABLA/VASC2, and ABLA/VASC3. The range of distribution of each of these three plant associations coincides to a large extent with a climatic region (Mitchell 1976) (fig. 1). The range of distribution of ABLA/VASC1 is centered in climatic region VI (central Rockies), ABLA/VASC2 in climatic region II (northern Rockies), and ABLA/VASC3 in climatic region I (Northwest). The Greater Yellowstone ecosystem includes the central portion of the range of distribution of ABLA/VASC2 and small peripheral portions of the range of ABL/VASC1 and ABLA/VASC3. The other seven spruce-fir plant associations of the Greater Yellowstone ecosystem have more restricted distribution and represent landscape-induced topoedaphic variations of the three plant associations controlled by the regional climate (Engelking et al. 1991).

Another aspect of representativeness assessment is determining which part of the regional intracommunity biotic variability is encompassed by the study area. Occurrences of the three regional types were assessed in terms of the regional biotic variability that they encompass. Table 4 compares the number of species (total and by life form) found over the entire range of distribution to the number of species found in the Greater Yellowstone ecosystem for all high-elevation subalpine fir forests together, and for each of the regional plant associations controlled by climate. The Greater Yellowtone ecosystem seems to have the most value as a core conservation area for ABLA/VASC2, which is at the center of its range of distribution there. In the Greater Yellowstone ecosystem ABLA/VASC2 contains up to half the species found within the regional range of distribution of the community regardless of life form. The Greater Yellowstone ecosystem would have value as a complementary area to other core conservation areas for ABLA/VASC1 and ABLA/VASC2, for it contains northern and southeastern occurrences of each, respectively. The Greater Yellowstone ecosystem contains less than one-third of the regional pool of species for these two types.

**Table 4--Total Number of Species and Number of Species by Life Form for Subalpine Fir in the Rocky Mountain Region (Alberta to New Mexico) and in the Greater Yellowstone Ecosystem (GYE)**

| | Total | Trees | Shrubs | Graminoids | Forbs | Others |
|---|---|---|---|---|---|---|
| Regional subalpine forests | 490 | 22 | 74 | 80 | 306 | 8 |
| GYE subalpine forests | 132 | 9 | 27 | 20 | 75 | 1 |
| Regional ABLA/VASC1 | 168 | 14 | 40 | 15 | 94 | 5 |
| GYE ABLA/VASC1 | 32 | 4 | 2 | 5 | 21 | 0 |
| Regional ABLA/VASC2 | 191 | 17 | 42 | 21 | 106 | 5 |
| GYE ABLA/VASC2 | 83 | 6 | 20 | 11 | 44 | 2 |
| Regional ABLA/VASC3 | 157 | 11 | 32 | 19 | 94 | 1 |
| GYE ABLA/VASC3 | 50 | 5 | 14 | 6 | 25 | 0 |

## Biotic and Environment Relations

Statistical modeling of biotic-environmental relations can be used to determine how representative the environmental conditions of a particular site are of the range of conditions to which a species responds. To illustrate this point, we selected a widespread western species, Gambel oak Quercus gambelii . The data used for this analysis consist of spatially referenced plots containing vegetation and environmental data measured on forested land in Arizona, New Mexico, and Colorado (Muldavin et al. 1990). Similar data collected at the Gray Ranch were added for a total of 883 plots. Presence or absence of Gambel oak and parent material were recorded for each plot. In addition, climatic attributes were estimated for each plot using a microclimate simulation model, MTCLIM (Hungerford et al. 1989). Climatic attributes included annual precipitation, precipitation in each of the four quarters of the year, average temperature, maximum July temperature, minimum January temperature, and annual radiation.

A statistical model predicting Gambel oak response to climatic attributes and parent material was developed using a powerful class of regression models (Bourgeron et al. 1993b) called generalized linear models (McCullagh and Nelder 1989). The model calculated a probability of presence of the oak for each plot location based on its environmental characteristics. Environmental factors found to be important in determining Gambel oak response were parent material, maximum July temperature, and first quarter (January to March) precipitation.

A contour plot (fig. 3) for locations on the parent material category "other igneous rock," the category on which the Gray Ranch plots occur, shows that in general the probability of the occurrence of Gambel oak increases with increasing maximum July temperature and winter precipitation. The model predicts a 33 percent chance of Gambel oak presence at the Gray Ranch, which is outside the region of highest probability of occurrence. This prediction is due to the relatively low winter precipitation at the Gray Ranch. These results indicate that the environment experienced by Gambel oak at the Gray Ranch is not likely to be representative of conditions required for its persistence.

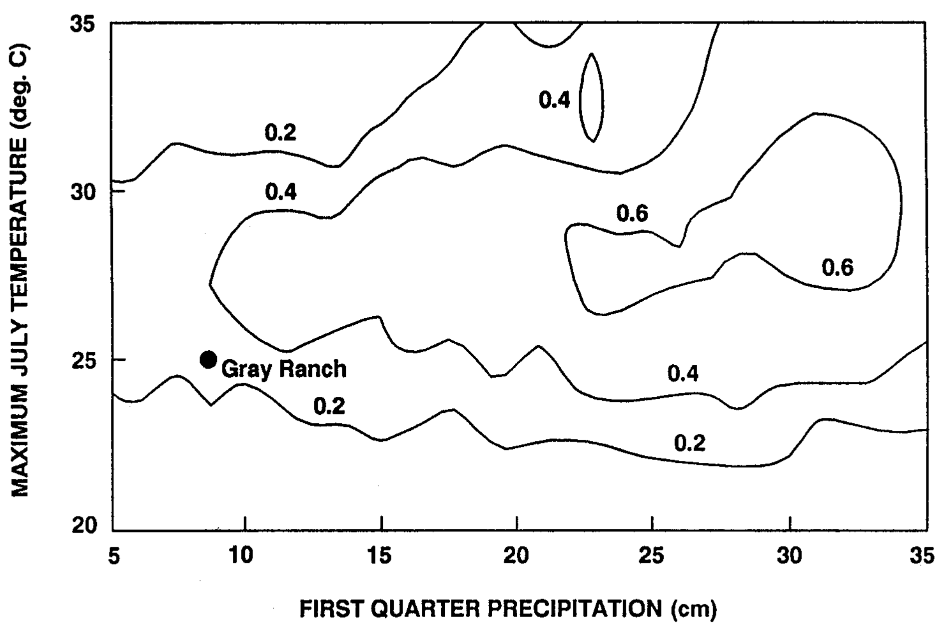



**Figure 3 - Contour plot of predicted porbabilities of occurrence of Gambel oak for locations on other igneous rock as a function of maximum July temperature and first quarter (January to March) precipitation for southwestern United States locations.**

Knowledge of a species' response to environmental conditions can provide valuable information in managing a site for the continued presence of the species. For example, model results suggest that a decrease in maximum July temperature could reduce the probability of continued Gambel oak presence at the Gray Ranch, while an increase in winter precipitation could increase its persistence probability. Managers could use this information to anticipate the effect of changes in climate on the species.

## Integration of Results

Gray Ranch--The methodology used to assess the representativeness of the Gray Ranch at several scales provided an understanding of the multiscale processes acting on the vegetation of the ranch. Plant species on the Gray Ranch must be adapted to broad-scale climatic conditions that include summer control of climate with highly variable timing and amount of precipitation. Topographic relief provides additional variability in climatic conditions at the landscape scale. The importance of climate in determining which species are represented at the Gray Ranch is reflected in the predominance of species with floristic affinities associated with broad tolerance in soil moisture requirements. Modeling results for Gambel oak placed the Gray Ranch within a context defined by maximum July temperature and winter precipitation in the southwest. Among forested locations measured, winter precipitation at the Gray Ranch was found to be low.

The analysis showed that the Gray Ranch captures a large amount of the biotic and environmental variability found in the Mexican Highlands ecoregion. The Gray Ranch was found to contain 21 percent of the plant communities and 19 percent of the physical environments in the ecoregion while occupying only 3 percent of the ecoregion's land area. The Gray Ranch ranks high as a conservation area under any of the three conservation values used: (1) rarity: it has a large number of unique physical environments and plant communities; (2) diversity: it has a large amount of diversity in biota (species and species groups) and physical environments; and (3) representativeness: the Gray Ranch appears to be representative of biotic and abiotic diversity patterns found in the ecoregion. Results from this floristic and vegetation analysis were used to define a core conservation area (no grazing) and managed conservation areas (controlled amounts of grazing). Preliminary recommendations also were made for reestablishing ecological processes disrupted by human activities, including fire in grasslands and woodlands, and increased stream flow regimes in riparian communities.

In light of the results, the sampling design and data collected were assessed as satisfactory. Improvement in the sampling design could have been achieved by using geographic information systems. After the study was completed, soil, elevation, geology, land use, and other maps were digitized. These databases will be useful for further field work at the Gray Ranch. Better geographic information systems regional databases for the diversity variables would have helped with the accuracy of the assessment as well. The lack of such databases emphasizes the general need to fill gaps in soil, climate, and vegetation databases (Bourgeron et al. 1993a, Stewart et al. 1989). Finally, if time and resources had not been limiting, the number of plots sampled should have been determined by the amount of environmental variability defined by the gradsect design.

Yampa River basin--The riparian vegetation of the Yampa River basin was largely unknown when the pilot project was initiated. The representation of two ecoregions within the study area can be seen in the patterns of the riparian vegetation. In the first division of plots in the TWINSPAN classification analysis, a group of 33 plots corresponded closely with the portion of the study area that is part of the Wyoming Basin ecoregion. These plots were characterized by alkali-tolerant species and generally sparse vegetation cover, despite the proximity of the plots to streams and rivers. A second major pattern is the importance of the elevation gradient in all analyses performed. This fact relates to the rapid increase in elevation across relatively short distances in the Rocky Mountain ecoregion.

The patterns of riparian vegetation and environments within the study area were complex, as revealed by the gradsect sampling design employed. The large number of plant species (373) and communities (38) identified by the quantitative analysis of samples is an indication of this complexity. Forty-seven percent of the riparian plant communities defined in this pilot project had not been previously described for Colorado. Examples of a large percentage of previously known communities were identified. Exploration of the relations between vegetation patterns and environmental gradients will allow future management and conservation decisions to be made in light of the distribution patterns of plant species and communities across segments of these gradients.

General after-the-fact comments on the work conducted in the Yampa River basin are similar to those for the Gray Ranch study. The use of geographic information systems databases would have helped tremendously. More plots would have contributed to a more precise documentation of the patterns of vegetation variability. The sampling design and data collected were, however, viewed as largely satisfactory, though the limitations of the study were acknowledged (Reid and Bourgeron 1993). After the pilot project was completed, adjustments were made in the sampling design and in the measurement of key variables to improve the overall accuracy and precision of the riparian inventories and interpretation (Kittel and Lederer 1993).

Greater Yellowstone ecosystem--A goal in determining the representativeness of the Greater Yellowstone ecosystem was to assess how adequately the ecosystem encompasses the range of variation of the high-elevation spruce-fir complex that is distributed regionwide from Canada to the southwestern United States. The Greater Yellowstone ecosystem contains 10 of 30 spruce-fir plant communities found in the Rocky Mountains. In terms of intracommunity variation, the Greater Yellowstone ecosystem is most representative of only one regionally distributed type of plant association (ABLA/VASC2). The range of distribution of this plant association is centered in the climatic region in which the Greater Yellowstone ecosystem is located. Occurrences of two other regionally distributed plant associations also found in the Greater Yellowstone ecosystem would complement core conservation areas for these plant associations located in their respective climatic regions. The Greater Yellowstone ecosystem is representative of the remaining seven plant associations in that it contains the landscape-induced topoedaphic variations that control their distribution.

The Blackfoot River bioenvironmental study contributed two insights. First, an ecologically meaningful picture of the study area in terms of moisture, temperature and fertility limitations of soils was provided to land managers. Second, the existing research natural area system, although extensive, was not found to be representative of all bioenvironments in the area.

The assessment of the Greater Yellowstone ecosystem in terms of the spruce-fir forests would have been more detailed if site-specific plot data of the vegetation and the environment had been available across the entire region. A gradsect sampling design of the spruce-fir forests contained in the Greater Yellowstone ecosystem would also help ensure that all aspects of the floristic and environmental variability of these forests are documented. These two points emphasize the need for standardized sampling procedures (Bourgeron et al. 1993b) and ecological databases (Jensen et al. 1993). The Blackfoot River study needs to be followed by field verification of the model outputs and by further refinement of the ecological models used to generate them.

## CONCLUSIONS

When making conservation recommendations, decisionmakers (private or governmental) are faced with the basic task of selecting areas of land that are most likely to represent the biological variability of large, poorly studied regions (Austin and Margules 1986). Conservation areas typically must qualify as being “significant,” “high quality,” “outstanding,” and “important.” These terms are too vague to be useful in the context of sustaining ecosystems and biodiversity patterns. Criteria such as rarity, diversity, and representativeness provide a yardstick by which abstract terms may be defined in a quantified and repeatable manner (Mackey et al. 1989). Furthermore, representativeness assessment leads to an understanding of the context for what is contained in various land areas. It requires the definition of regional patterns of environmental and biological variability, as well as of the key interactions among the biota and their habitats.

Representativeness assessment is a subset of ecosystem management. It includes an explicit, repeatable methodology for use in landscape evaluation and ecosystem characterization and analysis. Ecosystems are complex units subject to an integrated group of processes operating at various spatial and temporal scales. Managers ideally should understand each factor that affects the ecosystem and its biota before making management decisions. Such knowledge is acquired through the assessment of climatic and ecological regions, physical and bioenvironments, floristic elements, and vegetation patterns of the region.

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# Applying Landscape Ecology Theory To Integrated Resource Planning: Two Case Studies

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## ABSTRACT

Planning teams from two National Forests in Montana recently used landscape ecology theory in pilot resource analysis. These two case studies provide an interesting contrast in approaches even though the studies are subject to similar abiotic, biotic, and social factors. The Elkhorns Landscape Analysis Area on the Helena National Forest used a relatively extensive approach across a large, diverse area. The North Flints Landscape Ecology Unit of the Deerlodge National Forest used a more intensive approach across a smaller analysis area. The difference in intensity between the two studies led to a more detailed description of vegetation structure on the North Flints Unit, whereas the Elkhorns analysis encompassed a larger scale and involved several scales of analysis. Neither analysis examined scales beyond the National Forest (i.e., the regional context of analysis units was not assessed). The Elkhorns analysis was a cooperative multiagency effort that included the Bureau of Land Management and the Montana Department of Fish, Wildlife, and Parks, which cooperatively manage land and animals within the Elkhorns unit.

## INTRODUCTION

The evolving science of landscape ecology represents an enigma for many agencies and corporations attempting to integrate landscape ecology concepts into land management plans (Golley 1993). Pressures to maintain productivity, enhance biodiversity, and produce multiple resource values have risen to new heights, but solutions to these complex management problems are scarce. Landscape ecology represents one solution. Landscape ecology has been, however, little more than a concept with no guidance for implementation. As a concept, landscape ecology appears to be a sound basis for analyzing and eventually managing natural resources. Landscape ecology focuses on the structure, function, and change in the component ecosystems over time (Forman and Godron 1986). In a management context, this approach allows analysis and management of broad areas by using wide temporal scales and wide ranges of functions.

Two pilot or prototype projects were undertaken to develop methodologies and procedures for analysis and implementation of landscape ecology principles at landscape scales for the Northern Region of the USDA Forest Service. The Elkhorns Landscape Analysis Area is located within the Helena National Forest and some adjacent ownerships, and the North Flints Landscape Ecology Unit is in the Deerlodge National Forest. The projects are on neighboring National Forests in western Montana and thus have many similarities as well as some distinct differences, both in the area analyzed and analysis procedures used (fig. 1). The two landscape analysis areas are situated only about 50 air miles from each other but fall on opposite sides of the Continental Divide. One study area is nearly three times the size of the other, but similarities exist in the plant and animal communities.

This paper describes how the two projects, or case studies, attempted to incorporate landscape ecology theory into their analyses and plans. Comparing the approaches should allow extrapolation to other management efforts. Rather than focusing on the analysis areas or on the results from the analyses, this paper concentrates on the approaches used, and attempts to present a guiding framework for similar efforts in other regions.

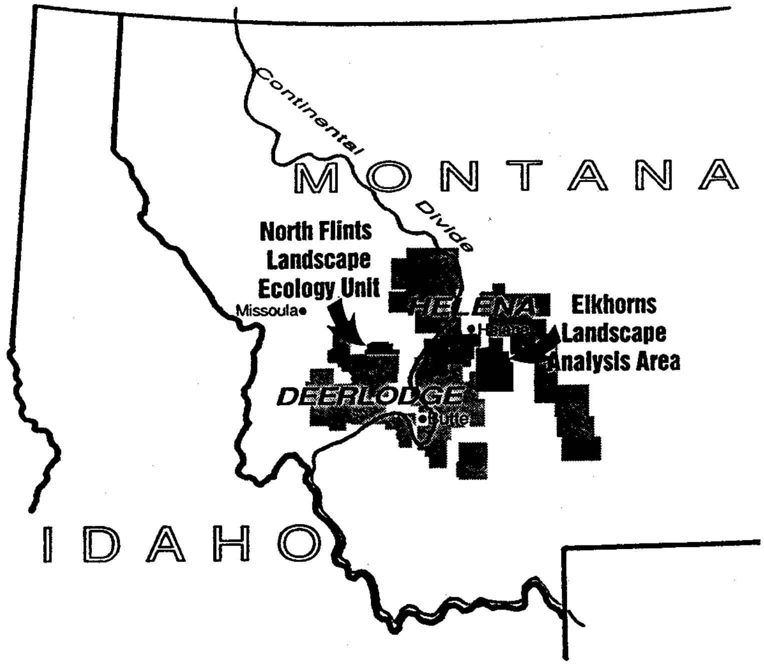


**Figure 1--Western Montana and locations for the Elkhorns Landscape Analysis Area and North Flints Landscape Ecology Unit.**

## Elkhorns Landscape Analysis Area

The Elkhorn Mountains Landscape Analysis Area is a 130,000-acre (52 600-ha) area in the Helena National Forest (fig. 1). This analysis area is the smallest of four landscape analysis areas identified in the Helena National Forest that range up to about 300,000 acres (120 000-ha). The Elkhorn Mountains are managed cooperatively by the Helena National Forest,, the Bureau of Land Management, and the Montana Department of Fish, Wildlife, and Parks. The analysis area encompasses the entire Elkhorn Range, which is located east of the Continental Divide and southeast of Helena, Montana (fig. 1), and is an island-type mountain range surrounded by valleys. Vegetation ranges from ponderosa pine (Pinus ponderosa) communities on the west side, to subalpine fir (Abies lasiocarpa) and lodgepole pine (Pinus contorta) in the high elevation central areas, to interior Douglas-fir (Pseudotsuga menziesii var. glauca) , true grasslands, and sagebrush (Artemisia tridentata var. vaseyana) communities on the east side.

Including multiple ownerships within the analysis provides an example of cooperative planning among agencies at landscape scales. In the Elkhorns study, a memorandum of understanding provided the impetus for cooperation between the Helena National Forest, the Bureau of Land Management, and the Montana Department of Fish, Wildlife, and Parks, and demonstrated that all three agencies were committed to the concept of ecosystem analysis. Cooperation has included shared databases, collaboration on analysis and documentation, and the design of future wildlife studies. The agencies share a common vision of the desired condition of the area because of the landscape analysis.

## Landscape Hierarchies

Developing hierarchical scales for analysis of landscape-level units represents one of the more difficult steps in constructing a landscape management plan. Because of the island-type nature of the Elkhorn Range and the relative isolation of this range from the nearby Rocky Mountains, the Elkhorn Range forms a logical landscape analysis area in a hierarchical framework. Within the Elkhorn Range, landtype associations were used jo group individual landtypes with similar features relating to geology, landform, landscape position, and potential vegetation. A landtype is a mapping unit with consistently repeatable landform features, geological parent materials, soils, and potential vegetation components (Bailey et al.1993, Wertz and Arnold 1972);

landtypes were mapped previously for the Helena National Forest. Eleven landtype associations occur within the Elkhorns Landscape Analysis Area (table 1).

**Table 1--Hierarchical Approach and Analysis Scales Used in the Elkhorns Landscape Analysis Area and North Flints Landscape Ecology Unit.**

| Elkhorns Landscape Analysis Area | Analysis scale | North Flints Landscape Ecology Unit |
|---|---|---|
| Helena National Forest | Regional | Deerlodge National Forest |
| Elkhorns Landscape Analysis Area (one of four areas in Helena NF) | Forest | North Flints Landscape Ecology Unit (one of 28 units on Deerlodge NF) |
| Ecological Landscape Units (four in Elkhorns Landscape Analysis Area) | Forest | Climax series or habitat type series (two in North Flints Unit) |
| Landtype associations (11 across Elkhorns Landscape Analysis Area) | Forest | |
| Landtypes/lifeform characterization | Project | Stand and disturbance area characterization |

Landtype associations were grouped into ecological landscape units to identify geographically and functionally connected ecosystems (see Hann et al. 1993). Ecological landscape units are, therefore, a grouping of landtype associations with similar potential vegetation, natural process regimes, and soil landform characteristics (table 1). Through delineation of areas with similar potential vegetation, disturbance patterns, soil, and landform characteristics, those ecosystems with predictable responses to natural processes such as fire, erosion, climatic regime, insects and pathogens, and herbivory were identified. Natural processes were defined as those occurring before the influence of Europeans. Four ecological landscape units were identified in the Elkhorns Landscape Analysis Area; these units ranged from about 30,300 to nearly 47,700 acres (12 200 to 19 300-ha). These ecological landscape units roughly corresponded to Forest Plan management areas.

## Natural Ecosystem Patterns and Processes

Identifying natural ranges of variability or "historical variability" (Swanson et al. 1993) was a priority for both the Elkhorns and North Flint analyses; however, the problem was approached in different ways. In the Elkhorn landscape analysis, the landtype association was the scale for describing ranges in natural vegetation, although these ranges were addressed by ecological landscape unit. Natural variability was typified by conditions existing before settlement by Europeans and included effects of Native Americans as natural processes. To ascertain the amount of information needed for describing natural conditions in the Elkhorns landscape analysis, wildlife was recognized as the resource requiring the most specific information. Wildlife information needs then defined vegetation information needs, which defined landtype and climate information needs (from landtype associations) as well as information needs regarding effects of pre-Eurpoean people on vegetation and wildlife.

Fire groups are a vegetation-dependent classification that indicate average intensity and frequency of fire and are assumed to describe natural (i.e., pre-European settlement) fire characteristics (Fischer and Clayton 1983). Average fire characteristics describe the likely disturbance intervals and types that initiate vegetation regrowth. For the Elkhorns landscape analysis, each landtype included only a few fire groups. The ecological landscape units, which consist of two to three similar landtype associations, therefore, are descriptors of average fire characteristics.

Vegetation was described for each landtype association in the Elkhorns Landscape Analysis Area. Because landtypes are based in part on potential vegetation, similar vegetation communities are found in each landtype. Pre-European vegetation conditions, therefore, were characterized in general terms: habitat types within landtype associations were mapped using photointerpretation, but vegetation development across time was not reconstructed. Instead, historical vegetation describes ranges of probable vegetation community structure and pattern based on natural processes before influence by European settlement, not specific structures and patterns at a given point in time.

Within each ecological landscape unit, the range of natural vegetation and existing vegetation was described for forest, grassland, and riparian vegetation types. Habitat type classification, based on potential climax vegetation (Pfister et al. 1977) and knowledge of influence of fire in habitat types (Fischer and Clayton 1983), was used for characterizing past composition of forest vegetation species. Structural characteristics of pre-European settlement stands were implied from information on fire frequency and intensity and fire scar data. Existing conditions were similar to natural vegetation conditions except for modifications in structure associated with fire suppression and domestic livestock grazing. Modifications of structure included fewer open stands, more single-storied stands, higher proportions of conifers, fewer aspen (*Populus tremuloides*), and more insect problems.

Observable changes in grassland-shrubland communities were the primary bases for analysis of pre-settlement and existing conditions. Changes in grassland communities since European: settlement differ with ecological landscape unit. For example, two ecological landscape units had the highest historic amounts of grassland-shrubland communities and experienced the greatest changes since settlement. These changes generally were the result of human settlement, fire suppression, and domestic livestock grazing. Changes included decline in spatial extent of grasslands, conifer and shrub invasion of grasslands, changes in grass species composition, soil erosion, declining productivity, and larger communities with less natural pattern diversity.

Presettlement and existing riparian conditions were characterized by landtype groupings based on geology and landform, and observable changes in the extent and structure of specific riparian plant communities were described. Riparian areas have generally declined in extent, function, and diversity due to grazing, fire suppression, erosion, and declines in some wildlife species (such as beaver (*Castor canadensis*)). These factors have changed stream channel morphology, altered species composition and structure, caused soil compaction, and exposed soil.

Natural ranges and existing conditions for soil, water, fishery, recreation, and wildlife resources also were assessed. These resources are not separate from vegetation resources, and clearly their condition relative to natural ranges of variability is tied to vegetation conditions and management. These resources were described in general terms for the entire Elkhorns Landscape Analysis Area rather than for each of the four ecological landscape units. Generally, these resources were found to exist in a state somewhat different than before European settlement. Grazing, mining, habitat losses, exotic species introductions, and loss of native species have led to modifications ranging from some soil conditions that are not reclaimable (such as mine tailings) to highly altered fish species populations.

The Elkhorns analysis reflected declining extents of grassland and riparian areas. Specific locations of riparian areas were described on the landscape; however, no other analysis of patterns was performed across the ecological landscape units. Instead, parameters were developed for describing the natural range of a given resource and were expressed as percentages or ranges, or both, for each ecological landscape unit. Ranges, for example, might describe size of grassland areas or percentages of Douglas-fir in certain mixed-species stands.

## Development of Desired Conditions

Desired future conditions generally involve various time frames, with ecosystem sustainability an ultimate goal. The desired conditions from the Elkhorns analysis were described as resource goals to be achieved in about a 50-year time frame. Statements describing desired conditions were designed to be specific enough to meet Forest Plan direction but flexible enough to provide some decision space for management.

Desired conditions were developed for all major biotic, abiotic, and social resources in the analysis area. Descriptions of desired future conditions were developed for each individual resource type. These descriptions were general in nature and did not address specific management unit area locations; for example, one goal for the entire landscape analysis area concerned restoring native fish and wildlife species to historical ranges where habitat exists or could be restored. After individual resource goals were developed, conflict resolution techniques (Daniels et al. 1993) were used to develop a consensus among resource interests. As a result of this action, landscape-level desired conditions were developed that described conditions which would be achieved over time through management practices.

At the Elkhorns, the ecological landscape units were the bases for the integrated descriptions of desired conditions. The desired conditions for each of the four ecological landscape units described vegetation composition and structure (by landtype association), wildlife species, and riparian conditions, as well as recreation features, values, and use patterns.

The relation of ecological landscape units to Helena National Forest Plan management areas determines, in part, where desired conditions will occur. Management area E-2, for example, was designated in the Forest Plan as big game summer range and as essentially roadless. In this management area, desired conditions in the landscape analysis included occupation by big game animals and no recreation facilities or access for motorized vehicles. Management areas of the Helena National Forest are relatively large (e.g., only four occur within the Elkhorns and no ecological landscape unit contains more, than two management areas); consequently, they provide general descriptions of desired resource flows and landscape patterns required to sustain those flows. Ecological landscape unit 4 also includes about 30,000 acres (12 000-ha) of Bureau of Land Management lands managed under direction of the Headwaters Resource Management Plan. These acres fall within six different management unit designations of the Headwaters Resource Management Plan. The management direction provided for these management units also influences patterns of desired conditions on the landscape.

## Implementation

The Elkhorn landscape analysis will be implemented through projects of about 30,000 to 40,000 acres (12 000 to 16 000-ha) that will fall under a single National Environmental Policy Act (NEPA) document. The wide range of activities prescribed through a single landscape-scale analysis is expected to provide much efficiency over traditional small scale analyses. Analysis of treatment effects on biodiversity, threatened and endangered species, or old-growth have not been properly addressed at small scales. Through identification of desired conditions at large scales, smaller projects can be implemented with minimal conflict between competing resource values.

The Crow Creek Project is the first project proposal for the Elkhorns Landscape Analysis Area. It identifies needs for an area encompassing two ecological landscape units and six landtype associations. Comparisons of existing and desired conditions form the basis for proposed management activities to create conditions similar to the desired condition or to move towards the desired condition.

A more detailed analysis of vegetation patterns was necessary for design of the Crow Creek Project. Within landtypes, vegetation map units were delineated based on lifeform (e.g., grassland, shrubland, or conifer). Habitat type, canopy closure, and colonization were characterized for each map unit. Comparison between existing life form abundance and distribution with desired conditions specified by landtype association, formed the basis for treatment needs within each landtype (table 2).

**Table 2--Percentage of Occurrence by Lifeform for Desired Condition, Existing Condition, Posttreatment Condition, and Acres Proposed for Vegetation Management on the Crow Creek Project Area[1]**

| | | | LIFEFORM | | |
|---|---|---|---|---|---|
| Ecological landscape unit | Landtype association | Condition/activity | Percent grassland | Percent shrubland | Percent conifer |
| 4 | 2 | Desired<br>Existing<br>Posttreatment | 60-65<br>57<br>60<br>(0 ac) | 5-10<br>16<br>16<br>(0 ac) | 5-10<br>27<br>25<br>(280 ac) |
| 4 | 4 | Desired<br>Existing<br>Posttreatment | 60-65<br>32<br>46<br>(0 ac) | 5-10<br>24<br>17<br>(874 ac) | 5-10<br>43<br>37<br>(724 ac) |
| 4 | 11 | Desired<br>Existing<br>Posttreatment | 85-90<br>17<br>75<br>(0 ac) | 1-5<br>26<br>15<br>(515 ac) | 0-5<br>57<br>10<br>(1717 ac) |
| 3 | 1 | Desired<br>Existing<br>Posttreatment | 30-33<br>21<br>22<br>(0 ac) | 1-5<br>12<br>12<br>(0 ac) | 65-70<br>66<br>66<br>(84 ac) |
| 3 | 3 | Desired<br>Existing | 20-25<br>57 | 1-2<br>4 | 60-70<br>38 |
| 3 | 9 | Desired<br>Existing<br>Posttreatment | 10-12<br>14<br>17<br>(0 ac) | 1-5<br>4<br>4<br>(0 ac) | 60-70<br>82<br>79<br>(348 ac) |

[1] Vegetation management treatments differ with ecological landscape unit and landtype association. No treatments are proposed for landtype association 3 due to its limited acreage.

Objectives for vegetation management include balancing the percentage of land occupied by a given lifeform; manipulating stand structures in dry Douglas-fir habitat types; increasing organic matter in grassland and shrubland soils; and changing herbaceous plant composition. These objectives will be accomplished through proposed actions which include vegetation manipulation (using burning to reduce sagebrush cover, reduce conifer colonization, and reduce understory conifer ingrowth in stands with old-growth characteristics), review of vegetative condition in livestock allotments, and improvement of riparian conditions (through livestock management, aspen regeneration, and development of standards for assessing riparian area health).

## North Flints Landscape Ecology Unit

The north end of the Flint Creek Range was identified as one of 28 landscape analysis areas in the Deerlodge National Forest in Montana (fig. 1). The delineation of the North Flints Landscape Ecology Unit was based on a combination of visual, hydrologic, and vegetation criteria. Nearly the entire unit can be viewed from Interstate 90, the primary east-west travel corridor in Montana. All drainages run north to the Clark Fork River, and the ponderosa pine at low elevations forms a relatively unique community on the Deerlodge National Forest. The unit encompasses about 46,000 acres (18 600-ha) and includes both Deerlodge National Forest lands and some small private holdings at lower elevations.

The North Flints Landscape Ecology Unit ranges from 4,800 to more than 9,350 feet (1450 to over 2850 m) in elevation and includes plant communities ranging from grasslands at low elevations to communities dominated by whitebark pine (Pinus albicaulus) at high elevations. Fire is the dominant natural disturbance process and, along with insects and pathogens, has created a variety of forest vegetation patterns and structures on the landscape over time.

## Landscape Hierarchies

The original 28 landscape ecology units delineated for the Deerlodge National Forest were an attempt to integrate geomorphic characteristics, hydrologic watersheds, vegetative communities, and visual characteristics into landscape-level management units. These units therefore represent a level in the management hierarchy between the project and a regional analysis scale that might encompass the entire Flint Range (table 1).

At the forest scale, the North Flint Landscape Ecology Unit was divided by elevation into two major habitat type series or climax series: the lower elevation Douglas-fir series, and the higher elevation subalpine fir series. Fire disturbance patterns were analyzed at a scale independent of habitat type series; however, the two series types studied have different fire disturbance regimes which support their use as criteria in mapping. Historical disturbance patterns at the stand level served as the lowest hierarchical scale of analysis.

## Natural Ecosystem Patterns and Processes

The North Flints analysis emphasized the description of historical vegetation patterns through an understanding of ecosystem functions and natural and human-caused processes affecting these functions. Historical vegetation maps were produced for 1880 (the year of last major stand replacement fire), for 1950 (a time before major timber harvest), and for 1991 (the current vegetation pattern as influenced by recent timber harvest and fire suppression).

Historical maps were constructed using a combination of aerial photo data, individual stand examination data, and fire history analyses. These maps identified aggregations of similar forest stands that probably originated from common stand replacement fires. Fire scars on residual trees were used to date understory burns and provide data on historical fire frequency and intensity (Arno and Sneck 1977). Stand age analyses were used to establish dates and spatial extent of stand replacement fires and to reconstruct historical vegetation patterns. Individual stand examination data were used to check origination dates and delineate recent management activity.

The landscape analysis area was divided by habitat type series based on elevation. The subalpine fir climax series represents just over half the entire unit. The vegetation maps also delineated nonforested openings and seven structural stand development stages: seedling (<20 year), sapling (20-40 year), immature pole (40-110 year), mature pole (110 year), mature two-storied (> 110 year), mature mixed-species conifer (>110 year), and mature Douglas-fir (>110 year). Stand structural stages are a means of classifying vegetation into logical units to define habitat for wildlife, hydrologic potential, and stand development pathways (O'Hara and Oliver 1992, Oliver 1992). Recognizing that an entire range of multiple resources is dependent, in part, on vegetation structure will allow management to create patterns of structures across entire landscapes to meet management objectives.

Fire histories indicated that major fires occurred in about 1880 and 1860. Fires earlier than these occurred, but complete evidence of spatial extent is missing due to later fires. Stand replacement fires ranged from about 7 to 270 acres (2.8 to 110-ha) in the Douglas-fir series before settlement. Additionally, low intensity fires periodically burned the understory, producing stands with open “parklike” structures. Following fire suppression, mature conifer structures were described as changing from open parklike structures in 1880, to structures of two-storied stands with dense understory, and mature pole structures in 1950 and 1991 in the Douglas-fir series (table 3).

**Table 3--Average Patch Size and Total Acres by Structural Type and Size Class for Each of the Three Analysis Dates for the Douglas-Fir Climax Series Portion of the North Flints Landscape Ecology Unit**

| | | 1880 | | 1950 | | 1991 | |
|---|---|---|---|---|---|---|---|
| Structure type and size class | Age (yr) | Avg. patch size (acres) | Total acres | Avg. patch size (acres) | Total acres | Avg. patch size (acres) | Total acres |
| Mature Douglas-fir | >110 | 196 | 6664 | 507 | 6088 | 423 | 6352 |
| Mature mixed conifer | >110 | 584 | 2925 | 460 | 4600 | 70 | 563 |
| Mature two storied | >110 | 38 | 113 | 78 | 1019 | 37 | 448 |
| Mature pole | 110 | 22 | 22 | 115 | 2410 | 83 | 4840 |
| Immature pole | 40-110 | 0 | 0 | 61 | 2921 | 0 | 0 |
| Sapling | 20-40 | 131 | 3142 | 0 | 0 | 52 | 2920 |
| Seedling | <20 | 53 | 3334 | 0 | 0 | 36 | 1330 |
| Nonforest openings | | 122 | 5068 | 67 | 3771 | 56 | 3608 |

Openings created by the last two major fires ranged from about 120 to 1700 acres (50 to 680-ha) in the subalpine fir series, burning a total of 15,000 acres (6000-ha). These fires left many individual trees and clusters of trees not burned, especially along the perimeter of a fire. Low-intensity understory burns were not an uncommon occurrence in these higher elevation stands, but trees with scars from more than two fires were not found. Exclusion of fire since 1880 in these areas has left most stands in mature pole structures (table 4).

**Table 4--Average Patch Size and Total Acres by Structural Type and Size Class for Each of the Three Analysis Dates for the Subalpine Fir Climax Series Portion of the North Flints Landscape Ecology Unit**

| | | 1880 | | 1950 | | 1991 | |
|---|---|---|---|---|---|---|---|
| Structure type and size class | Age (yr) | Avg. patch size (acres) | Total acres | Avg. patch size (acres) | Total acres | Avg. patch size (acres) | Total acres |
| Mature Douglas-fir | >110 | 0 | 0 | 94 | 1601 | 69 | 1303 |
| Mature mixed conifer | >110 | 113 | 3162 | 135 | 4739 | 143 | 4437 |
| Mature two-storied | >110 | 76 | 113 | 101 | 1307 | 75 | 1044 |
| Mature pole | 110 | 51 | 410 | 165 | 5760 | 840 | 13440 |
| Immature pole | 40-110 | 84 | 505 | 192 | 8238 | 32 | 255 |
| Sapling | 20-40 | 233 | 6991 | 9 | 9 | 101 | 950 |
| Seedling | <20 | 243 | 8772 | 3 | 3 | 34 | 983 |
| Nonforest openings | | 87 | 4251 | 57 | 3259 | 51 | 3200 |

Open areas (represented by nonforested openings structure class) have declined since 1880 and generally occur as smaller patches today than historically. This is coincident with fire suppression and changing patterns of grazing and herbivory. Historically, many large native mammals grazed in these openings.

Recent grazing use (over last 120-130 years) has been dominated by domestic livestock with subsequent changes in species composition of these grasslands, detrimental effects on soil and water resources on some sites, and an increase in conifers.

Riparian areas were described as having changed in composition to communities more typical of mid- to late-successional species. Lower elevation riparian areas have become increasingly dominated by

Douglas-fir and Engelmann spruce (Picea engelmannii) at the expense of herbaceous plants and hardwoods such as aspen, willows (Salix spp.), thinleaf alder (Alder incana) , and red-osier dogwood (Cornus stolonifera) . Higher elevation riparian areas have higher densities of conifers and more canopy layers.

The North Flints Landscape Ecology Unit analysis indicated a number of other interesting patterns across the landscape unit. Apparently, fires in low-elevation, Douglas-fir series stands tended to occur at greater intensities in draws and at lower intensities on slopes and ridges. Stands, therefore, were more likely to be younger and even-aged in draws, and multi-aged, open, and parklike in other areas. This mufti-aged pattern has been observed frequently in other Douglas-fir series stands in the region (Arno 1988, and Fischer and Clayton 1983). The higher burn intensity pattern in draws has not been observed locally but has been reported for pinyon pine (Pinus cembroides) stands in Mexico (Segura and Snook 1992). In the upper elevation areas, the opposite pattern is likely; consequently, the division of the landscape unit into Douglas-fir and subalpine fir zones in this analysis seems reasonable.

A second pattern observed in this study was the apparent increase in homogeneity of vegetation patterns. Between 1880 and 1990, both the low elevation and upper elevation habitat type series had increasing proportions of acres in mature stand structures, particularly the mature pole structures. These acres also tended to occur in larger blocks by 1991 (tables 3 and 4).

## Development of Desired Conditions

Desired conditions for the North Flints unit were centered around allowing natural processes and functions to assume their historical role to the extent possible given the current altered landscape. Maintaining a complete range of plant communities in various stages of development was a major component of maintaining natural processes and functions. When vegetation communities function within the natural range of variability, most other components of the ecosystem are also likely to function within natural ranges (Swanson et.al. 1993). Other landscape analyses, including the Elkhorns analysis, have shown that vegetation management plays a fundamental role in landscape-scale management. Information from historical vegetation maps concerning disturbance patterns and patch size was useful for incorporating appropriate management treatment scales into the description of desired conditions; for example, “natural openings and patch sizes will represent historical conditions to the extent possible. Unnatural fragmentation will be avoided.”

## Implementation

The North Flints Landscape Ecology Unit analysis will serve as guiding documentation for desired conditions over the entire unit. Individual documents will be used to refine specific projects used to reach desired conditions.

The French Gulch Environmental Assessment is the first analysis within the North Flints Landscape Ecology Unit. It encompasses about 14,700 acres (5950-ha) and proposes vegetation management treatment on about 350 acres (140-ha) to increase the extent of open-grown structures of Douglas-fir and ponderosa pine in the area. This treatment involves harvest and underburning. About 1,200 additional acres (484-ha) are proposed for underburning to reduce fuel loadings, create fuel breaks, and encourage regeneration of ponderosa pine and aspen.

The proposed management treatments represent modifications of more commonly used treatments that were largely driven by commodity production goals. Modifications include treatment of larger units, leaving larger trees as residuals, and more underburning. These modified treatments were driven by landscape-scale goals identified in the North Flints Landscape Ecology Unit.

The North Flints Landscape Ecology Unit was largely defined as a viewshed seen by travelers on a segment of Interstate 90. Interestingly, visual objectives were consistent with vegetation management goals because treatments are designed to occur in larger blocks, create natural patterns, and include more partial-retention structures. Such treatments meet landscape-scale objectives without being visually obtrusive.

## CONCLUSIONS

The teams involved with both the Elkhorn and North Flints analyses probably would describe the process used in their projects as evolving; therefore, the procedures outlined in this paper are not necessarily those the Forests will use consistently in the future but rather those developed in these prototype analyses. Accordingly, the teams acknowledge the usefulness of an adaptive management approach (Everett et al. 1993) to landscape evaluation. These projects and future projects are also likely to provide validation or amendments to existing forest plans. This point is important: management activity is being defined based on landscape-scale objectives concerning ecosystem sustainability, not on existing forest plan guidelines or annual sale quantities.

The two case studies provide an interesting contrast in approaches even though they are subject to similar abiotic, biotic, and social factors. The two primary differences in approach were the hierarchical scales used for analysis and the type of detailed analysis at the lowest hierarchical level; these two differences undoubtedly are related. Because there were only four landscape analysis areas within the entire Helena National Forest, there was a need to develop additional hierarchical scales (ecological landscape units, landtype associations) on the Elkhorns Landscape Analysis Area. Additionally, the limitations on the amount of analysis that could be performed on a single large unit had to be recognized. This relatively extensive approach has required additional detailed analysis as a precursor to implementation, but has also effectively defined where additional analysis should occur.

The North Flints Landscape Ecology Unit was one of 28 smaller analysis areas, that allowed intensive analysis at smaller scales (identification of actual stands) and fewer hierarchical levels. This detailed analysis allows relatively precise treatment scheduling that considers both spatial and temporal characteristics of vegetation pattern. Within the Deerlodge National Forest there may be a need, however, for further hierarchical organization and analysis of the 28 landscape analysis areas at a coarser scale.

Active participation by the Bureau of Land Management and Montana Department of Fish, Wildlife and Parks along with the Helena National Forest provided an impetus for multiagency cooperation, which will be necessary in other landscape analyses where intermingled ownerships exist. Even though all three agencies were committed to the concept of ecosystem management, all have different management objectives. These differences led to some interesting compromises, such as a desired condition for grasslands that was greater in spatial extent than the natural range of variability to meet Montana Fish, Wildlife, and Parks objectives for elk habitat.

Analysis results for both the Elkhorns Landscape Analysis Area and the North Flints Landscape Ecology Unit neither specified spatial location nor time frame for achieving desired conditions. Ecosystems are dynamic and only vague descriptions of desired conditions are possible without also identifying a specific temporal scale.

In neither the Elkhorns Landscape Analysis Area nor the North Flints Landscape Ecology Unit were hierarchical analysis scales considered that transcended the National Forest scale. Although both analyses included ownerships outside National Forest boundaries, these inclusions were not important deviations from the analysis scales used; for example, the Elkhorns included Bureau of Land Management lands and Montana Fish, Wildlife, and Parks management goals for lands within the Elkhorns range. The North Flints Analysis includes small private holdings that were part of both watershed and viewshed units. Future landscape analyses must examine the role of analysis units in larger systems encompassing entire regions and go beyond current administrative or political boundaries such as National Forests, states, or countries (Morrison 1993).

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# Assessment Techniques for Evaluating Ecosystem, Landscape, and Community Conditions

**W.Hann, R.E. Keane, C. McNicoll, and J. Menakis**

## ABSTRACT

This paper provides an overview of a systematic process for characterizing and assessing ecosystem and landscape structure, composition, and function. The process presented is useful for evaluating existing management systems and the design of management treatments that promote sustainable ecosystems. An analysis area in the Beaverhead National Forest of Montana is used to demonstrate the basic features of our characterization and assessment techniques. Results of this analysis provide an important framework for the development of management systems that conserve biodiversity, represent ecosystem processes, and produce resources and values to meet human needs.

## INTRODUCTION

This paper describes an analysis process appropriate for evaluating ecosystem and landscape conditions. The assessment methods presented are useful at different scales of hierarchical analysis. Techniques are demonstrated with data from a landscape and ecosystem-scale assessment of the Trail Creek Timber Sale in the Northern Region of the USDA Forest Service. The timber sale is in a 60,000-acre watershed in southwest Montana. An environmental impact statement (USDA Forest Service 1990) tiered to the Forest plan (USDA Forest Service 1986a) was developed for the project in 1990. A subsequent analysis was conducted to assess effects of the project on biodiversity (USDA Forest Service 1991). This paper expands on the prior biodiversity analysis by demonstrating the use of new techniques and methods developed and tested since that evaluation.

Terminology used in this paper concerning description of ecosystem and landscape conditions, and development of ecological land units, follows Bailey and others (1993) and Hann and others (1993).

### The Evaluation Process

The following is a list of the three general steps used in the analysis of ecosystem, landscape, and community conditions:

1. Characterize the general composition, structure, and process features of the ecosystems and landscapes within the analysis area.

2. Analyze data to assess changes in structure and composition elements and correlate such changes with previous management treatments.

3. Examine the ecosystem processes important for the area and their associated effects on ecosystem and landscape composition, structure, and rates of change.

