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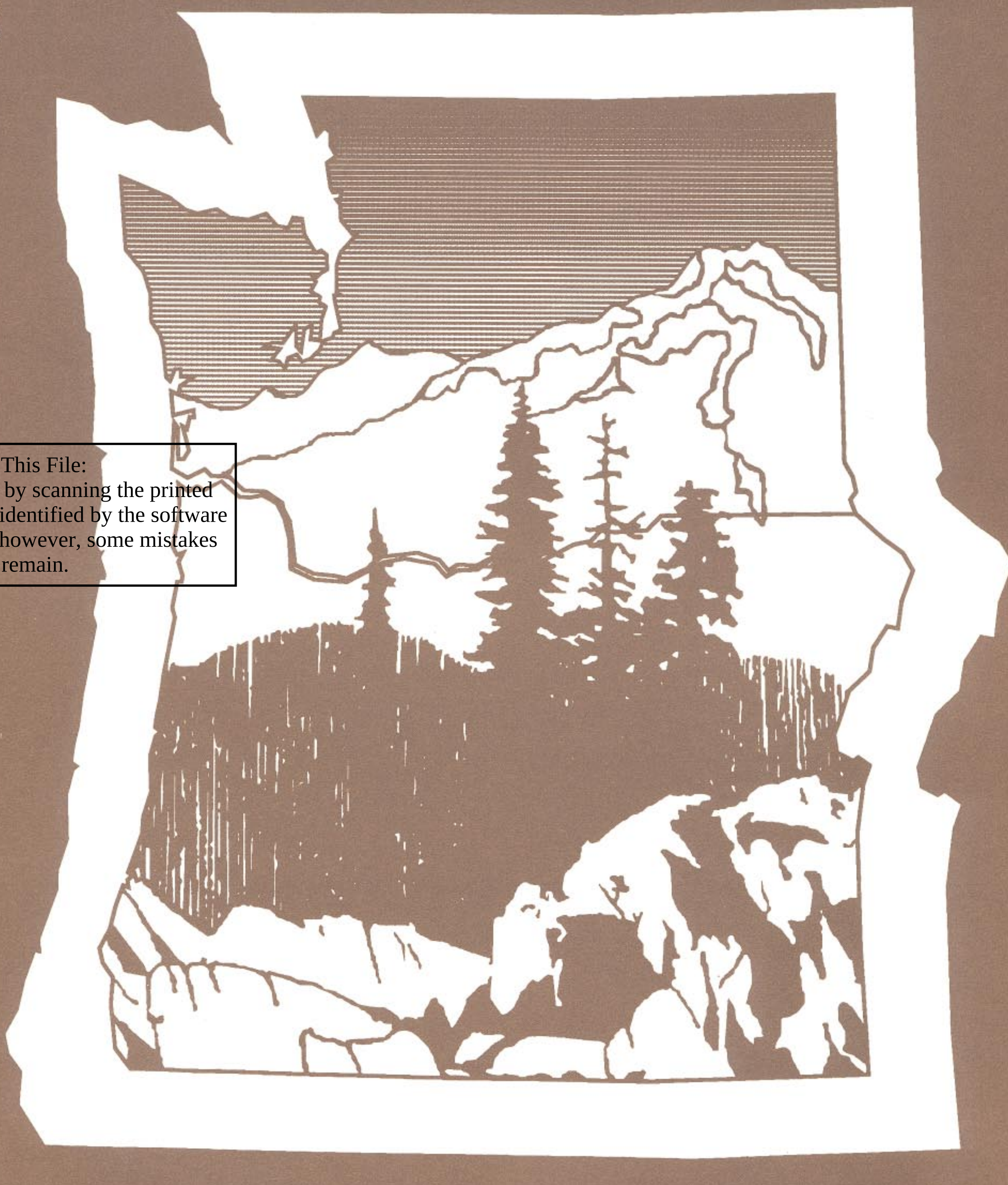

Technical Editors

M.E. Jensen is a regional soil scientist, U.S. Department of Agriculture, Forest Service, Northern Region, P.O. Box 7669, Missoula, MT 59807; and P.S. Bourgeron is a western regional ecologist, The Nature Conservancy, 2060 Broadway, Suite 230, Boulder, CO 80302.

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.

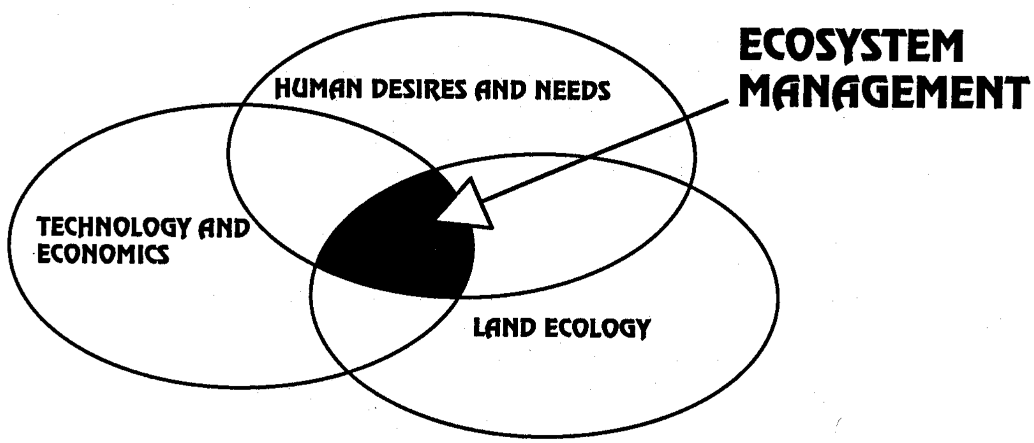

Figure 1--Conceptual framework for ecosystem management (adapted from Zonneveld 1988).

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--
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)
		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. We need to understand natural events and the effects of humans in our management [of the land, Forest Service people and public use]. 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.] 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. 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. 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]. 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.] 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.

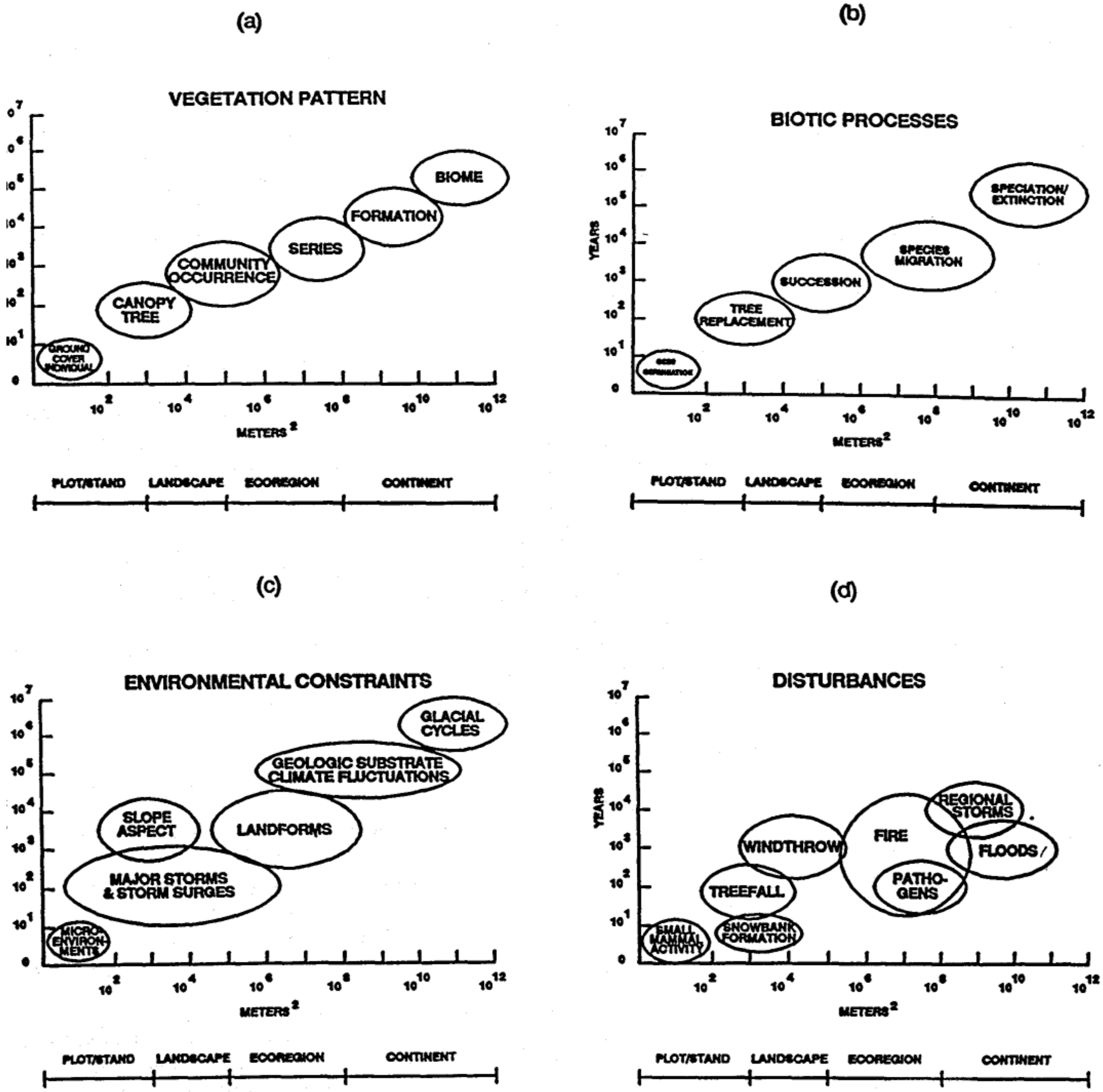

Figure 1--Spatial-temporal scaled patterns for (a) vegetation, (b) biotic processes, (c) environmental constraints, and (d) disturbances (modified from Urban et. al 1987).

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.