The process of characterizing landscape and ecosystem elements can be approached systematically. Characterizing these elements involves summarizing data concerning basic ecosystem and landscape processes, rates of change, and associated composition, structure, and function features. We use the following stepwise procedure for ecosystem and landscape characterization:

1. Stratify the analysis area by land system characteristics. Stratification begins at a coarse scale by using subsections to stratify broad landform, parent material, and climate characteristics. This stratification usually yields three to four strata for a watershed-size analysis area. The basic ecosystem processes controlling ecosystem composition and structure are then briefly described and used to develop criteria for defining and mapping landtype associations. For a given geographic area, landtype associations should be designed so that the number of dominant (> 20% of the area)

landtype associations does not exceed three. Our experience has shown that most important ecological parameters can be accurately described in a few strata. The delineation of few strata avoids unnecessary complication of an analysis and improves communication of results.

2. Stratify by potential vegetation. Groups of vegetation series are defined for subsection characterization, and series or habitat type groups are defined for landtype association characterization. The number of dominant (> 20% of the area) potential vegetation units described for an analysis area should not exceed three, to facilitate efficient analysis.

3. Evaluate historical vegetation and disturbance conditions. Refined information concerning changes during recent historic times are summarized for the analysis area. Such summaries should cover a full historical sequence of change in the vegetation (e.g., 100 years or more) to show the dynamics of changing patterns and successional stages. If possible, maps of historical vegetation conditions should be developed to facilitate subsequent pattern analysis. Initial delineations of dominant species, size class, and canopy closure for historical vegetation maps should be precise. Geographic information system software may be used to group types to answer different questions in subsequent analyses. Coarser-scale information should be summarized to analyze changes through geologic time. Information concerning the history of disturbance events in an analysis area is also synthesized and mapped when appropriate.

4. Develop ecological land units (ELUs). An ELU delineates potential vegetation strata within landtype associations to describe biophysical environments that do not change substantially across time in response to secondary disturbances. The ELU is the basic environmental stratification used to characterize ecosystem processes and associated rates of change for successional stages, biomass cycling, nutrient cycling, and photosynthetic production in most integrated resource analyses (Hann et al. 1993). Vegetation successional stages of ELUs are used to describe changes in the pattern and composition of an analysis area across time. The number of dominant (> 20% of the area) ELU strata for an analysis area should not exceed five.

5. Characterize successional stage composition and structure of an ELU. Various classifications can be used to describe the successional stages of an ELU in terms of composition and structure. Some ELUs (e.g., those of grassland or shrubland areas) may only have one or two stages, while other ELUs (e.g., those of forested areas) may have four or five. The scale of refinement of land systems, potential vegetation, ELUs, and successional stages should be appropriate to analysis needs and should be described in terms that can be understood by managers and members of the public. Simple classifications and mapping, that emphasize dominant landscape components and primary ecosystem processes, and that explain variability in composition and structure, will usually meet most analysis objectives.

6. Examine landscape connections. Landscapes must be evaluated in context of their environmental setting through multiscale analyses. Scales of analysis are selected based on connection elements appropriate to the analysis. For example, the landscape connections important to fire behavior include fuel conditions, lightning and human-caused fuel ignition potential, and weather patterns. In contrast, travel linkages for wide ranging animals, such as large predators (e.g., wolves and grizzly bears) or migratory birds commonly, include vegetation and landform characteristics of both adjacent and distant landscapes.

## Evaluation of the Trail Creek Area

The following is a description of how we used the principles outlined above in an analysis of the Trail Creek area of southwestern Montana.

**Step 1**. Three land system inventory subsections were identified for the Trail Creek area: (1) M20, moderately steep dissected mountain slopes in the upper elevations of the watershed, with coarse-textured soils developed on volcanic parent materials; (2) M22, gentle to moderately steep dissected mountain slopes in the middle elevations of the watershed, with coarse-textured soils developed on granitic parent materials;

and (3) M3, gentle to moderately steep dissected mountain slopes in the lower elevations of the watershed, with coarse-textured soils developed on metasedimentary parent materials. M20 encompasses about 20 percent of the analysis area, M22, 70 percent, and M3, 10 percent. Average annual precipitation for the area is about 20 inches and average annual temperature is about 35 degrees F.

Landtypes of the analysis area were grouped into landtype associations for analysis based on similar parent material and landforms. The dominant landtype association included mountain slopes, ridges, and benches on granitic parent materials, and comprised 51% of the analysis area. The second dominant landtype association (28% of the analysis area) included areas with similar landforms, but on mixed parent materials. The third largest landtype association (13% of the analysis area) included areas of mountain stream breaklands and drainageways on mixed parent materials. Mixed aspects and elevation ranges from 6200 to 7800 feet were characteristic of all landtype associations.

**Step 2**. Potential vegetation was classified by habitat type groups (Pfister et al. 1977). Habitat types were grouped based on similar patterns of successional stage development, which is strongly correlated to broad moisture and temperature regimes. The dominant habitat type group in the analysis area is the cold-moist group typified by the subalpine fir/grouse whortleberry habitat type. This group makes up 65% of the area. For the Trail Creek area the variability in potential vegetation is relatively low due to the limited variability of local climate, soil conditions, and landform. Differences in successional pathways and disturbance regimes are primarily a function of community position on landforms, rather than potential vegetation.

**Step 3**. Historical conditions were evaluated by mapping cover type, size class, and canopy closure from 1930s and 1950s forest inventory maps and aerial photos. The 1930s data were projected backwards 40 years to about 1890, just after the last major landscape-scale wildfire in this area. Historical lightning and human-caused fire ignitions, wildfire patterns, and insect-disease conditions also were mapped. The initial mapping of cover types, size class, and canopy closure provided an inventory of possible successional stages that could occur across time after disturbance.

**Steps 4 and 5**. ELUs were developed to provide a template for descriptions of rates of vegetation change, broad-scale successional stage compositions and structures, and disturbance regime characteristics. The ELUs of the analysis area are summarized below:

**ELU 1**. This ELU was found on cool wet habitat types of stream bottoms, draws, and cold air benches, with undifferentiated parent materials. This ELU comprises about 13% of the total analysis area and typically supports linear vegetation patches. Vegetation types of this ELU, prior to fire suppression, were typically dominated by pole-size and larger spruce-subalpine fir-lodgepole pine in the upper layer, subalpine fir in the understory regeneration layer, and a low layer of shrubs and herbaceous species. Pole and medium-size scattered dead snags occurred in the upper layer and down logs were common on the ground surface. In the pre-fire suppression era a number of disturbance factors were important. Historical disturbances included gap succession of individuals or groups of overstory trees from age; insects and disease; frequent low magnitude windthrow events; and creeping-torching fires that thinned individuals or groups of trees of all sizes on a 30-90 year interval within a given patch. In response to fire suppression, subalpine fir has increased relative to spruce in all layers, and layers have become much denser. Individual tree vigor has decreased, and insect and disease mortality has increased. Litter and duff layers have become deeper. With the accumulated fuel in all layers, potential fire behavior has shifted from a creeping-torching type process to a crown fire system that could have more severe fire effects. Timber harvest in this ELU was common in the 1950s in accessible areas. Larger spruce trees were selectively removed, which resulted in an increase of subalpine fir in harvested areas. Subalpine fir is more susceptible to stem rot and windthrow than spruce.

**ELU 2**. This ELU is found on cold-moist habitat types of gentle benches, slopes, and ridges, with granitic parent materials. It occupies about 48% of the analysis area and typically supports irregularly shaped vegetation patches. Vegetation types prior to fire suppression were typically dominated in the upper layer by relatively open, pole-size lodgepole pine. The lower layer was typically dominated by low-growing shrubs and herbaceous species. Pole-size snags were scattered in the upper layer, but down logs were usually sparse and litter and duff depths were shallow. The pre-fire suppression disturbance system was mixed. Chronic, low intensity mountain pine beetle activity caused minor annual death of pole-size and larger lodgepole pine. Dwarf mistletoe was common but had minor effects on vegetation composition and structure. Ground fires occurred at 30- to 50-year intervals and killed some saplings and a few of the larger trees. In response to fire suppression, the upper and understory layers of trees have become much denser. Individual tree vigor is lower and death from mountain pine beetle is increasing. Dead standing and down logs have increased, along with litter and duff layer depths. In response to fuel accumulation in all layers, fire behavior has shifted to crown fires. There has been some logging since the 1960s in this ELU. Traditional harvest systems have been used, including clearcutting, dozer slash piling and burning, and natural regeneration. There is no vertical structure remaining after this type of harvest, except a few seed trees or snags, and regeneration and succession are slow.

**ELU 3**. This ELU is found on cold-moist habitat types of moderately steep slopes and ridges with granitic parent materials. ELU occupies about 32% of the total landscape and supports irregular vegetation patches that follow steeper slopes and ridges. This ELU 3 is characterized by concave drainages and steeper mountain slopes that tend to funnel wind. Vegetation types (prior to fire suppression) typically were dominated in the upper layer by moderately open to closed lodgepole pine of sapling to pole-size. The lower layer typically had sparsely growing shrubs and herbaceous species. Pole-size snags were infrequent in the upper layer, down logs were usually sparse, and litter and duff were shallow. The pre-fire suppression disturbance system was mixed. The return interval of fire ranged from 30- to 50-years, with some fires acting as spotting or torching fires, which thinned sapling stands, and some fires acting as crown fires, which killed most of the overstory trees. Areas with low rates of fuel accumulation often missed one or two fire intervals. In response to fire suppression, the upper layer and understory layers of trees have become much larger and denser. Individual tree vigor is lower and death from mountain pine beetle is increasing. Dead standing and down logs have increased, as have litter and duff layer depths. Because of this fuel accumulation in all layers, the potential fire behavior has shifted from a mixed system to a crown fire system. There has been some logging since the 1960s in this ELU. Traditional harvest systems have been used, which usually included clearcutting, dozer slash piling and burning, and natural regeneration. Regeneration and succession are slow after this type of harvest and there is no vertical structure remaining after clearcutting except for a few seed trees or snags.

**Other ELUs**. Other ELUs in this landscape include dry Douglas-fir habitat types on lower elevation southern aspects; dry sagebrush-grass types on lower elevation southern aspects; poorly drained, wet willow riparian types on low-gradient stream flood plains; and very cold-high elevation types. These ELUs make up a relatively minor part of the total analysis area.

**Step 6**. ELUs 2 and 3 support lodgepole pine cover types that greatly influence spatial and temporal energy processes in this landscape. This influence is apparent for landscape-scale photosynthesis, successional biomass cycling, insect and disease activity, and for fire behavior and effects. Fire ignitions in these landscapes are driven upslope by the topography. In regional-scale cumulative drought conditions, fires are driven to the east by wind events into the analysis area. Such wind events, however, are generally not common during most fire seasons. Regional-scale cumulative drought conditions also are not common in this area because of the cold climate. This area seems to be protected from large wind-driven fire events by the Bitterroot, Beaverhead, and Pintlar mountain ranges that lie just to the west, south, and north, respectively.

The climate of the analysis area is generally cold with a short growing season. Photosynthetic growth is slow, but biomass accumulates at a faster rate than decomposition. Consequently, dead wood and litter and duff, that act as primary fuel sources, accumulate relatively rapidly. The mix of ground fire, thinning fire, and crown fires in the area historically provided patch mosaics that supported a diversity of adapted species and also buffered the area from major changes (e.g., landscape-scale crown fires). With the present amount of fuel accumulation, the area has a moderate risk for landscape-scale crown fire, depending on weather, fuel, and ignition conditions.

The spruce-dominated ELU 1 is not dominant in the analysis area in terms of energy processes. This ELU, however, is an important component in terms of landscape connections because it provides a relatively continuous overstory in stream bottoms. Such vegetation cover is important to the maintenance of riparian and aquatic systems as well as to species that use these habitats for cover from predators, resting, thermal protection, travelways, nesting, and feeding.

## Data Analysis Results

Succession can be displayed for an ELU by using stand development stages. We used the system developed by Oliver and Larson (1990) in this analysis. Information for each stage was summarized from data taken on sample plots following Northern Region procedures for ECODATA sampling and timber stand exam sampling (USDA Forest Service 1989, 1992). These data were summarized using the ECOPAC analysis program (Jensen et al. 1993) and the R1-Edit data management system (USDA Forest Service 1986b). Table 1 displays the composition of vegetation stages for ELUs 2 and 3. The main difference in successional patterns between ELU 2 and ELU 3 is due to differences in fire regime. The ground fire regime of ELU 2 sustains reinitiation stages for long periods. Low-intensity ground fires at 30- to 50-year intervals reduce ground fuels and ladder fuels, and maintain relatively open pole-size stands. Overstory lodgepole pines commonly show two to three scars and are usually killed by the fourth or fifth scar. This fire regime is effective in thinning larger trees older than 120 years and smaller trees less than 5 inches in diameter. In ELU 3, fires are typically low-intensity thinning fires or moderate-intensity crown fires. This mix of fire behavior exists due to landscape position differences and the influence of topography on local wind and fire behavior.

**Table 1--Composition of Successional Stages for ELU 2 & 3**

| Attribute | Initiation stage | Stem exclusion stage | Reinitiation stage | Old-growth stage |
|---|---|---|---|---|
| Cover type | Shrub-tree seedling-herb | Lodgepole | Lodgepole | Lodgepole |
| Upper layer size class | Low shrub | Sapling | Pole | Medium tree |
| Upper layer live canopy | Low | High | Moderate | Moderate |
| Upper layer dead size | Medium tree | Medium & sapling | Sapling | Medium tree |
| Upper layer dead canopy | Moderate | Low | Low | Low |
| Number of layers | 2 layers | 2 layers | 3 layers | 4 layers |
| Height of upper layer | 1 foot | 30 feet | 80 feet | 100 feet |
| Live basal area | 2 square feet | 50 square feet | 150 square feet | 200 square feet |
| Live trees/acre | 700 trees/acre | 1000 trees/acre | 285 trees/acre | 590 trees/acre |
| Dead trees/acre | 50 trees/acre | 70 trees/acre | 5 trees/acre | 7 trees/acre |
| Time between stages | 15-35 years | 25-75 years | 55-95 years | |
| Fuel model | 2(75)+5(25) | 8(100) | 2(60)+5(40) | 10(100) |
| ELU 2% of landscape | 0-5 | 0-5 | 60-90 | 5-40 |
| ELU 3% of landscape | 10-50 | 10-50 | 5-30 | 0-5 |

Forest cover type composition was summarized for conditions from 1890 to the present and projected based on proposed harvest activities until 1995 (fig. 1). Cover type composition has not changed substantially since the late 1800s. For this analysis area, each ELU has only one dominant successional cover type. In a more complex ELU, with a higher diversity of seral cover types, there may be more than one donimant cover type.

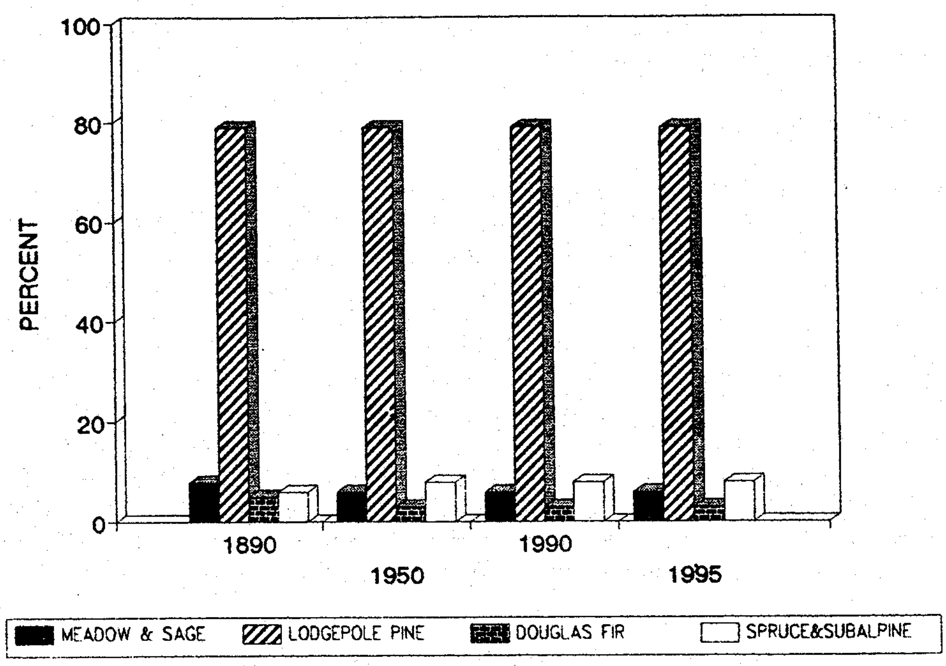


**Figure 1--Cover type composition for the Trail Creek landscape.**

Although cover types have not changed substantially across time in the analysis area, size class composition has changed substantially (fig. 2). For example, in 1890 there was a bell-shaped mix of size classes that shifted to predominantly pole-size stands by 1950. By 1990, many of these patches had moved into the medium-size tree class. Logging in the area between 1950 and 1990 has increased the seedling stage by a small amount, and there is a slight increase predicted in seedling stage related to the proposed harvest between 1990 and 1995. By 1990, the landscape has essentially missed at least two, possibly three fire events. Such fires would have underburned in ELU 2, thereby moving those stands from a medium-size tree class back to pole-size dominance. Crown or thinning fires in susceptible stands of ELU 3 would have moved those stands back to the seedling class or thinned the sapling class.

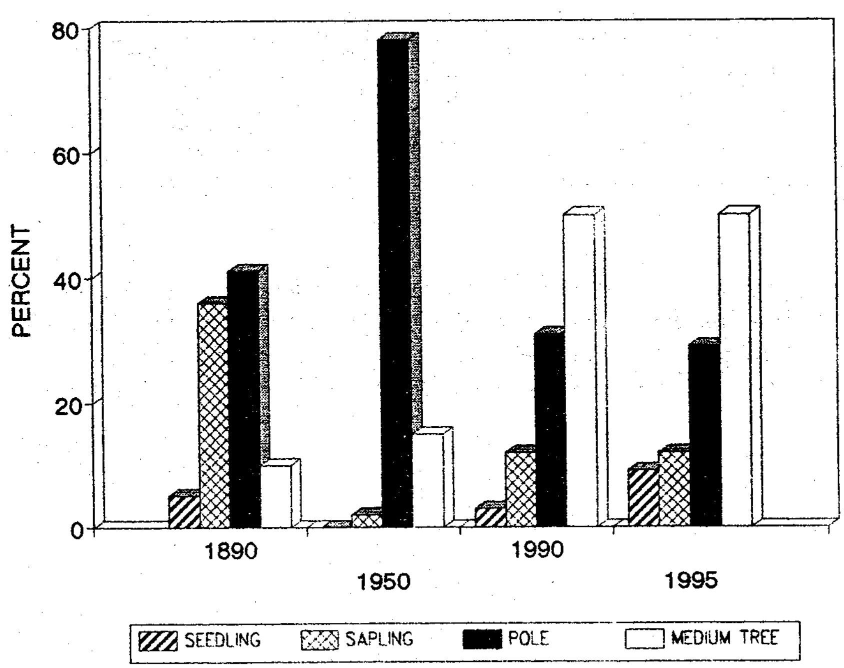


**Figure 2--Size class changes across time for the Trail Creek landscape.**

Temporal changes in tree maturity were examined by combining some successional stages to reflect similar structures and composition (fig. 3). Reinitiation and old-growth stages were grouped into mature spruce and lodgepole pine types for this analysis. Stem exclusion and initiation stages are grouped for all types because they have similar structure. Some important changes in maturity of trees, by species, and their associated successional stages has occurred in the analysis area. Not all vegetation types have changed substantially, however, for example, mature spruce have changed little at the landscape scale (fig. 3). The amount of mature lodgepole, however, increased moderately between 1890 and 1950 as a result of fire suppression, and has decreased somewhat since 1950 as a result of harvest. An additional major change in the lodgepole pine type (not described in fig. 3) has been a change in the structure of mature lodgepole on ELU 2. Underburning historically maintained parklike conditions in this landscape; however, multilayer structures are now dominant, a condition that typically supports a crown fire regime.

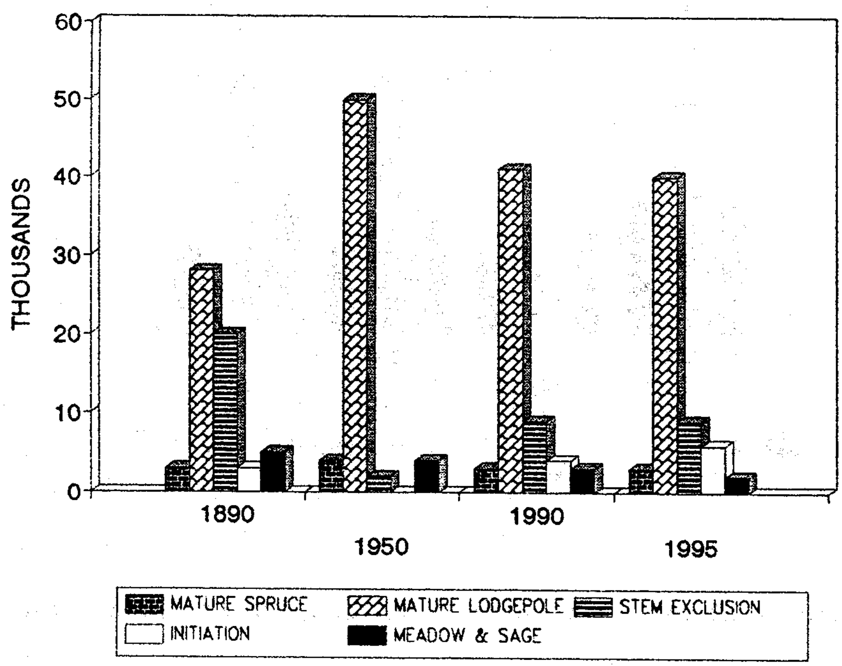


**Figure 3--Changes in amounts (acres) of successional stages for the Trail Creek landscape.**

The stem exclusion stage has steadily decreased since 1890, as it developed into mature lodgepole (fig. 3). The initiation stage was fairly low in 1890; however, it can be assumed that before 1890 (after the last major fire event), the initiation stage was a major component. We estimate that between 1850 and 1890 the initiation stage was probably around 20,000 acres, which is somewhat similar to the stem exclusion stage in 1890. The amount of the initiation stage has increased slightly as a result of harvest between 1950 to 1990. We predict this amount will continue to increase slightly (between 1990 and 1995) as a result of proposed harvest treatments.

The amounts of wet meadows and sagebrush in the analysis area have decreased slightly as a result of encroachment by lodgepole pine and spruce in the wet meadows and Douglas-fir and lodgepole in the sagebrush areas (fig. 3). Fires at 30- to 50-year intervals generally kept trees from regenerating in wet meadows and sagebrush areas during the pre-fire suppression period.

Change in the total amount of vegetation types does not fully describe change in landscape pattern dynamics To evaluate patterns, we examined landscape patch size, distribution, frequency, dominance, diversity, and contagion (fig. 4). In 1890, most vegetation patches were of the stem exclusion stage. As this stage developed into mature lodgepole, many of the mature lodgepole patches merged together. Consequently, by 1950, numbers of mature lodgepole and stem exclusion patches were greatly reduced in the analysis area. Initiation stage patches decreased due to successional development and the lack of new patches created by fire. Patches of meadows and sagebrush decreased from encroachment by conifers. By 1990, timber harvest of small but numerous patches had increased the number of initiation and stem exclusion patches. This trend will continue through 1995 if current proposals for traditional harvest unit size and numbers are implemented. The rapid increase in numbers of patches correlated with traditional timber harvest and roads is similar to what has occurred in other areas (Hansen et al. 1991). In contrast, the natural fire disturbance system for this area creates fewer but much larger openings that develop into large patches of mature forest.

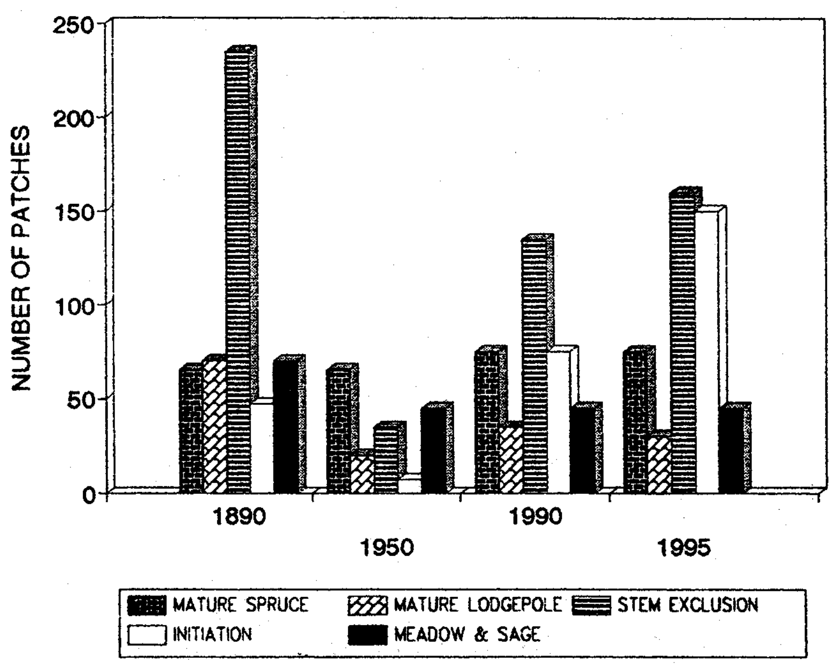


**Figure 4--Changes in number of patches in the Trail Creek landscape.**

Mean patch size is negatively correlated with the number of patches in a given landscape (figs. 4 and 5). In 1890, large patches of mature lodgepole were maintained by ground fires (fig. 5). These patches changed little in size and shape through pre-fire suppression time. Variability of patch size and shape is correlated primarily with landform shape and size. By 1950, as the stem exclusion stage developed into mature stages, the size of areas dominated by mature lodgepole increased substantially. This change was caused by the merging of adjacent stands of ELU 2 and 3 into the same condition. Given the natural fire regime, one fire event would have broken up this condition by crowning and thinning in susceptible stands of ELU 3 and underburning in ELU 2. The reduction of mature lodgepole patch size from 1950 to 1990 is due to timber harvests that have fragmented large continuous areas of mature lodgepole.

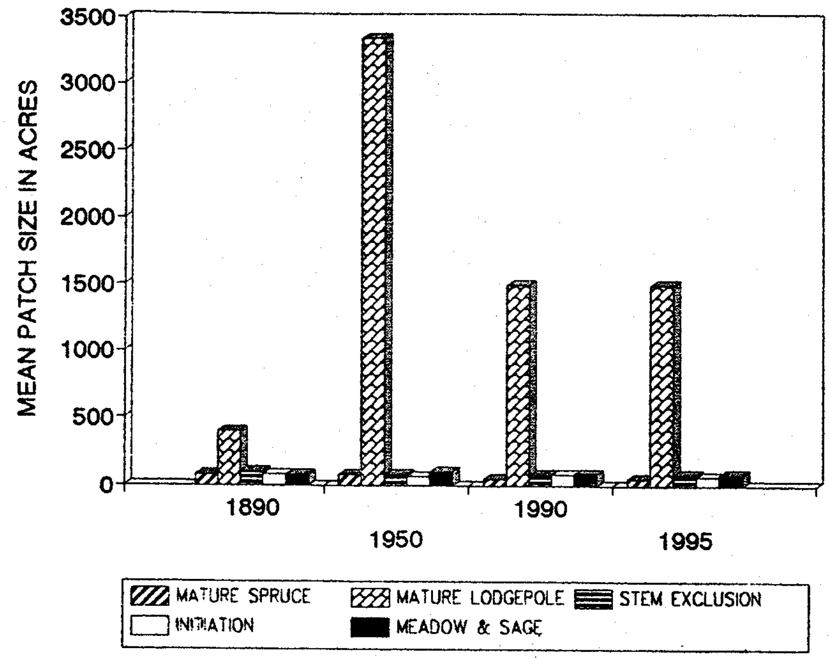


**Figure 5--Changes in size of patches in the Trail Creek landscape.**

Indices of landscape diversity, dominance, and contagion (Turner 1990, Flatherer 1992) were used to assess changes in patch dynamics (fig. 6). Changes in these indices indicate a reduction in vegetation type diversity at the landscape scale, which is correlated with the dominance of mature lodgepole between 1890 and 1950. The dominance index increased slightly for the same reasons. Contagion reflects association of elements within a vegetation type to the rest of the type. There is an increase in contagion from 1890 to 1950, which is associated with the merging of different successional stage patches into the same condition.

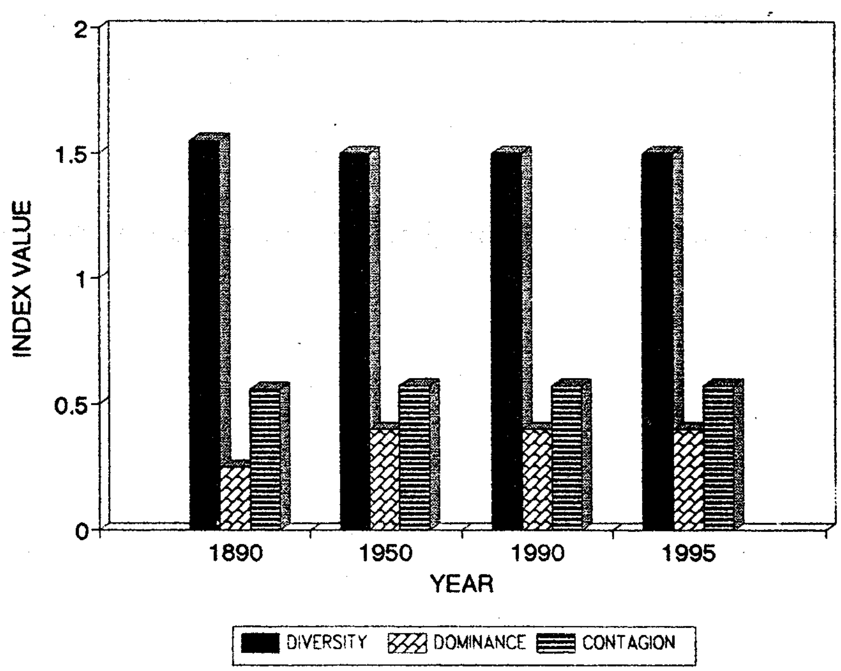


**Figure 6--Change in patterns of the Trail Creek landscape.**

Contrary to the old concept that managing for edge is good for wildlife, it is becoming apparent that many interior dwelling species are negatively affected by edge (Hansen et al. 1991, Hunter 1990, Morrison et al. 1992). These interior species seem to be much more sensitive to habitat loss than edge-dwelling species. Consequently, we examined the composition of edge and is change over time in the analysis area (fig. 7). Edge habitats increased steadily since 1950 due to small patch cutting patterns and road building associated with historic timber harvest. Road corridors substantially increase edge habitats due to their linear nature. Although roads and cutting patterns have substantially increased the amount of edge, the amount of interior forest also has increased dramatically as a result of fire suppression (fig. 8). Traditional roading and harvest systems, ff continued, will rapidly reduce interior forest areas. Because much of the mature forest is at high risk of a crown type wildfire, there is now a risk for loss of interior forest.

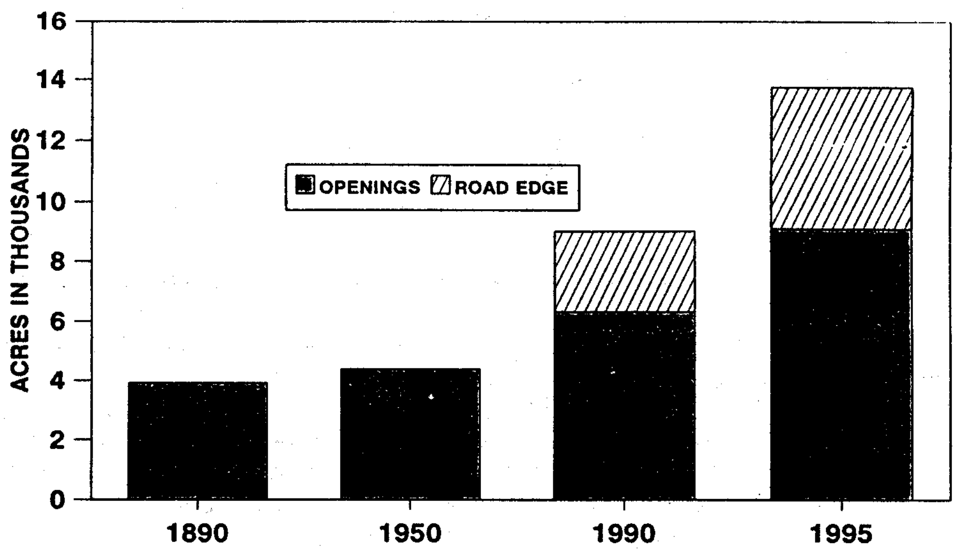


**Figure 7--Change in composition of edge for the Trail Creek landscape.**

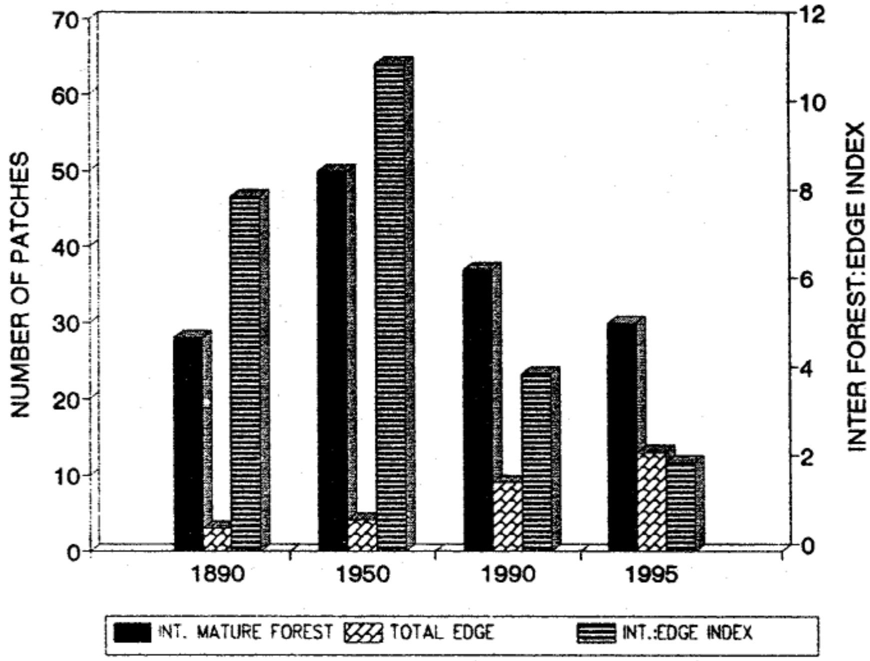


**Figure 8--Change in interior forest and edge in the Trail Creek landscape.**

In summary, it seems that many attributes of landscape composition and structure are within the natural (or historical) range of conditions for the analysis area. Nevertheless, rates of change related to traditional harvest systems, road activities, and fire suppression will rapidly move most conditions outside the ranges to which many species adapted. It is apparent that fire exclusion, road building, and traditional harvesting techniques are changing ecosystem structure and composition in the Trail Creek area.

## Assessment of Ecosystem Processes

Ecosystem processes are those events that change the form of materials across space or through time. A basic process is photosynthesis, which drives plant growth, development, and change. Other energy exchange systems include fire, weather, hydrology, nutrient cycling, insects and disease, mass flow, erosion events, animal movements, and the food chain. The monitoring and assessment of energy processes provides a method for assessing system balance, stability, and resilience. Assessment of process trends also assists the prediction of change in ecosystem composition, structure, and pattern.

The fire behavior regime of ELU 2 has shifted from a pre-fire suppression, ground fire behavior type to a crown fire type in both traditional harvested landscapes and fire suppression landscapes. Because of higher fuel loading, the intensity and rate of fire spread is generally higher in landscapes where fire has been suppressed and timber harvest has been minimal.

Ecosystem processes of ELU 3 also have changed. The shift in typical fire behavior is related to heavy fuel loadings that have accumulated across the landscape. Crown fires may now occur across larger areas, and the power of fire in heavy fuels may carry crown fires into areas occupied by ELU 3. The power of this type of fire can carry through the crown of riparian spruce or willow. Before fire suppression and the accumulation of continuous fuels, crown fires in types adjacent to ELU 3 typically dropped to the ground when they encountered the moist canopy of spruce or willow in riparian environments of ELU 3.

The distribution and form of biomass through time and across space is an important consideration in the evaluation of ecosystem processes and long-term productivity. We examined biomass components by successional stage (table 2) and compared the presettlement scenario of succession after fire with succession after traditional clearcutting. There are some improtant differences in distribution of biomass between the two scenarios. The initiation stage of traditional clearcutting is devoid of the standing dead materials common in fire-generated systems. In the natural system, after fire, this dead standing material falls with time and provides a steady, slow input of fresh, down, dead material that has slow decay rates. This material provides a steady flow of nutrients in the soil important to plant succession. After traditional clearcutting, down wood is left on the ground. If it is not burned, it rapidly decomposes, thereby resulting in a nutrient deficiency later in succession. If the material is burned, much of it will be consumed, and the large remaining charred wood will decompose very slowly. This results in a deficit throughout the successional development cycle.

**Table 2--Biomass Conditions for Successional Stages after Fire and Clearcutting Treatments in ELU 2 & 3**

**STRUCTURAL STAGE**

| Biomass component | Initiation tons/acre | Stem exclusion tons/acre | Reinitiation tons/acre | Old growth tons/acre |
|---|---|---|---|---|
| **Development after fire** | | | | |
| Dead standing | 28.3 | 13.8 | 3.4 | 6.1 |
| Dead down | 2.2 | 16.1 | 2.6 | 6.3 |
| Litter/duff | 0.1 | 0.5 | 1.4 | 4.6 |
| Total dead | 30.6 | 30.4 | 7.4 | 17.0 |
| Live biomass | 1.1 | 1.9 | 19.6 | 97.9 |
| **Development after clearcutting** | | | | |
| Dead standing | 0.4 | 0.1 | 0.1 | 12.1 |
| Dead down | 15.1 | 14.1 | 0.4 | 9.2 |
| Litter/duff | 0.1 | 0.5 | 2.1 | 5.9 |
| Total dead | 15.6 | 14.7 | 3.0 | 27.2 |
| Live biomass | 0.8 | 0.7 | 45.2 | 137.4 |

Without thinning and ground fires at frequent intervals after clearcutting, larger amounts of dead and live biomass accumulate in later successional stages. This accumulation lowers vigor of individual trees and increases insect and disease attacks. Under traditional clearcutting systems, there would eventually be a discontinuity in nutrient availability and a resultant decrease in soil productivity, probably after about two rotations.

Biomass amount was closely related to fire in the presettlement Trail Creek landscape. Death of overstory trees from each fire event also was closely correlated with fuel amounts and fire intervals. The FIRESUM model was used in this analysis to develop projections of fire intensity at 40- and 100-year intervals over a 500-year period by evaluating 10 stochastic projections and averaging their results (Keane et al. 1989). The 100-year-interval fire scenario generates an accumulation of fuels that results in longer flame lengths and increased tree death (fig. 9). In contrast, the 40-year scenario accumulates less fuels and supports shorter flame lengths, resulting in less tree death. Even with a 40-year interval, there are years that produce high-intensity fires, given dry fuels and wind. This fact is reflected in the variation displayed over the 500-year period (fig. 9). The average flame length for the 40-year-interval scenario is 5.8 feet with a standard deviation (SD) of 4.9 feet, and is associated with an average of 32 percent tree death (SD = 27%). The average flame length for the 100-year-interval scenario is 11.7 feet (SD = 7.4 feet), which is associated with an average of 65 percent tree death (SD = 41 %). Tree death in the 100-year-interval scenario will often reach 100 percent, but death in the 40 year interval is typically less than 50 percent. This demonstrates why fires with the 40-year interval, in either ELU 2 or 3, typically do not cause 100 percent tree death in the natural system. As fire suppression increased the interval between fires, however, both landscapes have developed fuel loads that result in high tree death when wildfire occurs.

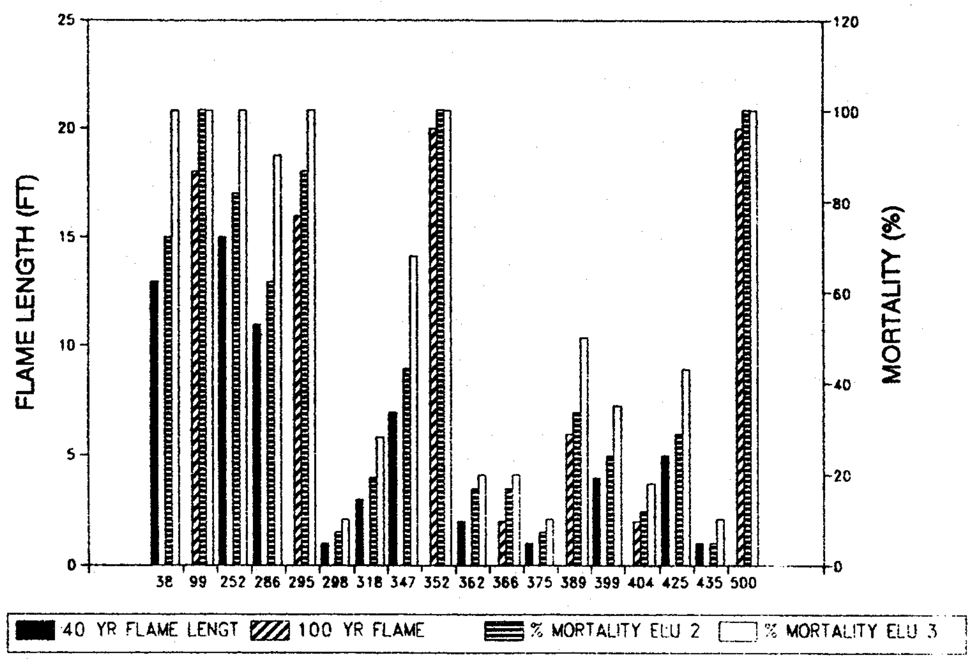


**Figure 9--Longterm projection of different fire intervals in the Trail Creek landscape.**

When information concerning fire behavior, fuels, insect and disease, and competition stress are summarized for the total landscape, a perspective for risk of disturbance events is obtained (fig. 10). Concurrent with maturation of the forest landscape and increase of continuous fuels, the crown fire risk in Trail Creek has increased fourfold since 1890. Increases in mountain pine beetle, mistletoe, and plant competition contribute to the increased risk of crown fire by creating increased dead fuels and flammable, dry, stressed trees.

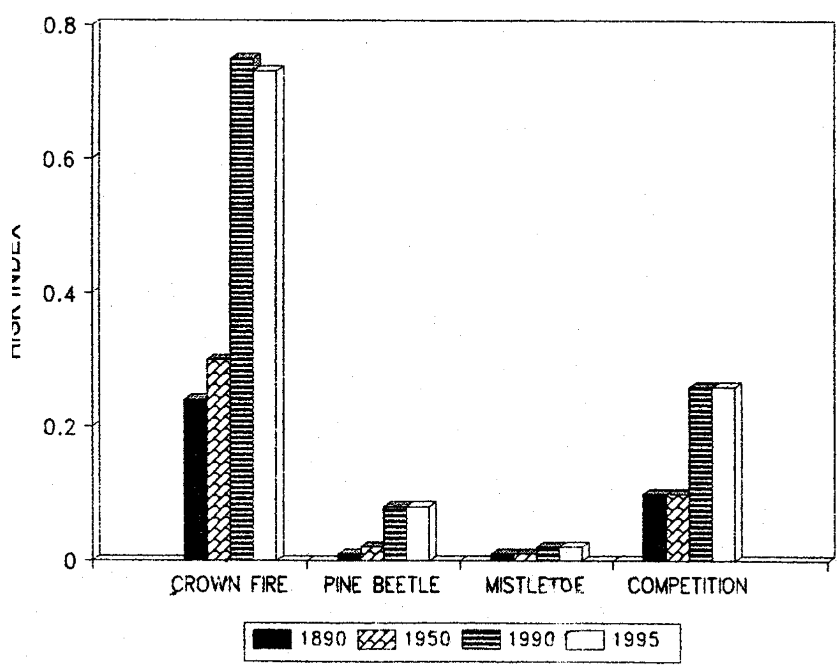


**Figure 10--Change in risk indexes for the Trail Creek landscape.**

An important aspect of assessing ecosystem process is the characterization of disturbance regimes across long time spans. Such information provides an idea of the range of conditions that could occur, given the limits of basic site conditions. We conducted such an analysis by using a model that projects stand growth and fire probabilities based on the ELU information provided in table 1 and the event risk information of figure 10 (Hann and Nygaard 1991). Results of this analysis were summarized for a 900-year period, allowing succession and fire probabilities to interact without interference. In 1930, fire suppression was initiated in our model simulation. The result was a shift in behavior from ground fire to crown fire as fuels accumulate (fig. 11). Plant and animal species have adapted to a certain range of natural disturbances. The rectangular boxes of figure 11 encompass this range. When fire behavior shifts substantially outside this range, certain species that have adapted closely to the disturbance, or associated composition and structural attributes, will be negatively affected. This important concept has implications to the conservation of biological diversity.

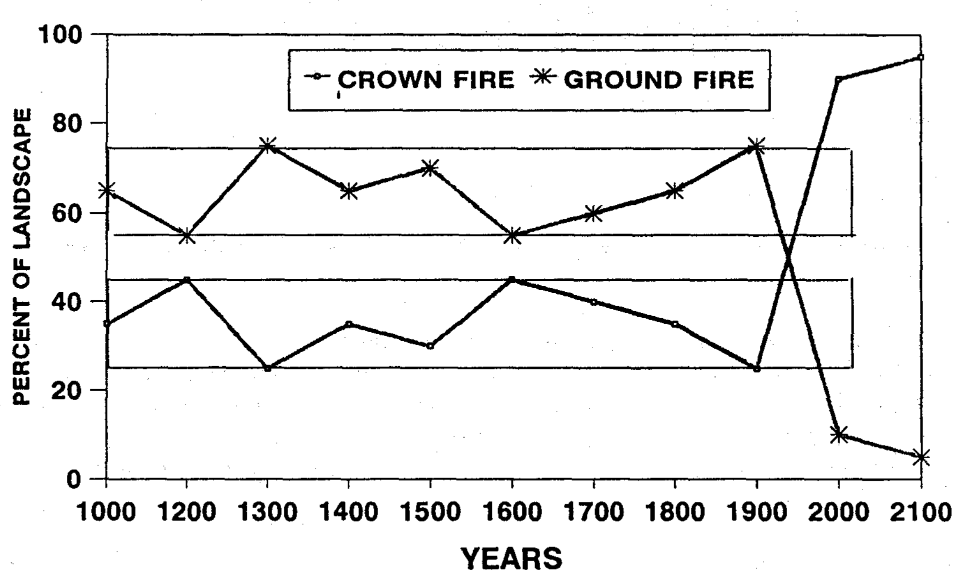


**Figure 11--Change in ground and crown fire regimes In the Trail Creek landscape.**

Hunter (1990) recommends a hierarchical strategy of conserving ecosystem processes, landscape diversity, community diversity, species, and genetic diversity as a general approach to biodiversity conservation. The assessment of ecosystem processes and landscape conditions has been covered in the previous discussion. A general assessment of community diversity was accomplished in this analysis through comparison of diversity indexes (fig. 12). The values presented in figure 12 were summarized for presettlement and postsettlement community types by using sample plot data and analysis methods of the ECOPAC system (Jensen et al. 1993). The dominance index for the total analysis area increased slightly from presettlement conditions to postsettlement time (fig. 12), which is due (in part) to the increase in mature lodgepole pine. The Shannon-Weaver index measures number of species and evenness of species abundance (Magurran 1988). This value decreased from presettlement to postsettlement conditions which was probably due to increases in lodgepole pine and subalpine fir in both forests and natural openings. The increases in structural diversity (fig. 12) indicate an increase in layers. This change is associated with the removal of ground fires in the study area that maintained open, parklike lodgepole pine stands, with simple two- or three-layer communities. Richness of vascular plant species has decreased from presettlement conditions to postsettlement time as fire dependent species, such as Scouler's willow (Salix scouleriana); Bicknell's geranium (Geranium bicknellii), and aspen declined. Diversity indexes are neutral indicators most appropriate for interpreting changes in community composition and structure (Magurran 1988).

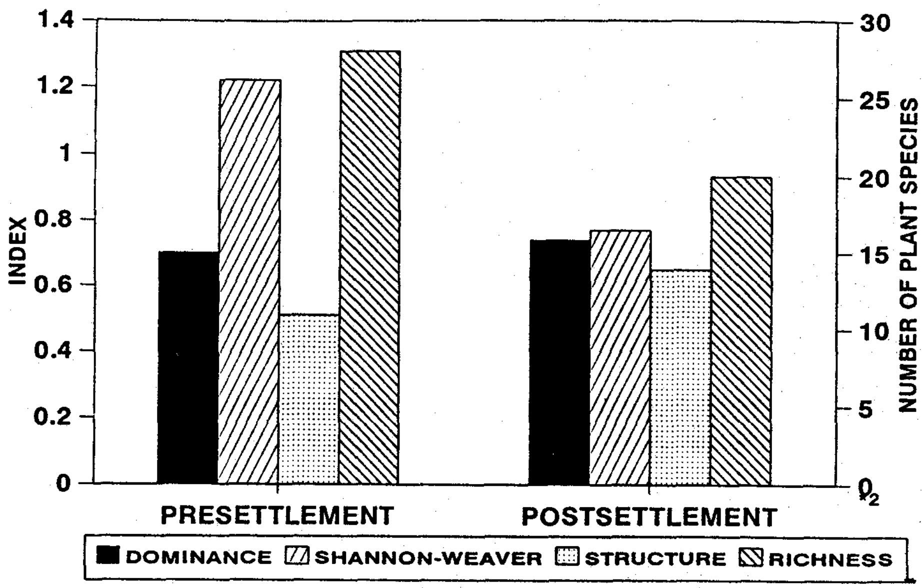


**Figure 12--Comparison of community diversity for the Trail Creek landscape.**

## CONCLUSION

Ecosystem management emphasizes sustainability (i.e., managing for sustainable ecosystem processes while producing a sustainable flow of values and resources for humankind). A principle ingredient in achieving both kinds of sustainability is the conservation of biodiversity. Our approach to conservation of biodiversity is based upon a solid understanding of the major aspects of ecosystem composition, structure, and particularly processes that operate across landscapes.

This paper describes a systematic process and set of techniques for assessing and characterizing ecosystem and landscape conditions and processes. The information presented should help land managers develop better landscape-scale and project-scale treatments and strategies for the maintenance of natural ecosystem conditions and processes.

## ACKNOWLEDGMENT

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# Multi-Scale Ecosystem Analysis and Design in the Pacific Northwest Region: The Umatilla National Forest Restoration Project

**A.J. Shlisky**


## ABSTRACT

This paper illustrates an application of the ecosystem structure-function model to understand and set objectives for landscape patterns. The subject area is the North Fork John Day River basin in the Umatilla National Forest, where long-term fire suppression has created conditions substantially different from those that would be present under a natural disturbance regime. Altered disturbance regimes have predisposed many ecosystems in the area to extensive damage from spruce budworm and other insect defoliators. A landscape analysis and design process is used to determine the natural landscape patterns and processes for the area; set objectives for future landscape patterns and prioritize watersheds for restoration; and develop project proposals that would accomplish restoration objectives.


## INTRODUCTION

Over the past 20 years, forests of the Blue Mountains in northeastern Oregon have been subjected to increasing damage by drought, fire, insects, and diseases. Restoration of these ecosystems has become a priority for affected Federal and State agencies, and private landowners. One restoration effort, applied on the North Fork John Day River (NFJD) basin of the Umatilla National Forest, used a landscape analysis and design process to evaluate current ecological conditions of ecosystems against their natural ranges of variability and develop priorities for restoration. As a first step, a largely ecological approach was taken, allowing for the integration of Forest Plan direction and for social and economic factors at a later time. The approach provides a framework for implementation of ecosystem management principles in other applications.

The strength of the process used for the NFJD project is its use of landscape ecology-based principles (Bourgeron and Jensen 1993). The project:

- Incorporates the natural sustainability of ecosystems in is foundation for objective setting.
- Incorporates landscape analysis procedures to address spatial, ecological, and management issues, allowing for future incorporation of social, economic, and Forest Plan objectives.
- Analyzes relations among ecosystem elements across a landscape, as well as relations between management activities and natural successional processes to drive the design and implementation of ecologically sound projects.

## Project Objectives

The NFJD project area encompasses about 913,000 acres, 551,000 acres of which are administered by the Umatilla National Forest. The basin is composed of 20 watersheds, ranging in size from 19,200 to 94,400 acres. The primary objectives of the project were to rank the 20 watersheds within the NFJD basin by their restoration needs and develop a list of ecologically based restoration activities. In general, the planning process was made up of three components: landscape analysis, ranking of watersheds by restoration need, and activity planning. Figure 1 illustrates how the process used here fits into the larger scheme of Regional and Forest planning.

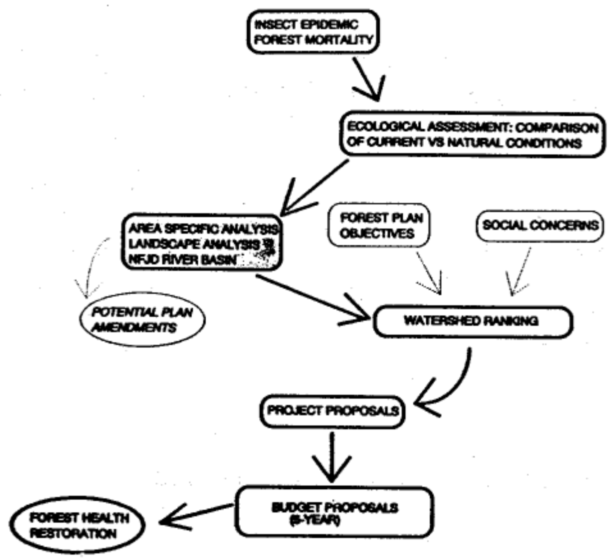


**Figure 1--The role of landscape analysis and watershed ranking for the NFJ13 Restoration Project in the context of Regional and Forest planning.**

## Landscape Analysis

Diaz and Apostol (1992) provide an applicable and effective method for landscape analysis and design for meeting resource management objectives, and-this process, in part, was followed for the NFJD project. In particular, the following elements from Diaz and Apostol were identified for the NFJD basin:

- Landscape structure (patch types, corridors, and matrix) and pattern
- Landscape flow phenomena
- Ecosystem functions for landscape flows.
- Natural successional and disturbance processes

In addition, to facilitate watershed ranking and project development, the following interactions were evaluated as part of the landscape analysis:

- The relations between natural successional or disturbance processes and human-caused disturbances
- The interaction between restoration activities in different patch types

## Patch Types, Corridors, and Matrix

The basic structures or elements of landscapes are patches, corridors, and matrix (Forman and Godron 1986). The characteristics and arrangement of these elements within a landscape determine how well an area functions as a system. The matrix is the most connected portion of the landscape (i.e., the most contiguous vegetation type), and through is relative area and pattern, it provides connectivity of habitat. Corridors are more linear landscape elements, which may function to connect patches.

A list of patch types and corridors identified for the NFJD basin project follows. The full list of 34 patch types was filtered to remove those defined by management direction as too small to be analyzed or not important enough at the time for meeting the forest health restoration objectives of the project.

The matrix of the basin is generally mature forest: multistory true fir (Abies grandis) at higher elevations and parklike ponderosa pine (Pinus ponderosa) at lower elevations. Patch types and corridors include:

| | |
|---|---|
| riparian corridors | old-growth stands |
| ponds/lakes/reserviors | grasslands-scablands |
| current late seral parklike stands | grasslands with juniper |
| late seral tolerant multistory stands | meadows |
| ponderosa pine, high density, low vigor | early seral stands |
| lodgepole pine, high density, low vigor | aspen |
| potential late seral parklike stands | roads |

## Flow Phenomena

Flow phenomena are those iems that use, or flow across, the landscape. Flow phenomena analyzed for the NFJD restoration project included:

| | | | | |
|---|---|---|---|---|
| water | wind | fire | elk and deer | sheep and cattle |
| people | fish | insects | birds and bats | large woody debris |
| sediment | seed | nutrients | carpenter ants | |

Analysis of insect flow focused on insects that have contributed to extensive tree death within the basin (e.g., spruce budworm and pine beetle). Carpenter ants were separated from the insect category to focus on their role as natural predators of spruce budworm. Large woody debris was assumed to be a limiting factor in the basin, and small woody debris was assumed to be less limiting and hence could be omitted as a flow phenomena for analysis at this scale. Birds and bats were lumped based on their similar roles as insectivores. Identification of flow phenomena and their relation to landscape structures facilitates the understanding of landscape-scale function.

## Structure and Flow Interactions: Landscape Functions

To apply a basic ecosystem structure and function model to the NFJD landscape, a matrix of interactions between patch types and flow phenomena was built to determine how the landscape functions as a whole. For the NFJD project, the focus was on the most important ecological functions to avoid excessive detail and remain within the scope of analysis. Where data were available, landscape structures and flows were mapped as overlays to aid in determining spatial interactions.

Five main types of ecological function were recognized:

- Capture - resources are brought into the system (e.g., through migration, photosynthesis). In table 1, "capture" is defined as a function of meadows and riparian areas because these habitats may attract animals from outside the system.
- Production - resources are manufactured within the system (e.g., through plant growth, reproduction, providing food or habitat).
- Cycling - resources are transported within the system (e.g., local migration).
- Storage - resources are conserved within the system.
- Output - resources leave the system (e.g., through migration, erosion).

Table 1 provides an example of some of the ecological functions described for patch types in the NFJD project area.

**Table 1--Subset of Patch Type-Flow Phenomena Interactions Described for the NFJD Restoration Project**

| | ------------------------ FLOW PHENOMENA ------------------------ | | | |
|---|---|---|---|---|
| Patch type | Elk and deer | Water | Fire | Fish |
| Meadow | Forage, capture, production | Storage, filter, cycle | Fire break | Production |
| Riparian | Water, travel, capture, production | Long-term storage | Water source | Production storage, travel |
| Parklike stands | Travel, optimal cover, forage | Long-term storage | Low intensity | ----- |

## Succession and Disturbance: Natural vs Human-Caused

To clarify management objectives for ecosystem restoration and to determine treatment priorities, relations among current ecosystem conditions, natural ranges of variability, natural successional processes, and human-caused disturbances were described. Natural ranges of variability of landscape elements were used as an ecological baseline for the development of restoration objectives and priorities for management.

The sustaining ecological systems approach was used to provide methods to describe the natural ranges of variability of landscape structures for any area (USDA Forest Service 1992a). The sustaining ecological systems approach is based on the concept that when systems are pushed outside the bounds of natural variability (Swanson et al. 1993) there is substantial risk that biological diversity and ecological function will be jeopardized and, therefore, ecological systems may not be maintained. Using the sustaining ecological systems process, a panel of resource specialists formed by the Regional Forester (Pacific Northwest Region) developed a broad assessment of current and natural ecosystem conditions. For selected landscape structures in the Blue Mountains, the range of conditions (or “natural range of variability”) that existed before effective fire exclusion and timber harvest was used as a baseline for natural ecosystem conditions against which current conditions could be compared (USDA Forest Service 1992b). Results of this analysis were used in the NFJD project. Determination of the natural range of variability of landscape elements for other applications can be accomplished by following methods from the sustaining ecological systems process (USDA Forest Service 1992a).

For the project area, the period from the mid-1800s to mid-1900s was accepted as a picture of natural successional change. Since the 1950s, a substantial increase in fire suppression efforts, harvest activity, and other effects altered natural ecosystem processes and structure. For the NFJD basin, for example, as determined by the Regional Forester’s panel, the natural range of variability of late seral ponderosa pine parklike ecosystems was between 50 and 70 percent of the basin in the true fir zone (parklike ecosystems are seral to fir in the lower elevations of the true fir zone in this region). Currently, parklike stands make up about 15 percent of the basin. Conversely, late seral multistory fir stands naturally made up between 20 and 50 percent of the true fir zone in the basin. Currently, these ecosystems make up close to 60 percent of the basin found in the true fir zone.

Successional and disturbance processes that maintain natural landscape structure and function also were identified. The generalized effects of current management activities, or human-caused disturbances, were then compared to the effects of natural processes. Some human-caused disturbances or management activities mimic natural successional processes (e.g., prescribed fire in an open grassland prone to shrub encroachment but naturally susceptible to frequent fire) and help maintain landscape elements within their natural range of variability spatially and functionally. Other human-caused disturbances may move elements farther from their natural ranges of variability (e.g., dispersed clearcuts in a landscape molded by long fire intervals and large fire sizes). By identifying human-caused disturbances that either mimic or hinder natural processes, actions prescribed to restore ecosystems could be designed or selected based on ecological principles.

Table 2 shows examples of natural disturbance processes, human-caused disturbance, and the generalized effects of these processes for two patch types from the NFJD restoration project.

**Table 2--Natural Disturbance Processes, Human-Caused Disturbances and Their Relative Effects on Two Landscape Elements in the NFJD Basin**

| | |
|---|---|
| **PARKLIKE PONDEROSA PINE STANDS** | |
| *Natural disturbance processes* | *Effects on patch type* |
| Fire | maintains (low-intensity, high-frequency fire) |
| *Human-Caused Disturbances* | |
| Fire suppression | reduces amount of patch, alters structure and composition |
| Harvest | reduces amount of patch |
| Grazing | reduces light fuels and fire spread, reduces amount of patch |
| **RIPARIAN CORRIDORS** | |
| *Natural disturbance processes* | *Effects on patch type* |
| Fire | alters species composition, introduces large woody debris and sediment, increases stream temperature |
| Streambank Erosion | recruits sediment and large woody debris |
| Windthrow | introduces large woody debris |
| Floods | maintains stream structure and fluvial processes |
| *Human-caused disturbances* | *Effects on patch type* |
| Harvest | reduces large woody debris source, disturbs streambank structure, compacts soil, creates sediment source, increases dead and down fuel (can increase fire intensity and duration), increases stream temperatures, increases peak water flows and decreases summer low flows, changes nutrient cycling, alters vegetation composition and structure |
| Grazing | alters species composition and structure, redistributes nutrients, increases streambank erosion, increases water temperatures (removal of vegetative cover), increases soil compaction, decreases water quality |
| Fire suppression | increases fuel loading |
| Road construction | increases sediment and stream temperatures, decreases bank stability, decreases area, obstructs flow phenomena (e.g., fish, large woody debris) |
| Water impoundments | captures sediment, obstructs flows, concentrates grazing and browsing, increases water temperature |
| Recreational development | decreases area |

## Patch Interactions

To integrate broad restoration objectives between patch types, the landscape-wide interactions between patch types and potential restoration activities were evaluated. Although the patch type-flow phenomena matrix (table 1) addressed interactions between patches and flows, thereby inferring landscape functions, a more thorough evaluation of functional and responsive interactions between patch types provided an opportunity for greater resource integration for use in refining management objectives and establishing priorities.

Two general concepts drove the evaluation of patch type interactions. First, it was assumed that the objectives for management of any given patch type were based at least in part on a desire to move toward natural ranges of variability for those patch types. Second, it was assumed that management treatment of any one patch type could affect the accomplishment of restoration objectives of another patch type. An understanding of these effects provided a foundation for integrated watershed prioritization and project development, which allowed for selection of restoration activities that are efficient and effective at the landscape scale.

Table 3 illustrates two examples from the NFJD project. The objectives shown represent a desire to move ecosystems closer to their range of natural variability. Other objectives, obtained from public input into the project plan (scoping) or the Forest Plan, will need to be addressed at watershed planning scales.

**Table 3--Interactions between Patch Types--Effects of Meeting Ecological Objectives for Two Patch Types In the NFJD Basin**

| PATCH TYPE | OBJECTIVES | TOOLS* |
|---|---|---|
| Grasslands with juniper invasion | Reduce invasion of grasslands by juniper | Fire<br>Girdle juniper<br>Cut juniper |
| | POSITIVE EFFECTS | NEGATIVE EFFECTS |
| | Increase grasses and shrubs, which may reduce effects on riparian corridors | Reduce juniper-type habitat |
| PATCH TYPE | OBJECTIVES | TOOLS |
| "Potential" parklike (currently invaded by fir) | Reduce area<br>Convert to parklike | Fire<br>Harvest<br>Reduce fuels |
| | POSITIVE EFFECTS | NEGATIVE EFFECTS |
| | Increase connectivity and amount of parklike patches<br>Increase forage outside riparian areas, increasing riparian shrub cover<br>Reduce fuels<br>Reduce host habitat for infesting insects | Loss of connectivity of potential parklike stands<br>Increase amount and size of early seral patches<br>Affects timing of water flows across large areas<br>Increase riparian shrub cover due to decreased competition (lower fuels) |

In other applications, ecological landscape information from the previous steps could be used to create a desired landscape design range for integration of management objectives (i.e., decision space), social issues, and economic concerns (Oliver et al. 1993). For the NFJD basin, landscape design was beyond the scope of the project. Watershed prioritization was the major objective, and information about landscape structure and function was used to develop a watershed prioritization strategy, refine restoration objectives, and develop sound tools for treatment.

## Watershed Ranking

Based on the extent of timber mortality in the Blue Mountains, there was a desire to prioritize areas needing treatment within the next 5 years and to identify areas to emphasize for the next 5-year budget horizon. The concept of "key" ecosystem elements (those indicative of forest health) along with the evaluation of patch type interactions described above, created the foundation for developing an integrated strategy to prioritize areas for restoration in the short term.

For each landscape or region, there are key landscape structures or functions that can be used as indicators of forest health. For the NFJD project, key ecosystem elements were identified by the Regional Forester's panel (USDA Forest Service 1992b). The landscape elements identified as key indicators of forest health in the NFJD basin, based on current conditions and natural ecological processes (USDA forest Service 1992b), were early seral, late seral parklike, and late seral tolerant multistory stands in the true fir zone; ponderosa pine and lodgepole pine (Pinus contorta) stands of high density and low vigor; available fuels; grasslands invaded by juniper (Juniperus occidentalis); riparian shrub cover; and bank stability.

For the NFJD basin, it was assumed that the highest priority key elements for management treatment were those furthest from their natural range of variability. The three key elements furthest from natural conditions were late seral parklike stands, available fuels, and riparian corridors (shrub cover and bank stability, specifically).

By using the patch interactions described above, effects were evaluated of treating each patch type to meet its respective ecological objectives on these three priority ecosystem elements. Through this process, patch types where treatment would bring the greatest return (i.e., have the most positive effects on treating the three priority elements) were identified. The results of this comparison are shown in table 4 as a relative ranking of key elements.

**Table 4--Patch Type Ranking, (see USDA Forest Service (1992b) for descriptions of key landscape elements)**

| Ranking | Key landscape elements |
|---|---|
| 1 | Potential parklike ponderosa pine stands (currently with a large fir component as a result of fire suppression) |
| 2 | Late seral tolerant multistory (fir), killed by spruce budworm |
| 3 | Parklike ponderosa pine stands |
| 4 | Riparian corridor |
| 5 | Late seral tolerant multistory (fir), live |
| 6 | Lodgepole pine, high density and low vigor |
| 7 | Grasslands being invaded by juniper |
| 8 | Early seral stands |

Ranking of key landscape elements was made based on those considered to have the greater positive effects for restoring the three high-priority elements listed above. Then, watersheds within the NFJD basin were prioritized by their apparent opportunity for the most gain; the 20 watersheds within the NFJD basin were ranked based on their proportion of each of the key elements. Key elements were weighted by their rank (1 having the greatest weight), and these weights were multiplied by the acreage percentage of each patch type within each watershed, and the results were summed by watershed to come up with a watershed "vegetation score." Lack of data on riparian vegetation by watershed resulted in the use of a "watershed condition" rating that served as an indicator for the riparian corridor element. This watershed condition rating was based on pools per mile, stream temperature, ground cover, seral stage of dominant riparian vegetation, road density, and an erosion risk factor. The ranking of watersheds by the vegetation score was further refined by using the watershed condition rating for each watershed. Watersheds with the highest ranking became the highest restoration priorities in the short term based on ecological principles.

## Activity Planning

A list of restoration tools then was developed for use in the NFJD basin. These tools were ecologically based and represented an emphasis on restoring natural ranges of variability in ecosystems, using natural successional processes and disturbance regimes and the effects of human-caused disturbances on these processes. To facilitate the development of restoration tools, patch type objectives were refined. The direct and indirect effects of each tool on accomplishment of restoration objectives were analyzed to aid in the future selection of projects for implementation. Prioritization of areas provides the basis for scheduling activities on a site-specific basis.

The refinement of restoration objectives, and the analysis of direct and indirect effects of activities (i.e., tools) on the achievement of these objectives are illustrated in table 5 for one key element in the NFJD basin. Refinement of restoration objectives clarified the ecological basis for activity planning, and provided a wide range of activities that could be implemented to address multiple objectives. By identifying the effectiveness of all tools on patch type restoration objectives, projects can be selected that are beneficial to the widest range of objectives.

**Table 5--Effects of Various Activities on Meeting Management Objectives for Potential Parklike Stands that Currently Support Fir Species due to Fire Suppression**

| | Objectives* | | | | |
|---|---|---|---|---|---|
| Activity | 1 | 2 | 3 | 4 | 5 |
| intermediate harvest | D** | I | I | - | I |
| group selection | D | D | I | - | I |
| seeding | - | - | D | I | I |
| understory burn | D | I | I | - | D |
| chipping | D | I | - | - | I |
| yard or pile material | - | D | - | - | I |
| timber stand improvement | D | I | - | - | D |
| tree and shrub planting | - | - | D | D | I |
| exclosure | - | - | D | I | I |
| soil compaction treatment | - | - | I | D | - |
| natural fire planning | I | - | I | - | D |
| allotment plan modification | - | - | D | I | I |
| road obliteration | - | - | I | D | - |
| natural recovery | I | D | D | I | I |
| spraying | - | - | - | - | - |
| fertilization | - | I | I | D | - |

* 1. Remove understory
2. Protect existing larch and ponderosa pine
3. Reestablish native vegetation
4. Improve soil productivity
5. Reestablish natural (historical) fire regimes

** D = direct effect and I=indirect effect of an activity on meeting an objective

## CONCLUSIONS

Principles of landscape ecology, ecosystem conditions relative to their natural range of variability, and ecosystem interactions were used to prioritize areas and refine objectives for ecosystem restoration within the North Fork John Day River basin. The analysis process described provides the framework for developing ecologically based management objectives that may be further refined by social, economic, and Forest Plan objectives at smaller scales of planning. Emphasis on the evaluation and use of natural ranges of variability of ecosystem elements in this process does not imply that these ranges are necessarily desired conditions. Managers can identify situations where management objectives require that ecosystem conditions be maintained outside ranges of natural variability; however, the effects of such actions on landscape function and long-term management costs must be described. Ecosystem management evolves the balancing of commodity production objectives with maintenance and restoration of ecosystem health. The analysis process presented assists this goal by placing ecological function and structure in the forefront of planning and project prioritization, when historically it has been more of a mitigation concern.

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# SECTION 5 - IMPLEMENTATION STRATEGIES

## Summary of Implementation Strategy Papers

This section contains a series of papers that describe logistical, economic, social, and theoretical considerations that should be addressed in the development of implementation strategies for ecosystem management. The first paper (Morrison) provides recommendations on how ecosystem management principles may be incorporated into the planning process of the USDA Forest Service. Morrison states that the agency's current approach to planning follows regulations specified in the National Forest Management Act of 1976, which reflect a traditional agricultural production model of forest management. Such regulations disaggregate complex forest and grassland ecosystems into single resources (e.g., water yield and indicator species) which are addressed as constraints on the dominant economic use (e.g., timber production). This disaggregation of ecosystem components into competing resources and the cumbersome economic modeling used in forest plan development has undoubtedly contributed to the unprecedented controversy concerning management of National Forests that has ensued since the mid-1980s. Fortunately, forest planning is a dynamic process that involves plan implementation, monitoring and evaluation, and amendment and revision. Each of these phases of the planning process are reviewed by Morrison, who suggests that the following actions be considered: (1) emphasize landscape ecology and conservation biology principles (section 2, this document) in Forest Plan Implementation-Integrated Resource Analysis efforts; (2) use the Integrated Resource Analysis process to share information with the public regarding ecological conditions and trends, and mutually develop desired landscape condition descriptions (see O'Hara et al., this document); (3) coordinate broad-scale analyses (regional, multiregional) with all affected parties, and report findings through the monitoring and evaluation phase of the planning process; and, (4) based on regional and forest-scale monitoring results, implement modifications and revisions to forest plans incrementally by ecologically delineated landscape units (see Hann et al., this document).

The second paper of this section (Grossarth and Nygren) builds upon Morrison's concepts by specifically addressing 11 issues that relate to the implementation of ecosystem management through the Forest Service planning process. These authors review established planning processes and techniques appropriate to ecosystem management implementation and also suggest new technologies and processes that should be considered. Given the fact that the forest planning process is the cornerstone for ecosystem management implementation, the suggestions offered in the paper are of particular value to land managers.

The third paper of this section (Hann et al.) offers specific suggestions concerning the use of hierarchical principles of landscape ecology in the forest planning process. These authors review basic ecosystem process and landscape ecology concepts, and discuss how these concepts are appropriate to the maintenance of biological diversity and ecosystem sustainability. The authors suggest that such-information can only be incorporated into land assessment if the scales of planning and ecological information are correlated and integrated.

Hann and others present a systematic hierarchy for ecological delineation of analysis area boundaries useful to mufti-scale planning efforts. These authors also discuss the use of ecological mapping units (Bailey et al., this document) and maps of current conditions (e.g., existing vegetation) in environmental assessment efforts. A comparison of their approach to land evaluation and traditional forest planning efforts demonstrates how concepts of ecosystem process, landscape ecology, and conservation biology may be incorporated into future planning efforts. These authors conclude that: (1) analysis area delineations for National Environmental Policy Act and National Forest-Management Act planning exercises should be based on ecological criteria; (2) planning assessments (at all scales) should emphasize an understanding of ecosystem processes and how they affect the composition, function, and structure of landscapes; (3) planning assessments should be based on ecological hierarchy theory (see Bourgeron and Jensen, this document) and should identify the range of conditions that influence system development; (4) management should emphasize representation of key ecosystem processes in landscape design; and (5) more emphasis should be placed on multi-regional scale ecological characterizations and assessments that are applied through smaller scale planning efforts (regional, forest, and project plans).

The fourth paper of this section (Ervin and Berrens) discusses critical economic issues related to the implementation of ecosystem management on National Forests. The authors state that sustainable ecosystem management involves the integration of ecological and economic theory. Key literature from a new subdiscipline of economics, ecological economics, is reviewed to illustrate the relation between economic system health and environmental health. Ervin and Berrens describe a framework for economic assessment of ecosystem management that requires specification of a system's minimum natural capital stock (i.e., the minimum biological and physical conditions that describe a sustainable path of products and services), and requires estimation of the benefits and costs of alternative ecosystem resource combinations that satisfy those conditions. Their economic analysis process allows the land manager to combine resources in a variety of ways to identify specified ecosystem values that have the lowest cost to current and future generations.

The fifth paper of this section (Montgomery) describes how perceptions and preferences of risk influence public demands on forest management policy. Montgomery suggests that the uncertainty of ecosystem management (both in product output and public demands) necessitates that land managers design policy options with the flexibility to respond to changes in social perceptions of risk and new biological and technological knowledge (i.e., adaptive management). Given the fact that public forest land managers are mandated to manage public lands for the "permanent good of the whole people," Montgomery argues that traditional economic strategies that allow private agents to respond solely to market incgntives may not provide optimal solutions to the needs of society. In terms of risk, the traditional market outcome fails in the following ways: (1) private agents pay too much for the abatement of private risk and too little for abatement of public risk; (2) private agents emphasize abatement of current risk rather than future risk; and (3) private agents accept too much risk in loss and too little in gain. Implementation of ecosystem management will reverse such allocation of risk. For example, minimization of the risk to future losses of such nonmarket forest resources as clean water and biodiversity may be achieved by managing within natural variability ranges. Because ecosystem management involves a transfer from abatement of private risk to abatement of public risk and a transfer of risk to loss from the future to the present, it appears to be an improvement over management by private agents. The socioeconomic ideas presented in Montgomery's paper are very appropriate to implementation of ecosystem management principles in the land management planning process.

Lippke and Oliver provide the sixth paper of this section, which describes a straightforward economic tradeoff system that could be used in ecosystem management. These authors suggest that nonmarket ecosystem products (such as biodiversity) are not well considered in most traditional economic systems because values are commonly not assigned to items that lack immediate usefulness. Global economic systems naturally seek to obtain the most production from regions with the greatest productivity, causing an extreme imbalance in biodiversity at the local scale. Given this fact, Lippke and Oliver propose that economic incentives (instead of regulations) are required if nonmarket ecosystem values are to be maximized on both public and private lands. The authors provide an example of timber management strategies from eastern and western Washington to illustrate how a tradeoff system could be implemented and become part of an efficient market system.

The seventh paper of this section (Daniels et al.) describes techniques that can be effectively used in the resolution of social conflicts that may arise from ecosystem management implementation. These authors draw upon the following sources of information in their examination of the potential for more collaborative approaches to forest policy formation: an examination of public participation and its ability to deal with conflict in general terms; an inventory of recent collaborative natural resource decisionmaking efforts in the Pacific Northwest; and a framework for understanding natural resource conflict. They suggest that previous public participation efforts of the Forest Service have often failed due to the self-reliance of the Forest Service in making decisions (i.e., as the size and complexity of natural resource issues increases, the feasibility of a single agency making adequate decisions decreases). Public participation is often structured as internal-external (us versus them) relations and occurs in a fairly rigid format. The agency commonly approaches participation exercises from a defensive stance by developing "bulletproof" decisions to avoid appeals and litigation.

Daniels and others recommend that collaboration (i.e., a process in which interdependent groups work together to affect the future of an issue of shared interests) is most appropriate to the resolution of social conflicts that may arise from ecosystem management. They suggest that collaboration differs from the traditional public involvement model of the Forest Service in the following ways: it is less competitive and more accepting of additional parties because they are viewed as potential contributors; it is based on joint learning and fact finding; it allows underlying value differences to be explored and joint values to be identified; it resembles principled negotiation based on its focus on interests rather than positions; it allocates the responsibility for implementation to as many parties as the situation warrants; and it is an ongoing process.

Adaptive management is reviewed in the eighth paper of this section (Everett et al.). These authors indicate that ecosystem management is an untested experiment in managing human environments and societal needs while sustaining ecosystem values for future generations. Accordingly, adaptive management is appropriate to the Forest Service because it promotes partnerships between society, land managers, and scientists to facilitate incremental refinement of ecosystem management strategies based 6n changing socioeconomic conditions and increased ecosystem understanding. It is based on the premise that uncertainties in estimating biological constraints and future societal desires limit our current ability to draft long-term ecosystem management strategies. Risk increases with levels of uncertainty and is recognized as part of the operating environment of adaptive management. The authors suggest that adaptive management recognizes risk and uncertainty by maximizing opportunities for informed decisions to redirect management. Adaptive management systems are highly accountable to defined goals and processes, despite their flexibility. They require quantitatively explicit hypotheses of their iy$tem structure and function, a clear statement of objectives, and a set of actions. The system not only tests that goals are being achieved but also inspects the validity of the basic assumptions used in the management design. Everett and others conclude that National Forest Systems and Forest Research agencies should evaluate the full potential of adaptive management (its strengths and weaknesses) in the management of sustainable ecosystems.

The last paper of this section (Oliver et al.) provides an integrated system for implementing ecosystem management that incorporates many of the ideas and suggestions presented elsewhere in this document.

# Integrating Ecosystem Management and the Forest Planning Process

**J. Morrison**

## ABSTRACT

The USDA Forest Service's land management planning process is the essential mechanism for translating the policy and concepts of ecosystem management into action. Traditional planning principles emphasized individual resources and provided only limited consideration of spatial relations. Ecosystem management requires that these principles be modified to include more effective consideration of landscape patterns and processes. Each of the three phases of the planning process (implementation, monitoring and evaluation, and amendment and revision) provide opportunities for integrating ecosystem management concepts into agency decisionmaking and public involvement procedures. The integrated resource analysis procedures of the implementation phase provide excellent opportunities for integrating landscape pattern and process considerations into Forest Service decisionmaking. Existing monitoring and evaluation procedures should be modified to more explicitly address broad-scale, muttiforest, ecological, and social factors. Integrated resource analysis and monitoring and evaluation provide the informational foundation for proposals to amend and revise forest plans to better integrate ecosystem management principles.

## INTRODUCTION

The USDA Forest Service's recent announcement of a new management philosophy that emphasizes an ecological approach to management of the National Forests and Grasslands perhaps represents the most important policy development for the Forest Service since expansion of the timber management program after World War II (Steep 1976). This policy announcement occurred during an explosion of conferences and publications attempting to define the concepts of ecosystem management (e.g., Keiter and Boyce 1991, Swank and Van Lear 1992), and its underlying scientific disciplines of landscape ecology and conservation biology (Bourgeron and Jensen 1993, Golley 1993). Portions of the American public also have increasingly used ecological concepts and emerging scientific principles to express their desires and concerns for the management of National Forests and the quality of their environment. This synergism of scientific developments and changing public values has inspired a series of judicial opinions, administrative actions, and legislative proposals that suggests a need for Federal agencies to give greater consideration to the ecological goals and consequences of their actions. These trends led to the announcement of a new Forest Service ecosystem management policy (Robertson 1992) on June 4, 1992.

The Forest Service has challenged itself to integrate this policy and its general principles (Overbay 1992) into the actual management of the National Forests and Grasslands. The essential mechanism for this transition is the land management planning process of the Forest Service. This planning process, defined by the National Forest Management Act (1976), and substantially influenced by the National Environmental Policy Act (1969), governs nearly all land management decisions of the Forest Service--from the broad strategic program of the Secretary of Agriculture to specific project decisions of District Rangers. To fully implement ecosystem management principles on National Forests and Grasslands, the principles must become an integral part of the Forest Service planning and decisionmaking process.

The background section of this paper briefly reviews the principles and procedures used to develop existing forest plans, and the limitations they pose for ecosystem management implementation. Sections on integrated resource analysis and monitoring and evaluation offer a two-part strategy for integrating ecosystem management into land management planning and decisionmaking. This strategy was suggested by the Chief of the Forest Service in his June 4, 1992, policy announcement, where he stated that Forest Service employees "need to take advantage of the flexibility within existing forest plans to practice ecosystem management. As forest plans need to be amended or revised they should reflect the . . . policy on ecosystem management" (Robertson 1992). As the first part of this strategy, the section on integrated resource analysis describes a process for incorporating ecosystem mangement principles into the identification and design of individual project propbsals that implement existing forest plans. The monitoring and evaluation section outlines possible planning procedures for evaluating ecological and social functions that operate at a forest-wide or multiforest scale.

## BACKGROUND: CURRENT PRINCIPLES OF FOREST PLAN DEVELOPMENT

The National Forest Management Act of 1976 is the principal statutory source of procedural guidelines and substantive standards governing the development and maintenance of forest plans. The central themes of the National Forest Management Act are the integrated consideration of multiple uses and values, regulation of timber management, and public participation in Forest Service decisionmaking (Gippert and DeWitte 1990). The Secretary of Agriculture promulgated regulations based on these statutory provisions most recently in 1982. These regulations (36 CFR 219), along with subsequent Forest Service directives, prescribe a detailed and complex process for the development and maintenance of forest plans (Wilkinson and Anderson 1985).

Several provisions of the planning regulations directly relate to the integration of ecosystem management and the forest planning process. First, the regulations establish separate planning requirements for each resource (36 CFR 219.14-.24). Second, the regulations prescribe detailed procedures for determining resource production potential, assessing the economic tradeoffs between different uses or values, and measuring the economic efficiency of alternative resource production mixes (36 CFR 219.12(e)-(g)). The primary focus of these requirements is the timber resource (Johnson 1992). Third, the regulations require that forest plans establish habitat objectives for individual "management indicator species" of fish and wildlife (36 CFR 219.19). Finally, the regulations require the preparation of regional guides-that provide objectives for individual resources, establish limitations on timber management methods and intensities, and address major regional management issues (36 CFR 219.9).

These characteristics of the current planning regulations reflect the traditional agricultural production model of forest management (Kennedy and Quigley 1993, Kessler et al. 1992, Shepard 1993). The current planning regulations disaggregate Forests and Grasslands into single resources, and biological communities are simplified into individual vertebrate species. The hydrologic system is represented as water yield and sediment production (36 CFR 219.23). These separate resources, and others such as cultural and historical resources (36 CFR 219.24), are addressed as constraints on the dominant economic use (Johnson 1992). Forest plan alternatives are evaluated based on the competing economic value of these individual resources. These procedures are designed to ensure that a primary consideration in forest plan development is the relative amount of projected output of these independent, competing resources.

This decisionmaking model embodies a long-established view of natural resource management and policy analysis (Majone 1989). The resulting forest plans are more comprehensive and based on more citizen involvement than any in the Forest Service's history. Nevertheless, the Forest Service's experience in developing over a hundred forest plans has exposed some substantial limitations of these procedures (USDA-Forest Service 1990).

The cumbersome economic methods of modeling and evaluation contributed substantially to the long timeframe (more than 5 years) and high average cost ($2 million) required to complete many forest plans (GAO 1987). The complexity of these procedures frustrated effective public involvement (USDA Forest Service 1990). Moreover, the disaggregation of Forest and Grassland elements (e.g., timber, wildlife, and water) into individual, competing resources for allocation may have contributed to the unprecedented controversy over National Forest management that has ensued since the mid-1980s (Daniels et al. 1993).

Some of the limitations of existing planning procedures have carried over into Forest Service efforts to implement forest plans. The emphasis on economic tradeoff analysis detracted from the Forest Service's ability to thoroughly address potential environmental consequences during forest plan development. The principal analytical tool used in forest plan development (the computer model FORPLAN) is limited in its ability to address spatial relations that are critical in evaluating potential environmental consequences (Johnson 1992; Turner 1993). Similarly, the linear programming structure of FORPLAN permits only very rough estimates of the effects on commodity output that result from standards and guidelines designed to maintain or protect nontimber resources. The responsibility for evaluating potential environmental effects, including cumulative effects, and for determining conformance with standards and guidelines has fallen to Ranger District personnel in their individual project proposal activities. These considerations, combined with further public comment, frequently lead decisionmakers to select alternatives that are smaller, or less intensive, than anticipated by the forest plan. As a result, the output projections of forest plans have often proven to be unrealistically high and the costs of project planning have skyrocketed.

Fortunately, forest planning is a dynamic process. Currently, most National Forests have completed development of their initial forest plan and are now primarily involved in the other phases of the process: implementation, monitoring and evaluation, and amendment and revision. Each of these planning phases encourages decisionmakers to consider new information and technologies, advances in our understanding of ecological, social and economic relations, and changing public demands. The following sections of this paper outline how the emerging concepts of ecosystem management can be integrated into this dynamic process.

## INTEGRATED RESOURCE ANALYSIS AND FOREST PLAN IMPLEMENTATION

Forest plan implementation is the prevailing planning activity on most National Forests. Implementation is the identification and design of permits, contracts, and other site-specific actions that will promote achievement of forest plan goals and objectives. Forest Service administrative Regions and National Forests have developed general procedures to ensure that the goals and objectives for all resources and values are given full consideration during forest plan implementation. These procedures are commonly referred to as integrated resource analysis. It is these procedures that offer the most immediate opportunity to merge many ecosystem management principles into the design of specific land management actions.

Although integrated resource analysis procedures differ among Regions and National Forests, they all share some basic elements. Such analyses usually are conducted at a landscape scale of 10,000 to 100,000 acres. Typically, the analysis procedures include evaluation of existing environmental conditions; description of various forest plan goals, objectives and standards (or desired conditions) for the area; identification of management practices that may move the landscape from the existing condition toward the desired condition; and a preliminary assessment of the potential environmental consequences of the management practices. The result of these procedures is identification of specific project proposals subject to the public involvement requirements of the National Environmental Policy Act (1969).

Several modest modifications to these general analysis procedures can greatly facilitate the integration of ecosystem management principles into forest plan implementation. First, many concepts and methods of landscape ecology and conservation biology (Bourgeron and Jensen 1993) can enhance the evaluation of existing conditions at the landscape scale. For example, employees of the Pacific Northwest Region have described methods (based on principles of landscape ecology and environmental design) for the effective consideration of spatial patterns, such as those affecting animal movement or the spread of fires, during project design (Diaz and Apostol 1992). This evaluation of patterns and processes provides integrated consideration of forest and grassland ecosystems and encompasses more scales of biological organization than the individual resource and species approach that has previously characterized the forest planning process (Hutto et al. 1987, Ricklefs et al. 1984). The consideration of patterns and processes also is more likely than the individual resource approach to reveal unaccepted threats to human values, such as increased risk of large fires affecting private property, risks of decreasing visual quality that may be associated with unprecedented insect and disease events, or risks of losing some fish or wildlife populations associated with certain landscape patterns.

An interdisciplinary team from the Forest Service, Northern Region, has described integrated resource analysis procedures for evaluating existing ecological conditions organized around the categories of ecosystem composition, structure, and function. This analysis framework also promotes a more complete consideration of basic ecological characteristics and processes during implementation than was achieved during forest plan development (USDA Forest Service 1992a). Furthermore, the procedures defined by this team supplement the evaluation of existing ecological conditions with an assessment of historical disturbance regimes, successional pathways, and the resulting variation in historical landscape composition and structure. For many landscapes, this evaluation framework provides an extremely useful context for determining the ecological trends associated with the existing landscape condition and its future trajectory (USDA Forest Service 1992b).

This broader temporal frame of reference helps the public and decisionmakers evaluate the probability that management objectives (as quantifiable, measurable goals) can be sustained over time. Research on historical fire regimes and successional patterns in the northern Rockies, for example, indicates that objectives to provide large contiguous acres of late seral forests cannot be maintained in some areas because they will become increasingly susceptible to insect and disease events of unprecedented geographic scale and intensity, and large, high-intensity crown fires that exceed human control capabilities and that seldom, if ever, occurred under historical disturbance regimes. Thus, the consideration of historical disturbance regimes in the integrated resource analysis process should result in management objectives that are more likely to be achieved and to avoid substantial, unintended ecological consequences.

Recent events and analyses in the Blue Mountains of eastern Oregon demonstrate the unintended consequences of establishing management objectives without considering historic disturbance regimes (USDA Forest Service 1991). These approaches to integrated resource analysis also may identify ways to design economic activities that simulate ecological processes, rather than disrupt them (O'Hara et al. 1993). These approaches also offer promise (in some instances at least) of reducing the often-perceived conflicts between preservation and use. These and other efforts to incorporate ecosystem management principles into the integrated resource analysis process can result in several tangible products that systematically integrate ecosystem management and the forest planning process.

## Ecosystem Management-Based Project Proposals Consistent with the Forest Plan

In many instances, applying landscape ecology based analysis procedures during the integrated resource analysis process, can result in project proposals consistent with forest plan goals and objectives as well as with ecosystem management principles. Forest plans are programmatic in that they establish goals, objectives, standards, and guidelines that often are general. Accordingly, the public and Forest Service personnel have flexibility in interpreting how forest plan decisions apply, or can best be achieved, at a particular location. In addition, forest plans typically do not specify the precise timing, location, or other features of individual management actions. Consequently, the plans provide opportunities for incorporation of ecosystem management principles into the identification and design of site-specific activities. For example, forest plans leave considerable discretion in the choice of silvicuftural system, logging method, and to a lesser extent, the size of harvest units. This discretion, in many instances, allows managers to propose activities that are consistent with the forest plan, based on a consideration of existing and historical ecological characteristics of the landscape. Such activities may also be designed to maintain or restore desired elements of landscape composition, structure, or function.

Ecological information and principles applied through integrated resource analysis procedures offer abundant opportunities to collaborate with other agencies, organizations, and scientists to evaluate historical, existing, and future characteristics of landscape composition, structure, and function. Perhaps more importantly, the discretion that forest plans reserve for site-specific decisionmaking provides managers with often underused opportunities to collaborate with the public. Managers should take advantage of the integrated resource analysis process to share with the public information regarding ecological conditions and trends. Managers also should take advantage of the discretion in forest plans to understand and respond to the public's desires and concerns in light of emerging ecological information. General consensus and acceptance are more likely if the public can participate in the identification and design of specific proposals, and not merely react for the first time to proposals designed exclusively by the Forest Service (Daniels et al. 1993).

## Ecosystem Management-Based Amendment Proposals for Geographic Areas

Integrated resource analysis procedures that apply ecosystem management principles also offer the opportunity to propose and evaluate forest plan amendments for specific geographic areas. Even though forest plans often provide flexibility for the identification and design of project proposals, normally they do not explicitly express or encourage many of the emerging ecosystem management principles. Forest Service decisionmakers, however, can use the integrated resource analysis process to describe how these principles can be incorporated into forest plans for geographic subunits of the forest.