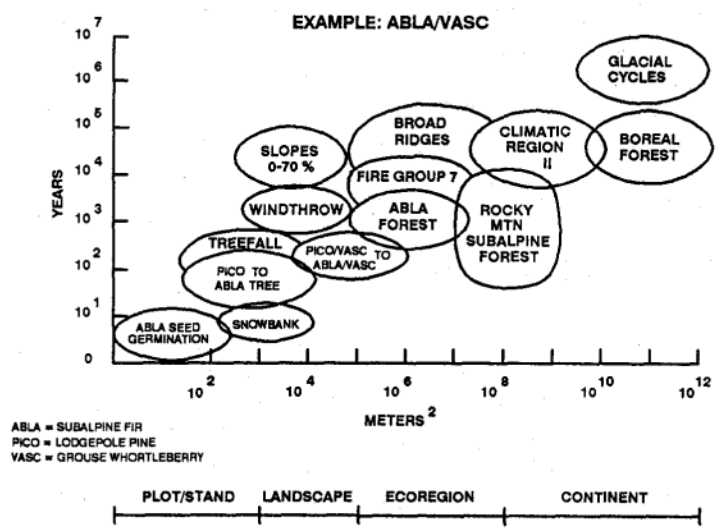

Figure 2--Hierarchical spatial-temporal representation of ecological relationships.

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).

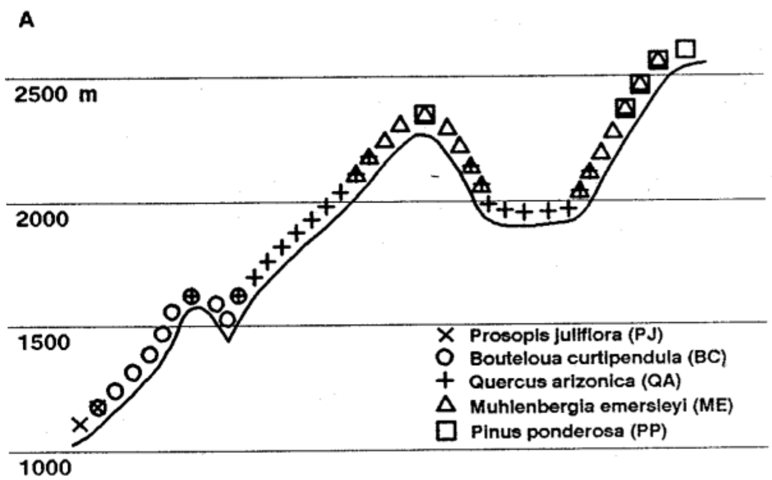
At the continental-global scale, the concept of the floristic or biotic province has been used to recognize large land units. This scale (among regions) leads to the delineation of regions assumed to be internally homogeneous. Biotic provinces (e.g., Dice 1943) are reasonably discrete large areas, usually 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.



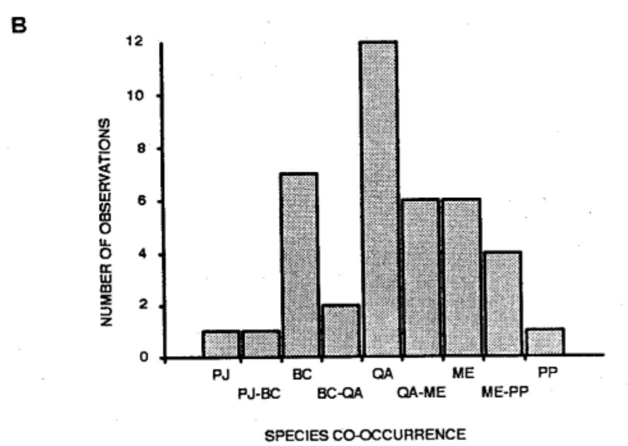


Figure 1A--Species distribution along a directional transect showing the spatial co-occurrence of species as a function of landscape pattern (data from Whittaker and Niering 1965); Figure 1B--Frequency of occurrence of species groups used to define communities.

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.

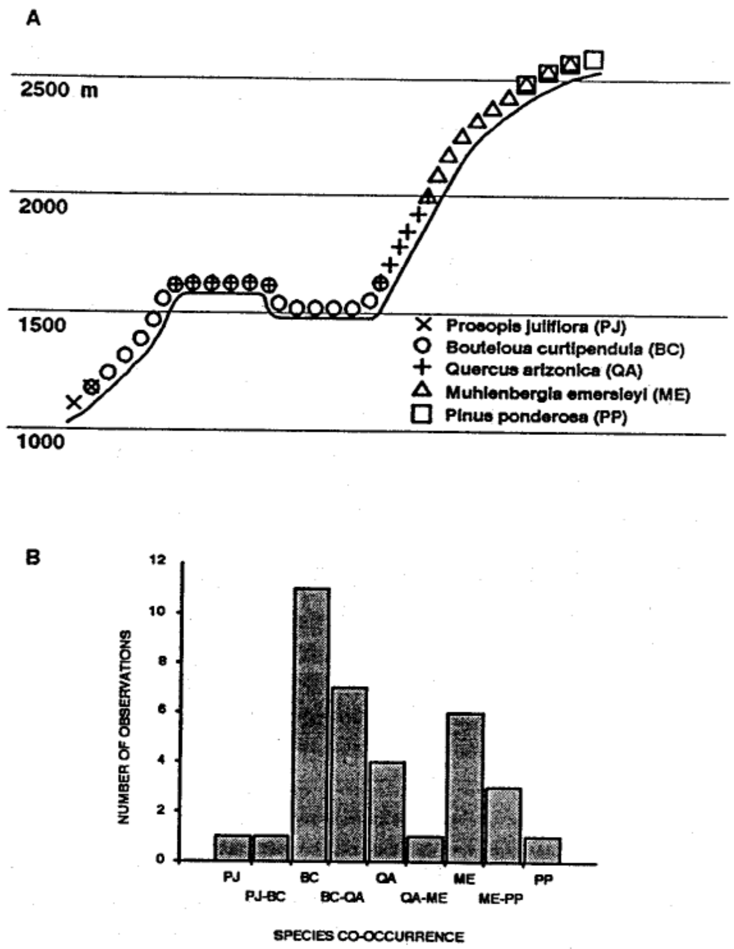


Figure 2--Species along a transect. Species and the elevation gradient are the same as In figure 1. Landscape pattern is different (see text).