Amendments can propose the delineation of geographic areas comprised of a group of contiguous management areas having similar ecological and social characteristics. This addition to the hierarchy of forest plan management direction is consistent with the ecosystem management principles of considering the hierarchical nature of ecosystems and evaluating management consequences at multiple spatial scales (Bourgeron and Jensen 1993). Several forest plans currently delineate geographic areas but do not establish unique goals and objectives for them (e.g., USDA Forest Service 1985). To better integrate ecosystem management and the forest planning process, forest plans may be amended, based on the results of integrated resource analysis and associated public involvement, to define goals that describe the ecological and social functions and purposes of management for specified geographic areas. In addition, decisionmakers could propose objectives that define the desired landscape pattern and structures to be maintained or achieved. The consideration of such potential objectives would be improved by an analysis of historical disturbance regimes (Swanson et al. 1993) and the resulting variation in landscape patterns experienced in the geographic area. These objectives form the basis for defining the precise steps necessary to work toward the goals and objectives for the geographic area. This approach promotes forest plans by explicitly addressing critical ecological processes and social values, in addition to the current emphasis on production of specific resources.

### Ecosystem Management-Based Monitoring, Evaluation Reports, and Database Development

Often, the information developed during integrated resource analysis will have broader application than just the identification and design of possible management activities, or the assessment of possible forest plan amendments. The analysis may identify ecological processes that operate at broader spatial scales or are poorly understood. Such information needs to be carefully preserved, shared with the public, and considered for forest plan amendment or revision. Annual monitoring and evaluation reports provide an excellent opportunity to preserve such information and present it to the public for consideration. This monitoring information and the public’s response to it, can be systematically preserved for later consideration or for later forest-wide or multiforest amendment proposals.

Integrated resource analysis procedures also allow databases to be incrementally supplemented with information essential for large-scale applications of ecosystem management principles. For some ecological variables, such an incremental approach to database development may be much more efficient, and therefore more feasible, than immediate attempts at extensive, high-resolution inventories.

These suggested applications of the integrated resource analysis process of forest plan implementation provide a foundation for integrating ecosystem management and the forest planning process. Such procedures gradually build information bases, and provide opportunities for greater dialogue between the Forest Service, other agencies and organizations, and scientists. The relatively small spatial scale of most integrated resource analysis efforts may reduce the complexity of ecological considerations so they are more easily understood by decisionmakers and the public than efforts to address simultaneously all potential aspects of ecosystem management at forest-wide or multiforest scales. In time, this increased understanding of ecological conditions and trends, and public desires and concerns, can be extrapolated across broad geographic and decisionmaking scales.

## MONITORING AND EVALUATION OF LARGE-SCALE ISSUES AND INFORMATION NEEDS

Despite its advantages, the spatial scale of integrated resource analysis presents some limitations in a thorough integration of ecosystem management and the forest planning process. Many ecological and social functions operate at spatial scales larger than those considered in the typical integrated resource analysis (e.g., dynamics of some plant and animal populations, geomorphic processes, air quality, and atmospheric processes). These large-scale issues and the analysis needed to address them frequently are deferred to integrated resource analysis procedures and project-scale decisionmaking. The analysis and evaluation of all ecological considerations at a single decisionmaking point often overwhelms the capabilities of Forest Service analysts, decisionmakers, and the public. The extensive spatial scale, analysis complexity, and amount of information often is simply too much to assimilate at one time. To partially alleviate these difficulties, the Forest Service could apply the concepts of ecosystem hierarchies (Bourgeron and Jensen 1993, Turner et al. 1993) to break these issues and analysis needs into logical pieces, and address them at the appropriate geographic and decisionmaking scale. These procedures present a more practical and efficient approach to integrating ecosystem management principles into forest planning than attempts to address all issues at one time at one decisionmaking scale.

Analyses at larger spatial scales are necessary to adequately address many ecosystem processes. Often, this will require coordination of analysis efforts among National Forests, Regions, and other land owners and land management agencies. Coordinated analyses of large-scale issues must become the norm, rather than the exception (Daniels et al. 1993).

Monitoring and evaluation of the forest planning process is a logical avenue for implementing muftiscale analysis procedures. Inventory and analysis efforts designed to address individual or small sets of ecological issues could provide many of the same opportunities for collaboration with other agencies, organizations, scientists, and the public as described for the integrated resource analysis process. These types of analyses also could result in the same type of tangible planning actions as identified by traditional integrated resource analysis efforts.

## Ecosystem Management-Based Forest-wide and Multi-Forest Amendment Proposals

The inventory and assessment of individual, broad-scale, ecological issues can be integrated into the forest planning process through monitoring and evaluation procedures. Annual reports commonly explain ongoing or planned analysis efforts, and are used to present results to decisionmakers and the public. These reports offer a mechanism for distributing analysis results for public review. These reports also ensure the effective preservation of information and its evaluation for future consideration.

The annual reports of monitoring and evaluation, and subsequent public comments, can help identify possible amendments to forest plan goals and objectives. Proposed amendments might suggest adjustments to existing forest-wide goals for maintaining or restoring large-scale ecological processes and rare ecosystem elements. In addition, such proposed amendments could suggest objectives that define in general terms the desired forest-wide or muftiforest landscape pattern to be maintained or achieved during the planning period. Public review of annual monitoring and evaluation reports, along with public consideration of the specific amendment proposals, ensure that the needs of people are fully considered in the decisionmaking process.

Some amendments to forest plans could be proposed for two or more National Forests simultaneously. Many characteristics of ecosystem composition, structure, and function can be addressed effectively only by considering land use patterns at spatial scales encompassing several National Forests and other jurisdictions and ownerships. For example, white pine blister rust (Cronartium ribicola) and recent alterations in fire regimes pose substantial risks to the distribution and abundance of whitebark pine (Pinus albicaulis) (Hann et al. 1993). Simultaneous amendment of multiple forest plans that encompass the scale of white pine blister rust, or other ecological characteristics or processes, would help coordinate management direction among agencies and land ownerships.

Simultaneous amendment proposals for multiple forest plans also could greatly improve the efficiency of the planning process. Higher level administrative units, such as Regional Offices, could conduct or coordinate much of the work necessary to evaluate these large-scale issues and alternative response strategies. This would relieve National Forest and Ranger District personnel from this burden and allow them to focus on ecosystem management needs that are best addressed at local scales.

## Foundation for Integrated Resource Analyses and Forest Plan Revision

Supplementing databases and presenting large-scale analysis results in annual monitoring and evaluation reports will greatly improve the efficiency and quality of integrated resource analysis efforts, and will help document environmental consequences of site-specific proposed actions required by the National Environmental Policy Act. If carefully prioritized, large-scale analyses could substantially reduce the burden on project-scale interdisciplinary teams to conduct numerous cumulative effect analyses. Most analyses needed to address possible cumulative effects could be met by incorporating into National Environmental Policy Act documents of projects the large-scale analyses presented in annual monitoring and evaluation reports.

Compiling of proposed amendments from integrated resource analyses and ecosystem management-based monitoring can facilitate forest plan revisions which are ordinarily done every 10 years. In 1991, the Forest Service issued an Advanced Notice of Proposed Rulemaking in which the Agency presented possible changes to existing planning regulations. This document proposed reforms to simplify and shorten the process used in major forest plan amendments and revisions. The preamble to the Advanced Notice of Proposed Rulemakng describes a process for major amendments and revision that allows incremental changes to a plan. In addition, the document proposes elimination of many, of the detailed economic tradeoff analyses currently required for major amendments and revisions. These suggested changes to the planning regulations may help the Forest Service to incrementally incorporate ecosystem managementbased goals and objectives into forest plans. The changes could reduce the complexity, procedural burden, and financial costs that would result if the Forest Service addressed the vast number of social and economic issues all at once at the scale of an individual National Forest.

Under this approach, the Forest Service would not need to reconsider in detail recently completed amendments arising from ecosystem management-based integrated resource analysis and monitoring efforts. Proposed revisions would focus on areas of forest plans that need to be changed as identified through the evaluation of monitoring results. Obviously, forest-wide and geographic area considerations not previously addressed through ecosystem management-based amendments should be the specific focus of proposed revisions. Data from integrated resource analyses and monitoring, previously shared with the public, would provide much of the foundation for the revision process.

## CONCLUSION

Recent scientific developments, evolving public expectations, and the Forest Service's announcement of its ecosystem management policy have created an undeniable demand for change in the way National Forests and Grasslands are managed. The challenge the Forest Service has imposed on itself is to respond quickly and effectively to this growing demand. The transition from general policy, principles, and concepts to the actual management of ecosystems, however, cannot be instantaneous for at least two reasons. First, the Forest Service is bound by law to comply with existing forest plans. Although the development of existing forest plans was guided by regulations that recognized ecosystem interrelations (36 CFR 219.1(b)(3)), these plans do not fully incorporate the amount of knowledge and technology embodied in ecosystem management. In addition, the Forest Service's tradition of making every effort to satisfy the expectations of Congress (as expressed in annual appropriations legislation) inhibits immediate disruption of ongoing programs. Therefore, the transition from general policy and principles to specific objectives and actions must be measured and systematic.

This paper presents a relatively simple strategy for integrating ecosystem management principles into the forest planning process. The defining characteristic of this strategy is a systematic and incremental approach for incorporating ecosystem management principles into the three phases of forest planning: implementation, monitoring and evaluation, and amendment and revision. At each of these steps (and their multiple spatial scales), traditional analyses of individual resources and species should be supplemented by thorough consideration of ecological patterns and processes.

Congress (by passing the National Forest Management Act) anticipated these types of changes to the planning process when it found that "the new knowledge derived from coordinated public and private research will promote a sound technical and ecological base for effective management, use, and protection of the Nation's renewable resources." The new ecosystem management policy of the Forest Service illustrates this congressional finding. Effective integration of this policy into the forest planning process represents a new and exerting challenge to the management of National Forests and Grasslands.

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# IMPLEMENTING ECOSYSTEM MANAGEMENT THROUGH THE FOREST PLANNING PROCESS

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## ABSTRACT

This paper describes the ways in which the USDA Forest Service land and resource management planning and environmental analysis processes fit the agency's new ecosystem management approach. Ten planning issues are addressed to describe established processes and techniques that now meet the goals of ecosystem management, and new technology and approaches that will meet those goals: General issues considered in this paper include incorporation of ecosystem perspectives in planning and environmental analysis; use of new technology, public participation, and adaptive management; and inclusion of social and economic factors in the planning process.

## INTRODUCTION

This paper synthesizes some basic concepts of the forest planning process and describes how the USDA Forest Service can, through forest planning, incorporate ecosystem management into its mission. We provide insights and recommendations on 10 topical issues and acknowledge that many of the concepts presented may appear radical to a traditional Forest Service work force. Agreement or disagreement with our synthesis and recommendations is unimportant to the primary objective of this paper, which is to stimulate new and creative approaches to ecosystem management through the Forest planning process. Forest planning is a dynamic process that must incorporate our evolving concepts of ecosystem management. Accordingly, as our understanding of ecosystems improves, we must continue to adapt our planning process and related documents. The following is a discussion of major topics related to ecosystem management and the forest planning process.

### An "Increased Understanding" of Forest Plans and Ecosystem Management

Are forest plans, ecosystem management, and the planning process understood? Do we need "increased understanding"? The answer to these questions depends on the understanding or basic desire of the individual and the tie in the individual's mind between forest planning and ecosystem management. There are perhaps as many "understandings" of forest plans as there are people active in the forest planning arena; therefore, a common definition for forest plans and the ecosystem management that they direct is needed.

Both the National Forest Management Act and the National Environmental Policy Act endorse an ecosystem approach to planning and project implementation.[1] A forest plan represents key decisions for long-term management of a National Forest; consequently, it must assure the diversity, sustainability, and productivity of forest ecosystems and should not be counter to this direction.

Perhaps the Forest Service should reaffirm the importance of ecological perspectives in plan development and clarify the forest plan's role as the cornerstone of good ecosystem management. A clear understanding is needed of the principles of an ecosystem approach[2] and the translation of these principles into plans that assure proper actions on the ground. An "increased understanding" by the Forest Service should incorporate the concept that ecosystems are key factors in all forest plans and the implementation of those plans. Individual projects should consider the entire ecosystem and the total cumulative effects of any proposals.

### Assuring That Forest Plans Address all Relevant Scales

As forest plans are made or implemented, line officers must assure that decisions recognize larger scale issues as well as project-scale considerations. A more detailed analysis of the ecosystem (from the large river-basin scale to the smallest of niches) may be needed at various landscape scales to assure that plans provide the necessary framework for ecosystem management and forest health. Although it is common to think of ecosystems as a range of scales, it is important to realize that social and economic values also occur at (and are different at) a wide range of scales. Listening to the public can provide insights that may lead to an imaginative combination of sustainable habitats, social values, and economic benefits. Such attention will help to assure that forest plans meet local, regional, and national desires and needs.

As broad ecological assessments are made and desired conditions are identified, Forest Service line officers must involve those individuals who manage other parts of the ecosystems. States, tribes, governmental entities, private landowners, commodity users, environmental groups, and individuals all have a part to play in ecosystem management and must be involved in the Forest Service's efforts to implement this management approach. Decisions do not need to be made jointly, however, because other agencies or private landowners do not have to commit themselves to the Forest Service's decisions about their lands. Often, the power of public dialogue will bring about decisions for the common good and will give local landowners a stake in the process. The public needs a sense of how a given National Forest fits into the whole watershed and a feel for the desired condition of private, county, state, and Federal lands.

According to National Forest Management Act regulations, the normal planning unit is a National Forest; however, National Forest boundaries may or may not follow logical ecological unit boundaries (Bailey et al. 1993). The current planning process provides two scales of National Environmental Policy Act decision-making: the forest scale and the project scale. National Environmental Policy Act decision project areas also commonly do not follow ecological units. Ecological classifications (such as plant series and association), watersheds, and land types commonly are used to define ecological boundaries and should increasingly be used to define forest plan and project analysis boundaries (Hann et al. 1993). Ecological units also are useful in developing and scheduling project proposals in integrated resource analysis (O'Hara et al. 1993).

## Incorporating Ecosystem Management Concepts into Forest Planning

Ecosystem management must be a major consideration in any adjustment of forest plans. Four major discussion areas are listed below that describe ways of integrating ecosystem, social, and economic objectives into the design of desired conditions. The concepts presented can be applied to all planning scales (e.g., regional, forest, watershed, and project scales).

- Developing information and an understanding of the nature of ecosystems--If ecosystems are to be managed, the terminology and common characteristics of their elements (e.g., the classification and mapping of existing and potential vegetation, geomorphic features, landscape patterns, and watersheds) must be understood and used in planning efforts. Such information can help assess current conditions versus potential conditions, natural variability, and sustainability of key ecosystem elements.

- Using goals, objectives, and direction for ecosystems--Each planning scale has a role in determining appropriate ecosystem goals and objectives. Forest plans provide long-term, broad-scale objectives and give directions that are used in detail at lower project scales. Goals and objectives can be addressed through planning alternatives and incorporated into Forest plans through standards and guidelines, management prescriptions, land allocations, plan activity schedules, and descriptions of the desired condition.

- Assessing effects on the ecosystem from management proposals--Scientifically accepted methods must be used to estimate the effects of forest management practices on the ecosystem. Studies describing the effects of management practices, landscape processes, and land uses on vegetative composition, ecosystem function, and natural variability are particularly appropriate to planning. Such information serves as a foundation for other types of effects analysis, such as species viability, tree growth, and water quality.

- Monitoring and evaluating ecosystem conditions--During plan implementation, key indicators are monitored to measure changes in ecosystem condition and to evaluate progress toward achievement of the desired condition. All three levels of monitoring (implementation, effectiveness, and validation) should be used to assess current ecosystem conditions and suggest adjustments to the plan.

## Desired Future Conditions for Ecosystem Management

Desired future condition is a concept used in National Forest management following direction specified in the following documents:

National Forest Management Act regulations (219.11) state “The Forest Plan shall contain the following....” (b) “Forest multiple-use objectives that include a description of the desired future condition of the forest or grassland....”

The draft Forest Service Manual states that desired future condition “portrays the land or resource conditions which are expected to result if the goals and objectives are achieved. The function of the desired future condition is to enhance understanding of the ultimate intent and result of implementing the goals and objectives and the associated management area prescriptions; standards and guidelines designed to achieve those goals and objectives; and as a communication tool. It may be described at the management area, total forest or some other geographical scale.”

The desired future condition represents the condition of the forest to be achieved through long-term plan implementation. The National Forest Management Act regulations refer to desired condition among the Forest plans goals and objectives; however, desired condition is neither a goal nor objective, but a long-term condition resulting from meeting plan goals and objectives. Desired future condition serves as a communication tool that describes a vision of the long-term condition of the land.

To be useful, the desired future condition must be a realistic vision and be consistent with the goals, objectives, standards, guidelines, and activity schedules contained in the plan. This vision must be socially acceptable, economically and technologically feasible, and ecologically sustainable (Jensen and Everett 1993). Accordingly, planning analyses should determine if the desired future condition meets these criteria, if plan direction will actually lead to the desired state, and if planning direction will lead to other consequences contradicting the description of the desired condition.

The desired future condition identifies a range of indefinitely sustainable conditions. Determining the bounds of the desired condition often requires identifying undesired conditions (e.g., what conditions could conflict with or negate the long-term goals and objectives described in the plan). A knowledge of landscape potential also is critical to the development of desired condition visions that promote long-term sustainability (Bailey et al. 1993). To obtain the desired future condition, existing conditions must be analyzed and actions taken to move to interim conditions that will eventually lead to the long-term desired future condition.

Desired future condition is a useful concept when contrasted with similar descriptions of the forest at other points in time. A description of the current or existing condition highlights the changes needed to meet the desired future condition. The expected conditions of the forest after the plan period (10 years) or plan horizon (50 years) can be used to highlight the changes expected from plan implementation.

## Incorporating Ecosystem Management in the National Environmental Policy Act Process

The National Environmental Policy Act process, when used correctly at the forest or project scale, can provide analysis, disclosure, and public involvement to help achieve desired ecosystem management. Federal agencies are required under the National Environmental Policy Act to “utilize ecological information in the planning and development of resource-oriented projects” (Section 102(2)(H), NEPA). The required analysis should include a description of ecosystem factors within the forest environment and disclose the effects of the alternatives on the productivity, diversity, and sustainability of the ecosystems.

In describing the environmental elements that wilt be substantially affected by proposed management alternatives, the interdisciplinary team should place those elements in the context of their ecosystems, their current conditions, and sustainability of those ecosystems.

The following steps should be used in any proposed forest or project plan to analyze the relation of the elements of affected ecosystems:

1. Identify the scale(s) most relevant to understanding the expected changes in an environmental element.

2. Discuss the range of natural conditions of an environmental element in its ecosystems.

3. Discuss the role of disturbance in these ecosystems, and the effect that natural or human-caused disturbance events have had on this environmental element.

4. Discuss the current and potential sustainability and productivity of the ecosystems involved. Discuss the current, prior, and prospective diversity of the ecosystems, and the role of diversity in their productivity and sustainability.

5. Disclose whether current conditions and the estimated effects are or will be within the range of historical conditions, and their contribution to the sustainability and productivity of the ecosystems and their diversity.

Descriptions of existing, expected intermediate, and desired future conditions can be made for various geographic analysis scales (e.g., regional, forest, watershed, and project site), as well as other planning attributes (e.g., management area, ecological type, and species range). Forest conditions should be described hierarchically, with higher scales addressing larger areas and longer time horizons. Portions of higher scale descriptions can be applied to the next lower scale of analysis. Lower analysis scales generally provide more detailed descriptions at finer spatial and temporal resolution.

## Using Adaptive Management to Increase the Flexibility of Forest Plans

Land managers and the public recognize that Forest plans must be updated or fine tuned more frequently than originally anticipated. Plan adjustments must be ongoing to reflect changing public concerns and needs. Generally, these adjustments can be made as a series of smaller actions through nonsignificant amendments. This type of adjustment strategy is called adaptive management because it attempts to adapt to ongoing situations instead of waiting for them to develop into major issues, which are harder to address or solve (Everett et al. 1993).

Forest Service line officers historically have been judged on their ability to produce targeted outputs. Under adaptive management, the line officer would be less concerned with production of a specified target and more concerned with using the flexibility of the Forest plan to reflect the current and changing needs of the ecosystem. This situation highlights the distinction between the environmental objectives of the Forest plan and the estimated amount of derived outputs from implementing those objectives. Accurate resource projections should be made, but plans should explicitly state that these projections are estimates and give a range of possible outcomes. Similarly, the Forest plan must allow the line officer the flexibility to respond to unforeseen events, such as a catastrophic fire or insect epidemic.

Adaptive management requires use of new tools and information which were not available a few years ago. Powerful computerized models can help define needed changes and options; however, these projections must be continually updated and revised. No single model, such as FORPLAN, can totally represent anything as complex as a National Forest. Several new tools enable managers to use adaptive management analysis of Forest plan data in a realistic time projection. Sometimes, a number of simple computer models working together in an integrated and changeable framework can be used to produce adaptive plans. These tools have a wide range of editing or analytical power so that a subunit or total area can be tested for a change in distribution of the objectives, a change in standards and guidelines, various thresholds, or exclusion of certain areas or types of information. Additionally, the ripple effects on other subunits within the total area may be studied by such tools.

## State-of-the-Art Computer Tools for Ecosystem Management

Over the last few years, many new advanced technologies have become available that enable line officers to make improved decisions on natural resource management. Many of these state-of-the-art computers and programs are appropriate to ecosystem management implementation because of their excellent quality, speed, ease of set-up, and operation. Such programs can do sensitivity testing, risk analysis, and probability analysis in minutes and hours, instead of days or weeks as in the past.

Some state-of-the-art programs are produced commercially; however, many are developed by Forest Service employees to meet their day-to-day needs. Examples of commercially available programs include:

Risk Analysis tools:

"Lightyear" from Lightyear Inc., Santa Clara, CA 95051
"Risk" from Palisade Corporation, Newfield, NY 14867
"Crystal Ball" from Decision Engineering Inc., Boulder, CO 80302

Simulation tools:

"DYNAMO" latest version by Stephan Boyce, Duke University, Durham, NC 27706
"VIP" Windows Version 2.0, University of California, Santa Barbara, CA 93102
"Vistapro-2" from Virtual Reality Laboratories Inc., San Luis Obispo, CA 93401

Several barriers remain that slow or hinder the aggressive use of available computer tools within the Forest Service:

- A tendency to do "analysis for analysis sake" rather than "analysis for decision support," with too much new technology being applied to old processes.
- The "more complicated models" syndrome where complex problems become even more complicated due to the technology used in analysis.
- The "one tool will solve all situations" syndrome, which doesn't recognize that several good programs often need to be linked with other good analytical tools to be used effectively.
- An attitude that a database, whether relevant or not, can somehow be jammed into a model and provide good information.

The solution to these barriers is good technology applied with a heavy dose of reality and practical sense. To retool and use state-of-the-art processes will require the following key items:

- Changing management and specialist attitudes concerning decision support services.
- Retraining analysts to build or use different models and to use tools such as simulation and spatial disaggregation.
- Showing how various techniques and tools fit together to address different management problems and objectives.
- Rehabilitating "FORPLAN junkies" to use the improved analytical tools and build simpler but better models.

## Incorporating Social Input into Ecosystem Management

Social (including economic) factors are central to all ecosystem decisions. These social factors must be explicitly stated to ensure that the management of ecosystems better meets social needs within the physical and biological constraints of a system. Two ways of accomplishing this objective are through the expression of preferences and through the analysis of foreseeable effects.

Preferences are reflected through public involvement, the prices paid in markets, and the development and application of laws and regulations (Montgomery 1993). The elements of scale, process, and organization are as relevant in social factors as they are in other aspects of ecosystem analysis. The importance of scale is seen in agreements that waste sites and tree harvest are needed (somewhere), but "not in my backyard." The variables of process (education, conflict resolution or intensification, trust, technology, and value) greatly affect preferences and how they are expressed. The organization of people and markets also affects the way preferences are stated and realized.

The second way of making social and economic factors explicit for ecosystem analysis is through identification of foreseeable effects. Impact assessment (social and economic) is a method that uses social science data and professional expertise to predict the results of management choices,(both for action and non-action). Scale (expressed as equity or distribution of effects), process (the analysis of markets, community coping structures, and history), and organization (markets, communities, and interest groups) also are the relevant factors in the analysis of future effects of human choices about ecosystems.

Impact analysis and the expression of preferences are parts of the Forest Service's planning,and National Environmental Policy Act processes; however, they may need some translation to make their role in ecosystem management clear and useful. Accordingly, greater attention and emphasis needs to be placed on impact analysis and expression of preferences as the principles of ecosystem management are applied to Forest Service plans and projects.

## Balancing Social and Economic Factors Within the Planning Process

Social and economic concerns, along with biological needs, must be balanced within the forest planning process (Jensen and Everett 1993). The following points are appropriate to the balancing of social, economic and biological values:

- Social and economic factors must be considered in the design and development of Forest plans. To incorporate these factors, managers must identify past trends and present socioeconomic conditions of the area(s) involved in the planning project. Analysis of management situations, socio-economic overviews, and environmental impact statements are examples of assessments that identify the social and economic (as well as political and historical) conditions relevant to decisions. Scale of the project or plan is important; whether a small ecosystem project or a multistate ecoregion. The scope of the ecosystem in terms of physical and biological information should first be considered in an analysis, then the effects that people have had or will have on the land should be considered. The effect of land resources on people also must be described.

- Public interests and concerns must be incorporated into the decisionmaking process for plans and projects. Ideally, forest users would be involved in all phases of the planning process, and therefore, would feel more committed to the final outcome. Public input and citizen involvement should not be limited to responding to a draft environmental impact statement. Instead, citizens should have a role in the project or plan from its conception. Additionally, citizens should be actively involved in the entire Forest planning process.

- Social and economic concerns should be incorporated into the evaluation of alternatives for each plan or project (relevant sections of the Forest Service Manual and Forest Service Handbook apply to social and economic impact assessment). Effects need to be fully disclosed to the public in the environmental impact statement so these effects can be considered during decisionmaking.

- Socio-economic factors should be included in the implementation and monitoring of plans and projects. Questions and concerns need to be evaluated concerning the actual impacts to the social and economic environment. The public and public agencies need to be involved in this evaluation, so that effects on local communities will be fully documented. Discussions of employment or money flows through a community are not enough. Instead, changes at regional and national (perhaps even international) scales that may affect communities must be considered. Such iems as interest rates, housing starts, air and smoke regulations, automobile use, industry changes, population shifts, and infrastructures may need to be evaluated.

The incorporation of socio-economic input in the planning process does not ensure balance. Only those involved in final decisions can assure a balance between people and nature, although this balance will likely be of short duration. The mutual sustainability of natural, social, and economic subsystems is a necessary goal that has often not been achieved.

### Using Forest Plan Adjustments as a Forum for Social and Economic Input for Ecosystem Management

Forest plan adjustments will be required in the near future to respond to forest health concerns, wildlife habitat needs, and changing public values relating to forest issues. Such adjustments can provide a unique forum for social and economic input to ecosystem management; however, this will be accomplished only by changing how social and economic information is gathered and used in the planning process.

Current adjustments to Forest plans generally incorporate social and economic considerations into the planning process by assessing how the forest affects people: this process identifies historical, demographic, economic, and social trends; classifies the public into affected groups; estimates each group's future wants and needs; and predicts how different segments of the public will respond to the goods, services, and uses associated with various management alternatives. Using this model, line officers often make decisions that require the community to change. This approach gives the appearance that the Forest is acting on people, rather than people acting on the Forest.

A result of past planning processes has been confrontation among user groups, and between national and local interests. Many conflicts were not resolved during the last round of Forest planning, nor will they be adequately resolved by plan adjustments if current approaches are used. Care is required to keep such arguments from becoming "humans versus nature" conflicts as ecosystem management is implemented. An improved approach to plan adjustment would use social and economic input as descriptors of the existing ecosystem condition.

Ecosystems are in their present condition because humans have used forests to fulfill their desires, needs, and values at local, regional, and national scales. These desires have influenced timber harvesting, grazing, fire prevention, recreation, preservation, and other forest uses. Existing ecosystem conditions can be evaluated for appropriateness given current social needs. Selected social and economic input can provide a direct link between ecosystem assessment and management.

## CONCLUSIONS

The Forest Service needs to emphasize ecosystem awareness and understanding and apply those concepts on the ground. Ecosystem management is a term that has different meaning to different individuals. Because of this disparity, more emphasis should be placed on developing common definitions and understandings of ecosystem management, both internally and externally.

The following factors should be considered in the application of ecosystem management concepts:

- Adaptive management means adapting to new information that may be obtained by monitoring and evaluating what is being done, how it is being done, and why it is being done, and then determining if changes are needed.

- Forest plans should be examined for possible adjustments that would facilitate better use of adaptive management concepts.

- Adapting is insufficient in itself to solve the problems of tomorrow. Research on basic ecosystem processes at various scales may provide information that will help avoid problems.

- Adaptive management must recognize that both ecological trends and events can lead to a need for change in Forest plans.

- A systematic approach can overcome problems associated with fragmented and piecemeal monitoring. A diagnostic approach to measuring (monitoring) the ecosystem may be required.

- A coordinated monitoring effort involving other land managers can increase the efficiency of landscape analyses at various scales.

- Research must regularly be coordinated with plan development and application. The entire Forest Service must become less fragmented to ensure that researchers and managers work effectively together.

- Ecosystems include humans and, therefore, include all values held by people. Such values must be considered in planning processes and management decisions.

- Reward people who provide excellent monitoring, evaluation, research, development, and applications and those who learn and provide excellence from what they learn. We need to celebrate excellence gained from learning.

## FOOTNOTES

[1] Federal agencies shall "utilize ecological information in the planning and development of resource-oriented projects" (Section 102(2)(H), National Environmental Policy Act). Forest plans will be based on the "recognition that the National Forests are ecosystems and their management for goods and services requires an awareness and consideration of the interrelationships among plants, animals, soil, water, air, and other environmental factors within such ecosystems" (36 CFR 219.1(b)(3)). The National Forest Management Act also requires the maintenance of diversity and productivity.

[2] See Deputy Chief Overbay's address at the National Workshop on Taking an Ecological Approach to Management, Salt Lake City, April 27, 1992 (USDA Forest Service, Washington, DC). The principles he presented are: diversity and sustainability; dynamics, complexity, and options; desired future conditions; coordination; integrated data and tools; and integrated management and research.

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# Land Management Assessment Using Hierarchical Principles of Landscape Ecology

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## ABSTRACT

This paper provides examples of incorporating concepts of landscape ecology and conservation biology in ecosystem management guidelines. A hierarchical framework for delineation of analysis areas and assessment of ecological conditions is discussed. Examples of assessment within and among scales are provided. Five recommendations are made to land managers regarding the implementation of similar assessments for ecosystem management.

## INTRODUCTION

Ecosystem management includes management for sustainable ecosystems and landscapes (sensu Costanza 1992) that are not in conflict with human values, needs, and demands. Conserving biodiversity is the foundation for sustaining native ecosystems (Hunter 1990). To conserve biodiversity and keep ecosystems resilient, managers need an understanding of how these systems have evolved and developed over recent and geologic time. In our experience, the simplest way to achieve this management goal is to understand ecosystem processes. These processes have shaped landscapes across time, and have been the primary driver of ecological functions and species adaptation.

There is a sound basis for the conservation of biodiversity. Diverse ecosystems and landscapes provide for a wide range of human values, uses, and management options. Species that appear to have no value for humanity may actually have great value in the future (Ehrlich 1988, Farnsworth 1988, Iftis 1988). Examples of values include habitats for other species, indicators of environmental quality, toxin filters, restoration, environmental buffers, genetic sources, and so on. The biotic component of ecosystems is constrained by the range of natural variability of ecosystem processes. By understanding and managing for the range of natural variability, we can provide conditions that will maintain ecological patterns (Bourgeron and Jensen 1993, Swanson et al. 1993).

The primary objective of this paper is to illustrate how some basic ecosystem, community and landscape ecology concepts may be used in assessments of land management options. We accomplish this objective by developing a hierarchical ecosystem-based analysis framework to conduct assessments at all scales of the planning process. We contrast results from these assessments with results from traditional Forest plan and project-scale environmental impact statements. Finally, we provide five recommendations for ecosystem management to land managers.

### Ecosystem Process and Landscape Ecology Concepts

The hierarchical nature of ecosystems and landscapes is central to ecosystem management (Bourgeron and Jensen 1993). A landscape is an ecological system that is iself a cluster of connected, interacting ecosystems (Bailey et al. 1993, Forman and Godron 1986, Turner 1989). A broad ecological system (such as the forest ecosystem of the Rocky Mountains) can contain many landscapes, each made up of clusters of ecosystems of finer resolution. The term “process”, in relation to ecosystems and landscapes, is commonly used when referring to ecological events that cause change. This term has been used to cover a broad array of biological and physical phenomena (Forman and Godron 1986, Ricklefs et al. 1984, Turner 1989, Urban et al. 1987).

For the purpose of this paper, we broadly define ecosystem processes as the biological and physical events that create change within ecosystems and across landscapes at all scales of ecological hierarchies. Processes or events that interrupt or substantially change other processes are called disturbances. Disturbances are characterized by their frequency, intensity, and duration (Oliver and Larson 1990, Pickett et al. 1989, Turner et al. 1993, White and Pickett 1985) For example, disturbances can be high-frequency/low-intensity, low-frequency/high-intensity, with characteristic durations, and any other combination of frequency, intensity, and duration. A good example of a low-frequency, high-intensity, short-duration disturbance is a crown fire that shifts the vegetation from late successional forest to standing dead and down blackened trees with ash on the soil surface (Agee 1990). There is, in this example, a relation among frequency, intensity, and duration. The interval between events is typically long (40 to 500 or more years) because of the time needed to produce the amount of biomass that will carry this kind of fire. An example of a high-frequency, low-intensity, short-duration disturbance is a cool surface fire in a forest that is subject to periodic underburns (e.g., 5- to 15-year intervals). This type of fire typically causes only small changes in species composition and conversion of biomass. The cumulative effect of such high-frequency, low-intensity events on vegetation, however, may be just as pronounced as crown fires. Disturbance characteristics are important to landscape dynamics (Bourgeron and Jensen 1993, Turner et al. 1993).

Ecosystem processes fall into three categories: biotic, hydrologic, and physical. Examples of biotic processes include: herbivory of plant material by vertebrate and invertebrate animals at different trophic scales; insects and diseases that reduce plant vigor, cause plant death, and affect nutrient cycling; movements of animals and their physical and biological effects on energy and genetic flow within ecosystems; and cycling of plant and animal matter through decomposition and soil development. Humans are also part of biotic processes. Hydrologic processes include surface erosion and deposition, floods that change stream channels, springtime rain-on-snow runoff events, snow avalanches, evaporation, streamflow, and subsurface flow. Physical processes include fire, mass-wasting, windstorms, hurricanes, cyclones, drought, high moisture periods, volcanic eruption, and glaciation. Physical, biotic, and hydrologic processes interact with each other. For example, many erosion events in the northern Rocky Mountains are associated with summer rain storms that put out fires in steep terrain. The intensity and size of such erosion events often determine whether plant succession is primary or secondary.

Knowledge of the history of system development is important for ecosystem management (Costanza 1992, Hunter 1990). Ecological systems have developed through time in response to a range of ecological conditions and specific sequences of events, referred to as the range of natural variability (Swanson et al. 1993). When both conditions and events are changed and fall outside the range of natural variability, the result is usually different ecosystems and landscape configurations. These new ecological systems are often called “out of balance” or “unhealthy.” These terms are largely human-centered and infer a value judgment with respect to the desirability of the ecosystems and landscapes. This human-centered component is useful to management (Swanson et al. 1993).

A good example of a system that is out of balance in the northern Rocky Mountains is the western white pine seral forest. In such forests, western white pine (Pinus monticola) has been killed by the introduced disease, white pine blister rust (Cronartium ribicola). For many of the western white pine seral forests, there is no insect- or disease-resistant tree species that can replace western white pine after it dies of rust infection. These forests are usually dominated by root disease-susceptible Douglas-fir (Pseudotsuga menziesii) that eventually die or are burned by wildfire and then replaced by shrubs. Generally, the loss of native species contributes to the shift from one ecological system to another. Other examples of ecological systems that changed state in response to the loss of native species include: subalpine and timberline forest ecosystems that are responding to the on-going loss of whitebark pine (due to white pine blister rust and lack of fire); and changes in the dynamics of wild ungulate populations in response to the loss of large predators (e.g., the wolf). Although the examples are of species that can be studied and the effects of their loss evaluated, we know little about the role of many other species that may be very important to the sustainability of ecosystems. This uncertainty, and the need to conserve our options for the future, provides strong support for the principle that conservation of processes and biodiversity is central to ecosystem management.

## Ecosystem Process and Conservation Biology Concepts

Conservation of biodiversity includes conserving patterns (e.g., species or communities), habitats, and processes (Bourgeron and Jensen 1993, Hunter 1990). A management consequence of the hierarchical structure of ecological systems is use of the coarse- and fine-filter concepts (Bourgeron and Jensen 1993, Hunter 1990). The coarse- and fine-filter concepts assume that management of ecosystems and landscapes (the coarse filter), in ways that represent conditions under which these ecological systems have developed, will maintain patterns of biodiversity without having to manage individually each ecosystem-landscape element, such as species (the fine filter). The choice of the coarse- and fine-filters is a function of the ecological scale of interest, which is itself dependent on the management objective and the ecological system of interest (Bourgeron and Jensen 1993).

The hierarchical structure of ecological systems provides a template for a hierarchical ecosystem management approach. This approach also applies to species and their genetic diversity. Because of loss of habitat and connectivity, and death from human causes, many species have become rare or extinct, or are on the edge of extinction. For these species, we must monitor, evaluate, and manage them at species and population scales. The community, ecosystem, and landscape scale of ecological organization are generally referred to as the coarse-filter conservation strategy. The species, population, and genetic scales are often called the fine-filter conservation strategy (Hunter 1990). Both approaches. are necessary to meet the objective of ecosystem management. Neither approach replaces the other, or is more important, because the approaches complement each other for the conservation of all elements of ecosystems and diversity.

## Hierarchical Planning Scales and Ecological Analysis Area Delineation

To apply landscape ecology and conservation biology to National Forest and Grassland assessments, the scales of planning and ecological information must be correlated and integrated. National Forest and Grassland land management planning and reporting have historically been conducted at multiple planning and administrative scales. These scales include international, national, regional, National Forest, and project scales (table 1). Assessments of environmental effects have traditionally been conducted at the project scale and not at higher scales in the hierarchy. Assessments of ecosystem processes, landscape relations, and biological diversity have traditionally been ignored or dealt with in a cursory fashion, and certainly have not been conducted in association with the various planning scales.

**Table 1--Planning and Administrative Scales In Relation to Ecological Analysis Area Delineation**

| Planning level | Administrative level | Terrestrial delineation | Aquatic delineation |
|---|---|---|---|
| International agreements | Washington DC | Domain | Continental river basin |
| Resource planning assessment | Washington office | Province or subprovince | Regional river basin |
| Regional assessment & policy | Regional office | Section | River basin |
| Forest plan | Regional office/national forest | Landscape analysis area | Watershed group |
| integrated resource analysis | National forest | Watershed | Watershed |
| Project plan | Ranger district | Ecosystem polygon | Stream reach |

In the Northern Region, we have developed a systematic hierarchy for ecological delineation of analysis area boundaries that is applicable to different planning scales. Such delineations are used to bound analysis areas for basic assessments of management alternatives, and to evaluate direct and cumulative effects as directed by the National Environmental Policy Act and National Forest Management Act. The relations between traditional planning scales and ecological analysis area delineations are presented in table 1. The upper scale of the planning and assessment hierarchy is very broad and is used to address global issues between nations in international agreements. Administratively, this scale is handled by cabinet-level government in Washington, DC. Terrestrial analysis area delineations at this scale use some type of ecoregional mapping system, such as domain or division (Bailey 1978, 1982). For aquatic delineations, continental-scale river basins are typically used to bound the analysis area (e.g., eastern North American versus western north american river basin drainages). Map scales used at this scale of planning typically range from 1:10,000,000 to 1:30,000,000.

The next scale in this hierarchy is resource planning assessment. This scale of assessment is conducted at the USDA, Forest Service, Washington Office. The province or subprovince scale of ecological map unit design (Bailey 1978, 1982) is used for terrestrial delineations, and regional river basins are used for aquatic delineations at this scale. The Rocky Mountain forest province and the Columbia River basin are examples of ecological analysis areas that would be used at this scale of assessment. Map scales typically range from 1:2,000,000 to 1:5,000,000.

Regional planning and assessment are conducted at the next scale of the hierarchy. This scale is critical to ecosystem and landscape characterizations. Assessment at this scale should be coordinated between regions or forest zones because they provide a consistent framework for further ecosystem characterization and assessment at the National Forest scale. This scale of analysis is commonly used to evaluate priorities for management, rare species, identification of disfunctioning ecosystems, risk assessment, and identification of research needs. Terrestrial delineations of analysis areas at this scale follow section criteria (e.g., areas of similar regional climate and potential vegetation). Section delineations of the Northern Region have been developed using Arno's (1979) "Forest Regions of Montana" and modifications of Bailey's (1982) ecoregion maps. Mapping is correlated at the Regional Office to provide consistency between Regions and National Forests and ensure linkage with the next higher scale of the analysis hierarchy.

Section maps delineate areas containing similar and predictable variation in productivity, species distribution, disturbance regime, and response to management. Such section maps provide an efficient template for interpreting the monitoring and inventory data collected during smaller scale forest planning and integrated resource analysis efforts. Sections can be aggregated for broader scale assessment. Aquatic delineations at the regional assessment scale follow river basins as defined by the U.S. Geological Survey (1980). Aquatic and terrestrial analysis area map scales usually range from 1:100,000 to 1:1,000,000 in regional scale assessments.

The forest plan scale of assessment and planning is conducted by the Forest Supervisor's Office. This scale involves more than general ecosystem characterization and assessment. Terrestrial analysis area delineations follow landscape analysis areas or groups of landscape analysis areas at this scale of planning. Such areas are frequently delineated by using refined climate zonation, landform juxtaposition, watershed orientation, and valley bottom to mountain top relation criteria. Each mapping unit is characterized by specific regimes of ecological processes. Ecosystem and landscape dynamics are predictable within a landscape analysis area. Aquatic analysis delineations use watershed groups identified by hydrologic unit codes on 1:24,000 to 1:1,000,000 scale maps. Mapping criteria and delineation for terrestrial and aquatic systems are coordinated by the Regional Office to provide consistency between Districts and Forests.

In forest-scale planning, landscape analysis areas are commonly grouped to facilitate descriptions of ecosystem conditions and processes across connected landscapes. Such groupings may cross boundaries of sections (the next upper-scale of the hierarchy) for the analysis of broad issues. For example, all landscape analysis areas in the Pintlar, Beaverhead, Centennial, Sapphire, and South Bitterroot ranges of Montana may be grouped to look at the connectivity of landscapes for grizzly bear travel zones between the northern Continental Divide, Selway-Bitterroot, and Yellowstone grizzly bear recovery areas. Such landscape analysis area groupings are appropriate for analyses that encompass several sections. Landscape analysis area grouping is not commonly used for larger areas because landscape analysis area information is too detailed. Scales of mapping for landscape analysis area groups can range from 1:100,000 to 1:1,000,000.

The next scale of planning and assessment in table 1 is integrated resource analysis. Planning and assessment at this scale are usually coordinated by the Supervisor's Office, but may also include Ranger District personnel. This analysis scale is critical for monitoring and implementing forest plan goals and objectives, and commonly includes description of the kinds and location of projects that will be implemented (Morrison 1993). Integrated resource analyses have traditionally emphasized the evaluation of effects associated with the production of commodity targets (e.g., timber sales, livestock grazing, mineral extraction, and oil and gas leasing). There has recently been a general shift away from this issue-oriented approach. Instead, assessments are now more ecologically oriented and identify multiresource use and ecosystem restoration potential based on sustainable ecosystems criteria.

Third-order watershed boundaries are commonly used to delineate analysis areas at this scale of planning. Landscape analysis areas watershed groups may also be used for more general planning. Refined descriptions of ecosystem composition, structure, and patch dynamics are achieved at this scale of planning. Such information can be, used to successfully predict effects of proposed actions. Environmental Impact Statements or Environmental Assessments may be developed to document project alternatives. Coordination of data, inventories, cumulative effects assessments, and projects is best achieved if terrestrial and aquatic hierarchies are integrated. Mapping scales appropriate to this level of analysis typically range from 1:24,000 to 1:1,000,000.

The project scale of planning is conducted by the Ranger District and is critical to ecosystem management. All planning at the upper scales of the hierarchy presented in table 1 is meaningless if the activities conducted at the project scale are not effective. For terrestrial systems, the analysis area delineation for projects is usually an ecosystem polygon (e.g., vegetation stand, prescribed fire unit, and wildlife improvement unit). For aquatic systems, project areas are commonly delineated by stream reaches, or lake types. Projects implemented at this scale are commonly prioritized in the forest plan or integrated resource analysis scales of the planning and assessment hierarchy based on their potential for maintaining ecosystems. Project assessment, design, and monitoring is done on a site-specific basis (i.e., the map scale of assessment typically ranges from 1:4,000 to 1:24,000). Local cumulative effects are often evaluated at this scale; however, the bounds for cumulative effects analysis depend on the types of attributes being assessed. Most terrestrial and aquatic components are typically assessed within the project area.

The analysis hierarchy presented in table 1 provides a framework for prioritizing landscapes for assessment. The framework also facilitates consistent identification of the type of management direction that may be required for ecosystem maintenance. Information derived from the multiscale analyses of ecosystem conditions is appropriate to forest plan monitoring and evaluation, and may provide a basis for forest plan amendment or revision.

## Characterization of Ecosystem Potential

Each of the ecological analysis area delineations presented in table 1 requires description of biological and physical potential to be of use in planning. A hierarchy of ecological classification and mapping information should be used to describe such potential (Bailey et al.1993). The ecosystem potential hierarchy used in analysis area description by the Northern Region is presented in table 2. This hierarchy was developed by integrating the suggestions of Jensen (1992) and Hann (1992a,1992b), with some modification and additions based on Pfister (1991) and subsequent field testing. The ecosystem potential hierarchies displayed in table 2 are used to describe the inherent productivity, rates of change, dominant ecosystem processes, and successional dynamics of an ecological analysis area delineation.

**Table 2--Relation of Analysis Area to Indicators of Ecosystem Potential**

| Terrestrial analysis area | Potential vegetation | Land system inventory | Ecosystem process |
|---|---|---|---|
| Domain | Class or subclass | Province or subprovince | General biomass cycle and climate regime |
| Province or subprovince | Series group | Section | Rate of biomass cycle and climate variation |
| Section | Series | Subsection | Ecosystem dynamics and cycles of change |
| Landscape analysis area | Series or habitat type group | Subsection or landtype (LT) association | Successional rate structure/age<br>Probability of change |
| Watershed | Habitat type or group | Landtype or LT association | Successional rates<br>Composition/structure/age<br>Probability of change |
| Project | Habitat type or phase | Landtype or LT phase | Specific processes<br>Community dynamics<br>Treatment pathways |

The potential vegetation hierarchy follows the classification system and nomenclature described by Daubenmire (1968), Jensen (1992), and Pfister (1991). At the domain scale, potential vegetation classes (forest) or subclasses (coniferous forest) are used. At the province scale, series groups are used. A series is defined as the dominant upper layer climax species (Daubenmire 1968). Series groups are usually constructed by life zone settings: valley, montane, lower subalpine, upper subalpine, timberline, and alpine. Series level potential vegetation classifications are commonly used to describe section analysis areas, and they also can be used to describe landscape analysis areas and watersheds. Habitat type groups are also used to describe landscape analysis areas and watersheds. At the project scale, habitat types (i.e., climax overstory and understory indicator species), habitat type phases, or ecological sites (Jensen 1992) are used in area descriptions. The following example of a potential vegetation classification illustrates the hierarchical nature of the Northern Region system: forest > coniferous forest > lower subalpine series > subalpine fir series > subalpine fir/grouse whortleberry habitat type > subalpine fir/grouse whortleberry habitat type pine grass phase (Pfister et al. 1977).

Land system inventory is also used to describe ecosystem potential in the Northern Region (table 2). Land system inventory is based on landform, soils, geology, potential vegetation pattern, climate, and hydrologic criteria (Wertz and Arnold 1972, Northern Region 1976). The upper scales of this system (i.e., section and above) are correlated with Bailey's ecoregion hierarchy (Bailey 1978, 1982). These scales emphasize broad regional climatic zones that are further modified by land-surface form (Hammond 1970) or other climatic criteria (Trewartha and Horn 1980, Bailey et al. 1993). In the Northern Region, we have found that refinement of sections is best achieved through recognition of elevation zones in vegetation series distribution (Arno 1979). The subsection scale of the land system inventory emphasizes differences in landforms (Holdorf and Donahue 1990), geomorphic process, parent material, and precipitation. Landtype associations are commonly constructed by grouping similar adjacent landtypes based on interpretation objectives (e.g., hydrologic function, productivity, fire behavior, and succession pathways). Different landtype association maps may be developed for an area depending on specific management needs. Landtypes delineate relatively similar areas in terms of landform and parent material; however, there is often considerable variability in habitat types and soil patterns in such units. Landtype phases are used to refine this variability at the project design scale of analysis. In the Northern Region, land system inventory mapping units have generally proven effective in delineating areas with similar processes (e.g., fire, weather, and hydrologic flow) and local biotic population patterns.

Ecosystem processes are also used to characterize ecosystem potential (table 2). At the domain scale of analysis, biomass cycle is commonly defined in terms of dominant biomass types and rates of change. Domains can then be compared, and areas with relatively rapid rates of accumulation and high probability of disturbance events can be identified. For example, areas with potential for stress from drought would tend to have high probabilities for major fire events if they also had high accumulations of biomass. At the province scale, biomass cycles are correlated with general processes and climate in more refined analysis description. Section analysis scales are sufficiently detailed to permit description of disturbance types, regimes, and cycles. For landscape analysis areas and watershed groups, disturbances can be identified and correlated with amounts and rates of change in vegetation composition and structure. Specific functions can also be identified and probabilities of change estimated. At the project scale, specific disturbances can be characterized and projects designed to represent processes and the associated ecosystem components that disturbance events acted on. Ecosystem developmental pathways can also be predicted at this scale and estimates made of future composition, function, and structure (e.g., description of successional pathways).

Ecosystem process characterizations should always be done in terms of patch, ecosystem and landscape dynamics, and how these processes change through time. Such characterizations are required if the detection of important relations concerning hierarchical scales, climatic changes, role of previous patterns, and environmental constraints are to be understood (Bourgeron and Jensen 1993, Clark 1989, Johnson and Larsen 1991, O'Neill et al. 1989, Pickett et al. 1989, Turner 1989, Turner et al. 1993, Urban et. al 1987). At the domain and province scales, these attributes can only be assessed in a cursory fashion. At the section scale, landscapes can be sampled and evaluated for pattern dynamics. At finer scales, the complete landscape can be assessed.

Our experience suggests that a review of the literature and a complete description of a landscape area are often required to characterize ecosystem processes. Integrated resource or project analyses that use a simplified approach with minimal field sampling will usually produce erroneous disturbance regime descriptions. Such descriptions can contribute to the scheduling of management treatments that are in direct conflict with ecosystem processes and the adaptive strategies of local biota. A detailed approach to characterizing landscape structure and pattern is provided by Arno and others (1993). We strongly recommend that the first step in ecosystem analysis involve a general description of the process regimes, which can be obtained through literature review, looking at the complete array of local information, and talking with experts.

## Characterization of Historical and Existing Ecosystem Conditions

Some of the more common types of information used in ecological assessment of existing and historical conditions are displayed in table 3. Live vegetation is generally classified by life forms (e.g., trees or shrubs) or by broad cover type groups (e.g., hardwoods) at the domain scale of analysis. At the province scale, cover type groups are recognized as a specific group of dominant species, such as cottonwood-aspen or spruce-subalpine fir. At the section analysis scale, refined cover types also are used to describe existing live vegetation (e.g., lodgepole pine). At the landscape analysis area scale, cover types are usually further stratified by size class of the upper layer, and often by canopy closure of the upper layer (e.g., moderate cover, pole-size lodgepole pine). At the watershed scale, live vegetation can be further refined by dominance type (i.e., dominant species of the upper and lower canopy layers) and then subdivided by size class and canopy closure. Cover types are often adequate for most general watershed scale analysis needs. At the project scale, data concerning species composition, density by size class, canopy closure, and layer types are used to group vegetation into community types. Such site-specific data can be used to predict changes (i.e., vegetation succession) under different scenarios of management.

**Table 3--Relation of Analysis Area to Indicators of Historical and Existing Conditions**

| Terrestrial analysis area | Live vegetation components | Dead organic components | Successional stage | Ecosystem event-history | Species distributions |
|---|---|---|---|---|---|
| Domain | Life form or cover type group | Probability of occurrence | Early & late seral stages | Major process events by decade | Species & groups |
| Province or subprovince | Cover type group | Probability of occurrence | Seral stages, structural stages | Dominant process events by decade | Dominant and rare species |
| Section | Cover type | Probability of occurrence | Seral stages, structural stages | Mix of process events by decade | Species and populations |
| Landscape analysis area | Cover type, size class, canopy closure | Standing dead class down dead class litter/duff class | Seral stages, structure/age | Dominant process events by year | Seasonal and home range populations |
| Watershed | Dominance type, size class, canopy closure | Standing dead class down dead class litter/duff class | Seral stages, Structure/age, community type | Mix of process events by year | Seasonal and home range populations |
| Project | Species composition, density by size, canopy closure | Standing dead wood down dead wood litter/duff | Seral stage, structure/age, community type | Monitor effects of process events by year | Monitor habitat locations and populations |

The dead organic component of the landscape can be more important to some species than the live vegetation component. This information is commonly lacking in most environmental assessmdnts. At the domain, province, and section scales of analysis, broad probabilities of occurrence of dead organic material are associated with different disturbance regimes. The landscape analysis scale is sufficiently detailed to allow identification of standing dead tree size and down dead tree size classes, and amount or kind of litter and duff. Similar, but more refined, estimates can be made at the watershed scale of analysis. At the project scale, these components can actually be measured and used in the prescription process.

Successional stages are commonly classified by using early, mid, late, and potential vegetation terminology, when plant community age can be evaluated. When age cannot be determined, floristic similarity terminology of low, moderate, high, and potential vegetation are used. Such classifications are developed by comparing age, vegetation composition, and structure of the existing vegetation to the assumed potential vegetation community of a site. Successional stage classifications can be used at all scales of the analysis hierarchy. The successional stage classification system we have found most useful follows Oliver and Larson's (1990) system of classifying structural stages and cohorts in relation to stand developmental history. Stages of vegetation development in this system include initiation, stem exclusion, reinitiation, and old growth. Although this system was designed for forest vegetation, it seems to work equally well when generalized to shrub and herbaceous vegetation. The successional stages can be further refined by identifying the dominant species, number and kinds of cohorts (i.e., individuals that have regenerated after disturbance), and disturbance regimes. Seral community type classification are developed in the Northern Region (Hann 1992b) for use at the watershed and project scales.

Disturbance history can be characterized at all scales of the analysis hierarchy. At the domain scale, major events (e.g., regional drought cycles) that affect regional hydrology and vegetation pattern are commonly described. At the province scale, the characterization is refined by identifying fire and flood regimes. Maps of these processes can be drawn at the section scale. At the domain, province, and section scales, maps are drawn using decade summaries for each type of disturbance. At the landscape analysis area and watershed scales, major disturbances and associated historical information are commonly mapped and described on an annual basis. For project scale assessments, actual effects are evaluated and monitored.

Species distributions and habitat relations can also be described by hierarchical scales. At the domain scale, information is displayed by species groups. At the province scale, dominant and rare species are described and correlated with habitats and habitat use. At the section scale, populations of wider ranging species can be displayed. At the landscape and watershed scales, the seasonal distribution, and home ranges of selected species populations can be described. At the project scale, species habitat use and population numbers are commonly assessed.

## Comparison of Different Hierarchical Planning Scales

Most project areas, such as the Trail Creek Watershed (Hann et al. 1993), are small compared to higher scales of the planning hierarchy. Table 4 presents an estimate of conditions in the Trail Creek area in comparison to those at higher scales in the analysis hierarchy. The examples provided in table 4 are just a short list of the kinds of conditions that could be referenced across analysis hierarchies. In this example, the Trail Creek area has fairly low priority for management compared to other areas within the Beaverhead Mountains section. A similar conclusion can be made concerning the priority of the Beaverhead Mountains section within the northern Douglas-fir subprovince. This type of broader scale evaluation of landscapes, processes, and management issues provides an efficient way of prioritizing management, inventory, and research needs. Future quantitative assessments that use consistent ecosystem maps and classifications will help replace the rating system displayed in table 4 (i.e., low, medium, and high) with actual resource values.

**Table 4--Comparison of Conditions In the Trail Creek Analysis Area to Coarser Hierarchical Scales.**

| Condition or risk | Trail Creek watershed | North Big Hole LAA | Beaverhead Mountains section | Northern Douglas-fir subprovince |
|---|---|---|---|---|
| Number rare species | Low | Low | Low | Low |
| Exotic species | Low | Medium | Medium | Medium |
| Fuel accumulation | Medium | Medium | Low | Medium |
| Fire wind | Low | Low | Medium | High |
| Large fire risk | Low | Low | Medium | High |
| Insect/disease risk | Low | Low | Medium | High |
| Erosion risk | Low | Low | Medium | Medium |
| Soil productivity loss | Low | Low | Medium | Medium |
| Climate change risk | Low | Low | Medium | Medium |
| Pollution risk | Low | Low | High | Low |
| Hydrologic function loss | Low | Low | Medium | Medium |
| Aquatic function loss | Low | Low | Medium | Medium |
| Wetland/riparian loss | Low | Low | Medium | Medium |
| Landscape departure | Low | Low | Medium | Medium |
| Process departure | Medium | Medium | Medium | High |

Table 5 summarizes the conditions or risks of the northern Douglas-fir subprovince and compares them to conditions of risks in other sections of the Rocky Mountain Forest Province. A similar comparison is made between the Rocky Mountain Forest Province and other conifer forest provinces of temperate latitudes in table 6. These comparisons are based on literature review, national reports, and large-scale maps. This type of coarse-scale assessment provides information useful in prioritizing management and research plans, (e.g., rare species recovery plans, fuel management plans, and watershed restoration strategies). This assessment also provides an efficient mechanism for analyzing landscape connectivity and cumulative effects that are not efficiently dealt with at finer scales of analysis.

**Table 5--Comparlson of Conditions within the Northern Douglas-fir Subprovince to Other Subprovinces**

| Condition or risk | Northern Douglas-fir | Grand fir Doulgas-fir | Southern Douglas-fir | Ponderosa pine Douglas-fir |
|---|---|---|---|---|
| Number rare species | Low | Moderate | Low | Moderate |
| Exotic species | Low | Low | Low | Moderate |
| Fuel accumulation | Medium | High | Medium | Medium |
| Fire wind risk | Medium | Medium | Low | Low |
| Large fire risk | Medium | High | Low | Low |
| Insect/disease risk | Medium | High | Medium | Medium |
| Erosion risk | Medium | Medium | Low | Low |
| Soil productivity loss | Medium | Medium | Medium | Medium |
| Climate change risk | Low | Medium | Low | Medium |
| Pollution risk | Low | Medium | Low | High |
| Hydrologic function loss | Medium | Medium | Medium | Medium |
| Aquatic function loss | Medium | High | Medium | Medium |
| Wetland/riparian loss | Medium | Low | Medium | High |
| Landscape departure | Medium | High | Medium | Low |
| Process departure | High | High | High | High |

**Table 6--Comparison of Conditions between Forest Provinces of Temperate Latitudes**

| Condition or risk<br>Number rare species | Rocky Mountain Forest<br>Low | Pacific Forest<br>Medium | Willamette-Puget Forest<br>Medium | Sierra Forest<br>High |
|---|---|---|---|---|
| Exotic species | Low | Low | Medium | High |
| Fuel accumulation | High | Medium | Low | Medium |
| Fire wind risk | Medium | Low | Low | High |
| Large fire risk | High | Low | Low | High |
| Insect/disease risk | High | Low | Low | Medium |
| Erosion event risk | High | Low | Low | Medium |
| Soil productivity loss | Medium | Medium | Medium | High |
| Climate change risk | Low | Medium | Medium | High |
| Pollution risk | Low | Medium | Medium | High |
| Hydrologic function loss | Medium | Medium | Medium | Medium |
| Aquatic function loss | Medium | High | Medium | Medium |
| Wetland/riparian loss | Medium | Low | Medium | High |
| Landscape departure | Medium | High | High | Medium |
| Process departure | Medium | High | High | High |

## CONCLUSIONS

Landscape ecology and conservation biology are in their early stages of development as the Forest Service embarks on a new era of research and adaptive management (Everett et al. 1993). Accordingly, researchers and managers will need to work cooperatively to develop new technologies to manage for sustainable ecosystems and biodiversity. Although many of the theoretical issues and assessment techniques applicable to ecosystem management are undergoing continual development, the basic concept of managing for sustainable ecosystems and biodiversity by using ecological hierarchies is sound. Management treatments can be developed that recognize the range of historical conditions and processes with which landscapes evolved. Descriptions of the major ecosystem processes within assessment areas provide a basis for evaluating the ecological conditions of landscapes. Although it may not be refined (Swanson et al. 1993), the characterization of the range of natural variability is much simpler and easier than the traditional evaluation of management alternatives that frequently have no common objectives or criteria. Traditional planning approaches underemphasize the limits and potential of ecosystems and landscapes. Traditional assessment approaches have forced managers and scientists to predict outcomes for an almost unlimited array of treatment combinations given typical multiple-use values and concerns. The ecosystem process approach places bounds on the desired conditions that can be expected within an analysis area by identifying the limits of the ecosystem in terms of composition, structure, function, and diversity.

In summary, we provide the following recommendations to land managers concerned with ecosystem management implementation:

- The delineations used for bounding National Environmental Policy Act and National Forest Management Act analyses (at all scales) should be based on ecological criteria to improve consistency in assessment, management, and coordination with adjacent landowners.

- Assessments (at all scales) should emphasize the understanding of ecological processes and their effects on landscapes and ecosystem composition, function, and structure. Such assessments should use an ecological hierarchy (as described in this paper) and identify the range of conditions under which ecosystems developed.

- Key ecosystem processes should be emphasized in management plans. Ecosystem level monitoring efforts should contrast existing conditions to the historic range of conditions.

- Planning and assessment should be conducted at appropriate scales to achieve efficiency in analysis, inventory, monitoring, and implementation. More emphasis should be placed on regional-scale ecological characterization and assessment, and such knowledge should be applied through management at lower scales of the hierarchy.

- The hierarchical analysis system presented should be used as an efficient way to conduct across-scale comparisons of landscape descriptions. Such comparisons can assist in the identification of international, national, regional, and local priorities for ecosystem restoration.

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# Critical Economic Issues In Ecosystem Management Of A National Forest

**D.E. Ervin and R.P. Berrens**

## ABSTRACT

New developments in natural resource science and environmental economics are providing an analytical framework and principles that can be used in implementing ecosystem management. First, there is a growing public consensus that desired long-term ecological conditions must be specified by the public policy process with the aid of science to ensure adequate natural resources for future generations (i.e., establish a minimum natural capital stock). Second, ecological systems should be analyzed and managed to reflect the interdependence of long-term economic health and ecosystem health. Invoking these premises, economics provides critical information about ecosystem use and nonuse values and tradeoffs to help policy makers and resource planners select among alternative input and output paths. The economic problem is to achieve the ecosystem goals with the highest value or the lowest social cost or both. Multidisciplinary research is required to implement this economic approach.

## INTRODUCTION

An important consequence of interconnected ecological systems is that management of products or services one-by-one is rarely optimal. Changes in the harvest rates of a resource usually have negative or positive effects on other resources. An obvious example is the reduction in old-growth forest habitat and biodiversity as increasing amounts of the forest are cut.

There is a corollary to ecological interconnections in economic systems. Because increases in one economic activity usually decrease resources available for other uses, private firms and governments consider all outputs simultaneously. A farmer deciding what combination of crops to grow illustrates this economic decision process. This decision process requires that land managers and the public consider the economic tradeoffs or opportunity costs as they make choices among outputs.

Most importantly, there is a growing consensus that sustainable ecosystem management requires the synthesis of both ecological and economic processes (Jensen and Everett 1993, World Commission on Environment and Development 1987). Indeed, a new subdiscipline, ecological economics, has emerged based on these fundamental tenets (Costanza 1992). Although the subdiscipline is immature, the literature is identifying key issues that must be addressed in ecosystem management. Chief among them are the recognitions that long-term economic health depends upon environmental health and vice versa. That is, management of the environment and the economy cannot and should not be analyzed and managed as separate systems. Traditional economic approaches to natural resource and environmental management have not sufficiently incorporated ecological system effects and linkages into the economic system. We describe various principles of environmental economics in this paper and suggest how these principles may be used to evaluate alternatives for resource allocation under an ecosystem management philosophy.

### Sustainable Resource Management

Management of a complex National Forest embodies many of the issues and principles emerging from ecological economics. First, minimum biological and physical conditions must be specified that describe a sustainable ecological path for the many forest products and services (Common and Perrings 1992). Such an exercise presumes society wishes the National Forest or a similar natural system to continue indefinitely. Sustainable resource conditions require assumptions of what intergenerational equity should be. The sustainability conditions may specify the maximum amount and spatial pattern of harvest rates, the minimum instream flow during summer months, the degrees of plant and animal diversity, and so on.

For large natural systems, the specification of these minimum conditions takes the form of a constant "natural capital stock" (Pearce and Turner 1990). Equation 1 displays this relation:

$$\text{TNK} \geq \text{NRK(t)} + \text{RRK(t)} \qquad (1)$$

where: TNK = total natural capital
NRK = nonrenewable resource capital
RRK = renewable resource capital
t = time period t

Because the full social (private plus external) costs of extraction and use of many natural resources have not been paid by current or past generations, some argue that the present natural capital stock is the appropriate minimum.

The underlying argument is that maintenance of the total natural capital stock will provide future generations the opportunity to maintain their economic and social welfare through time, including adaptation to uncertain environmental and social phenomena in the future. This approach is referred to as the "strong" natural resource sustainability approach. Because all forms of natural capital comprise the stock, including renewable and nonrenewable resources, there may be substitution between different forms of natural resources across space and through time to maintain the stock amount. That is, exhaustion of nonrenewables must be accompanied by growth in the capacity of renewable resources to maintain the constant total natural capital stock. Or, improvements in technological efficiency in the use of the remaining nonrenewable stock must offset depletion so that expected demands will be served.

A "weak" natural resource sustainability theory asserts that the goal is to maintain a constant total capital stock through time, comprised of natural and human-made forms of capital as shown in equation (2).

$$\text{TK} \geq \text{NRK(t)} + \text{RRK(t)} + \text{PMK(t)} + \text{HK(t)} \qquad (2)$$

where: TK = total capital
PMK = physical human-made capital
HK = human capital

This approach allows substitution between natural resources and human-made technology or human capital (training) as long as the constant total stock condition is not violated. Defenders of the "strong" sustainability approach argue natural capital provides several roles or functions that human-made capital cannot, such as providing base material for scientific discovery by future generations to adapt to uncertain and irreversible environmental conditions (e.g., ozone depletion and climate change) (Pearce and Turner 1990).

The metric for converting different resources to a comparable basis for calculation of the natural capital stock is not yet devised for most situations. Ecological scientists are working on this problem, however, with the intent to provide a common unit of measurement that captures the contributions to ecological health from each of the diverse resources. This common measure is a critical scientific question that requires progress before economists can provide useful economic tradeoff information for ecosystem management questions.

## Economic Issues and Contributions

It is the selection of the mix and amount of natural resource combinations that requires the introduction of economic factors and processes. Valuation of resource effects provides tradeoff comparisons that can be used in making resource management decisions. Forests produce a number of outputs and services that may be sources of either benefits or costs (Pearce 1992). These outputs and services include the following:

- Harvest of existing trees or increasing forest cover increases timber and other derived wood products and can add new recreational value in the latter case.

- In situ biological diversity may be increased with increased forest cover (depending upon displaced land use) or decreased with timber harvest.

- Landscape values may be enhanced or diminished.

- Watershed protection from soil erosion and flood peaks may be enhanced by increased forest cover or reduced by harvesting, road building, and so on.

- Water supplies may be enhanced by harvest or reduced by increased forest cover.

- Higher standing timber stocks may increase the deposition of airborne sulfur oxides and nitrogen dioxide and reduce their transport (acidification stripping), but may increase water pollution from these compounds.

- Increased forest cover or reduced harvest levels can reduce $CO_2$ emissions and lower the potential magnitude of the greenhouse effect.

- Economic security may be enhanced from reduced costs of interruption of foreign timber supplies.

- Rural community integrity may be enhanced through increased forest cover and sustainable harvest patterns.(Such an effect may be thought of as reducing the human and physical adjustment costs associated with nonsustainable resource use patterns.)

For alternative ecosystem management approaches, all of these benefits should be calculated. Note that an increase in one benefit (e.g., watershed protection) may decrease or increase another (e.g., timber harvest or watershed supplies). That is, the list of possible effects include both positive effects and negative effects or costs. Once the capital and operating costs of forest management are included, the benefit and cost streams of the ecosystem management approach can be compared. Recall that the benefit-cost comparison takes place only for those resource configurations that satisfy the constant natural capital constraint imposed by a sustainable ecosystem approach, if such a sustainable path is chosen by policy makers.

## Valuation of Ecosystem Effects

Economic analysis is one of many sources of information for the social evaluation process of national forest planning. Economic values are only one type of “assigned values” (Brown 1984). Decision criteria that are based on economic values, such as efficiency and benefit-cost tests, reflect a particular utilitarian philosophy. There is no prima facie evidence, however, that economic assessment is antithetical to the developing concept of ecosystem management.

Ecosystem management itself is likely to reflect a spectrum of policy options. For example, several alternatives may exist for attaining a given index of biological diversity or ecological health. Moving towards ecosystem management will undoubtedly alter the mix of goods and services provided by a National Forest. Accurate assessment of these tradeoffs poses numerous challenges; however, economic tools can aid in making these difficult social choices. A critical first step is the distinction between different economic value components, which are values held by the individual for the resource, but not inherent in the resource.

One conceptualization or taxonomy of economic values can be described as follows:

- Direct use values (e.g., timber harvest or recreation).

- Indirect use values (e.g., watershed protection or ecological services).

- Option value (refers to the value individuals place on the potential use of a resource).
- Bequest value (refers to an individual's preference for bequest to future generations).
- Existence value (independent of any use of the resource).

A number of similar value taxonomies exist, and the subtle distinctions are not important. The critical distinction for the decision process is between those goods and services whose economic values are (or are not) fully captured in market prices. For example, timber products may have a direct use value that is more or less accurately reflected by market prices. Recreation may also have a direct use value, but minimal or nonexistent fees in incomplete markets do not accurately reflect this value. Nonuse values, by definition, have no discernible link to market behavior. Missing or incomplete market values, however, do not imply the absence of economic value. In the absence of efficient pricing, specialized techniques must be used to assess these values in a manner commensurate with more conventional commodities (e.g., timber sold in a market).

The measurement of total economic value refers to the attempt to assess the combined component values of an environmental asset or resource system (Pearce 1993, Peterson and Sorg 1987, Randall 1991). In the same way that physical resource functions can be interconnected, the economic values for the various goods and services produced are interconnected. A valid total economic value measurement must account for this interconnectedness (e.g., substitution and complementary issues, and non-additivity of component parts). While the concept of total economic value is generally accepted by economists, systematic attempts to measure-total economic value in a regional policy or planning context are still rare. Kristrom (1990) provides an empirical example for forest preservation in Sweden.

In the last several decades, economists have developed and refined a battery of techniques for assessing the economic value of nonmarket goods and services (Braden and Kolstad 1991). These techniques can be broken down into two basic types. The first type, indirect approaches, relies on observed behavior to infer values. Examples include the travel cost model in which the relationship between visits and travel expenditures is used to infer the value of a recreational site. Another example is hedonic pricing methods which attempt to decompose the value of market goods; say recreational real estate adjacent to a national forest, to extract embedded values for environmental assets. The second type of assessment, direct approaches, uses a variety of survey-based techniques to directly elicit preferences for non-market goods and services. Although there are a number of variants of the direct methods, the most common is the contingent valuation method. Indirect and direct techniques share a common foundation in welfare economics where measures of willingness-to-pay and willingness-to-accept compensation are taken as the basic data for individual benefits and costs.

From a measurement perspective, "passive" or nonuse values (i.e., option, existence and bequest) are the most problematic component of total economic value. The contingent valuation method is the only technique in the economist's tool kit for assessing these values. The topic of existence values for environmental assets is one of the most controversial subjects in all of environmental economics (Bishop and Welsh 1992, Edwards 1992, Kopp 1992, Rosenthal and Nelson 1992). Substantial evidence shows that many individuals will contribute to environmental organizations, and express willingness-to-pay to preserve environmental assets on contingent valuation method surveys, with no expectation of current benefit or future use of the resource. Evidence that existence values exist is something less than arguing that they can be measured on a sufficiently comprehensive and reliable basis for use in formal decisions (Castle and Berrens 1993, Rosenthal and Nelson 1992).

Because of the public controversy concerning the measurement of existence values, a blue-ribbon panel, containing several Nobel laureate economists, was convened by the US Department of Commerce's National Oceanic and Atmospheric Administration (NOAA) in 1992. This panel met to provide guidance on promulgating regulations (pursuant to the Oil Pollution Control Act of 1990) that would explore the use of the contingent valuation method in measuring lost or nonuse values in natural resource damage cases. The potential for assessing nonuse values through application of the contingent valuation method was essentially reaffirmed by the NOAA panel, provided rigorous guidelines are followed (Arrow et al. 1993).

Examples in the Pacific Northwest of the application of contingent valuation method in measuring existence values can be found for old-growth preservation and spotted owls (Hagen et al. 1992, Rubin et al. 1991), and Columbia River salmon (Olsen et al. 1991). Some economists remain skeptical that existence values can be reliably measured for endangered species and irreversibility problems (Castle and Berrens 1993, Stevens et al. 1991). Such economists turn from the traditional benefit-cost framework towards more ecologically conservative decision criteria such as the “safe minimum standard” (Ciriacy-Wantrup 1952).

Empirical estimates of nonmarket use values are substantially less controversial, and are also important to the planning process. Hundreds of site-specific studies valuing recreational services and environmental quality have been completed, and documented by a voluminous literature. Natural resources of the Pacific Northwest have been the focus of numerous studies. Recent examples include Berrens and others (1993), Donnelly and others (1990), Fried (1993), Johnson and Adams (1989), Johnson and others (1990), Morey and others (1991), and Olsen and others (1992). Viewed in the aggregate, the numerous valuation studies over the last several decades document the considerable economic worth of nonmarket goods and services. Many of these studies, however, are characterized by outdated techniques or changing circumstances.

An emergent issue in the valuation literature is the need for developing an acceptable protocol for transferring values (Brookshire and Neil 1992, Walsh et al. 1992). “Typical” nonmarket values for recreation, fish, and wildlife are commonly incorporated in the USDA Forest Service planning process under the Renewable Resource Planning Act (RPA) of 1974, as amended (Duffield 1989). In some cases these “RPA values” differ greatly from state-of the-art primary data studies (Olsen 1989). A further criticism of the RPA values is the failure to incorporate nonuse values (Duffield 1989). It is difficult to envision how current RPA values can be used to assess the full value of such nonmarket services as biological diversity (which may grow in importance in an ecosystem management perspective).

The need for valuing nonmarket goods and services arises independently of planning considerations for ecosystem management and sustainability concerns. As forest planning objectives change to accommodate a new set of issues, the need for de novo valuation studies may increase (i.e., standard RPA values may be poor indicators of the economic benefits and costs produced by forest quality changes under an ecosystem management regime). The process of developing a holistic framework for approaching ecosystem management (Kennedy and Quigley 1993) may require that valuation studies also co-evolve to aid critical management decisions. For example, explicitly linking valuation techniques to physical resource functions, through bioeconomic models, remains an important research area (Adams et al., 1990). Linking valuation measures to indices of biological diversity is largely uncharted territory.

Finally, documentation of nonmarket benefits for planning, and possibly resource damage liability cases, is not the only purpose for conducting valuation exercises. Valuation studies can also play a critical role in determining the potential for appropriation, that is, in converting some portion of nonmarket values into public revenues (Pearce 1993). For example, increased use of fees for recreational access and services may help to defray lost timber revenues. Furthermore, dispersed recreational use can be complementary to biological planning objectives and recreational value studies can play an important role in establishing fee schedules (Adams et al. 1989, Adams and Berrens 1993). Notably, there have been a number of recent arguments, in local print and television, for increased use of recreational fees on Pacific Northwest forests.

In summary, economists have a variety of valuation techniques that can be used to select among policy choices concerning environmental assets. Applying these techniques to ecosystem management issues is a new challenge, and the techniques may need to be recalibrated to handle emergent policy questions. Interdisciplinary cooperation appears to be necessary for successful design of future valuation experiments.

## Applying the Economic-Ecosystem Management Approach

The framework presented for economic assessment of ecosystem management requires a specification of “minimum natural capital stock,” and then the articulation and estimation of the benefits and costs of alternative ecosystem resource combinations that satisfy stock constraints. This approach focuses attention on the services that flow from the National Forest, and not on the management inputs required to produce those services.

Estimation of relevant use and non-use values requires a variety of methods, some of which (e.g., the contingent method) are controversial. As noted, some authors argue that existence values are not reliable and, therefore, should be implemented through “safe minimum standard” protection of relevant resources. Other authors continue to estimate existence values under the contingent valuation method approach. The critical factor regarding appropriate use and reliability of the contingent valuation method is whether the respondent is familiar with the resource and service in question. Preferences for unfamiliar goods may be poorly defined or unstale. In a broader social science context, sensitive economic valuation estimates (e.g., the level of protest responses) can provide clues to social values.

Once relevant values are estimated, benefit and cost streams (discounted at an appropriate rate) can be aggregated into the present value of net benefits. Selecting the appropriate discount rate is an important task. Some argue for omitting or artificially lowering the discount rate because it may cause resource degradation for future generations. More reasoned analyses, however, show the discount rate can exert both positive and negative effects on resource stocks (Pearce 1993). Moreover, the discount rate is a critical (intertemporal) price in our market economy that cannot be suppressed.

The largest discounted net benefit or lowest net cost ecosystem approach may be preferred on social valuation grounds. Social decisions, however, often involve intragenerational and intergenerational equity questions which cause policy makers to select resource use paths different from the highest valued approach. Calculation of the values associated with each approach allows decisionmakers to observe the implied tradeoffs of selecting resource use patterns for equity reasons. If not all benefits can be monetarily valued, then net monetary values can be supplemented with quantitative and qualitative descriptions of resource effects. Strategies for choosing can be devised with a combination of monetary and non-monetary measures, such as maximum net benefit value with a minimum resource degradation constraint.

This choice process selects the desired resource outputs and services from the National Forest system; however, those outputs or services might be produced with a variety of input patterns, including spatial and temporal vegetative cover. Unless the inputs produce desired services in themselves, the focus on total outputs from the National Forest system allows the managing agency to combine the forest resources in a variety of ways to achieve specified ecosystem values at the lowest cost to current and future generations.

## CONCLUSIONS

Several important steps in conducting economic-ecosystem analysis emerge from this discussion:

- Obtain and clarify the minimum natural capital stock conditions for the ecosystem under study.
- Construct economic-ecological models that consider the effects of ecological systems on economic performance, and the relation of economic health to ecological values.
- Assign economic values to all quantifiable forest outputs and services affected by alternative ecosystem management paths, including use and nonuse components susceptible to credible monetary values.
- For those effects not susceptible to monetary measurement, describe the quantitative or qualitative effects, or both, of alternative management paths.
- Estimate the discounted net benefits of the alternative paths using an appropriate social discount rate that reflects the expected future return on capital.
- Summarize qualitative and nonmonetary effects to demonstrate implied ecosystem tradeoffs, and use this information in monetary effects analysis.
- Focus management attention on the desired ecosystem outputs and services that will be produced. Allow inputs to be arranged in a least cost fashion, rather than specifying or regulating input levels (e.g., vegetation patterns).

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# Socioeconomic Risk Assessment and Its Relation to Ecosystem Management

**C.A. Montgomery**

## ABSTRACT

Public land management agencies have a mandate to manage for the "permanent good of the whole people." One component of the permanent good is the amount, type, and timing of exposure to risk. I argue that private agents responding solely to market incentives fail to produce the "best" allocation of risk, and consider how forest policy might address that failure. Ecosystem management is assessed in terms of its risk compared to risk associated with a free market. Although management within the bounds of natural variability means less uncertainty about the future state of forest ecosystems, it means more uncertainty about current amounts of commodity production. This allocation of uncertainty might appear to be an improvement over free markets; however, policy makers must ask whether ecosystem management overshoots the mark.

The "permanent good of the whole people" depends on what people think they want. In this paper, I examine how risk perceptions and preferences are formed. Perceptions and preferences about risk have direct and powerful implications for public demand for forest policy. Ecosystem management is, in part, a response to strong public aversion for a particular type of risk--the possibility that current land management practices will result in the unraveling of ecological systems upon which life depends.

Faced with uncertainty, both in outcomes of management and in demand for forest resources, forest policy makers would do well to design policy that can respond to change in social perceptions of risk and to new biological and technological knowledge. Ecosystem management seems to be an attempt to do just that.

## INTRODUCTION

The goal of normative economics (the economics of what should be) is to guide government policy makers who are presumably striving for the maximum well-being for their constituency. If markets functioned perfectly, there would be no legitimate role for government in the management of resources. It is widely recognized, however, that markets fail in important ways--both in the allocation of resources between uses and in the allocation of resources across time. Hence, forest land is publicly owned and the USDA Forest Service has a mandate to manage for the "permanent good of the whole people and not for the temporary benefit of individuals or companies." [1]

One of the beauties of the market is that it functions automatically, so underlying relations between demand and supply can remain unknown. If government intervenes in markets in an effort to improve market allocations (the Forest Service intervenes through public ownership and management of some of the forest resource), these relations must be made explicit. A large volume of information must be accumulated, and the dynamics of choice must be understood.

Recent Forest Service policy reflects the importance of uncertainty in public land management through its emphasis on adaptive management (Everet et al. 1993). In particular, the current policy directive of ecosystem management has important consequences for the magnitude, timing, and type of risk faced by public land managers and the public. This paper describes the efforts of economists to understand a particular aspect of choice--choice under uncertainty. The paper also describes the effect of ecosystem management on the allocation of risk when compared to resource management approaches where private agents respond solely to free market incentives.

## Risk in Forest Management

Public land managers make decisions that have implications for future forest productivity[2] without certainty of what the result will be. An activity is said to be risky when the outcome is uncertain. The magnitude of the risk depends on the nature of the possible outcomes and the likelihood of their occurrence. What kinds of uncertainty do public forest managers face?

The outcomes of management activities are not often known because natural disturbances such as fire, disease, and geological events affect outcomes. Not only is it impossible to predict the timing of these disturbances but only in a few cases do we have data from a long enough period to allow us to describe their occurrence as stochastic processes (Swanson et al. 1993). Also, incomplete knowledge of how biological systems work and their response to management activities and disturbances is the source of further uncertainty.

Likewise, public forest land managers can only predict how consumers value many forest resources. Consumer demand for forest commodities, such as timber, is observable because consumer preferences are revealed by their market behavior. But consumer demand is not as readily apparent for a whole array of forest goods that are not exchanged in markets, and the methods used for predicting consumer interest are not refined.[3]

This basic uncertainty about demand is aggravated by the passage of time. One view of consumer demand is that individuals use resources to produce such basic commodities as security and comfort, and varied and fulfilling life experiences. Demand for specific resources depends in part on their perceived usefulness in producing these basic commodities--and thus on consumer awareness about the characteristics of the resources. As consumers learn more about forest ecosystems, their perception of the role of forest resources in the production of recreation, health, and security changes. New technologies such as pharmaceuticals, jet travel, and home videos also affect the perceived usefulness of forest resources. As a result, policy makers must predict what consumers want now and in the future.

## Economics of Uncertainty

In the standard economic model of choice under uncertainty, individuals do not know which outcome will occur, but they do know the potential outcomes and their likelihood of occurrence. In this model, individuals choose actions that maximize their expected well-being.[4] This model is widely taught in business and finance programs as a normative model for decisionmaking under uncertainty. The model has come under intense criticism because experiments in which subjects are offered choices among various outcomes and probabilities of occurrence indicate either that people are irrational (which is heresy to economists) or that the model is missing something important.

These experiments reveal behavior that differs in systematic ways from that predicted by the expected utility model. For instance, people evaluate potential losses and potential gains differently. People prefer a certain payment over an uncertain gain of the same expected value, and yet they prefer an uncertain loss over a certain cost of the same expected value.[5] People are said to be risk-averse in gain and risk-loving in loss. If the question can be framed either as a gain or as a loss, public opinion may be manipulated. In addition, people tend to give more importance to outcomes that have high losses or gains and low likelihood of occurrence than the expected utility model predicts.

In one extension of the basic model, probabilities are replaced by decision weights.[6] These weights differ from true probabilities in the following ways: they don't necessarily follow the laws of probability, they are based on subjective assessment of probabilities which are influenced bur not defined by objective probability estimates, and they reflect the effect of subjective probabilities on choices. In addition, individuals base their choices on evaluation of changes from a reference point (usually their current situation), thereby allowing losses and gains to be treated differently. This model may be more credible in its description of how individuals choose in the face of uncertainty. Hence, this model may be richer in its ability to provide insights about current and future public demand for forest resources than the basic expected utility model.

A key point is that individual choice is based on perceptions of riskiness. These risk perceptions often differ from the objective probability estimates obtained by experts. The importance of perceived riskiness depends on the characteristics of the outcome (likelihood and dimension of potential loss or gain), the current situation of the individual, and individual preferences about risk. Some cognitive psychologists (Tversky and Kahneman 1974) study how people think about risk and thus how their perceptions of riskiness are formed. People are influenced, in part, by their exposure to a particular outcome; that is, can they visualize it, has it happened to anyone they know, and how recently and how frequently have they been made aware of it in the media, by conversation, or by experience? The more people have been exposed to an outcome, the higher is their subjective assessment of its risk. In addition, people tend to overestimate the likelihood of very low probability events.

Given their subjective estimate of the likelihood and severity of an outcome, people still display preferences for the types of risk they may be exposed to. Slovic (1987) suggests two factors that affect how people feel about risk: a dread factor and an unknown factor. A high-dread factor is for risk of catastrophic loss and fatalities (e.g., nuclear accidents). A high unknown factor is for risks that are not well understood and may involve a delay between an event and is outcome or cumulative effects (e.g., exposure to toxic substances). People are especially averse to risks that rank high in both of these factors even if their subjective estimate of the likelihood of occurrence is low.

## Market Failure and the Role of Government Policy in the Allocation of Risk

Because individual well-being depends, in part, on risk, demand for forest resources depends, in part, on their role in the abatement of risk and the distribution of risk across time. Policies that affect forest productivity can affect the amount and the timing of risk. Normative economic models of social well-being generally rest on the assumption that government is benevolent--that it tries to choose policy that improves the well-being of its constituents. The doctrine of consumer sovereignty upon which these models are built states that individuals know best what will make them happy. As a result, government must rely on consumer demand to evaluate the well-being of is constituents.[7] But even in public choice models, in which policy-makers act in their own self-interest by maximizing budgets or job security, consumer demand defines the political environment and the pressures that policy makers face. In fact, forest policy is too often adopted without a full understanding of the importance and the nature of consumer demand for forest resources.

One issue is the optimum amount of risk for society. Do individuals, responding independently to market feedback, choose the ‘best’ amount of risk? Probably not. As the number of individuals that share the benefits of a policy increases, the ability to spread the loss (should it occur) increases. For example, a 3000-acre fire may cause a total loss of wealth for a small landowner, whereas for a shareholder in a land-holding corporation, the loss may be a small fraction of individual wealth. Federal land is owned by the entire U.S. population. Thus, the U.S. Government, acting in the interest of society, should be more neutral to risk than private landowners are (Arrow and Lind 1970). In fact, many believe that the U.S. Government should be completely risk neutral, thereby making decisions based on the expected value of outcomes.

There are markets for some types of risk. One can pay a certain price for insurance to avoid a possible private loss, or one can purchase a chance at a gain in securities markets. Just as some goods are difficult to exchange in markets, however, there are some types of risk that are not marketable. Such risks commonly have the character of public goods. A public good is produced in a fixed amount and individuals can neither vary the amount of consumption to match their own preferences, nor can individuals who do not pay be excluded from consumption of the good. Examples of public goods include clean air, clean water, and national defense. Examples of public risk include risk of health impairment from exposure to dirty air or toxic chemicals, risk of nuclear war, and risk of deterioration in the quality of life resulting from loss of species diversity. Abatement of public risk is a public good. Just as markets in goods allocate too many resources to the production of private goods and too few to the production of public goods, markets in risk allocate too many resources to the abatement of private risk and too few to the abatement of public risk.

How should probabilities enter into the comparison of risky alternatives? Should government respond to social perceptions of well-being based on subjective probabilities and decision weights for risk, or should government play “father knows best” and substitute the best objectively obtained estimates of probabilities that are available? Perhaps government can do a little of both by altering subjective probabilities and decision weights through education (or propaganda) campaigns and by using objective probability estimates while acknowledging such phenomena as dread and unknown risk. In any case, constituent pressure on government will be related to social perceptions and preferences about risk.

Also at issue is the timing of risk. To compare alternative management activities that result in potential losses and gains at different times, we compute the net present value of each alternative by discounting future gains and losses to the present.[8] The result is that future gains and losses count less than current gains and losses. The discount rate is the rate at which an investment must grow in value for an individual to be willing to give up current consumption for future benefits. The higher the discount rate, the less future gains and losses weigh against the present. Is the appropriate discount rate the same for private agents (responding to market incentives) and the government (managing for social well-being)? Probably not. Most economists believe that the social rate of time preference (the discount rate that weighs the well-being of future generations in a way that represents our communally held values) is lower than the private discount rate (Baumol 1968 and Tresch 1981).[9] If so, society values the well-being of future generations more highly than market behavior indicates. Thus, the market fails by imposing too much cost or risk of loss on future generations and too little on current generations.

To illustrate the importance of the discount rate in management decisions, consider the following example. Suppose that there are two possible plans for responding to wildfire. Plan A is to reduce short-term risk to property and timber production by putting wildfire out as soon as possible. Plan B is to reduce long-term risk to forest health and to biodiversity by allowing wildfire to play is natural role in the maintenance of forest ecosystems (in other words, by letting some fires burn). Each plan results in different outcomes in the present and in the future. The choice between these two plans may be represented in the following extremely simplified model (table 1). Hypothetical dollar values and probabilities of occurrence of outcomes are provided in table 1 to display present and future (100 years from now) affects of two different plans for wildfire management. If plan A is chosen:

- In the present period, there will be a 1-percent chance that wildfire will cause a 1-billion dollar reduction in timber production.

- In the future period, the accumulation of fuel and the changing age class structure of the forest will result in an even higher likelihood of loss to wildfire--a 10-percent chance of a 1-billion dollar reduction in timber production. There will be a high probability of a small loss in nonmarket forest values such as clean water or wildlife habitat--a 90-percent chance of a loss of 100,000 dollars. There also will be a very small probability that the change in some ecosystems will result in catastrophic effects on the ability of the forest to support life--a 0.05-percent chance of a 10-trillion dollar loss.

If plan B is chosen:

- In the present period, there will be a higher likelihood of a large loss to wildfire than with plan A--a 5-percent chance of a 1-billion dollar reduction in timber production. There will also be a high chance of a small loss to managed fire--a 90-percent chance of a 100,000 dollar loss.

- In the future period, the likelihood and magnitude of loss are the same as in the present period because the forest structure will be the same. Nonmarket values will remain unchanged.

**Table 1--Simple Model of Intertemporal Tradeoff in Risk of Loss**

| | | Program A | | | Program B | |
|---|---|---|---|---|---|---|
| | | Timber | Amenities | | Timber | |
| Loss in current period | Likelihood | 1% | --- | --- | 5% | 90% |
| | Magnitude | $1 billion | --- | --- | $1 billion | $100,000 |
| Loss in future period | Likelihood | 10% | 90% | 0.05% | 5% | 90% |
| | Magnitude | $1 billion | $100,000 | $10 trillion | $1 billion | $100,000 |
| Present value of expected loss | d = 0.04 | $111 million | | | $51 million | |
| | d = 0.10 | $10 million | | | $50 million | |

In table 1, two plans are compared by using two different discount rates. The Office for the Management of the Budget uses a discount rate of 10 percent to emulate private investment decisions. With a 10-percent discount rate, the present value of the loss for plan A ($PV_A$) is $10 million, and plan B ($PV_B$) is $50 million. Plan A (protection of current timber production) is the preferred option. The Forest Service uses a discount rate of 4-percent for investment analysis to represent social preferences about tradeoffs across time. With a 4-percent discount rate, $PV_A$ is $111 million and $PV_B$ is $51 million. Plan B (protection of future nonmarket values and future timber production) is the preferred option. The key factor in the different outcomes of the analyses with different discount rates is the expected present value of the future catastrophic loss associated with plan A. With the high discount rate, the expected present value of the catastrophic loss is negligible, and so we are willing to impose that risk on future generations. With the low discount rate, the well-being of future generations matters more and we avoid risk of catastrophic loss in the future.