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).

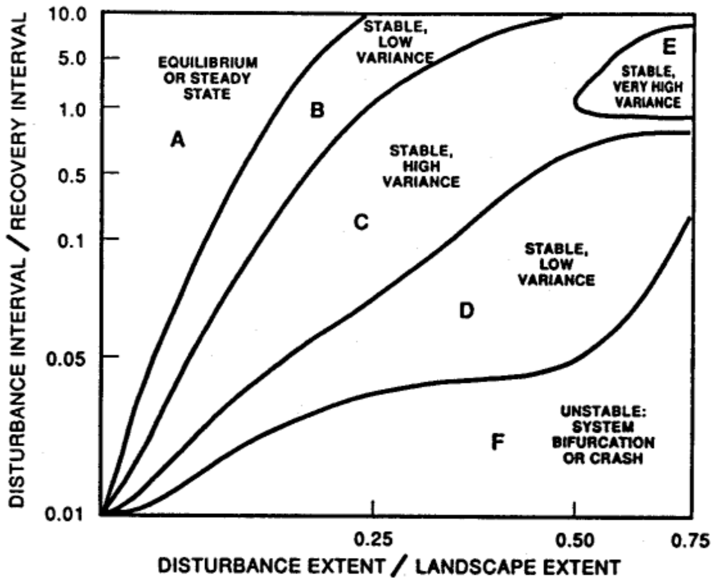

Figure 1--State-space diagram of temporal and spatial parameters that Illustrate regions with qualitatively different landscape dynamics (from Turner et al., In press).

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

F.J. Swanson, J.A. Jones, D.O. Wallin, and J.H. Cissel

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.

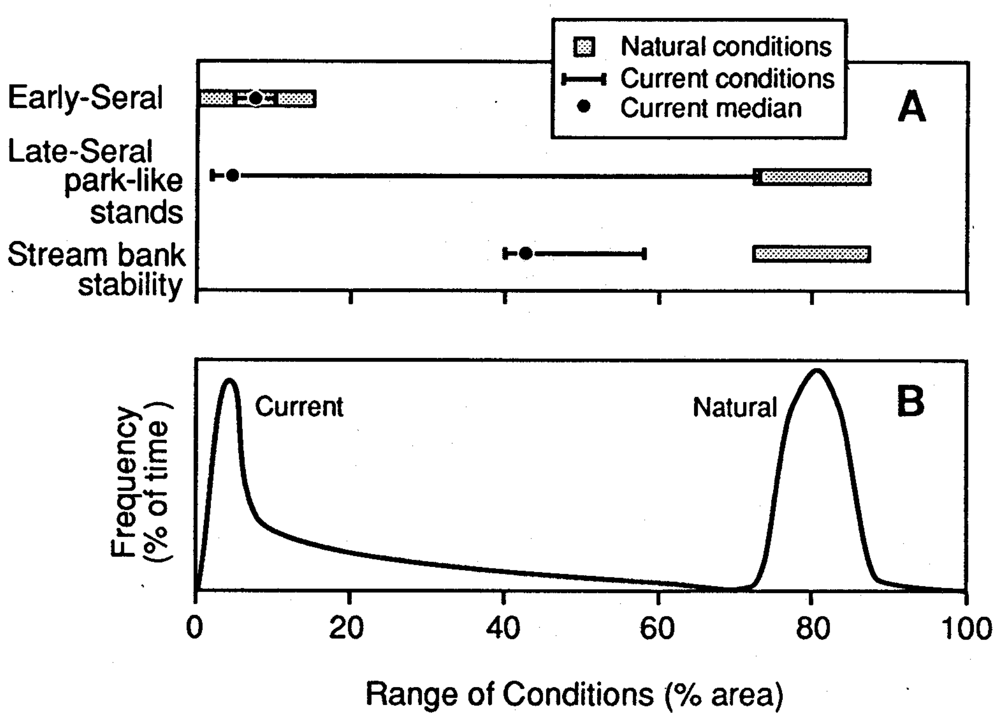

Figure 1-Example display of range (A) and distribution (B) of ecosystem conditions. (A) Range of conditions for natural and current conditions are adapted from Silvies River area In Caraher and others (1992, p. 22); (B) Shows hypothetical distribution of ecosystem conditions of late seral, park-like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