In summary, because of its ability to diversify losses and spread gains, government should be willing to accept riskier prospects of gain and be more willing to pay to avert risk of loss than private agents responding to market incentives. Government should move from abatement of private risk to abatement of public risk. It is not obvious how government should measure risk (e.g., by the best objective probability estimates or by social perceptions of riskiness), but it is clear that social perceptions and the weights individuals give to different types of risk effect the political arena in which agencies operate. Finally, government should move risk of loss toward the present and away from the future.

## Ecosystem Management and Risk

Ecosystem management can be summarized as a broadening of the scale of management perception (e.g., from the stand or drainage to the landscape) and an increased effort to manage within the range of natural variability (Jensen and Everett 1993). Patterns and characteristics of forest types that have resulted from naturally occurring disturbances provide a range of variability useful in making land management decisions (Swanson et al. 1993). Ecosystem management should reduce uncertainty about long-term outcomes of management activities because actions will be taken that mimic natural processes, the outcome of which we have had some opportunity to observe.

For wildfire, ecosystem management translates into a move away from immediate fire control regardless of cost (e.g., the “10:00 a.m. policy”) toward an increased use of prescribed fire to maintain stand structure, and a willingness to occasionally let wildfire burn while monitoring is progress. More frequent, low intensity fires are likely to result; however, the risk of catastrophic fire will be reduced as fuel buildup is removed and stand structures return to a more varied state. For timber production, ecosystem management translates into a move away from fixed targets for annual timber sale volume, toward a tolerance for variation in timber sale offerings that reflect variation in forest conditions.

Is ecosystem management a “good” policy? Although no such judgment is made here, I do point out some considerations that should be made in the evaluation of ecosystem management and other forest policy, focusing on the allocation of risk. What makes good policy? If you know where you are now and where you want to be, good policy moves you in the right direction. The previous section gives a description of the direction that policy ought to take if we start from a pure free market allocation of risk. The United States has a mixed economy, resulting from government intervention in the operation of markets when it is deemed necessary or expedient for the achievement of social goals. The role of the Forest Service in managing the National Forests is one such example. In the discussion that follows, I make no assessment of where we are now relative to a free market allocation or socially optimal allocation of risk. Instead, I identify the direction that I believe ecosystem management takes with respect to risk, but make no judgment as to whether it is the “right” direction.

First, with ecosystem management there is a tradeoff in risk across time. By allowing more variability in the landscape through the use of management activities that mimic natural processes, the ability to control outcomes in the short term is reduced. In other words, short-term risk is increased. By maintaining the healthiness and robustness of forest systems, long-term risk resulting from uncertainty about the long-term implications of management activities is reduced. Thus, there is a transfer of risk away from the future toward the present.

Second, the abatement of public risk is a public good, and the abatement of private risk is a private good. The policy of immediate fire control was motivated by a desire to reduce risk of loss in the value of standing timber and structures--a private risk. Increased use of prescribed fire and occasional use of wildfire will increase the likelihood of such, losses. Fixed targets for timber sale volume reduce uncertainty about short-term timber supply. Increased variability in timber sale offerings increases private risk to investors in the wood processing industry. In the long-term, however, it is likely that the losses to fire that do occur will be smaller, and the ability of the forest to sustain some amount of timber harvest will be more certain than with the current management strategy. Again, this change in fire policy represents a transfer of private risk across time--away from the future to the present. Further analysis is required to determine whether the net result is more or less private risk overall. On the other hand, an array of public goods are at risk under current management practices (e.g., human values associated with ecosystem and species diversity and the stability of forest systems in their life-supporting role). By managing within the bounds of natural variability, ecosystem management reduces risk of loss of these public goods. Hence, there is a shift from abatement of private risk to abatement of public risk.

Because people evaluate potential losses and potential gains differently (i.e., they are risk-averse in gain and risk-loving in loss), a reference point must be identified before the desirability of changes in risk can be assessed. Whether a change in resource policy is viewed as a gain or a loss (and to what parties) depends on how the status quo is defined (Bromley 1992). Suppose that the reference point is that wood processors have the right to relatively certain short-term public timber supply and the claims of “environmentalists” to benefits of ecosystem diversity and stability are not weighted as heavily. In this case, risk associated with timber production is risk in loss and risk associated with future forest resource benefits is risk in gain. Risk-loving individuals (timber producers) pay too little to abate risk of loss and risk-averse individuals (environmentalists) pay too much to abate risk in gain. Because a “good” risk-neutral government policy favors abatement of risk in loss over abatement of risk in gain, ecosystem management moves in the “wrong” direction. Now, let’s frame the issue differently. Suppose the right to stable and varied forest ecosystems is preemptive. Then risk associated with future forest resource benefits is risk in loss and risk associated with timber production is risk in gain. In this case, a “good” risk-neutral government policy favors abatement of risk in loss (of ecosystem stability and diversity) over abatement of risk in gain (of short-term timber supply). Hence, ecosystem management moves in the “right” direction. The key point of this discussion is to highlight the importance of framing issues in terms of preemptive rights and gains versus losses on the outcome of normative policy analysis. Because media plays a large role in the framing of issues in the public mind, there is substantial potential to manipulate public opinion about resource policy.

Finally, should government rely on objective estimates of probabilities to assess riskiness, or should it respond to social preferences about risk that depend on subjective probabilities and weights? That question goes unanswered here. It is important, however, to recognize the role social perception and weighting of risk play in the demand for changes in forest management that are embodied in ecosystem management.

First, consider social perceptions of riskiness. Scientific understanding of biological systems and the interrelatedness of their components has increased dramatically in recent years (Golley 1993, Turner et al. 1993). Evidence is accumulating that continued human manipulation of ecosystems may have large-scale and long-term effects such as global warming, massive loss of biodiversity through the extinction of species, and acid precipitation. This knowledge has made its way into the social consciousness through frequent and, at times, sensational media coverage. People fear that such activities might destabilize the global life support system and thus present a threat to human life. In the forming of subjective assessments of riskiness, individuals tend to overestimate the likelihood of infrequent, catastrophic events and their outcomes if they have frequently or recently been made aware of them.

Individual feelings about risk also depend on the nature of the risk. The risk associated with human tampering with ecosystems is perceived as sensationally catastrophic and widespread--unravelling systems, global deterioration in the quality of human life or even in the ability of the planet to sustain human life. The dread-risk factor is high. Also, people are more aware than they were of the complexity of biological systems and the incompleteness of our knowledge about them. The uncertain-risk factor is also high.

Considering that people tend to form high subjective estimates of the likelihood of such events and that people are strongly adverse to this type of risk, it is not surprising that prophecies of doom have historically played a role in the formation of forest policy. The National Forest system was established, in part, because of fear of the implications of potential timber famine for national security. Fire control policy in the early 20th century was a response to public horror at the huge forest fires of the era (Shephard 1993, Dana and Fairfax 1980). In the post-World War II era, the catalyst for dread was the fear of nuclear war and communist domination. Forest policy was peripheral to that fear and turned to the support of continued economic growth and response to the demands of an increasingly affluent society.

Starting in the 1970s, concern about the consequences of rampant human population growth drew attention to the ability of the planet to sustain life. The first concern was food production, but issues of environmental degradation and biological system stability soon moved to the forefront. The attempt to maintain healthy and varied forest ecosystems that are consistent with naturally occurring patterns is one way to reduce risk associated with such a catastrophic outcome (Jensen and Everett 1993). Public demand for forest management practices such as ecosystem management that attempt to sustain ecological systems and protect life-supporting forest resources such as clean water can be seen, in part, as a reaction to this dread.

## CONCLUSION

Public land management agencies have a responsibility to manage public land for the “permanent good of the whole people.” To fulfill this responsibility, policy makers must understand what would happen if forest land management were left entirely in the hands of private agents responding solely to market incentives, and how that outcome might fall short of the optimal outcome for society.

In terms of risk, the market outcome fails in the following ways: private agents pay too much for the abatement of private risk and too little for the abatement of public risk, private agents pay too much for the abatement of current risk and too little for the abatement of future risk, and private agents accept too much risk in loss and too little in gain.

Implementation of ecosystem management will result in transfers in the allocation of risk. Management within the bounds of natural variability and for healthy and diverse ecosystems will mean less risk of future loss of such nonmarket forest resources as clean water, species preservation, and the stability of biological systems on which human life depends. There will be more variability in the current amount of annual timber harvest, as well as a higher likelihood that the forest will be able to sustain some amount of timber harvest in the long term. There will be more risk of loss of current timber volume to fire, but the maintenance of stand structures that are consistent with a history of fire will reduce the risk of catastrophic fire in the future. Because ecosystem management means a transfer from abatement of private risk to abatement of public risk and a transfer of risk to loss from the future to the present, it represents an improvement to management by private agents. Whether ecosystem management means a transfer of risk in loss to risk in gain depends on how rights to forest resources are assigned, or in the absence of legal assignment, how rights to forest resources are perceived by the public.

It is necessary to qualify this discussion with the following observations. Even though it seems that ecosystem management is an improvement to a private market-driven outcome in terms of the allocation of risk, it is not possible on the basis of this discussion to determine whether it is an improvement over the status quo. Government agencies have been in the business of forest resource management for a long time, and private forest management practices are heavily regulated in some states. We are far from management of the National Forest resource by private agents responding solely to market incentives, and past policy represents an effort to correct for the failures of markets.

How much correction is necessary? Is ecosystem management a movement in the right direction, or have we overshot the mark? These are questions that policy makers must face. An essential element in the decisionmaking algorithm is the nature of demand for forest resources. In terms of risk, policy makers must understand social perceptions of risk, how those perceptions are formed, their relation to objective risk estimates, and how people feel about the risks as they understand them. Perceptions of risk depend on exposure to and understanding of a particular risk. These perceptions are subject to manipulation through control of information that is transmitted through public media. Perceptions of risk also depend on how issues are framed (e.g., as losses or as gains). Public feelings about risk depend on aspects of risk that are easily sensationalized. Thus, social perceptions of and preferences about risk are volatile. Such feelings are likely to change as the collective attention span is exceeded, as new information and technology become available, and as the public changes and learns. At times, public land managers may feel buffeted in the wind of public opinion. In the face of such uncertainty, it may be tempting for public land managers to label social risk perceptions and preferences as irrational, and rely instead on the best available objective estimates of risk. It is preferences, however, that define social well-being, and to ignore them is to ignore the mandate under which these agencies operate.

Faced with uncertainty, both in the outcome of management activities and in the current and future demand for forest resources, forest policy makers would do well to design flexible policy options that can respond to change in social perceptions of risk and to new biological and technological knowledge. Ecosystem management implementation through adaptive management strategies (Everett et al. 1993) seems to be an attempt to do just that.

## FOOTNOTES

1. Gifford Pinchot, in a 1905 letter written to himself, in the name of the Secretary of Agriculture Wilson, stating the principles under which the new forest reserves would be managed (Dana and Fairfax 1980).

2. Forest productivity in this context includes the production of nonmarket goods such as biodiversity and scenic vistas.

3. The three most widely used methods for valuing nonmarket forest resources are the travel-cost method, hedonic pricing, and contingent valuation. The travel-cost method and hedonic pricing a; 1 limited in application to use values that are reflected in observable expenditures. Contingent valuation is used to measure non-use values by survey and is highly controversial. For a survey of methods, see Pearse (1990).

4. This model is known as the expected utility model and is attributed to Von Neumann and Morgenstern (1944). For a good description of this model, the issues associated with it and criticisms of it, see Schoemaker (1982). Utility is economists' ordinal measure of individual well-being. Consumers choose between courses of action, each of which generates an array of possible outcomes and associated probabilities. Consumers choose the one with the highest expectation of well-being (highest expected utility) as follows:

$$\max EU(a_i) = p_1{*}U(x_1) + ... + p_n{*}U(x_n)$$

where:

| | |
|---|---|
| $a_i$ | is a course of action |
| $U(x_n)$ | is a measure of well-being associated with a possible outcome |
| $p_n$ | is the probability of that outcome occurring |

5. In a much cited example provided by Tversky and Kahneman (1981), subjects were asked to imagine an outbreak of an unusual disease that is expected to kill 600 people. One group was asked to choose between a certain gain (save 200 lives) and an uncertain gain (save 600 - 1/3 probability, save no lives -2/3 probability) with the same expected value. The subjects overwhelmingly preferred the certain gain of 200 lives. The same choice was posed to another group, but framed as a choice between a certain loss (lose 400 lives) and an uncertain loss (lose no lives - 1/3 probability, lose 600 lives - 2/3 probability). The subjects overwhelmingly preferred risk--to take a chance that there might be no loss of life.

6. Prospect theory is a descriptive economic model of choice under uncertainty developed by two psychologists, Kahneman and Tversky (1979). Machina (1987, 1989) provides surveys of objections to the standard model and other attempts to expand it to fit observed behavior.

7. Consumer demand is not synonomous with market demand. Because it is not possible to express some preferences through market transactions, market demand is an incomplete manifestation of consumer demand.

8. If the gains (losses) are nonmarket goods (bads), such as species conservation or air pollution, we assign values as best we can by using the techniques described in footnote 3. Present value is computed as follows:

Present value = (value at time t)/(1 + discount rate)

where the discount rate is a positive number between zero and one.

9. Private agents will only invest if the expected rate of return exceeds the cost of borrowing (the market interest rate) plus taxes on earnings. The social discount rate should reflect the willingness of society to postpone consumption in return for future benefits to society. The private discount rate is believed to exceed the social discount rate for the following reasons:

a) Future generations cannot make their preferences known in current markets.

b) Individuals tend to be pessimistic in terms of their own futures, overestimating the likelihood of their own demise at any point in time.

c) The costs of operating a capital market require that savers be paid a lower rate of return than borrowers are charged.

d) Returns to capital investment are taxed.

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# An Economic Tradeoff System for Ecosystem Management

**B. Lippke and C. Oliver**

## ABSTRACT

Those resources that are owned in common, such as air and water are considered to have no value until they are exhausted and consumers must pay producers to provide the resource. Local ecosystem values, such as biodiversity, are not well represented in the economic system; however, forest management can increase biodiversity and ecosystem amenities, changes which appear to be consistent with the changing values of the public. Unfortunately, the traditional use of regulatory mandates to achieve such attributes increases costs and reduces returns, thereby discouraging investment. Regulatory constraints on timber production also are likely to increase harvests in less productive regions. With their lower productivity and processing efficiency, these less productive regions also are likely to contribute to even greater environmental stress.

If incentives for ecosystem management are awarded by competitive bidding, an efficient response by landowners in an area can be achieved without groups of owners making allocation decisions in violation of Federal trade laws. An example of management goals to increase biodiversity and forest health on eastside forests results in management costs that are not much higher than the cost savings realized from the reduced unemployment that is related to the jobs created. Internalizing the values of biodiversity and other ecosystem amenities into the market system by contracts for management to reach specific goals provides effective protection for biodiversity and other nonmarket values while offsetting some of the employment losses in timber-dependent communities.

## INTRODUCTION

Economic systems produce goods and services that are valued by consumers. Producers require that the value of the output be higher than the cost of the input, and this margin is the source of savings to be invested in future growth, which increases consumers' standard of living. Commonly owned resources, such as air and water, are considered to have zero value until their supply is limited. Then, consumers will pay producers to provide the resource, bottled water, for example. Consumers in Los Angeles would likely pay much more for cleaner air than consumers in Seattle because they have substantially less of it. If a pricing system is not developed for a common resource that becomes scarce, consumers are denied the opportunity to consume as much of that resource as they would like relative to other goods.

A limitation of the economic system in producing the highest net value (made up of market and nonmarket goods) to its consumers is in the determination of the proper values to place on common resources, and how to ensure these values are reflected in the economic system. If no value is placed on resources, they may be overly used and exhausted.

Failure to protect common resources is not attributable to producers making poor economic decisions; it is because proper values have not been determined for these resources. Values of resources will differ geographically and may be hard to determine. If too high a value is assigned to a common good, such as a pollution tax in a remote area, too few market goods will be produced relative to the availability of the common good. Regulatory standards, especially when applied uniformly on a broad regional basis, will probably overstate the values of the resource in some regions and understate the values in other regions, thereby causing inefficient production. In this paper, we present an economic incentive strategy for incorporating non-market values into ecosystem management.

### Economic Systems With Nonmarket Values

Local ecosystem values like biodiversity are not well considered in the economic system because values are not placed on items that lack immediate usefulness (e.g., species that may become endangered). The Endangered Species Act, however, puts an infinite value on the existence of each species in its designated habitat within the United States. This Act mandates protection of one species at a time with no consideration of tradeoffs, such as endangering species elsewhere in the world by substituting the forests of one region for another. The protection or enhancement of biodiversity could be internalized into the economic system, thereby facilitating the same efficiency as production of other market goods.

As applied to forest management, current economic competition commonly results in the highest production of timber from the most productive timberlands and relatively little production from low productivity forests (Sedjo and Lyon 1990). Therefore, the number of harvested acres is minimized, and this should correspond to minimal global environmental degradation. Without other barriers and regulations, economic efficiency in forest growth and production will generally correspond to the least environmental consequences on a global scale (Lippke 1991). The global environment may best be served by intensive forest management, including clearcutting (for various commodities) in the most productive places, and less intensive commodity extraction (e.g., selective cutting or “high grading”) or no activities at all in less productive places. This scenario may not be desirable, however, for some local and regional environments and local economies.

As a specific example, ecosystem management in eastern and western Washington can be viewed as a microcosm of the global situation. Western Washington has productive soils for tree growth, whereas soils in eastern Washington are less productive. Within eastern Washington, there is a gradient of productive lands to less productive lands. It is economically efficient to site prepare, plant, and harvest stands at relatively short rotations in western Washington or on the more productive lands in eastern Washington, because the high growth rates and ease of regeneration allow an economic return for the investment. Diverting productive land from this use by using uneven-age management, thinning, leaving cull trees standing, and extending rotations would increase costs and shift management to less productive areas. If all the forests of western Washington were managed intensively and for highest economic efficiency, however, there would be little, if any, remaining old-growth stands on the west side, and biodiversity would be decreased.

In eastern Washington it is usually economically feasible to extract timber on an uneven-aged basis as trees become merchantable; however, it is not generally economical to invest in site preparation, planting, weed control, felling cull trees, or other even-age activities except on the most productive acres. If all stands were managed for economic efficiency in eastern Washington, most stands would be of uneven age and contain many old-growth features. Stand-initiation structures (Oliver and Larson 1990) would occur primarily where uncontrolled fires had burned the previous stand. Stem-exclusion structure stands (Oliver and Larson 1990) would largely be overcrowded and susceptible to insects and fires, because it is not economically attractive to precommercially thin such stands. Where frequent ground fires had burned, there also would be stands with parklike structures. Except for the parklike structures, the economically efficient landscape described above resembles the present landscape in eastern Washington.

The stand structures produced by economic practices in western Washington are largely open with a dominance of stem-exclusion structures and little old growth, just the opposite of eastern Washington. Maintaining these structures locally may involve a tradeoff between local biodiversity and global environmental quality, because one may be reduced where the other is increased. Reducing efficient management in productive areas to preserve biodiversity may promote timber harvest from less efficient areas, or substitution of more energy-intensive, polluting materials such as steel, aluminum, plastics, and concrete for wood products (Koch 1991). Managing forests to produce desirable structures for biodiversity in less productive areas will require costs (labor and energy) not balanced by the value of timber and other products produced and marketed in such areas.

The challenge for ecosystem management in western Washington, therefore, will be to encourage enough uneven-age management, thinning, leaving cull trees standing, and extending rotations to maintain a viable number of the plant and animal species, patterns, and processes not otherwise maintained through traditional management for maximum economic efficiency. The challenge for ecosystem management in eastern Washington starts from the same premise, that is, to encourage enough of the operations that would not normally be maintained as a part of economically efficient management. Because it is economically efficient to manage forests intensively in western Washington but not in eastern Washington, the stand structure changes that must be paid for to maintain all ecosystem patterns and processes (biodiversity) are nearly opposite in the two areas. For example, creation of old-growth structures must be targeted in the west and stand initiation structures in the east.

A method is needed to resolve tradeoffs between maintaining local ecosystem integrity by preserving all natural patterns and processes in an area (Swanson et al. 1993), and maintaining global environmental values by not increasing global pollution or endangering ecosystems elsewhere as a consequence of local prescriptions.

## Incorporating Ecosystem Values Into the Economic System

The value of endangered species, wood, and substitute products for wood could be reflected in the economic system by requiring all countries to protect all endangered species and to pay taxes for pollution or avoid pollution altogether. Imposing such regulations in only some countries simply increases the price and rate of harvesting in other regions, or increases the use of wood substitutes, with a net degradation of the global ecosystem (Lippke 1991).

It is impossible to require the direct beneficiaries (the global public) to pay these costs in the current global economy because the costs cannot be imposed throughout the world. If imposed in one region or country and not another, these costs raise prices locally and shift production to another region, which may be less efficient at producing the desired product. The effect would be to improve the environment of one region or country at the expense of is economy and standard of living, and decrease the quality of the global environment. For example, for each acre not harvested in western Washington because of environmental costs, 15 acres of forests in Russia would have to be harvested to produce the equivalent wood product volume. Part of this difference derives from less productive acres and part from an inadequate infrastructure to reduce waste (Lippke 1991). Because the harvesting would be less efficient, consumers would pay more and there would be less savings and reinvestment.

If a country or region attempts to mandate standards to create certain nonmarket forest values, the net result may include higher costs and less investment in the forest, increased wood supplied from other regions with less efficiency, and environmental degradation as a result of the lower efficiency. Attempts to internalize the costs at a local level will cause substitution with competing suppliers and, hence, a similar result. If a forest manager is provided incentives to produce stand structures needed for biodiversity, it is possible to produce market outputs and these nonmarket outputs without substantial substitution from competing suppliers (Lippke 1992). Encouraging management practices that are economically unattractive requires incentives to produce for nonmarket values. The incentives should be provided by those who benefit. The overall challenges of ecosystem management will be to determine the tradeoffs among benefits to the local and global environments and costs to U.S. citizens, and to make the incentives as efficient as possible.

Under present economic conditions, the private landowner receives no compensation for providing these public values, which are not incorporated into the global economic system. On public lands, the costs of providing nontimber outputs has not been calculated, and probably could not be constructed from existing data. Maintaining these otherwise uneconomical structures will require an investment by the landowner, and this can be promoted by incentives on private lands or budget allocations on public lands. Regulations may also be used to achieve this objective. Incentives, as different from regulations, encourage the landowner to perform activities not otherwise personally beneficial but which benefit the public. An incentive system for increasing the productivity of degraded soils on private lands through tree planting was extremely successful in the southeastern United States in the 1950s (USDA Forest Service 1988).

## A System of Allocations

Managers must determine how to maintain ecosystem values not presently reflected in the economic system. Who should pay and how much should be paid? Maintaining these values requires activities that are presently not economically efficient. Regulations requiring such activities would increase the costs of management, thereby driving production to less efficient regions. There is also a need to maintain ecosystem values across all landowners in an area, so adjacent landowners work efficiently and in concert. Landowners are, by law, not allowed to collectively allocate resources, thereby making such agreements difficult.

Because regulations are inefficient from a market economics perspective, those public bodies charged with protecting ecosystem values could achieve better results at less cost to the consumers if they purchased the ecosystem benefits through open market transactions rather than by regulatory processes. A public organization chartered to make these purchases could be responsible for ecosystem management. This organization would be responsible for determining the needs, costs, and tradeoffs of different levels of ecosystem management and for implementing a program to purchase ecosystem values. Costs and tradeoffs would include management of wildlife and fish populations, forest stand structures, and landscape patterns. The ecosystem management organization would be given a public budget based on the estimated cost of various policies, which would be used to both manage wildlife populations and contract with forest management organizations to provide the correct forest structures.

Land managers of publicly owned forests should provide the same function as private forest managers, whose objectives are to maximize revenue and profits from the forest. This approach would allow relative global efficiency in producing both timber and other values obtainable in the market. Forest management organizations would sell their services to the highest bidder, which might be commodity production or ecosystem management, or more likely a mix of the two. Often, the two objectives would be complementary, thereby reducing the costs of obtaining both timber and ecosystem values (Oliver 1992). For example, commercially thinning a Douglas-fir stand in eastern Washington is not economical, because it might cost as much as $300 more per acre (more than its market value) to remove and haul the wood. If thinning the stand had a positive effect on the ecosystems, however, the ecosystem management organization would have to pay only the $300 per acre to achieve its objectives, as opposed to as much as $900 per acre to cut and remove the wood if it has no value. The forest management organization would receive returns for timber by selling it in the timber market and returns for producing ecosystem values by contracting with ecosystem managers to manage stands to achieve their targets.

The public would obtain the largest increment in ecosystem values possible if the request for ecosystem production were put up for open bidding. The forest manager who can produce those ecosystem targets at the lowest incremental cost, perhaps for much less than the $300 average cited above, will be able to offer the lowest and winning bid. As a consequence of the bidding process, ecosystem managers can learn what it costs to produce ecosystem values in different regions and obtain higher ecosystem outputs for less cost.

In the same way, other ecosystem values might be given specific budgets, for example, employment opportunities, recreation, and specific endangered species. The various values to society, their costs, and tradeoffs, would be clear to society by the separate budgets and by determining the cost to purchase those values.

Conflicts over values would be minimized by forest managers being able to bid for contracts from several value-managing organizations. Budgets allocated by the Federal government for each value would be supplemented by state and local governments where local support was sufficient to allocate resources to that value. In some cases the benefits are likely to be only local, thereby necessitating local support.

The process for achieving different values would become efficient, because each organization responsible for a value would be responsible for achieving as much as possible on a limited budget. Because the cost of achieving each value becomes well established in the process, political decisions can focus on how much of a given value is enough given its cost. Under current systems, we do not know what it costs to produce ecosystem values.

This system would allow public and private lands to be coordinated, as private landowners as well as public land managers would find it economically efficient to bid for the service of providing certain ecosystem structures and processes. The system avoids the need for land managers to collaborate on who does what, because they would bid competitively for values in the market.

## Example of Allocation System in Eastern Washington

An example of the proposed allocation system is presented in figure 1, which displays a landscape of 15,000 acres typifying the distribution of a isting stand structures for non-Federal land in eastern Washington (Oliver 1992). Historical operations on this landscape involved partial cutting of 20 percent of all timbered stands in a 10-year period; this is assumed to be responsible for the large number of stem-exclusion structure stands now present. In this example, five valued target stand structures are proposed. These structures include the four structures described earlier (Oliver and Larson 1990), plus a parklike structure typical of areas containing low intensity, frequent fires on which the goshawk and other species depend.

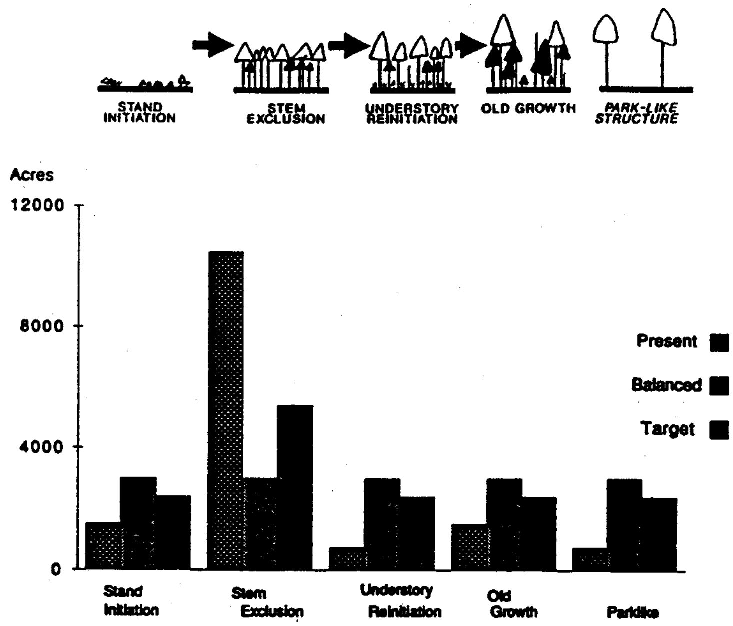


**Figure 1--Targeting for a more balanced distribution of stand structures.**

A possible target distribution for this landscape would have each structure at a minimum of 80 percent of an equal (20 percent) area of the landscape; therefore, each structure should cover at least 2400 acres [80 percent*(1/5)*15,000]. To reach the minimum target, stands would be converted from the existing dominant stem-exclusion structure as follows:

- Stand-initiation structure: Some stands in the stem-exclusion structure would be clearcut; one-half would be broadcast burned; and all would be replanted to suitable species.

- Stem-exclusion structure: Because there is a surplus of these stands, some stands would be left in this structure.

- Understory-reinitiation structure: Some stands in the stem-exclusion structure would be precommercially thinned; one-half of these treated stands would be underburned with a ground fire.

- Old-growth structure: Some stands in the stem-exclusion structure would be commercially thinned; one-half of these treated stands would be underburned with a ground fire.

- Parklike structure: Some stands in the stem-exclusion structure would be treated similarly to a shelterwood cut, thereby leaving dominant overstory trees at wide spacings throughout the area; one-half of these treated stands would be underburned with a ground fire.

The distribution of the present and targeted stand structures is characterized in table 1. The excess acres in the stem-exclusion stage become the source for better management by manipulating them to maintain other structural stages on the landscape.

Ecosystem management may require more activities and labor than historical management. The net returns and labor for these two types of management compared to the example landscape are compared in table 2 (costs estimates provided by Forest Service silviculturists). While the gross return to ecosystem management is greater than present management, the net economic return is less. Ecosystem management, however, would employ more labor and provide a greater tax return than historical management. Selling management contracts to produce these changes in stand structures moves the forest towards the target structure at minimal cost.

The advantages of ecosystem management over historical forest management are the benefits to the ecosystem. For the public to achieve these benefits, the costs are the marginal increase in costs over historical management. As shown in table 3, if similar activities were spread across 10 years and no other benefits were considered in achieving ecosystem management, the cost of achieving the designated landscape structures for ecosystem management over the 15,000-acre example landscape would be $3.6 million higher than present management. But the cost is only $1.1 million over the savings from reduced unemployment if the operations take place in rural timber-dependent communities where unemployment has become high and persistent. Federal and state tax payments also would be greater as would county payments from the Federal forests.

**Table 1--Stand Structure Distribution and Biodiversity Targets**

| | Stand initiation | Stem exclusion | Understory reinitiation | Old growth | Parklike |
|---|---|---|---|---|---|
| Present distribution of structures (%)<br>Area in 15,000-acre landscape | 10<br>1500 | 70<br>10500 | 5<br>750 | 10<br>1500 | 5<br>750 |
| Balanced distribution of structures (%)<br>Area in 15,000-acre landscape | 20<br>3000 | 20<br>3000 | 20<br>3000 | 20<br>3000 | 20<br>3000 |
| Target distribution of structure (%)<br>Area in 15,000-acre landscape | 16<br>2400 | 36<br>5400 | 16<br>2400 | 16<br>2400 | 16<br>2400 |
| Acres to be converted | 900 | -5100 | 1650 | 900 | 1650 |

**Table 2--Example Acres Managed, Costs, and Person-Days of Employment for Present Vs. Ecosystem Management Based on USDA Forest Service Data**

| Eco Management Treatment | Price/ MBF* | Cost/ MBF | Cost/ acre | Return/ acre | Person-days/ acre | # acres | Total cost | Total return | Net $ |
|---|---|---|---|---|---|---|---|---|---|
| Clearcut | 200 | 150 | 1500 | 2000 | 3 | 900 | 135,000 | 180,000 | 450,000 |
| regenerate | | 0 | 400 | | 2 | 900 | 360,000 | 0 | -360,000 |
| bd-cst burn | | 0 | 200 | | 0.8 | 450 | 90,000 | 0 | -90,000 |
| Precommercial thin | | 0 | 240 | | 1 | 1650 | 396,000 | 0 | -396,000 |
| Commercial thin | 120 | 180 | 900 | 600 | 2 | 900 | 810,000 | 540,000 | -270,000 |
| Underburn | | 0 | 100 | | 0.4 | 2550 | 1550,000 | [illegible] | -255,000 |
| Heavy shelterwood | 200 | 160 | 1280 | 1600 | 3 | 1650 | 2,112,000 | 2,640,000 | 528,000 |
| Fire protection | | | | | | 0 | 0 | 0 | 0 |
| Total | | | | | | 9000 | 5,373,000 | 4,980,000 | -393,000 |
| Present Treatment<br>partial cut | 230 | 100 | 600 | 1380 | 2 | 30,000 | 1,800,000 | 4,140,000 | 2,340,000 |

*MBF = thousand board feet of timber

**Table 3--Comparisons of Total Costs, Returns, and Employment for Ecosystem and Present Management**

| | (% in millions) | 1 Present mgt | 2 Eco mgt | Difference 3=2-1 eco-present |
|---|---|---|---|---|
| a | Total returns | 4.14 | 4.98 | 0.84 |
| b | Total costs | 1.8 | 5.37 | 3.57 |
| c=a-b | Net return | 2.34 | -0.39 | -2.73 |
| d | Forest jobs (person days) | 6000 | 14,280 | 8280 |
| 3=d*(120) | Forest labor cost | 2.02 | 3.64 | 1.62 |
| i=(d+f)*(40) | Cost if unemployed (at $40/day) | 1.01 | 1.21 | 0.81 |
| j=c+i | Net return plus direct unemployment saved<br>Net return plus direct and indirect unemployment saved | 3.35<br>4.36 | 0.82<br>3.24 | -2.53<br>-1.1 |

Marginal cost of ecosystem above direct and indirect unemployment savings is about $110,000 per year for 10 years, an incremental cost of $27 per person day.

Money also could be allocated for managing wildlife and fish populations and for restoring riparian areas to the extent that these values are not balanced directly through biodiversity goals. Such costs can be treated in a manner similar to the previous upland forest structure example. For example, the market price for maintaining either wildlife or fish populations may be greatly reduced if there are incentives for producing biodiversity because several of these values may be complementary. If these additional goals are implemented through competitive contract bids, market pricing will help determine the best mix of contracts. Market contracts can produce an efficient joint production decision more easily than individual planning groups can learn how to coordinate their implementation plans.

## Tradeoff Between Ecosystem Risk and Effort to Reach a Balance

Table 3 shows the marginal costs of ecosystem management if the target proportion of structures in our example were achieved. Because of random events and population changes not associated with forest structures, there is probably no more than an 80 percent probability that stable ecosystems will be achieved Continuing historical management will result in relatively low portions of several stand structure types such that only a 10-20 percent probability of stable ecosystems might be assumed. Figure 2 illustrates a schematic relation between the cost of ecosystem management and stand balance, which is related to the probability of achieving stable ecosystems. A linear relation to the single-point cost estimate provided in table 3 was assumed in constructing this figure. Alternative management treatments might lower total costs when the degree of balance being targeted is reduced. This relation, although presently very crude, could be estimated with increasing precision as our knowledge and methods of analyzing ecosystems increases. This illustration suggests that the risk of unstable ecosystems will decrease as the balance of structures increases; however, costs should also be expected to increase nonlinearly, especially as efforts are devoted to approaching a perfect balance rather than targeting a tolerance region, such as within 20 percent of balance.

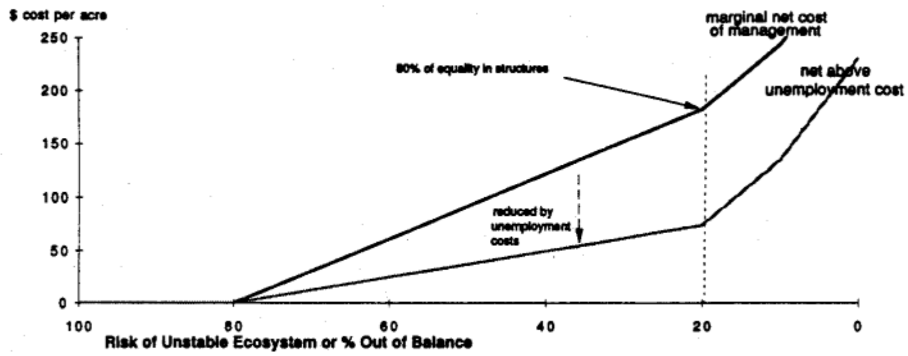


**Figure 2--Cost of reducing unstable ecosystems.**

As illustrated in figure 2, the net cost of managing our 15,000-acre landscape example is just over $150 per acre for 80 percent of balance and could be reduced to about $75 per acre for a target of 50 percent of balance. Adjusting these costs for the savings from reduced unemployment lowers the cost to about $75 per acre for 80 percent of balance and only $30 per acre for a target of 50 percent balance. There also may be other lower cost prescriptions to achieve roughly the same amount of protection that could be even more attractive, especially given the high cost estimates provided for some proposed treatments. Benefits from fire suppression also might be factored in as a reduced cost in this example.

Knowledge of these cost relations allows the public to allocate efforts to ecosystem management based on a knowledge of tradeoffs between costs and risks (Montgomery 1993). Such relations can be developed by landscape analysis units (Morrison 1993) to determine where to allocate limited resources most efficiently.

## The Synergistic Value of Managing Ecosystems for Other Values

Table 3 compares historical and ecosystem management scenarios and shows the link to unemployment, which has become a serious problem in rural, timber-dependent communities. Presently, unemployment is a substantial cost to the public; it is estimated from the economic report of the President to be $40 per day per unemployed person. Ecosystem management, in our example, directly employs about 56 more person-years per 15,000 acres than historical management, or about 168 more person-years when indirect transportation and support service jobs are included. As a consequence, $1.7 million in public funds for unemployment would be saved if such jobs are targeted for the high-unemployment, rural, timber-dependent communities. In addition, the people are employed at $30,000 per year (including overhead), which is a much higher standard of living than unemployment brings, and employed people contribute to tax revenues rather than being a drain on them. The cost of purchasing biodiversity can be very low where there are already significant expenditures being channeled to the unemployed.

Other values, such as recreation, future higher quality of wood (and concomitant increased employment through pruning and secondary manufacture), and increased forest protection also could be considered in managing synergistically for ecosystem management. The above example is overly simplified because it does not include either the investment value of the various operations or the reduced cost of forest protection by managing for more stable and healthier forest ecosystems.

## CONCLUSION

The establishment of separate budgets and organizations for the various objectives of forest management to be achieved through market mechanisms allows public and private land managers to coordinate for best ecosystem results. Both public and private land managers become “service” organizations producing both timber and nontimber outputs, while maximizing income from the land in this scenario. Consequently, both public and private landowners within a landscape unit could compete (through competitive bids) for funds to manage their vegetation stands for ecosystem values, employment, and other public values.

Such funding allocations create needed incentives for land managers, while competitive bids help ensure that funds are spent as efficiently as possible.

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# Managing Ecosystems and Social Conflict

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## ABSTRACT

Ecosystem management has arisen in part as a response to social conflicts; it will address some and raise others. Ecosystem management will challenge decisionmaking institutions. This paper reviews some of those challenges, presents basic assumptions about social conflict and ecosystem management, and discusses ways that land management agencies' traditional public participation techniques may not be adequate. Collaborative negotiation is presented as a potential forum for ecosystem management and the recent upsurge in collaborative decisionmaking in the Pacific Northwest is reviewed for possible lessons.

## INTRODUCTION

Implementing ecosystem management on Federal lands will pose several substantial challenges, particularly where forest health has deteriorated. Technically, it will be challenging to define ecosystem processes and measure them across the relevant spatial and temporal scales (Bourgeron 1993, Turner et al. 1993). Economically, it will be challenging to capture financial values without undue negative effects on nonmarket values (Lippke and Oliver 1993). Administratively, it will be challenging to marshall the personnel, resources, incentives, and interagency coordination to perform the tasks at hand (Kennedy and Quigley 1993). Lastly, it will be a tremendous social challenge, because implementing ecosystem management will test the ability of many groups to work together and deal with differing goals that exist for Federal lands (Shepard 1993).

Both the recent motivation and direction for ecosystem management have been shaped by social forces. The motivation by Federal land management agencies to convert to an ecosystem management philosophy has sprung, at least in part, from the public controversy that has characterized Federal land policy during the last 30 years (Daniels and Carroll 1993, Kennedy and Quigley 1993, Overbay 1992, Shepard 1993). It is clear that ecosystem management has a substantial grounding in recent scientific advances in conservation biology and landscape ecology (Golley 1993), but the timing, intensity, and focal points of the ecosystem management debate have been profoundly altered by social forces.

Ecosystem management has been defined as the use of ecological knowledge to produce desired values in ways that sustain the diversity and productivity of ecosystems (Avers 1992, Overbay 1992). Thus, the concept of "desired" is central to the direction established by this definition, and desired by whom and for what must be addressed. Indeed, Zernold and others (1992) concludes that "collaboration and public participation are essential if we are to provide an array of functioning ecosystems that accommodate social needs which range from places for solitude to intensively managed production systems."

Not only is it clear that the recent interest in ecosystem management has emerged in part from social controversy, but also that its implementation will create new conflicts. Ecosystem management requires that effects be considered across scales of time and space, transcending social institutions, such as property rights and ownership (Morrison 1993). Old ways of doing business will no longer suffice, and the constituencies that have developed around longstanding land management practices will be under pressure to adapt.

In recent years, the dominant means of settling public lands disputes have been either litigation or quasijudicial administrative appeals. Such contentious methods of handling disputes expend much goodwill, energy, time, and money. These methods produce winners and losers, may leave fundamental differences unresolved, and potentially please few or none of the parties. The stakes involved in "us versus them, winner takes all" confrontations compel groups to fortify positions and encourage competing claims for natural resources that, if met, may not be consistent with ecosystem health. It seems fundamentally flawed to attempt to resolve differences in a divisive manner when trying to manage holistically; the technique does not fit the task. Moreover, there is no guarantee that politically crafted compromises between extreme demands are biologically feasible, and they therefore may compromise important ecosystem functions.

This paper examines the potential for more collaborative approaches to forest policy formation, with a particular emphasis on meeting the challenges inherent in ecosystem management. This examination draws on three sources of information: (1) an examination of public participation and its ability to deal with conflict in general, and the unique demands of ecosystem management on public lands; (2) an inventory of recent collaborative natural resource decisionmaking efforts in the Pacific Northwest; and (3) a framework for understanding natural resource conflict.

## Understanding Natural Resource Conflicts and Disputes

Effectively managing social conflicts related to ecosystem management requires a fundamental understanding of natural resource conflicts and disputes. The following discussion provides the foundation and framework for such understanding and relies on four assumptions about natural resource conflict and disputes.

**Assumption 1:** Ecosystem management involves both conflicts and disputes. As environmental mediator Gerald Cormick (1982) notes, a natural resource conflict occurs "when there is a disagreement over values or scarce resources." Disputes arise as part of a natural resource conflict; encounters involve "a specific issue over which the conflict in values is joined." Cormick explains that "the settlement of a dispute is achieved when the parties find a mutually acceptable basis for disposing of the issues in which they are in disagreement, despite their continuing differences over basic values." Natural resource conflicts are ongoing and reflect different values, interests, and fundamental views of the relation between people and the natural environment (Henning and Mangun 1989, Hunter 1989). Disputes are identifiable issue-specific episodes that arise within that conflict (Crowfoot and Wondolleck 1990). Consequently, enduring conflicts need to be managed; disputes arising within them require settlements. Natural resource managers and organization representatives must understand both the disputes and the conflicts. For example, conflicting views and philosophies over the importance of old-growth preservation will continue over the long term, but timber harvest, stream buffer, and salvage disputes require decisions in the short term.

**Assumption 2:** Natural resource conflicts and disputes are complex. Natural resource disputes arise within some context that can be defined by issues, parties, interdependence, and strategy. Natural resource management, such as by the USDA Forest Service, is a context of great importance and complexity, particularly to people in the Pacific Northwest. Issues define one area of complexity. Natural resource controversies are rarely single issue; disputes typically include many interrelated matters. A dispute over the removal of timber salvage, for example, may involve visual effects, fish and wildlife habitat, forest health, and industry needs. Natural resource dispute settlements are not easy: there often is no quick fix or simple formula for success.

**Assumption 3:** Natural resource agencies and organizations must address conflicts and disputes. Conflicts and disputes are an inevitable part of ecosystem management. The commitment by the Forest Service to multiple use, for example, implies the diversity of interests and ideas on how forest resources should be used. Regardless of the management philosophy and policy the Forest Service (or any other natural resource agency) embraces, those divergent views and values will remain. These views and values typically seem to compete with one another, thereby making dispute settlements difficult. Consequently, some organizations and managers may become frustrated by conflicts and disputes and see them as negative and something to block or avoid. Seeing conflict as negative and something to avoid encourages crisis management rather than constructive management of disputes. Natural resource agencies and organizations, rather than relying on a "conflict crisis" approach, should emphasize a "conflict collaboration" approach. In so doing, agencies embody what organizational psychologist Dean Tjosvold (1991) calls "the conflict-positive organization." In such an organization, people value diversity in individuals and opinions, seek mutually beneficial outcomes to problems, are empowered to deal constructively with conflict, and take stock of and reflect on how they and the organization handle conflict. To become conflict-positive, natural resource agencies and organizations, through their structure, culture, and the attitudes of individuals, should support constructive conflict and dispute management approaches (Tjosvold 1991).

**Assumption 4:** Effectively managing natural resource conflict requires good analysis and strategic choices. Managers and representatives from natural resource organizations need to understand conflict and disputes and enact strategies to deal constructively with those disputes. Analysis demands forethought and time, both often denied when crises erupt. Crises encourage knee-jerk reactions, rather than well-evaluated options and constructive, planned strategic choices. In crises, land managers often rely on “trained incapacities,” habitual methods of responding, even when they know these methods may not work (Folger and Poole 1984).

To handle natural resource conflicts and disputes, managers and representatives need to analyze situations carefully and evaluate the alternatives. Such analysis and strategic choice benefits from addressing the following nine natural resource dispute assessment categories:

- What are the ISSUES in the dispute?
- What are my party’s GOALS AND INTERESTS?
- Who are the PARTIES and STAKEHOLDERS in the dispute?
- What are the RELATIONS AND POWER concerns?
- What is the dispute negotiation SETTING?
- What STRUCTURAL factors warrant attention?
- What is the dispute’s SETTLEMENT POTENTIAL?
- What is the preferred STRATEGIC ORIENTATION?
- What NEGOTIATION INTERACTION is desired and expected?