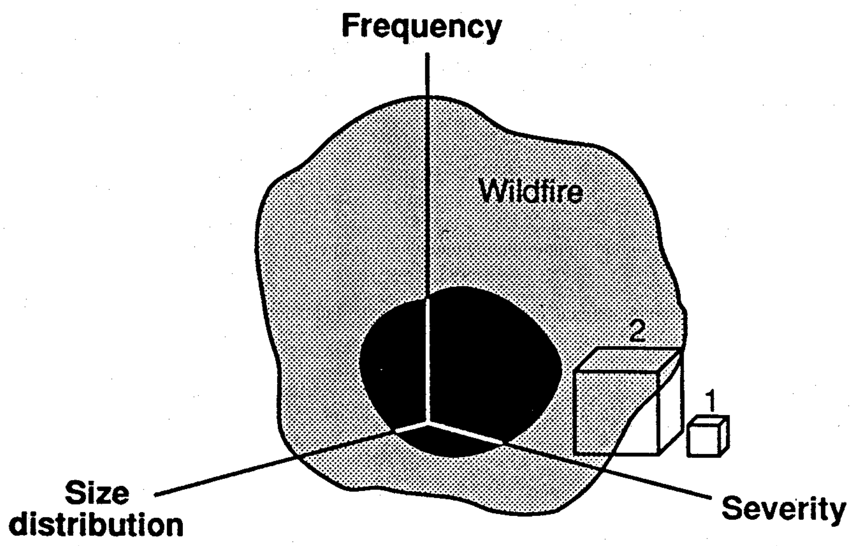

Figure 2--Hypothetical representation of a natural disturbance regime with the large, irregular "cloud" showing a probability distribution of wildfire events interpreted from dendrochronologic observations. Box 1 represents a management system of dispersed clearcuts with broadcast burning. Box 2 represents 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.

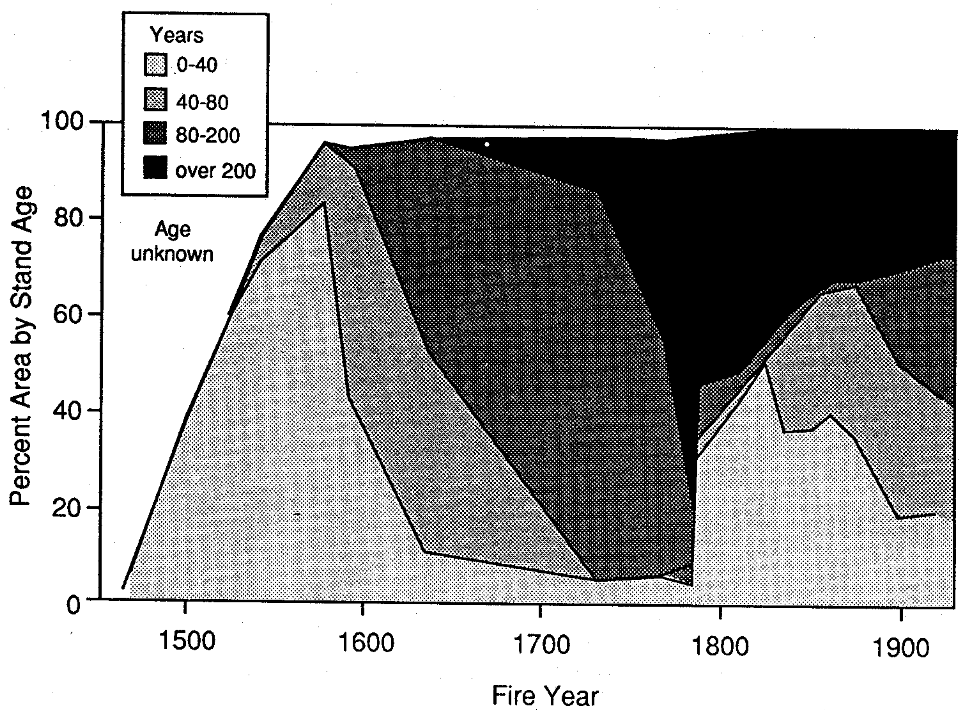

Figure 3--Percentage area in various stand age classes for the non-wilderness portion of the Augusta study area (15,650 acres), Willamette National Forest, based on dendrochronologic reconstruction of wildfire history. Note that the assumption that burned areas experienced stand replacement fires results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

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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 land-scape 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.

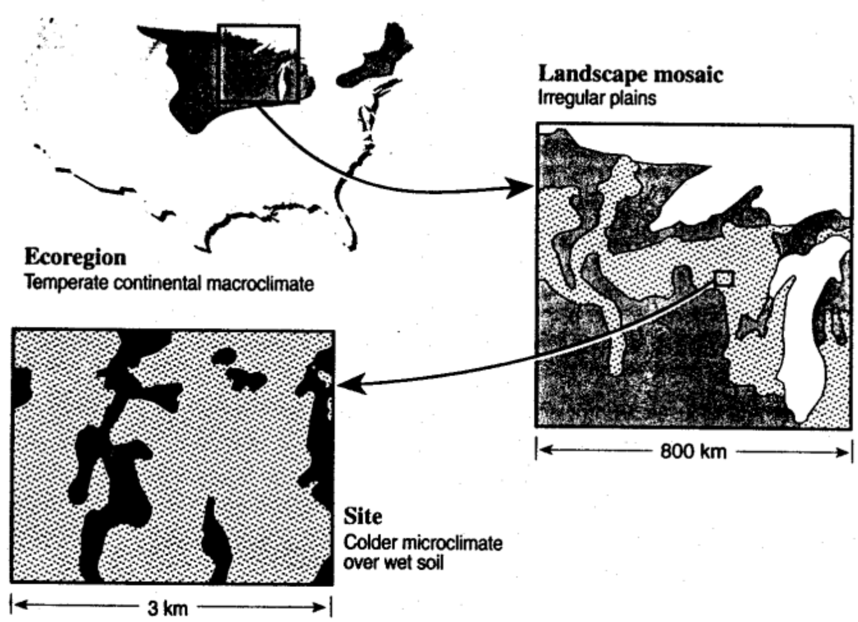

Figure 1--An example of ecosystem maps at different scales.

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) Suborder (Boroll)	Class (forest) Subclass (coniferous forest) Formation (Temperate Mesophytic forests)
Meso	Great Groups (Cryoboroll) Subgroups (Lithic Cryoboroll)	Series (grand fir)
Micro	Family (clayey, Lithic Cryoboroll Phase of family (eroded phase)	Plant association (grand fir/ginger) 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 sub-orders 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.

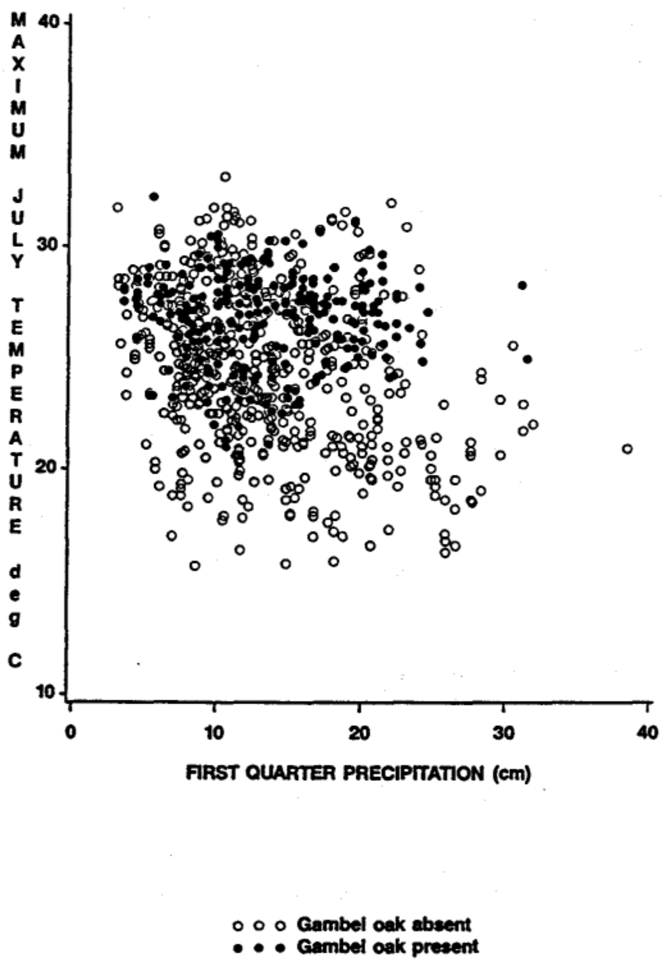

Figure 1-- Distribution of southwestern forested plots with and without Gambel oak in the environmental space determined by the climatic attributes of maximum July temperature and first quarter (January to March) precipitation.

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).

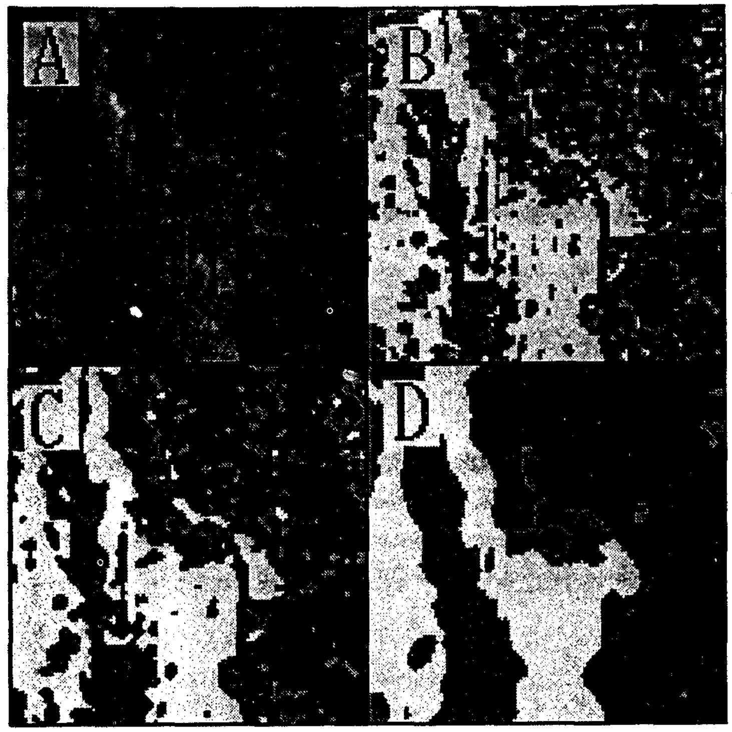

Figure 1–Changes in the association of cover types with the scale at which habitat density is measured: (A) Aerial photograph of woodlands (dark) and grasslands (light or medium tones); (B) initial cluster analysis of the photograph, showing three distinct cover classes including woodland (black), rock 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.