Each category offers a series of analytical questions. As part of dispute-settlement planning, natural resource organization managers and representatives can use the category areas and those specific questions they find relevant to better understand the challenges and opportunities within any specific dispute. The complete framework for analyzing natural resource conflicts and disputes and its specific questions are available from the authors on request.

## Understanding Agency Public Participation as a Form of Natural Resource Dispute Management

If the quality of ecosystem management will depend, at least in part, on the quality of conflict management, are the conflict management techniques of the Forest Service adequate to the task? In general, Federal land management agencies and the public currently interact through formal public participation processes where agencies are assumed to make decisions and the public to provide comment. The specific legal requirements for public participation in agency decisions come from several sources: the Administrative Procedures Act of 1946; the National Environmental Policy Act of 1969 and the regulations pursuant to it written by the Council on Environmental Quality; the Resources Planning Act of 1974, as amended by the National Forest Management Act of 1976; the Federal Advisory Committee Act of 1972; Federal Land Planning Management Act of 1976; and numerous legal precedents. Accordingly, most Federal agencies have explicit public involvement mandates.

It is clear that public participation has a major effect on how conflicts over agency decisions evolve, and there are several ways in which these effects can be played out. Ideally, public participation provides a forum where the scientific information and values of the public and the agency can be integrated so the final decision is viewed as both desirable and feasible by the broadest portions of society. It can make agency decisionmaking processes transparent, and allow the public and the courts to see the extent to which the agency has taken a hard look at issues. Less well done, public participation can serve to raise expectations that cannot be fulfilled, or to gloss over fundamental value differences.

There are some who contend that the Forest Service handles public participation poorly more often than it handles it well. They argue that, at best, the Forest Service uses the results of public participation to make marginal changes in decisions, and at worst uses them to sugarcoat decisions already made. Based on data from the RARE II process in several states, Mohai (1987) contends that statistical support is lacking for the Agency’s contention that public comment would be a factor in roadless area allocations. Using his personal experiences as an environmental advocate in southern Oregon, Brittel (1991) argues that the Forest Service has used public participation, and, indeed, its entire planning processes, to rationalize and substantiate decisions that already are made.

Several ways to evaluate the effectiveness of public participation processes are available, but an obvious one is the number of postdecision appeals that result. Other things being equal, a public participation program that produces fewer appeals per decision is better. The link between appeals and public participation is that appeals are a measure of discontent with both process and outcome. To the extent that public involvement can alter the outcome, or show the public why the outcome might be acceptable and that the decision process was thorough and deliberate, the frequency of appeals should fall.

In recent years, the Forest Service decision process has been virtually choked with appeals. The number of appeals ballooned into the thousands in the late 1980s. These appeals became such a burden that, in 1992, the Forest Service suggested a rule change to substantially restrict,the number of people who could appeal, and the types of decisions subject to appeal. In addition to appeals, there is other direct evidence of discontent: a recent survey of public participants in National Forest planning shows that 43 percent were "somewhat to very dissatisfied" with the planning process in which they had participated, 55 percent reported frustration with the Forest Service planning process as a whole, and 72 percent felt the agency unfairly favored some interests above others (Dixon 1990). Another review (Blahna and YontsShepard 1989) revealed that some National Forests were conducting public participation activities that did not contain features that the standard literature on public participation contends are crucial to success.

In sum, the number of appeals in recent years and the general tenor of the discourse related to Forest Service management indicates little support for Forest Service decisions among a vocal and powerful segment of society. It, therefore, becomes important to consider the role that public involvement may be playing in creating that discontent, and the extent to which changes in public involvement could improve the agency's relations with a broad range of the public.

## Limitations of Traditional Public Participation

Several factors of the Forest Service public involvement process seem to have contributed to the current administrative and judicial gridlock the agency is experiencing. First, as the size and complexity of the natural resource issues increases, the feasibility of any single agency making adequate decisions decreases. The conventional approach has been for the Forest Service to make decisions about its lands after modest consultation with other agencies, and for the Bureau of Land Management, National Park Service, and various state agencies to do likewise. Unfortunately, the analysis of cumulative effects required by National Environmental Policy Act, the judicial requirements for the management of endangered species, and the interest in ecosystem management make the single-agency decision approach increasingly questionable. Overbay (1992) lists six principles of ecosystem management, one being coordination across ownerships. This is an explicit recognition of the extent to which current agency structures and ownership patterns are at best insufficient and at worst incompatible with ecosystem management. Even the seemingly straightforward management of big game species, which are owned by the state but which commonly winter on private lands and summer on Federal lands, seems to defy an agency-by-agency, owner-by-owner strategy.

Public participation often is structured as an internal-external, us versus them, relation. In that context, strategies of both the agency and the public likely become competitive rather than collaborative, centered around the distributive allocation of a fairly fixed set of resources. It is very difficult in such a situation to develop incentives for innovative problem solving that can incorporate and integrate the parties' interests (Wondolleck 1988). Thus, any emergent creativity occurs in spite of the structure of the public participation, not because of it.

In addition, public participation occurs in a rigid format. Because the agency's public participation is largely the result of external mandates, there is a considerable body of legislation, regulation, and case law collectively defining the adequacy of public participation (U.S. Congress, Office of Technology Assessment 1992). Usually those requirements are crafted in terms of specific periods for public comment, each with a minimum number of days and a minimum number of local papers in which legal notices must be published, and so on. Quite understandably, a common agency response is to comply with those minima and not undertake additional or different kinds of public participation, lest it create additional delay or an unforeseen procedural error. Going beyond the letter of the law is not precluded, but there seems to be little incentive for taking that risk. Public involvement practices, therefore, are not situation specific, as much of the literature suggests would be helpful.

Finally, a catch-22 comes from agency personnel focusing on the appeals and litigation process. Fear of having decisions challenged or overturned creates a defensive stance, where the strategy becomes one of crafting “bulletproof” decisions. Unfortunately, this orientation often is perceived as suspicious and confrontational by interest groups, in turn increasing the likelihood of adversarial relations and ultimately the very appeals that motivated the Forest Service behavior initially.

The public participation process of the Forest Service is not successfully managing the conflict inherent in land management decisions (USDA Forest Service 1990). The reasons for this lack of success may be linked to three factors: aptitude, motivation, and structure. It may be that Forest Service personnel are inadequately trained for the difficult task of managing multiparty, multi-issue conflict. A second explanation is that there is not sufficient motivation to manage the conflict, perhaps because the Forest Service can more readily achieve its goals by appearing to compromise between irreconcilable interest groups (O’Toole 1988). A third is that the complexity of public lands conflict may overwhelm public participation as a conflict management structure.

The most likely explanation for the mixed success of public participation is a combination of the three factors. Unfortunately, the research reported in this paper can contribute little insight about aptitude or motivation and thus attention turns to structural issues, such as, what changes to Forest Service public participation practices would result in greater ability to address the complex conflicts facing the agency? This question is particularly timely because the complexity of conflicts will no doubt increase as agencies shift to ecosystem management. Perhaps the most promising alternative would be a shift away from a competitive “us against them” model toward a more collaborative “us against the situation” perspective.

## Structuring a Collaborative Orientation

Collaboration is a process by which interdependent groups work together to affect the future of an issue of shared interests (Gray 1989). As a relation between an agency and the relevant public, collaboration differs considerably from the traditional public involvement model. The following is a list of six major attributes of collaboration that differentiate it from traditional public involvement procedures:

- Collaboration is less competitive and more accepting of additional parties in the process because these parties are viewed as potential contributors more than as potential competitors.
- Collaboration is based on joint learning and fact finding; information is not used in a competitively strategic manner.
- Collaboration allows underlying value differences to be explored, and there is the potential for joint values to emerge.
- Collaboration resembles principled negotiation, because the focus is on interests rather than on positions.
- Collaboration allocates the responsibility for implementation across as many participants in the process as the situation warrants.
- Collaboration is an on-going process; the participants do not meet once to discuss a difference and then disperse. Collaborations may have limited life spans, however, if the issues that brought the participants together are resolved.

These distinctions between collaboration and public involvement can be encapsulated into two philosophical differences. The first is that the agency cannot adequately address the issues at hand by working independently. There may be any of a number of additional resources that the agency needs that collaborators might bring to the process: different perspectives on the problem at hand and potential solutions, understanding of rapidly changing social values, scientific data, political clout, agreement and coordination of other agencies and private land owners, finances, volunteer labor, and so on.

The second philosophical difference is that collaboration is based on cooperation, whereas public involvement has evolved to be centered on competition. Although there was no a priori reason for public involvement to develop along a competitive orientation, it, nonetheless, has. Public involvement is firmly embedded in the adversarial comment-appeal-litigate-legislate mentality that characterizes much of public lands politics. A call for collaboration is not a starry-eyed proposal ignoring the current venom and rancor; rather, it raises the possibility that the energy currently devoted to competitive behaviors can, in some instances, be channeled into developing new approaches to resource management.

As discussed above, because the focus of land management is changing from specific resources (stands of trees or herds of big game) to ecosystems, collaboration seems better suited to the planning and implementation tasks than does traditional public involvement. Collaboration arranges the relations among the stakeholders in a manner that more closely matches the resources and responsibilities that each brings to the process.

Theory suggests two keys to shifting the relations in public land management away from competition and toward collaboration: correctly select those situations where collaboration is an appropriate strategy, and structure the process so that it is easier and more rewarding to cooperate than to compete. Not all situations are amenable to collaboration. The complexity of natural resource conflict implies that there are many reasons in any given setting why it is not realistic to expect collaborative behaviors to emerge. Some research indicates that, in fact, collaboration may be successful in the minority of cases (Amy 1987, Buckle and Thomas-Buckle 1986). It also is unrealistic to merely announce that a collaboration is beginning and expect the current relations and patterns of behavior to change. Collaboration is going to require innovative decision-building structures, and they will have to be designed with considerable attention to the incentives they create. If these structures do not create clear rewards for collaboration and disincentives for competition, there is no reason to expect much change.

One final note on the nature of collaboration is that it does not expect the participants to set their self-interests aside, and the success of collaboration does not hinge on their doing so. Quite the contrary, participants are expected to clearly voice their interests and energetically work to achieve them. The key is that their efforts are oriented not in opposition to those of their fellow participants, but in concert. It is therefore necessary to create an environment where exploring differences is not hindered but encouraged. If differences are not openly addressed, they may fester below the surface and become the impetus for discontent with the process and dissatisfaction with the results.

## Case Study Research

If the public participation procedures of the Forest Service are not up to the task of managing the conflicts inherent in public land management, and a more collaborative approach seems to have more potential (at least theoretically), case study research should reveal two trends. First, the number of decision processes that develop outside the traditional single agency-public participation model should increase. To the extent that agency personnel and the public recognize that their needs are not being met by business as usual, they will pursue other forums. Second, such processes should develop a more collaborative tone, and those that are indeed collaborative should have some advantages over those that evolve more competitively. The collaborative tone should arise if participants recognize the extent to which their interests are interrelated; if participants are able to achieve a collaborative process, positive effects will be apparent, assuming our understanding of land management politics and negotiation theory is correct.

With these thoughts in mind, we initiated an inventory and analysis of the recent cases of negotiation on natural resource policy in the Pacific Northwest. We have had only a brief time to perform this research (roughly 10 weeks), and because many of the cases are still on-going, it is inappropriate to label them as successes of failures, or even collaborative or competitive, at this time. Most have, in fact, succeeded in at least some small way (at a minimum producing a shared understanding of stakeholders' interests), and even the most widely touted cases probably failed to reach their potential. Several cases have resulted in recommendations or management proposals and are currently progressing through administrative, legal, or political processes. Whether these cases ultimately will be upheld as legal, if they will be adopted and implemented, and if they produce the desired results are unknown. The discussion that follows provides a brief description of the most general trends and important features of the case studies we reviewed. Descriptions of each case study and the sampling procedures employed are available from the authors on request.

## Discussion of Cases

The sheer number of natural resource negotiation cases (n=56) that we identified lends support (however qualified at this point) to the notion that parties interested in natural resource issues are turning to alternative means of decisionmaking. If traditional public participation and decisionmaking processes were adequately integrating the public's values and generating viable proposals, it is hard to see how this number of cases could arise outside those venues.

The complex nature and variety of cases examined here make summary statistics questionable, other than sorting cases by issue as shown in table 1. Despite the exploratory nature of this analysis, it provides direction and reference for future research efforts; in the interim it offers a glimpse into the extent and nature of current efforts to manage natural resource disputes. We examined the extent to which these cases were collaborative by examining the six features of collaboration discussed above (i.e., access afforded to stakeholders, joint learning and fact finding, exploration of value differences, interest- versus position-based discussion, shared implementation responsibility, and life cycle).

**Table 1---Distribution of Cases by Primary Resource Issues[1] [2]**

| Issue | Site-specific | Policy |
|---|---|---|
| Natural resource management and public land use: | | |
| Forest management: | 9 | 3 |
| Native sites | 1 | |
| Wilderness | 1 | |
| Watershed management | 4 | |
| Fisheries | 1 | |
| Oil and gas development | 1 | |
| Range management | 2 | |
| Water resources: | | |
| Water quality | | 1 |
| Water supply (irrigation, groundwater) | 2 | |
| Air quality: Agriculture | 1 | |
| Land use: Rural planning and growth control | | 1 |

[1] Table format adapted from Bingham (1986).

[2] Many cases contain a mix of causes, effects, and stakeholders; they, therefore, may defy precise categorization (e.g., the dispute involving irrigation was focused on irrigators' possible degradation of fish habitat).

## Stakeholder Access

This issue involves three fundamental questions: Who is included, at what time, how, and for what apparent reason? How are new participants treated? How do size restrictions affect the process?

Most of the cases in our research reflected an inclusive philosophy rather than an exclusionary one. Outright exclusion of stakeholders, when it arose, was manifested in several ways: the parties were omitted from an invitation list or were not among those hand-picked for working groups (e.g., Guiding the Course case study); or the rules of collaboration and issues addressed excluded meaningful participation by the parties (e.g., preservation-oriented parties attempting to influence a primarily grazing-oriented process, such as in the Leslie Ranch case study). In these cases, the sponsor played a major role in determining the agenda. Not surprisingly, the excluded parties often pursued other avenues of recourse to influence the outcome and often had to be dealt with in the future. In addition, many of the excluded or disenfranchised parties expressed extreme hesitation at participating in any future efforts of similar nature (High Country News 1992). So, not only did exclusion jeopardize the long-term success of the collaboration-management strategy, it alienated an interest group and soured the party on the concept of collaboration in general. Usually the issue of how to include new participants was not an important factor. Either the mediator-convener had been very thorough in assembling critical parties, so much so that the issue of neglected parties did not arise, or the party was omitted as described above.

Theory predicts that parties base their involvement in collaborations on the results they expect. In those cases where a party perceived the dispute was influenced by a stakeholder with different interests or with a history of “unfavorable” decisions or untrustworthy processes, the party was less likely to participate in discussions (Trout Creek, Leslie Ranch). There were cases, however, where a dissatisfied party invested considerable resources in the discussions, even after concluding that the process probably would not satisfactorily address that party’s interests (Trout Creek, TFW).

## Joint Learning and Fact Finding

This issue features two questions: To what extent does joint learning occur? How is information transferred or manipulated?

In most cases, parties shared certain types of information rather freely (Leslie Ranch, Spencer Project, Pine Eagle, many others). This information was usually factual and related to a specific task (e.g., how to best preserve ecosystem integrity while still producing a commodity). Parties were willing to invest time in discussions that debated factual issues of resource management and seemed interested in joint problem-solving, as long as their proposals were taken seriously. Some parties generally believed that the public did not understand their concerns, so they were eager to tell their side of the story. Disparate interest groups rarely shared information on other avenues they might use for influencing the outcome of the discussions. All parties could pursue options outside the group, and often they would conceal this information or strategies to maintain a collaborative posture (Shasta Costa, Bonner’s Ferry). At Bonner’s Ferry, local environmentalists communicated with the regional representative of a national conservation organization living in another state. This communication resulted in a visit from the regional representative that altered the tone and outcome of the discussions.

## Exploration of Value Differences

If parties advanced beyond the issues of trust, jurisdiction, and factual differences, the next stumbling block was values-based differences. In cases described by participants as collaborations, the values of individual participants were explored early in the discussions or were inferred from a narrative that parties provided describing the kind of ecological and social settings they desired, a kind of “visioning process” (Spencer Process, Pine Eagle, Shasta Costa, and others). Emphasis usually was placed on respect for others’ values, and parties were encouraged to concentrate on interest- and fact-based problems. Interestingly, agency representatives often had a difficult time expressing their organizational values; rather, actions were attributed to “policies” or “mandates.”

## Discussion of Interests Versus Positions

Are the natural resource and forest dispute discussions primarily interest based, reflecting integrative-collaborative strategies, or position based, indicating distributive-competitive strategies?

Our examination of the dispute-management processes and the content of these cases indicates that interest-based discussions have a greater likelihood of achieving some substantive progress toward settlement than do position-based discussions. Of the 27 cases studied in depth, 13 moved substantially from position-based to interest-based discussions; of these, 10 settlements satisfied all major stakeholders. Four cases remained fixed in position-based strategies and ended in failed negotiation. Two cases began as interest-based discussions and continued as such, ending in agreement. Process and settlement information regarding the other cases is incomplete.

In the Pine Eagle Group case, which used an interest-based emphasis, conservation groups were willing to compromise on early positions (e.g., no logging in roadless areas) in favor of final plans that they perceived as meeting the interest of the larger ecosystem. This case is noteworthy in that it involves primarily National Forest land, and even though the District Ranger involved has been transferred and replaced twice, the process has continued.

Conversely, in the Shasta Costa case study, which featured positions, some stakeholders were committed to a position of not allowing entry into roadless areas (after representing this attitude to constituents) and could not or would not discuss harvest in these areas regardless of ecosystem considerations. This also happens to be a case where the relevant Forest Service line officers have either retired or transferred, but the effect of these personnel changes has been criticized by stakeholders.

Other cases involved a shift from positions to interests. For example, in central Oregon, stakeholders disagreed about desirable boundary lines for the proposed National Monument at Newberry Crater and began to lobby for their preferred boundaries (positions). By breaking the boundary area into physiographic sections (fractionating the boundary issue) and asking parties to lobby for their interests, the mediator moved the group through consensus on the entire boundary in one marathon session.

## Shared Implementation Responsibility

The case studies we reviewed revealed some interesting insights on the issue of implementation. The premiere issue of contention often was who had jurisdiction and implementation responsibility. The issue of jurisdiction responsibility seems most pronounced in disputes involving Federal lands. It is one thing for a District Ranger or Forest Supervisor to allow a collaboration to proceed; it is entirely another matter to truly delegate decision authority to it. Therefore, potential problems can arise when the collaboration yields an outcome that the line officer rejects: the agency appears inconsistent at best and duplicitous at worst. A recent situation at the Willamette National Forest provides a perfect example of this pitfall. A public participation group was established to advise the Forest's planning team after the Warner Creek fire. The group never reached a consensus about timber salvaging, and the Supervisor proposed a timber salvage amount higher than most of the group supported. This seems to have produced substantial negative reaction in the press and in segments of the local population. The topic of delegating authority to a collaboration is touchy, and it seems that participants develop a sense of ownership in the outcome that may exceed what has been granted them.

## Collaboration Life Cycle

The following life cycle questions emerge as important: What is the effect of time considerations on the involvement of interest groups? What is an average length of time for a successful collaboration? What sequence of techniques substantially contributes to a satisfactory outcome?

When convinced that collaborative effort was in their best interest, participants usually were prepared to see it through to completion. Typically, for important projects (large physical areas, precedent setting cases, amendments to, or drafting of Forest plans, and so on) all participants could commit substantial time and resources to the effort. Even in those cases that can not be considered successful (e.g., Shasta Costa), parties that eventually broke off or were disenfranchised were willing to participate substantially, despite their misgivings about the endeavor. It should be noted, however, that dissatisfied parties from these types of cases, and anyone in communication with these parties, may abstain from similar future discussions (Brittel 1991).

These cases indicate considerable variation in the duration of site-specific natural resource collaboration efforts, similar to that found by Bingham (1986) in another set of environmental dispute cases. Certain attributes, in addition to those already mentioned, can be identified as potential ingredients for productive and long-lived collaborations. Based on our review process, the following sequence of successful social conflict resolution techniques are apparent: (1) appropriate resource commitment by at least one party; (2) early consultation with and possible constant involvement by an impartial facilitator or mediator; (3) "introductory" social sessions (presentations by all parties, value identification, discussions, ground rules, and so on); and (4) formation of task-oriented working groups and presentations by experts viewed by all parties as credible.

## CONCLUSIONS

Perhaps the fundamental tenet of faith in land management is that it is a field of applied biology. Such a view is a simplification of potentially grave significance. Obviously forests, grasslands, and riparian areas are biological, but their vexing management problems are as much social and political as biological. One result of the historical fascination with the biological and physical aspects of land management is that biological factors often are better understood than social aspects. Fortunately, there are many parallels between the two, and the latter can be informed by a knowledge of the former.

Ecologists recognize that biological diversity is important to most land management and may in fact be an inherent part of healthy ecosystems. But by the same token, diversity of social values and interests is equally inherent in ecosystem management on public lands, and perhaps as important. To the same degree that land managers strive to work in concert with biological processes because they cannot truly be controlled, so must they labor to use potentially overwhelming and diverse social forces. The management of wildlands must include practices cognizant of ecosystem structure and composition; however, the same consideration of social structure and composition must go into the design of the discourse by which natural resource policies are crafted and implemented.

In sum, a simplified model of ecosystem management that addresses only biological issues does not appear robust enough to supply adequate amounts of goods and services while maintaining ecosystem health. Accordingly, new approaches to managing the social debate about ecosystem management need to be used in its implementation.

What kinds of structures for the agency-citizenry relation should be explored as part of this process? If our assessment of the situation is correct, the missing features in the current public participation model are joint decisionmaking based on an integrative orientation, and shared implementation of the decision (i.e., collaboration). The features of a collaborative model deal with the deeply held value differences and multifaceted complexity of natural resource conflicts more completely that do any alternative design.

Although collaborative approaches may have the highest likelihood for progress, our research into the on-going collaborations in the Pacific Northwest show that they are not easy, nor do they guarantee success. Neither the long history of divisive politics nor the existing competitive institutions and incentives will be washed away by a few grassroots efforts to cooperate. Even so, recent collaborative efforts provide important lessons.

This research had neither the time nor resources to examine the on-going collaborations to the extent they deserve. One of the greatest impediments to initiating collaborative processes may be the participants' inability to predict how the process will evolve and with what results. One way to allay the doubts such uncertainty creates would be to have a set of case histories and guidelines about collaboration. If such information was available, every new collaboration would not reinvent the wheel unnecessarily and could build on the lessons of previous attempts.

Another research area of potential benefit would be a series of operational experiments, actual attempts at collaboration where the goals are not only substantive progress but also new information about the process. In this sense, the experiments would become demonstration projects very similar to the biological demonstration projects with which the Forest Service has considerable experience.

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# Adaptive Ecosystem Management

**R. Everett, C. Oliver, J. Saveland, P. Hessburg, N. Diaz, and L. Irwin**

## ABSTRACT

A management approach is needed to deal with the uncertainties of implementing ecosystem management. Adaptive management is appropriate to this need, because (1) it is based on the concept of management as an experiment, (2) it accepts uncertainties, (3) it requires quantification of objectives, (4) it emphasizes a stated understanding of system operation, and (5) it provides a rapid feedback and evaluation loop for redirection of experiments. Adaptive ecosystem management promotes the integration of social, economic, and ecological issues in land management planning and their expression in landscape patterns of resource values.

## INTRODUCTION

Sustainable ecosystem management (Overbay 1992) is an untested design formulated to provide for transitory values and expectations of society while maintaining or enhancing long-term ecosystem fitness and integrity for future generations. To respond to the dissatisfaction of society with commodity-oriented public land management and to reverse trends in declining health of National Forests, the USDA Forest Service proposed and adopted the paradigm of sustainable ecosystem management (Jensen and Everett 1993). The enormous task now at hand is to figure out what ecosystem management means and how it can be accomplished.

Although land managers have learned much in 90 years of resource management, knowledge of ecosystems and their kinds, component species, interactions, temporal and spatial scales, and management, is rudimentary. There are species in ecosystems that have never been identified, and processes still unknown. Nonetheless, these unnamed species and processes may be vital to ecosystem structures and flows (Diaz and Apostol 1993).

To better understand ecosystem management, the process of adaptive management has been suggested and widely embraced. This paper discusses some of the merits of an adaptive management approach to ecosystem management. In an adaptive management process (Walters 1986, Walters and Hilborne 1978, Walters and Holling 1990), society, public land managers, resource management professionals, and scientists enter into a partnership to regularly reshape management goals, redefine objectives, and redirect management actions in response to changing socioeconomic information and evolving biological, physical, chemical, and environmental conditions. Resource management decisions have always been and will continue to be made with a measure of uncertainty. Land managers can begin to minimize that uncertainty by continuously improving the knowledge base of decisions. The increasing complexity and interaction of environmental and social issues assure knowledge will continually need to be updated (Sonntag et al. 1980).

### Dealing With Ecological and Social Uncertainty

Land managers have limited but increasing knowledge of ecosystem structures, processes, interactions, and species. For this reason, their capacity to manage wisely is also limited. The coarse-filter management approach of Hunter (1991) recognizes this lack of knowledge and experience, and prescribes the conservation of plant communities and whole ecosystems as a means of conserving known and unknown species, options, structures, and processes contained therein. Coupling the coarse-filter management approach to a hierarchical landscape framework extends this concept of “conservation of options out of uncertainty” to each important temporal and spatial scale.

### The Relation of Society to Resources is Complex

Although land managers know that social values and expectations interact with biological and physical capacities, economics, and technology (Jensen and Everett 1993, Zonneveld 1988), exact mechanisms are still unknown. Social and economic communities at national, international, regional, and local scales directly and indirectly participate in the resource management decision processes (Kennedy and Quigley 1993, Shepard 1993). Rates of resource extraction and resultant landscapes are social decisions, but decisions must be constrained by biological realities (Swanson and Franklin 1992). Not only is managing within biological constraints essential from practical and scientific points of view, it also is increasingly important to society that the effects of current management practices do not adversely affect options and quality of life for future generations (Kessler et al. 1992).

An approach to public land management is needed that recognizes that although social and biological uncertainties exist, land managers can advance more sustainable ecosystems. A management approach designed for constantly changing systems is appropriate to ecosystem management; “variability, not constancy, is the feature of ecological and social systems that contributes to their persistence and to their self-monitoring and self-correcting capabilities” (Sonntag et al. 1980). A management system that recognizes that people are integral parts of ecosystems is necessary as human influence continues to expand locally and globally. The adaptive management process accepts uncertainty and suggests that a continuous learning process is needed to redirect management as new socioeconomic, biological, or political information is available (Holling 1978). Adaptive management requires active participation by society in the design, implementation, monitoring, and redesign of ecosystem and landscape management experiments.

## Adaptive Management Using a Modular Approach

The expected goals of management will not be achieved within a management system if any of the natural, societal, or other components are incorrectly understood or organized in the management system (i.e., the system will be dysfunctional). Feedback is cumbersome in complex systems if only the final result is monitored. Instead, a system can be designed in modules (Oliver et al. 1993) so that monitoring can be done within and between modules continuously. The modular approach addresses error in a management experiment at the module level and allows the needs for decisions and information to be identified, isolated, and improved without disrupting the entire management system. The modular approach can be extended down to project-scale work, where individual ecological functions (elk habitat, wood production) are represented as modules of ecosystem management (Covington et al. 1988).

## Increased Focus and Flexibility of Adaptive Ecosystem Management

Adaptive ecosystem management assumes that uncertainties in current and future ecological requirements, and societal values and expectations, limit our ability to draft long-term ecosystem management strategies. Uncertainties include variability in biological and social systems, errors of estimation, bias in projection models, and ambiguous questions (Marcot 1986). “Not only is the science incomplete, the {eco}system itself is a moving target, evolving because of the impacts of management, and the progressive expansion of the scale of human influences on the planet” (Walters and Holling 1990). Historical changes in management emphasis of Pacific Northwest forests from protection of water rights and yields, to livestock grazing, to timber harvest, to threatened and endangered species, and currently, to sustainable ecosystems, clearly demonstrates the transient nature of the preferences of society in use of National Forest lands.

Risk of failure or error increases with increasing uncertainty. “The willingness to risk failure is an essential component of most successful initiatives. The unwillingness to face the risks of failure--or excessive zeal to avoid all risks--is, in the end, an acceptance of mediocrity and an abdication of leadership” (Shapiro 1990). Adaptive management addresses risk and uncertainty by increasing opportunities to redirect management with new information.

Although flexible, adaptive management is accountable to defined goals, Marcot (1992) describes a simple risk analysis model that could be useful for the selection of treatments (fig. 1). When the “risk of error” (the likelihood that projected outcomes will fail), and “risk of loss” (the significance of failure) are both high, projected goals should be revised or intensively monitored. “Risk is a measure of the probability and magnitude of an adverse effect, and risk assessment consists of identification, estimation and evaluation” (Saveland 1985). Each adaptive management process will have associated risks, but risk to ecosystems is present with or without human intervention.

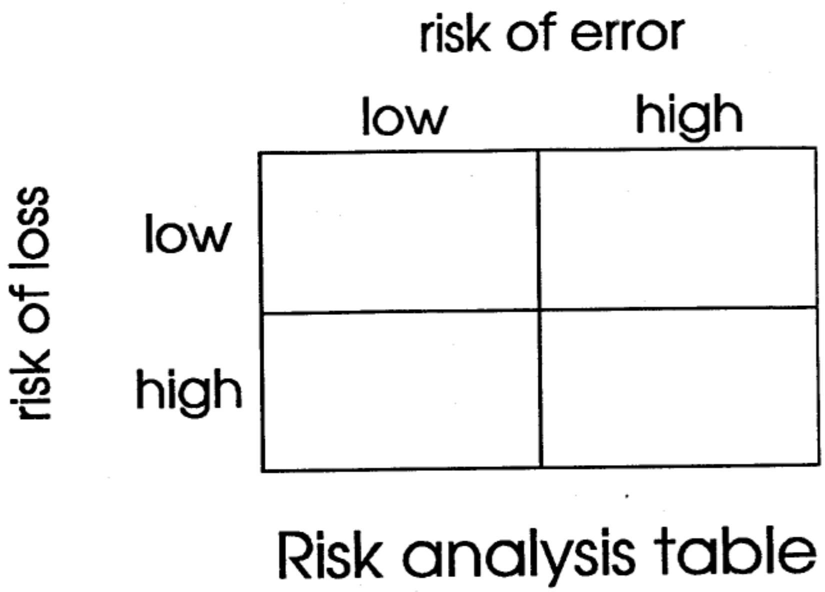


**Figure 1--Risk analysis table used In assessing potential for error and Importance of making an error In adaptive management (Marcot 1992).**

Adaptive management does not require that managers and scientists have a perfect understanding of ecosystems, but it does require quantitatively explicit hypotheses about system structures and processes, a clear statement of management goals and objectives, and a set of targeted actions (Holling 1978, Walters and Holling 1990, Baskerville 1985). Adaptive management not only tests whether goals are achieved, but also assesses the merit of the goals and the validity of the primary assumptions of the experimental design. Although the achievement of management goals may not have meaning beyond a particular case, the validation of assumptions can have far-reaching value. Opportunities for extrapolation can be enhanced if analyses are linked to a standard ecological classification (Jensen et al. 1991, O'Hara et al. 1993), and if basic ecological processes are acknowledged by such classifications (Bailey et al. 1993).

Baskerville (1985) provided nine steps to adaptive management of forested ecosystems:

1. Establish measurable goals for management
2. Explicitly define cause-and-effect relations for natural and management-induced processes
3. Design sets of actions that will achieve the goals of management
4. Implement management actions
5. Periodically assess progress and cause-and-effect relations
6. Compare actual system performance with forecasted performance
7. Evaluate the appropriateness of goals and forecasts of system performance; refine the conceptual model, redesign goals, and develop new management actions if the model and goals require adaptation
8. Implement new actions
9. Return to step 5 for reiterative evaluation

Active and Passive Adaptive Management

Walters and Holling (1990) described three approaches to managing ecosystems where outcomes of management actions are uncertain: deferred, passive, and active adaptive management. Each approach bases adjustments to management actions on an ongoing learning process; the rate of learning and the degree of accountability to management goals, however, increases from deferred, to passive, to active adaptive management. With deferred adaptive management, no action is taken until systems are sufficiently understood to predict outcomes with fairly high certainty. Passive adaptive management is a process of observation and adjustment according to an assumed correct management trajectory (Walters 1986). The management model is not viewed as experimental, but as correct until proven otherwise. The failure of passive adaptive management lies in its limited discovery along a single route and the relatively high risks of failure and loss.

Active adaptive management views each goal and management action as an experiment. This process tests hypotheses to maximize short-term information gains (redirection), and long-term definition of goals. Alternative pathways can be evaluated for a single management emphasis, and multiple hypotheses can be tested simultaneously (Walters and Hilborne 1978). For example, to create wildlife habitat for a threatened species (such as the northern spotted owl or red-cockaded woodpecker), Irwin and Wigley (in press) suggest simultaneous implementation of alternative landscape- and stand-scale options concurrently with an apparent preferred strategy. Irwin and Wigley suggest that multiple working hypotheses avoid bias toward any one approach. Information gain per unit of time increases, and conservation of known and unknown limiting or declining resources will more likely occur by this approach.

## Adaptive Management and Feedback

Kessler and others (1992) suggested that feedback concepts that describe ecosystem processes should be extended to adaptive management systems, especially where feedback allows for adjusting management actions to maintain ecosystem integrity. Adaptive management depends on negative and positive feedback in the reiterative evaluation of both the continued desirability of management goals and progress toward their achievement. Clear evaluation criteria must be used that will trigger reevaluation of management direction (Marcot 1992). Marcot used the example of a management decision to change locations of habitats allocated to woodpeckers when sites were not occupied for 3 years.

The reiterative approach causes management execution and adaptation systems to make progress on goals, even if goals change with time (Baskerville 1985). A case in point was an assessment of habitat requirements for northern spotted owls that, through a process of reevaluation, became increasingly focused on potential conflicts between owl habitat requirements and sustainability of associated ecosystems. It became apparent (Hessburg 1992, Hessburg and Everett 1992, Hessburg and Flanagan 1992) that unless larger landscape issues of increased hazard of fire, insect outbreaks, and disease epidemics were resolved, maintenance of owl habitat was moot (Everett et al. 1992, Irwin and Wigley, in press).

Adaptive ecosystem management promotes an information-rich environment and a rationale for routinely monitoring and evaluating social, political, and biological environments. Feedback loops with society exist to some extent, via forest and project plan scoping activities; participation in project design, analysis, and review; special public forums; and in worst case scenarios--litigation and legislation. The latter actions tend to stifle adaptive management because management actions are reduced to the knowledge available at the time of legal action. Still, an expanded social feedback mechanism for adaptive ecosystem management within the National Forest System has not been defined (Oliver et al. 1993).

Adaptive ecosystem management depends on an evolving understanding of cause-and-effect relations in both biological and social arenas. In the social arena, communities interested in the issues must be identified, and their values and expectations understood (Daniels et al. 1993, Montgomery 1993). Although social and biological components of ecosystems are ill-defined, land managers must at least explicitly state hypotheses and proceed via a reiterative process toward developing management models. If a management model operates outside a range of socioeconomic acceptability (Jensen and Everett 1993), the model must be reconsidered, or if the model is constrained by biological realities, society must be informed of the infeasibility of the goal.

The character of societal values of society has been effectively expressed in the implementation of both the Wild Horses and Burros Act of 1971, and in documents such as the Northern Spotted Owl Interagency Scientific Committee Report (Thomas et al. 1990). In both instances, the affected human community selected a putative biological conservation approach and described the desired characteristics of those ecosystems. In the Wild Horses and Burros Act, nonnative equids were protected. In the case of the Interagency Scientific Committee Report, a native owl species was protected, and there was provision for an adaptive management approach; a process was created to allow revision (with some difficulty) in response to the development of new information.

## Timeframes of Adaptive Management

Evaluation of ecosystem responses to management actions may take decades for large-scale landscape experiments, even for rapid response variables; and replication will be required to eliminate time-treatment interactions. Long-term modeling of alternate landscape patterns and rates of change should be examined though a transition rotation and at least one fully managed rotation under an adaptive management process (Swanson and Franklin 1992). In some instances, these timeframes will easily exceed 300 years.

## Thresholds of Evidence

Adaptive management is based on experimentation that continuously adjusts ecosystem and landscape goals and management actions. How do scientists and managers know, however, when it is no longer fruitful to pursue an experiment? Decisions on the continuation or adjustment of adaptive management experiments are made on the basis of thresholds of evidence (Saveland 1991). If a preponderance of evidence suggests that a critical threshold has been exceeded, then redirection is required. Thresholds that are too low are oversensitive to potential adverse effects, and unduly restrict some resource uses. Conversely, thresholds that are too high are insensitive to adverse environmental effects. A balance between these two thresholds is needed.

Every management decision has a measure of associated uncertainty that is based on conditional probabilities: that forecasted effects occurred (hit); that nonforecasted effects occurred (missed); that the absence of effects was correctly forecasted; or that effects were incorrectly assumed (false alarms). Management practices are adjusted when adverse effects occur, and decisionmaking processes are reevaluated when nonforecasted events occur, or forecasted events are absent.

Probabilities for each kind of outcome are expressed in receiver-operating characteristic curves (Saveland 1991, Saveland and Neuenschwander 1990). The plot of correctly observed responses against false alarms determines the shape of receiver-operating characteristic curves (fig. 2). A decisionmaker can select a balance point along a receiver-operating characteristic curve to reflect a conservative, neutral or high-risk strategy for recognizing adverse effects.

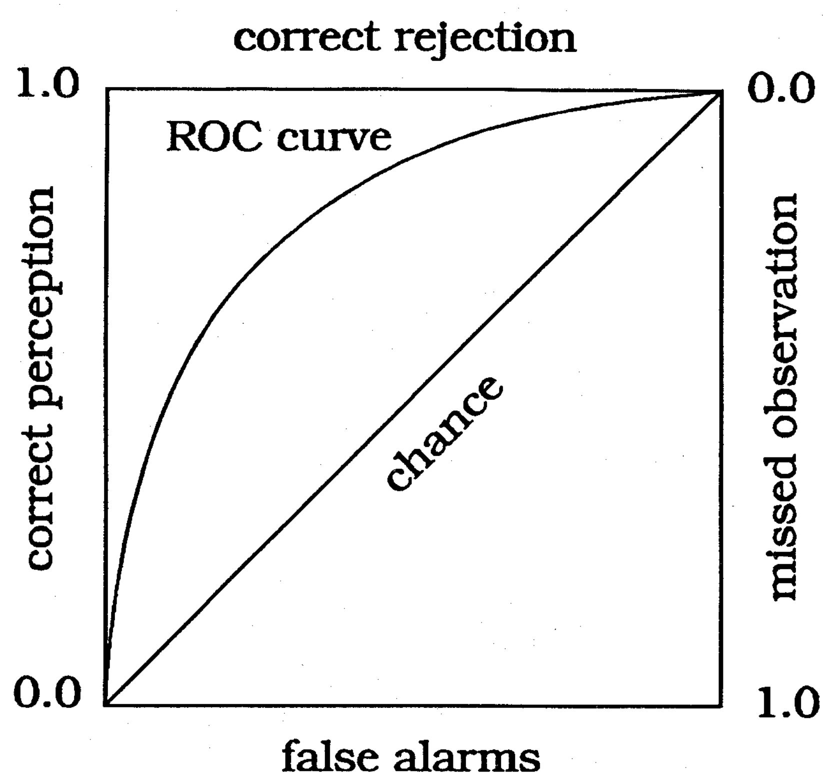


**Figure 2--Probabilities for making correct decisions on the presence or absence of important effects as expressed in receiver-operating characteristic curves (Saveland 1991, Saveland and Neuenschwander 1990).**

The simple risk analysis model described in figure 1 (Marcot 1992) could also assist decisionmakers in selecting a balance point. For decisions where both risk of error and risk of loss are high, a conservative balance point is warranted. A more liberal balance point can be adopted where both risks are low. Society, too, by expressing its values and expectations, suggests a kind of balance point for recognizing adverse effects associated with management experiments (Montgomery 1993). Historically, the Forest Service has maintained a liberal balance point, and has failed to detect significant adverse effects on ecosystems (Shepard 1993). This fact contributed, in part, to the creation of the National Environment Protection Act process. Under a conservative strategy, thresholds are reduced, and the probability of identifying adverse impacts is increased. The potential for false alarms is increased as well. The balance point has changed in recent years from an emphasis on correct rejection of no adverse impacts (minimize false alarms), to an emphasis on correctly identifying adverse effects when they occur (minimize misses). The future holds the prospect of increasing hits and correct rejections through adaptive resource management. The recent use of historical ranges of variability for terrestrial and aquatic ecosystem attributes (Minshall 1993, Swanson et al. 1993) to evaluate current landscape structures and processes is an attempt to move the balance point to a more conservative position that denotes potential ecosystem sustainability.

## Conserving Ecosystem Patterns and Processes

Given the complexity of ecosystem structures and processes, it might be asked which variables should most influence ecosystem management decisions. Because vegetation stand type and geographic location influence timber availability for harvest (Baskersville 1985), wildlife habitat suitability (Morrison et al. 1992), and conservation strategies (Hunter 1991), the use of variables that indicate landscape patterns and processes seems a reasonable common ground for social, ecological, and economic interests. Management of landscape patchwork for ecosystem stability and wildlife habitat was introduced in the Pacific Northwest in the 1980s (Franklin and Forman 1987) and has caused some shift in management practices to reduce forest fragmentation (Swanson and Franklin 1992).

Oliver (in press) suggests that landscapes should be managed like diversified investment portfolios, where both biological diversity and economic stability are enhanced by maintaining variety in landscapes, by managing stand structures, sizes, and geographic locations in a dynamic balance across each region and subregion. No single design is desired or practical in this system. Instead, landscape designs are constrained by diversity requirements at various scales and the inherent limitations of the environment. Diaz and Apostol (1993) suggested management of landscape pattern to express the values and environmental constraints of society as part of forest and project planning processes.

Covington and other (1988) developed a linked, geographic information and decision model system known as the Terrestrial Ecosystem Analysis Model to implement project-scale planning. In their example, alternative landscape scenarios were provided for improving elk habitat. Landscape patterns reflected the desired 40-percent cover and 60-percent forage ratio of Thomas (1979), and were constrained by steep slopes, no adjacent harvest units, and a set annual operating budget for stand treatments. Multiple landscapes emphasizing preservation, economics, or elk habitat could be evaluated simultaneously, or a single optimum could be evaluated for all products and ecosystem values (a systems approach to adaptive management) (fig. 3-a, b, c, d).

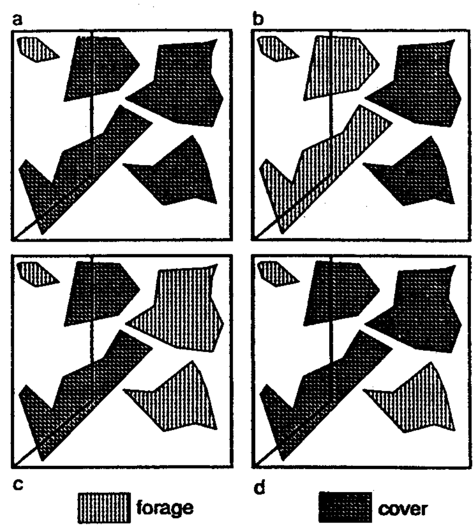


**Figure 3--An array of hypothetical landscapes defining potential management actions to increase elk forage habitat. Four options are shown: option a is no action, b is Increase forage base through the most cost-effective timber harvest adjacent to existing roads, c is Increase elk forage In dispersed off-road sites, and d is option c but within annual budget constraints (See Covington et al. 1988).**

Whether any or all of these landscapes are sustainable is unknown. A reference point, where these landscapes fit within the "historical range in variability" for structure and disturbance regimes (Swanson et al. 1993), would be desirable. Using historical ranges of variability to evaluate current landscape conditions (Caraher et al. 1992, Hann et al. 1993, O'Hara et al. 1993) provides an estimate of ecosystem conditions that were sustained before management activities of this century, and may aid in designing sustainable ecosystems of the future, with appropriate caveats (Swanson et al. 1993). We agree with Kay (1991) "that it is not possible to identify a single organizational state of a system that corresponds with ecological integrity. Instead, there would be a range of organizational states for which an ecosystem is considered to have integrity." Using historical ranges of landscape variability aids in the description of a range of alternative sustainable states, not a single sustainable state. When there is good scientific evidence (biological and social sciences) for the validity of alternative sustainable states, these should be included as sustainable ecosystem management options.

If historical ranges of variability are used to identify alternative sustainable ecosystem states, an adaptive management approach is essential because it is not possible to define all sustainable (scnsu Overbay) states according to a limited set of historical observations. Resource flows that met historical needs may not be adequate as human populations increase and expectations for commodity and amenity resources change. Environmental, soil, and climatic conditions also may change in the course of a management experiment. Rigid adherence to any landscape design would be detrimental because of unexpected events such as insect and fire catastrophes, and the ongoing evolution of ecosystems. Ecosystems should be thought of as evolving rather than fixed, with the ranges of variation for all ecosystem attributes flexing accordingly. Passive adaptive management approaches that depend solely on historical ranges of variability are constrained by the circumstances of a history that is never precisely repeated; active adaptive management allows the testing of all promising alternatives (Walters and Holling 1990). Because analysis of historical conditions can be confounded by unknown management and environmental effects, Walters and Hollings (1990) advocate investment in large-scale landscape experimentation.

Coarse-filter and hierarchical approaches to landscape management (Hunter 1991) provide a means to conduct management experiments at many scales, with a measure of confidence that resources, processes, and species will be conserved across space and through time. As diverse patchworks, landscapes provide a wide range of potential tests of treatments. By applying treatments across years, multiple experiments may be monitored simultaneously as suggested by Irwin and Wigley (in press), and time-treatment interactions will be minimized.