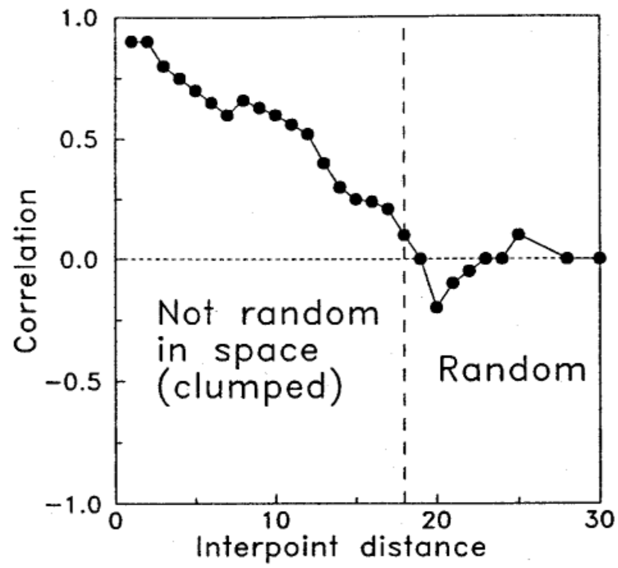

Figure 2-Theoretical relation between lag distance and correlation. A critical lag distance (vertical dashed line) may be found below which landscape patterns are correlated and above which measured quantities are randomly associated.

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.

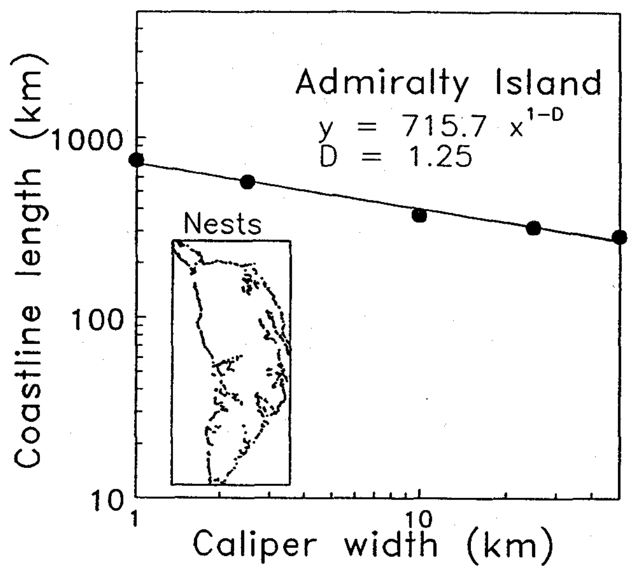

Figure 3--Fractal geometry of Admiralty Island. Estimated coastline length decreases with Increasing caliper width or decreasing resolution. The inset shows a map of eagle nests tracing the coastline.

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).

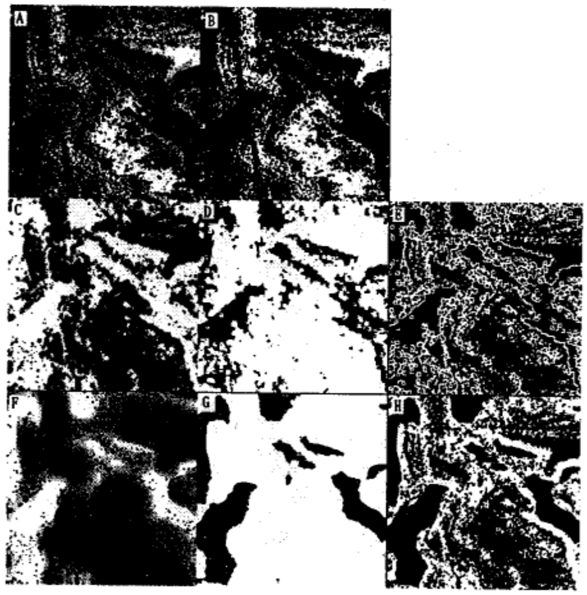

Figure 4--Visualization of forest thinning at two scales: (A) Digital image of woodlands showing trees in dark tones, (B) classification of photograph to provide a map of trees (black); (C) tree density measured within 7 x 7 cell windows is proportional to brightness, (D) candidate tree cells for thinning 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.

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Timothy Keitt and Jeremy Worley assisted with computer programming, and Colleen Hatfield assisted with geographic information system development. Funding was provided by the National Science Foundation (BSR-9058136, BSR-9107339, BSR-8811906) and the Electric Power Research Institute, Sevilleta LTER Publication 42.

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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).