## Adaptive Ecosystem Management and Monitoring

The point of project monitoring under the National Environmental Policy Act of 1976 was to assess whether management actions were implemented as designed. Evaluation (also in the sense of the National Environmental Policy Act) determined whether goals were achieved, and whether a management design worked as anticipated. Under adaptive ecosystem management, a similar logic is applied, but monitoring not only assesses the effectiveness of management actions in maintaining or enhancing ecosystem integrity and fitness, it also evaluates the congruence of management actions with social values and expectations. Monitoring strategies are integral to active adaptive management strategies because they provide the feedback to adjust further experimentation. Monitoring must address each of the major social and biological variables that are germane to an ecosystem management experiment, both internal and external to a treatment area. The degree of resolution of monitoring should reflect the scale of treatments, and the spatial and temporal scales of effects (Urban et al. 1987). As an example, if land managers were evaluating an area of riparian vegetation, they would want to monitor storm flows and low flows as disturbances to that vegetation, even though changes in water flows would be controlled by influences outside the study area.

Ecosystem management experiments must be based on clearly articulated hypotheses of the workings of various components of ecosystems. Response of ecosystems to management action must be articulated as testable hypotheses, which in turn must address each working assumption. Monitoring of each management experiment allows validation or invalidation of assumptions, and acceptance or rejection of hypotheses. Such feedback can be used for future experimentation.

## Changing Landscape Characteristics on National Forest System Lands

The forest landscape analysis and design process developed by Diaz and Apostol (1993) provides a mechanism to input landscape pattern objectives into forest planning and landscape design. Target landscapes developed by interdisciplinary teams are only provisionally identified as the “desired future condition” because they are developed from limited knowledge and information. New information from ongoing landscape experiments will revise landscape management goals and target conditions. Landscape designs are developed through analysis and interpretation of social and biological needs and potential (fig. 4). Landscape objectives are shaped from local and subregional social values; national, regional, and subregional social and political goals; and regional, subregional, and local biological, environmental, and physical constraints. The landscape configurations established by the Interagency Scientific Committee report (Thomas et al. 1990) are an example of regional and subregional constraints, where conservation strategies for a threatened or endangered species are a vital part of local landscape designs. Design criteria are also incorporated from the bottom up from local community needs and values (Quigley 1992) and from biological constraints at local landscape- and stand-scales.

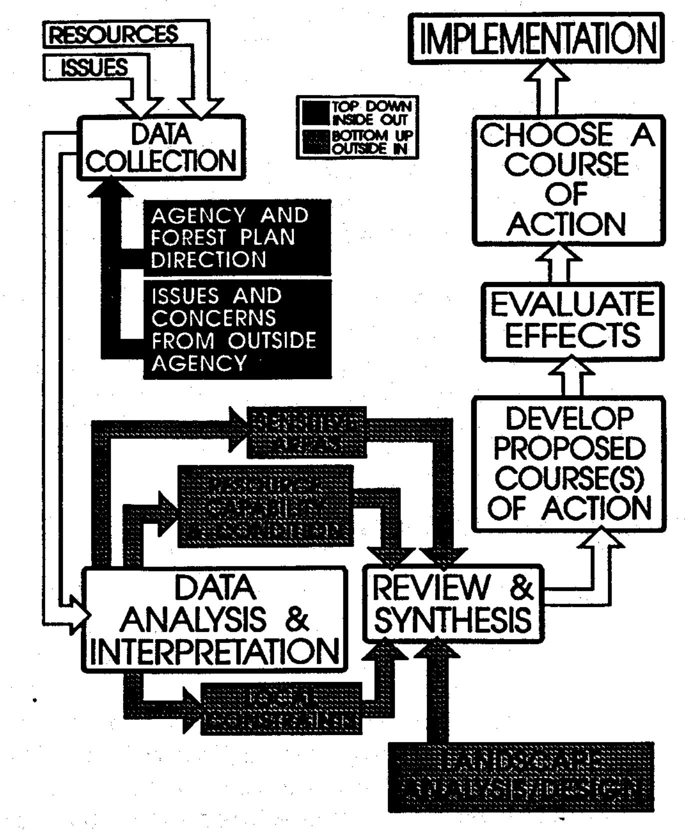


**Figure 4--A process for designing landscapes to meet biological, economical, and social interests on National Forest lands (Diaz and Apostol 1993).**

Landscape pattern objectives are determined by evaluating landscape structures and flows, and their interrelations (fig. 5, Diaz and Apostol 1993). Landscape structures (elements and patterns) determine process flows within and across landscape analysis units. The roles of disturbances and succession are defined in the context of landscape patterns and processes. For example, Covington and others (1988) provide a landscape analysis example that relates the design of elk habitat for internal process flows to timber harvest designs that emphasize resource flows outside the analysis area.

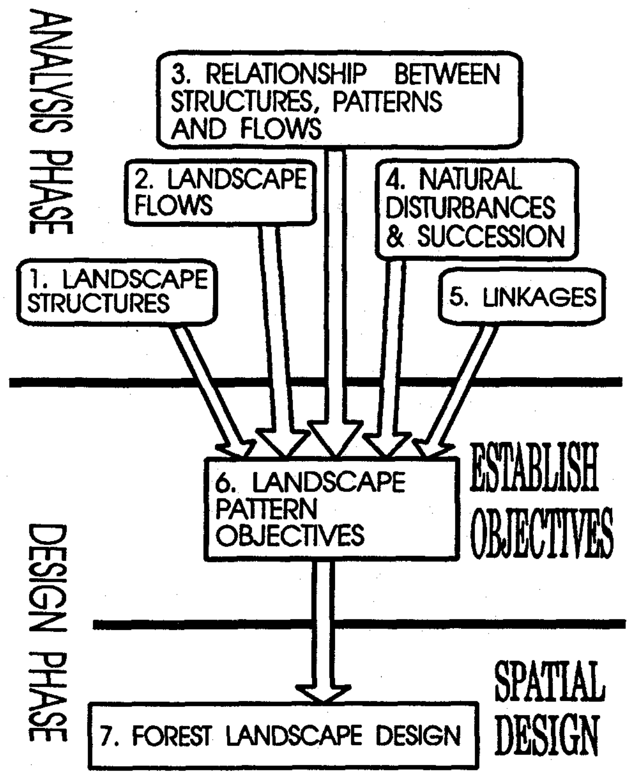


**Figure 5--Structural and process considerations in defining landscape pattern objectives and final landscape design (Diaz and Apostol 1993).**

## Linking Resource Managers with Research

Landscape experiments in adaptive ecosystem management are of such size and complexity that they can only be done collaboratively among land and resource managers, and scientists (Kessler et al. 1992). Past failures to develop policies, attitudes, and management directions that were socially and environmentally sound are linked to the historical inability of decisionmakers, scientists, resource managers, and society, to effectively interact (Sonntag et al.1980). The lack of an experimental framework to integrate new knowledge from associated disciplines has further limited effective interaction among biological, physical, and social science disciplines.

The adaptive environmental assessment and management method of Holling (1978) provides a formalized process to link ecological research with management via monitoring (Sonntag et al. 1980). This formalized approach uses computer simulation to reveal the features of alternative management scenarios, and forces participants to state their assumptions and understanding of ecosystem structures, processes, and interactions. Less formal but effective approaches were used in the development of the Interagency Scientific Committee report (Thomas et al. 1990), and the subsequent Scientific Assessment Team (Thomas et al. 1993). The Interagency Scientific Committee conservation strategy for the northern spotted owl . (Thomas et al. 1990) acknowledges the need for an adaptive management approach involving research and resource management personnel. National, regional, and local communities interact to define sustainable conditions for the owl and for other ecosystem inhabitants (Thomas et al. 1993). The next step is the implementation, monitoring, and adjustment of land management plans (Forest Plans) as grand scale experiments into ecosystem management (Kessler et al. 1992; Diaz and Apostol 1993; Morrison 1993).

## Social and Political Realities of Adaptive Ecosystem Management

Recognizing political realities is one very important key to progress in adaptive management. “New Forestry,” “New Perspectives,” and “Ecosystem Management” directives (Kessler et al. 1992) better align land and resource management with the values and expectations of society and with political realities (Shepard 1993). The adaptive management process must be aware of political realities at global, national, and local scales. The adaptive management process must take into account the broad array of public communities (Kennedy and Quigley 1993), and it must be interactive with the scientific discovery process rather than react to it. At the same time, the science components of adaptive ecosystem management must be unconstrained politically if they are to reflect good science. Short-term political and economic reasoning can violate long-term ecological and social values (Zonneveld 1988).

# CONCLUSIONS AND RECOMMENDATIONS

“It is the hallmark of scientists, but the lament of decisionmakers to qualify conclusions with uncertainties” (Marcot 1992). Sustainable ecosystem management is a large-scale experiment with an information base that is always limiting. Adaptive management provides a means to deal with uncertainties in ecosystem management by accepting each management action as a learning experience. Uncertainty is also dealt with by constantly reevaluating management goals and objectives, and social and biological cause-and-effect relations in light of new social, political, physical, and biological sciences information.

We have suggested that landscapes and their patterns can provide useful coarse-filter variables for adaptive management and monitoring. Much debate can be expected about which variables should be selected to monitor ecosystem and landscape conditions. There will be uncertainties on the sensitivity of thresholds for accepting or rejecting adverse management effects. Although actual ecosystem responses may be measured scientifically and with considerable precision, evaluation of the importance and consequences of each response is very much a social process.

Some landscape management experiments will take decades to complete, but information flow can be expedited through simultaneous study of a wide range of landscape constructions across diverse landscapes. Studies should be tied to characteristic ecological landscape units (Bailey et al. 1993, Hann et al 1993) to increase the range of applicability of experiments, and should avoid the characteristics of traonwoal agriculture plot studies where each experiment was unique.

Adaptive ecosystem management is the art of intelligent experimentation, of constantly creating new biological and physical science to integrate with transient social values in a mercurial political arena. Adaptive management is about experimentation on a grand scale, and it is also about making hard choices in socioeconomic, political, and biological arenas. People want and expect many things from ecosystems. In the long term, satisfying all expectations would exceed ecosystem capacities; consequently, all interests cannot be satisfied. Accordingly, sustainable ecosystem management involves compromise, and very long-term investment.

The management and research branches of the Forest Service should consider the full potential of an adaptive management approach to sustainable ecosystem management. Adaptive management strategies for ecosystems and landscapes should come from the collaborations of all relevant social communities, and Forest Service management and research personnel. Teams should collaborate on the concepts, designs, implementation, monitoring, evaluation, and feedback steps of the adaptive management process. To this end, administrative and funding barriers between the research and administrative organizations should be minimized, and mechanisms should be created to foster these types of collaborations above all others.

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# A System for Implementing Ecosystem Management

**C.D. Oliver, W.H. Knapp, and R. Everett**

## ABSTRACT

Understanding ecosystem patterns and processes has changed from assuming a stable system to recognizing constant change. Ecosystem management can maintain ecosystems in a condition compatible with natural processes and human values. Mimicking natural processes (disturbances, regeneration, and others) through management can help avoid and mitigate large undesirable disturbances. Analyses can identify and refine critical ranges of variation in ecosystem patterns and processes which management should achieve. Analyses also can help society refine is objectives by assessing consequences, costs, risks, tradeoffs, and efforts needed to achieve the stated desires. The time, type, and place of operations required to maintain ecosystems in desired states across time also can be identified through analyses. A decision support system is needed to assist these analyses. This paper presents such a support system, which uses both intuitive and analytical procedures that may be constantly refined through adaptive management.

## INTRODUCTION

Forest management has historically been concerned with producing and sustaining specific commodities (e.g., water, grazing, and timber). Ecosystem management is an explicit acknowledgment of a fundamental shift in human values, ecological knowledge, and perceptions of the relations of people to ecosystems. In ecosystem management, the first priority is to maintain ecosystems within limits compatible with both present human needs and the capacity of the ecosystem to provide these and future needs. Instead of managing each stand to achieve independent objectives, ecosystem management strives to achieve objectives across a landscape by managing stands through coordinated means. Management systems, tools, and organizations are being developed to deal with this shift. Most elements of ecosystem management are neither new nor unique, and forest managers are already using many of them; however, the primary focus on maintaining viable ecosystems will make a profound difference in how forested areas are treated.

This paper attempts to give organization and direction to this developing management system. A decision support system is presented that will help coordinate current and future knowledge of natural ecosystems, human societies, technology, various and changing objectives, and local infrastructures. This system can be altered to take advantage of technology that is available and appropriate (Oliver et al. 1991).

### Changing Perspective on Ecosystems and Management

Forests are constantly changing as a result of natural and human-caused disturbances and subsequent regrowth (Pickett and White 1985, Oliver and Larson 1990). The forest landscape (the aggregate of individual stands) changes as its component stands experience disturbance and regrowth. Such change alters habitat suitability, resistance of stands to disturbances, usefulness of stands for timber and other products, aesthetic appeal, and other values (Oliver 1991). Consequently, there is no single stand structure that will provide all ecosystem values, even if it were possible to keep stands from changing through tree growth and disturbances.

Attempts to preserve natural processes by simply leaving forested areas alone may not ensure total system sustainability because a diversity of structures or species may or may not be maintained, depending on the size of the area and the sizes and patterns of natural disturbances. The effects of large natural disturbances on landscapes may be more extreme than in the past because of management practices such as fire suppression and reduction and dissection of forested areas (Franklin et al, 1986). Such large disturbances are unacceptable to a society that desires commodities, and other values from timbered lands.

## Managing Ecosystems

Ecosystem sustainability and compatible human values can be achieved by actively managing to ensure that all natural patterns and processes are maintained across the landscape (Hann et al. 1993, Swanson et al. 1993). Management activities that may be required to achieve this objective include:

- Reduce (or mimick through management practices) the size and frequency of extremely large disturbances and other processes which may preclude other natural patterns, processes, or species. Examples of such large processes include very large natural fires which historically have covered hundreds of thousands of acres, extremely large floods, insect and disease outbreaks, or rapid animal or plant population increases or declines.

- Mitigate adverse effects where these large processes do occur. It will not be possible to prevent all large natural disturbances; for example, nothing could have prevented the volcanic eruption on Mount St. Helens of May 18, 1980. Forest landscapes will continue to reflect large-scale natural disturbance patterns as well as managed patterns which emulate natural processes. It may be desirable to restore a balance of forest structures and processes across the landscape soon after these processes occur because forests sometimes do not regrow naturally for many decades or centuries after a large disturbance (Oliver and Larson 1990). As a mitigation, some forest stands within a large disturbed area could be recreated much quicker through replanting. These stands would then serve as refugia for species requiring forest cover and so increase the diversity of habitats within the area.

- Avoid these large natural disturbances and other imbalances of patterns and processes through proactive management activities which mimic natural processes. Thinning and controlled burning reduce biomass; consequently, they may be used to mimic and reduce the adverse effects of insects, diseases, and natural fires. Replanting trees, shrubs, and herbs; creating snags, and controlling imbalances in animal populations can also help ensure that all patterns, processes, and species are maintained.

Forests of eastern Oregon and Washington can grow far more biomass than they can naturally sustain for an extended time. Before European settlement, natural disturbances such as insects, diseases, windthrows, and especially fires, periodically reduced the accumulation of biomass. At some times and in some forest types, these processes occurred in large-scale events such as stand replacement fires which burned across entire landscapes. In other forest types and at other times, frequent ground fires consumed accumulations of both dead and live biomass without destruction of entire stands of trees.

By reducing stand density, these fires tended to limit the incidence of bark beetles. By favoring tree species such as ponderosa pine (Pinus ponderosa) and western larch (Larix occidentalis), fires tended to reduce the natural susceptibility of defoliator insects such as western spruce budworm (Choristoneura occidentalis) and tussock moth (Lymantriidae Orgyia sp.).

Management practices such as thinning, well-designed harvest activities, and prescribed burning can effectively mimic these natural processes and bring stands into sustainable structures and species compositions. On a larger scale, such activities also can be used to design landscapes that reflect historical natural patterns and ecological processes.

Across each landscape unit, therefore, various ecosystem patterns and processes can be maintained through management operations. These operations would mimic and mitigate undesired natural processes to maintain landscape patterns and processes within a range compatible with human uses.

## Operations Needed for Ecosystem Management

The various management operations required for ecosystem management may include many conventional practices used in timber production, as well as new practices to provide other patterns and processes. Conventional timber operations include both even-age and uneven-age harvesting methods. Even-age harvesting (clearcutting, seed tree cutting, and shelterwood cutting) mimics large windstorms and natural “stand replacement” fires and reburns. These harvest methods create large openings and parklike stands used by many species. Single or mixed-species stands that grow after even-age harvesting can create various single canopy or layered canopy forest structures (Oliver and Larson 1990). Uneven-age harvesting can also create and maintain layered canopy forest structures used by other species.

Thinning traditionally has been used to promote growth of large trees, and also may be used to reduce outbreaks of insects, diseases, and fires. Thinning for programs not necessarily justified by timber revenue can be expanded because thinning alters the balance of stand structures and their usefulness for various plant and animal species. Broadcast burning of harvested areas can be used to minimize the spread of large natural fires. Artificial regeneration has been done extensively in the past with trees (to grow new forests) and grasses (to prevent erosion). Regeneration, however, can be expanded to other species (e.g., riparian willows) as desired. Wildlife population management has historically emphasized game species control by protecting animals from overhunting via hunting licenses and tags, by moving animals into areas of low populations, by controlling predator species, and by managing the vegetation. These practices may be expanded to other species as well.

New operations may need to be developed to mimic and mitigate natural ecosystem patterns and processes. Some of these operations have begun on a limited scale and could be refined and made more efficient. These operations will require creativity and may include:

- Artificially burning unhealthy stands or underburning (burning with cool, ground fires) the small trees within overly dense stands;
- Artificially creating crooked, forked, hollow, living and dead trees and downed logs for use by some wildlife species;
- Culturing of various shrubs, herbs, and nontimber tree species to maintain their presence;
- Managing streamside areas for the correct mixture of light and shade, organic detritus, large logs, and sediment;
- Mitigating past areas of siltation caused by poorly designed roads and mining activities;
- Increasing meanders in streams where past channeling has reduced these meanders; and
- Changing grazing patterns, especially near streams.

The extent of these operations, and the resulting patterns and processes they create across the landscape, will depend on the present ecosystem condition, the creativity of the management operations, and the effort the public is willing to invest in ecosystem management.

## Funding of Ecosystem Management

Historically, most operations were funded directly or indirectly where they could be justified for either timber management or fire control. Ecosystem management will be successful if these efforts also can be done where they cannot be economically justified on the basis of timber production or reduction of wildfires.

The amount of money that could be spent for ecosystem management in eastern Oregon and Washington could be extremely large. A rational means of capital allocation will be needed. One approach would be to examine the ecosystem to determine where critical elements and processes are far outside their historical range of variability. In these areas, managers could evaluate the values at risk and identify key structures and processes to be managed for. The greatest values at greatest risk would have high priority for funding ecosystem management activities.

Another consideration is the benefit received compared to the energy expended. Those activities resulting in greater ecosystem benefit for least cost would have high priority. One way to define the benefit to the ecosystem is to determine how much the activity would move the critical element or process toward the historical range of variability. These two approaches could be combined to develop a rationale for allocating capital.

It may be necessary to designate relatively small, critical areas to be managed for key structures and processes if little money is available. Areas outside of these, where limited funds were available for management for any purposes, could be managed extensively. For example, areas of overly dense stands susceptible to fires could be subdivided along natural fire breaks or artificially created fire breaks. These areas could be either burned artificially or allowed to burn naturally. Similarly, where little money was available, large stream and river areas with siltation problems could be allowed to continue silting until a new natural, geologic equilibrium was reached, while other areas were maintained in more controlled conditions.

If more funds were available for ecosystem management, intensive management areas could be expanded (e.g., areas with prescribed fire). In addition, mitigation and protection could be done in extensive areas. For example, refugia of various forest structures could be protected within the areas to burn; or, large logs and similar devices could be used to control the sediment in steams where active siltation was occurring. Even more available funds would allow more active management, smaller extensive areas, and more proactive management.

Managing for other values (in conjunction with basic ecosystem pattern, process, and species objectives) will reduce the costs of ecosystem management where other values are compatible (Lippke and Oliver 1993). Compatible values may include timber production, employment, fish production, and recreation. The added cost of modifying management to meet these other objectives may often be less than the cost of managing for just the patterns, processes, and species themselves. Such an approach would necessarily give primary consideration to ecosystem values.

## Achieving Ecosystem Management

Deciding to implement ecosystem management is still a long way from achieving the goal. Accordingly, the following questions must be addressed by the land manager:

- What is the range of variation in patterns, processes, and species population sizes to be achieved? The spatial and temporal aspects of these ranges need to be described so managers will know whether to favor or discourage certain patterns, processes, and species. A knowledge of natural variability range is useful in making this assessment (Hann et al. 1993, Swanson, et al. 1993).

- How can all patterns, processes, and species be managed for, especially when they are not all known? Managing for too many things will overload any present abilities to manage. The coarse-filter approach to biodiversity maintenance is useful in this situation (Hann et al. 1993, Bourgeron and Jensen 1993).

- How can the values of society be determined and incorporated into ecosystem management, especially if these values are changing, and it is not certain how biologically achievable these values are (Montgomery 1993)?

- How can patterns, processes, and species balances actually be achieved through time and across the landscape? The ability to manage the various changes and project the consequences of action and inaction through time and space needs to be coordinated more closely than has historically been possible (Everett et al. 1993).

## Determining Appropriate Patterns and Processes

Maintaining all patterns, processes, and species through ecosystem management requires that some measurable range of patterns, processes, and species be described in desired future condition statements for Forest plans (Morrison 1993). Defining this range is difficult because not all patterns, processes, and species are known (or may ever be known), and because these patterns, processes, and species vary among areas. Waiting until knowleage is more complete before ecosystem management is implemented will probably lead to further imbalance in ecosystems (either away from many natural processes if historical management is continued or away from many human values if all management is suspended). Plant and animal species do not stop growing, dying, and burning; and floods, fires, and windstorms do not stop when all management is suspended.

In the face of uncertainty, ecosystem management can begin by reducing the presently obvious and extreme imbalances of patterns, processes, and species populations. While these extremes are being reduced, the desired range of patterns, processes, and species can be improved through the adaptive management process (Everett et al. 1993, Walters 1986).

One conservative way of determining an appropriate range of patterns and processes to be maintained across the landscape is to use a historical variability approach (Swanson et al. 1993). It is almost impossible for any pattern or process to be universally bad, because some species will always benefit. For example, certain species use each stand structure, and many rely on several structures for various activities (e.g., food gathering, protection, and raising young). Too much of one structure will mean too little of another structure, however, and will endanger species that use the diminished structure. Ecosystem patterns and processes (including species population sizes) have fluctuated dramatically across space and time as a function of disturbance and regrowth. This fact must be acknowledged in land management.

The ecosystem is defined and described by is elements, that is, the critical processes, functions, structures, and patterns that are the habitats of indigenous plants and animals. Ecosystem management focuses on the whole rather than on the parts of the system, and assumes that, by taking care of the critical elements of the broad system, the component parts (the plants and animals within the system) have a high liklihood of being sustained. This approach is contrary to many modern-day management dictates, which focus on individual species management (i.e., a fine-filter approach). There are literally thousands of plant and animal species, some of them unnamed and unknown, that inhabit even the simplest forest ecosystem. For ecosystem management to succeed, both coarse-filter and fine-filter approaches to biodiversity maintenance need to be employed (Hann et al. 1993).

The vegetation structures within an area have always fluctuated widely and may have been necessary for species survival. For example, species requiring interior forest structures would primarily migrate (and avoid inbreeding) when most of an area contained closed-canopy forests. Alternatively, species requiring open areas (e.g., butterflies and bighorn sheep) would primarily migrate when most of an area was in open structures. When most of an area contained one structure, species favored by other structures would exist in refugia within the area or would be excluded, only to migrate back when suitable structures returned.

By using historical records, it is possible to determine the natural range of variability in patterns and processes across each landscape area. This pattern has been used as a template for managing ecosystems by filtering out the extreme variations and by using this conservative variation as the desired condition for ecosystem management (Hann et al. 1993).

Desired patterns and processes are ranges that reflect the natural variability of the landscape and presumably provide species habitat needs. This variation is often referred to as a decision space. Other values and products could be managed in combination with ecosystem management as long as the variations of patterns and processes are maintained within the desired decision space (Jensen and Everett 1993). As patterns and processes within each small area (e.g., 5000- to 15,000-acre landscape units) are managed with fluctuations within the decision space, adjacent landscape units can be coordinated to allow parallel or contrasting fluctuations of patterns, as needed (Oliver et al. 1992).

## Managing for All Patterns and Processes

Trying to keep track of all patterns and processes on all scales is an impossible task, and there is concern that some patterns, processes, or species will be lost if all patterns and processes are not managed for. On the other hand, there are many patterns, processes, and species in even a simple ecosystem.

Natural resource scientists can use historical records, climatology, system modeling, and other techniques not yet envisioned to identify critical elements and processes for the ecosystem and to define their historical range of variation, by using a coarse-filter approach to winnow out extremes (Noss 1987, Hunter 1990). When applied to a landscape, these ranges of variation give a first approximation of a sustainable ecosystem. They also define the biological bounds of management decisions by defining the decision space within which social, political, and economic factors would operate.

This decision space would provide a means for describing the desired condition for the landscape where the critical elements (patterns and processes) are within the acceptable range of variation. In a dynamic landscape, several patterns and structures could be developed that would meet the desired condition. There would not be just one approach to ecosystem management.

The focus of management has historically been at the stand scale, and the landscape was assumed to function satisfactorily as the aggregate of stands. The initial focus for ecosystem management is at the landscape (or watershed, river basin, or larger scale). Once the desired patterns and structures are defined for the landscape, stand treatments can be developed to move from the present condition to the desired condition for the landscape.

Converting the various requirements of critical elements to common terms which can actually be measured and altered through protection and management will help achieve management objectives. For example, if a given forest structure or landscape pattern influences a critical element, the requirements from the forest could be expressed in the common terms of stand structures, landscape patterns, and operational constraints (Washington State D.N.R. 1990, Oliver et al. 1992). Similar common, measurable criteria can be developed for aquatic and riparian habitats (Minshall 1993) and for various predator-prey populations (Jurgensen et al. 1993).

## Determining the Objectives of Society

Incorporating the objectives of society into ecosystem management is difficult because the public and private landowners have various objectives. In addition, these objectives change with time, and the complexity of ecosystems makes it difficult to determine if certain objectives are compatible, or even achievable (and at what cost) (Montgomery 1993).

A primary focus of ecosystem management is to maintain patterns, processes, and species so that options to emphasize other objectives will not be precluded (Jensen and Everett 1993). For example, species with no presently perceived economic value should be maintained because they may be critical for survival of other species and because they may have a future direct value to humans (e.g., the recent use of Pacific yew for medicinal purposes).

Even with the decision space of maintaining the natural patterns, processes, and species of ecosystems, society will need to make choices of both what effort is to be expended to manage ecosystems and what values are to be emphasized within the decision space.

Society’s values are translated into management objectives through a two-part process: determining what the desires of society are, and determining what effort, tradeoffs and risk society is willing to undertake to achieve the desires. This process is reflected in the U.S. Legislative process by having both enabling and budgetary legislation. A similar, iterative process can also be used at other times and at other management scales within a hierarchy.

It is not always obvious how realistic the initially stated goals (desires) of society are when managing complex ecosystems. The exact consequences are never certain; however, projecting the consequences by using scientific knowledge, models, and other analysis techniques helps make the consequences more realistic. Goal achievement is complicated by its feasibility, risk, effort needed, and tradeoffs with other desires. This determination is actually part of the goal-development process; however, in complex management systems, it is generally institutionalized as a separate step with feedback to the policy objective (fig. 1).

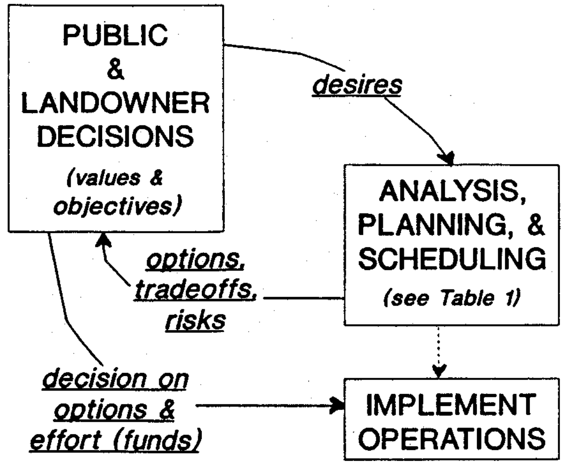


**Figure 1--Ecosystem management flow of information-decisions. Public desires are analyzed to determine consequences, risks, tradeoffs, and efforts needed to achieve them. The public then makes informed choices of values and objectives, which enable operations to be implemented. Analysis and planning-scheduling contain intuitive and technical components (table 1).**

A system for determining tradeoffs is imperfect because the consequences and importance of different objectives is not well known. The economic system is commonly used to bring all objectives to a common value. This system is also imperfect; however, it can be used to determine how much effort will be needed, and how much the landowner (public or private) is willing to spend or forgo to achieve an objective.

Historically, forest management has avoided decisions of how much effort to expend to achieve an objective by using timber revenues to promote many objectives (e.g., access; fire, insect, and disease protection; employment; recreation). This use of timber revenues is being questioned where timber revenues alone cannot justify management activities and where the most economically efficient method of timber management seems in conflict with other objectives (Shepard 1993). Recent analyses of the positive and negative effects of timber management on other objectives ("TSPIRS;" USDA Forest Service 1989-1991) are beginning to indicate the tradeoffs of various costs and objectives. Recent regulations on private forest lands with the objective of promoting public values are having the opposite consequences (Oliver 1993). Alternatively, carefully expressed incentives can promote the desired objectives (Lippke 1993, Lippke and Oliver 1993).

A coordination system (i.e., Decision Support System) that analyzes the various tradeoffs implicit in ecosystem management can help estimate the feasibility, tradeoffs, and risks associated with various public desires. Such a system is presented later in this paper.

## Achieving Ecosystem Management

Maintaining the various patterns, processes, and species within a desired balance will require coordination of various natural processes and human activities across time and space. This coordination will require knowledge of natural and social sciences as well as technical information on present natural conditions, infrastructures, and operational capabilities. Coordination also will require a reiterative process of input and feedback from various sources.

It is impossible for a single person or group of people to consider all facets of complex systems. Such top-down planning also does not allow flexibility in changing operations to correct errors or to achieve changed objectives (Reich 1983). It is only through physically working with the resources at the operational level that the feasibility, effort, and tradeoffs of achieving various objectives can be estimated. This bottom-up knowledge needs to be transmitted to policymakers, usually in an iterative process, so the correct balance between what is desired, what is achievable, and the costs, risks, and tradeoffs can be made.

Although bottom-up processes tell how much and with what tradeoffs the various values and products can be achieved, the desired balance of tradeoffs needs to be determined from a top-down process. Much of the U.S. economy, for example, has been based on the top-down objective of allowing a free-market approach to aggregating preferences. Integration of the top-down and bottom-up aspects of management is needed and can be achieved through a decision support system.

## Analytical Versus Intuitive Aproaches to Decisionmaking

Coordinating complex activities has been viewed from two approaches: the rational-comprehensive (or systems) approach and the successive limited comparisons (or muddling through) approach (Lindblom 1972). At its extremes, the rational-comprehensive approach assumes the values, tradeoffs, and means of achieving the values are perfectly known; the manager needs only to calculate the most effective means of achieving the values. At the other extreme, the successive-limited comparisons approach assumes. the manager should make decisions based on agreement wherever it can be found (Lindblom 1972), for immediate conflict resolution without extended considerations of values and consequences.

Management actually uses a combination of the two methods. Advances in knowledge, technology, and education are allowing analytical tools to be integrated more fully into the coordination and decisionmaking processes; however, analyses and models need to be viewed as tools to aid human judgment. Human judgment is necessary to decide when and where analytical tools are appropriate, to apply and interpret the analytical tools, and to make decisions alone where appropriate analytical tools do not exist.

Many elements of the systems approach are already incoporated into resource management. The reliance on information, knowledge, and analysis techniques to aid human judgment will increase as the need to make informed choices increases and as more understanding of natural sciences and political, social, and economic sciences develop. Ecosystem management will probably become increasingly analytical, and acknowledgement that a perfectly rational-comprehensive approach will never be achieved.

## Decision-Support System for Ecosystem Management

Figure 1 shows an overall flow of information in a decision-support system for ecosystem management. Much of the analysis and planning is done within the center box, with feedback to both policy makers and (through their determining of objectives) those responsible for management implementation.

Feedback to policy makers commonly involves information on costs, tradeoffs, risks, and consequences of various expressed desires. Policy makers can then restate the desires or express the amounts of effort, risk, and tradeoff to be expended in implementing management. Coordination of the planning process is used to determine the appropriate type, timing, and location of operations to achieve stated objectives. Although feedback and planning processes have two different outputs, objectives are achieved more effectively if the two processes are combined. In this way, tradeoffs, consequences, and risks will be realistic, and planning will be compatible with stated desires and objectives.

Effective organization of the analysis, design, and planning process has been intensively studied (Simon 1960, Dieter 1991, Kranzberg 1984, Bennis 1966, Blau and Schoenherr 1971, Cleland and King 1968, and Ackoff 1974). The intent of effective organization is to allow a smooth flow of information, which ensures that all critical areas have been considered. Effective organization processes have been incorporated into systems engineering, the planning process, systems management, and the National Environmental Protection Act process.

The effective organization process can be divided into steps or modules, where certain analyses are performed within each module. An example of such modules for ecosystem management planning and analysis is presented in table 1. These modules facilitate a process whereby each module can be constantly improved without disrupting the entire planning process. In the example provided, information flows among modules for effective analysis and scheduling, and changes are made within and between modules through adaptive management to ensure objectives are achieved. Modules may be constructed using expert judgment; however, this judgment needs to be updated as improved information and analytical tools become available.

**Table 1--Example of Modules (and Submodules) for Analyzing and Scheduling Ecosystem Management**

MODULE: Convert objectives to measurable criteria

SUBMODULE: Wildlife criteria
SUBMODULE: Fish criteria
SUBMODULE: Plants
OTHER SUBMODULES: Other ecosystem values, possibly including timber production, grazing, employment, etc.

MODULE: Determine existing patterns/species/processes

SUBMODULE: Various inventory data
SUBMODULE: Organizing information (e.g., GIS system)

MODULE: Project changing patterns, processes, species, time

SUBMODULE: Project stand structure changes
SUBMODULE: Project landscape pattern changes
SUBMODULE: Project wildlife populations
SUBMODULE: Project fish population changes
OTHER SUBMODULES: Project other ecosystem value changes (timber production, employment, grazing, etc.)

MODULE: Project alternative management options for ecosystems
SUBMODULE: Silvicultural decision keys
SUBMODULE: Options for management of wildlife, fish, etc.
SUBMODULE: Feedback to previous module to project changes

MODULE: Determine costs/tradeoffs of various alternatives

MODULE: Implement chosen operations to achieve ecosystem management

A systematic flow of information among modules can be established that facilitates an effective planning and analysis process. This information flow can be changed as other flows prove to be more efficient (i.e., as determined through the adaptive management process). Information flow between modules can be effective if each module receives and supplies information in a designated customer-supplier role. This flow of information is similar to (or compatible with) the critical measures described earlier in this paper.

The modular process for implementing ecosystem management (table 1) can most efficiently be used if the following suggestions are observed:

- Organize management at several geographically hierarchical scales. Identify goals at the national scale and sequentially expand such goals at the regional, subregional, and local scales, based on feedback from the local scale to the national scale concerning tradeoffs and feasibilities.

- Focus objectives on ecosystem conditions to be maintained, and leave decisions of operations needed to achieve the conditions and their output as close to the operational phase as possible. Especially in ecosystem management, specific techniques to achieve objectives will differ in time and space with biological, social, and technological conditions and knowledge. Although the specific goal of managing for certain ecosystem values needs to be maintained, the techniques for achieving those values will differ dramatically. In this way, ecosystem management will use a portfolio approach tf nanagement, with each local area managed flexibly to achieve an overall goal (Oliver 1991, Gottfried 1991). Coordination can remain efficient if management is flexible enough to change techniques when conditions change (Reich 1983).

- Focus management on designing and maintaining an effective management process by using the degree to which the stated objectives are achieved as one measure of the efficiency of management.

- Maintain a feedback process through which policy makers can make realistic decisions about the consequences, tradeoffs, and effort spent to achieve various stated objectives.

- Maintain an efficient mechanism for comparing conflicting local and global ecosystem values and other human values.

The modular process for a decision-support system begins with limited knowledge and expands in effectiveness through an adaptive management process. The adaptive management process (Everett et al. 1993, Walters 1986) is similar to the continuous quality improvement process (Deming 1982, Feigenbaum 1951 and 1983) because adaptive management continually compares the expected and received output from each step (module) of the process. Where differences exist, adaptive management determines the causes of the differences and corrects such causes (i.e., it uses improved information in a reiterative manner).

## CONCLUSIONS

Many parts of the analysis and scheduling process discussed in this paper are used by public and private forest management agencies. Modern technologies such as satellite imagery, geographic information systems, stand and landscape projection systems, silvicultural decision keys (O'Hara et al. 1990), and tradeoff analytical systems are being incorporated (e.g., Boyce 1985). Other parts, such as wildlife and fish input-output models, need to be developed.

Until such technology is more widely available, managers will use generalized analysis based on maps and aerial photographs. This level of resolution will be satisfactory for initial assessments of ecosystems at the landscape scale, but validation of conditions and development of operational plans will require more sophisticated systems.

Careful analyses may suggest that certain current laws and regulations prohibit ecosystem management. For example, controlled fires may be needed to avoid uncontrollable ones; however, such controlled fires may presently be incompatible with smoke and air-quality guidelines. In addition, a proactive approach for saving endangered species and avoiding species endangerment may be necessary. This approach would involve more active management to increase habitat for endangered species and to avoid other species and ecosystem values from becoming endangered.

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# APPENDIX A

## Glossary

**Adaptive management** - implementing policy decisions as science-driven management experiments that tests assumptions and predictions in management plans.

**Allowable sale quantity (ASQ)** - the quantity of timber that may be sold from the area of suitable land covered by the Forest Plan for a time period specified by the plan. This quantity is usually expressed on an annual basis as the average annual allowable sale quantity.

**Autocorrelation** - the correlation between measurements obtained from points or windows that are separated by a fixed lag distance.

**Bioenvironments** - combinations of environmental factors to which the biota respond directly (e.g., temperature), or consume as resources (e.g., nutrients).

**Biological diversity** - the variety of life and its processes, including the variety in genes, species ecosystems, and the ecological processes that connect everything in ecosystems.

**Biome** - large subdivision of the terrestrial ecosystem characterized by its physiognomy (e.g., grassland), including total assemblage of plants and animals.

**Biota** - the animal and plant life of a particular region.

**Broad scale** - encompassing a wide area.

**Classification** - the assignment of points, or sample units, to a finite number of discrete types, usually based on an analysis of many variables (e.g., vegetation classification, soil classification).

**Climate** - generalized statement of the prevailing weather conditions at a given place, based on statistics of a long period of record. Includes seasonality of temperature and moisture.

**Climatic climax (climax vegetation)** - the relatively stable vegetation that terminates on zonal soils.

**Community** - an assemblage of species at a particular time and place.

**Composition** - the constituent elements of an entity; for example, the species that constitute a plant community.

**Connectivity** - condition in which the spatial arrangement of land cover types allows organisms and ecological processes (such as disturbance) to move across the landscape. Connectivity is the opposite of fragmentation.

**Conservation biology** - the body of knowledge that deals with the careful protection, utilization, and, planned management of living organisms and their vital processes to prevent their depletion, exploitation, destruction, or waste.

**Continentality** - tendency of large land areas in midlatitudes and high latitudes to impose a large annual temperature range on the air temperature cycle.

**Corridor** - landscape elements that connect similar patches through a dissimilar matrix or aggregation of patches.

**Critical threshold** - point along a gradient of variable X where a small change in X produces a rapid change in response Y.

**Cumulative effects analysis** - an analysis of the effects on the environment that result from the incremental effects of a proposed action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency (Federal and non-Federal) or person undertakes such other actions.

**Curvilinear statistical models** - statistical models that include higher order terms (e.g., quadratic, cubic) in equations.

**Decision-support system** - an organized array of tools that policy makers and managers can use to select a course of action, often but not necessarily a formal model.

**Desired future condition** - a portrayal of the land or resource conditions that are expected to result if goals and objectives are fully achieved.

**Disturbance-recovery regime** - natural pattern of periodic disturbances, such as fire or flooding, followed by a period of recovery from the disturbance (e.g., regrowth of forest after a fire).

**Ecosystem** - all of the organisms in a given place in interaction with their nonliving environment.

**Ecosystem management** - the careful and skillful use of ecological, economic, social, and managerial principles in managing ecosystems to produce, restore, or sustain ecosystem integrity and desired conditions, uses, products, values, and services over the long term.

**Ecosystem sustainability** - the ability to sustain diversity, productivity, resilience to stress, health, renewability, yields of desired values, resource uses, products, or services from an ecosystem while maintaining the integrity of the ecosystem over time.

**Equilibrium** - oscillation around a central position; for example, condition in which the relative frequency and spatial pattern of land cover types remain relatively constant for a specified period of time.

**Extent** - the breadth of a study or the length of a time series.

**Extrapolation** - prediction outside the range of values included in a data set.

**Forest plan (Forest Land and Resource Management Plan)** - a document that guides all natural resource management and establishes management standards and guidelines for a National Forest, embodying the provisions of the National Forest Management Act (1976).

**FORPLAN** - a linear programming system used for developing and analyzing forest planning alternatives.

**Fractal** - a pattern, which upon magnification, reveals statistically similar geometry; the relation between broad- and fine-scale features of a fractal are related by a power law.

**Fragmentation** - division of a large land area (e.g., forest) into smaller patches isolated by areas converted to a different land type.

**Function** - the flow of mineral nutrients, water, energy, or species.

**Generalized linear models** - a class of statistical models that share properties such as linearity and a common method for computing parameter estimates. Linear regression and analysis of variance are special cases of generalized linear models.

**Gradient** - a gradual change with distance.

**Grain** - one aspect of ‘scale’; the threshold below which pattern or structure cannot be distinguished.

**Habitat** - place where an animal or plant normally lives, often characterized by a dominant plant form or physical characteristic.

**Heterogeneity** - variation in the environment over space and time.

**Hierarchy** - a sequence of sets composed of smaller subsets.

**Home range -** the area visited by an organism during the course of daily activity.

**Integrated resources management** - the simultaneous consideration of ecological, physical, economic, and social aspects of lands, waters, and resources in developing and implementing multiple-use, sustained-yield management.

**Interpolation** - prediction within the range of values included in a data set.

**Lag -** the distance between two subregions in geostatistical analysis.

**Landscape -** a heterogeneous land area composed of a cluster of interacting ecosystems that are repeated in similar form throughout.

**Landscape ecology -** a study of the structure, function, and change in a heterogeneous land area composed of interacting organisms.

**Macroclimate** - climate that lies just beyond the local modifying irregularities of landform and vegetation.

**Matrix** - the most extensive and connected landscape element that plays the dominant role in landscape functioning. Also, a landscape element surrounding a patch.

**Metapopulation -** population structure in which individual populations exist on patches and are dynamic in space and time.

**Mode** - value occurring most frequently in a series of observations.

**Model validation** - testing of a model by comparing model results with observations not used to develop the model.

**Multiple Use-Sustained Yield Act (MUSYA)** - authorizes and directs that the National Forests be managed under principles of multiple use for outdoor recreation, range, timber, watershed, and wildlife and fish purposes, and to produce a sustained yield of products and services, and for other purposes. This Act does not affect the use or administration of the mineral resources of National Forest lands or the use or administration of Federal lands not within National Forests.

**National Environmental Policy Act (NEPA)** - an act that encourages productive and enjoyable harmony between humans and their environment; promotes efforts to prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of humans; enriches the understanding of the ecological systems and natural resources important to the Nation; and establishes a Council on Environmental Quality.

**National Forest Management Act (NFMA)** - a law passed in 1976 as amendments to the Forest and Rangeland Renewable Resources Planning Act that requires the preparation of regulations to guide development arising from the Resources Planning Act.

**Natural variability** - range of the spatial, structural, compositional, and temporal characteristics of ecosystem elements during a period specified to represent “natural” conditions.

**Patch** - ecosystem elements (such as areas of vegetation) that are relatively homogeneous internally and that differ from what surrounds them.

**Pattern -** the spatial arrangement of landscape elements (patches, corridors, matrix) that determines the function of a landscape as an ecological system.

**Physical environments** - combinations of environmental factors to which the biota respond indirectly; for example, elevation, landform, geological substrate, or soil type.

**Physiography** - landform (including surface geometry and underlying geologic material).

**Plant association** - a kind of plant community represented by stands occurring in places where environments are so closely similar that there is a high degree of floristic uniformity in all layers.

**Plant class** - subdivision of the biome based on dominant growth form and cover of the plants that dominate the vegetation.

**Plant subclass** - subdivision of a class based on morphologic characters, such as evergreen and deciduous habitat, or on adaptation to temperature and water.

**Plant formation -** subdivision of a subclass based on size, shape, and structure.

**Plant series** - subdivision of a formation based on individual dominant plant species of the community.

**Potential vegetation** - vegetation that would develop if all successional sequences were completed under present site conditions (e.g., habitat type).

**Process** - change in state of an entity.

**Random variable -** a function over the sample space about which it is possible to make a probability statement.

**Replication** - repeated sampling under similar conditions to improve the validity of statistical analysis; for example, in gradsect methodology, sampling in similar environments located in different geographical areas.

**Representativeness** - a criterion for assessing how adequately an area of interest represents the range of biological and biophysical variation in a region.

**Scale** - the level of spatial resolution perceived or considered. Also, spatial proportion, or the ratio of length on a map to true length.

**Stand structure -** the physical and temporal distribution of plants in a stand.

**Stochastic process -** a collection of random variables defined on a common probability space.

**Succession** - a directional composition change in an ecosystem as the available organisms modify and respond to changes in the environment.

**Taxa -** objects grouped on the basis of similarity of properties (e.g., species).

**Wilderness Act of 1964** - establishes a National Wilderness Preservation System to be composed of Federally owned lands designated by Congress as wilderness areas; these areas shall be administered for the use and enjoyment of the American people in such manner that the lands will be unimpaired for future use and enjoyment as wilderness, and to provide for the protection of these areas, the preservation of their wilderness character, and for the gathering and dissemination of information regarding their use and enjoyment as wilderness.

**Window -** a subregion of a map or a subsection of a time series from which measurements are obtained, usually for comparison with other windows.

# APPENDIX B

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This document provides land managers with practical suggestions for implementing ecosystem management. It contains 28 papers organized into five sections: historical perspectives, ecological principles, sampling design, case studies, and implementation strategies.

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