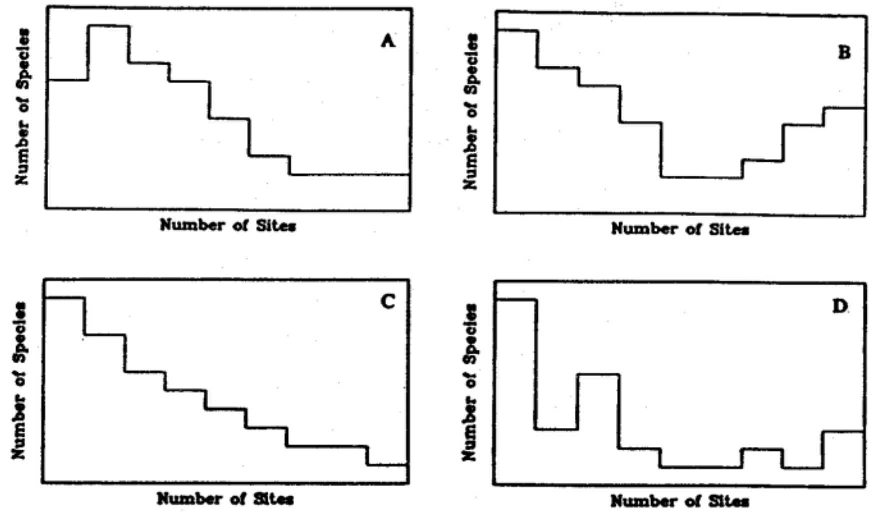

Figure 1-Number of species found In different numbers of sites based on four models (see text): A) Levins (1969), B) Hanski (1982a), C) Brown (1984) and D) Kolasa (1989).

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).

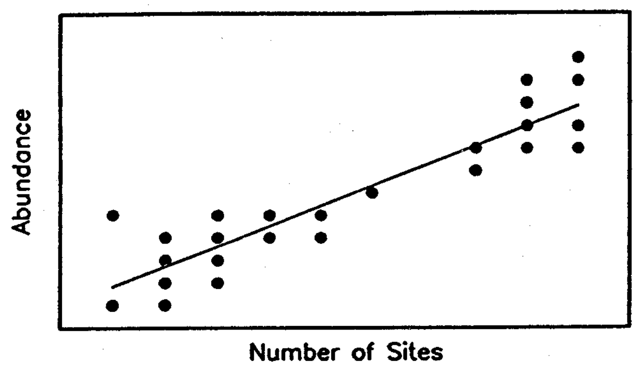

Figure 2-Relation between abundance of a species at a site and the number of sites the species occupies as assumed by Hanski's (1982b) core and satellite species model.

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

G.W. Minshall

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).

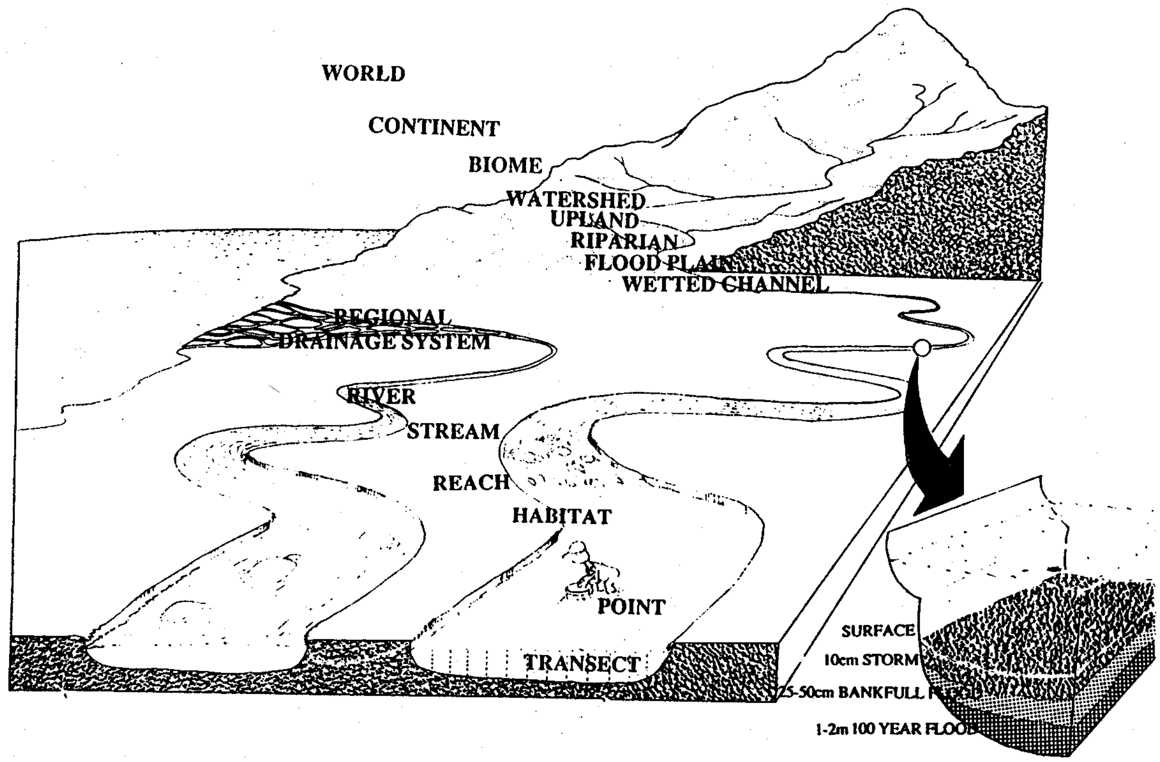

Figure 1--Spatial boundaries of flowing water ecosystems (from Minshall 1988). Inset illustrates how boundaries below the surface of the streambed may differ with flow, but the depth dimension also will differ from place to place (e.g., pools versus riffles).

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.

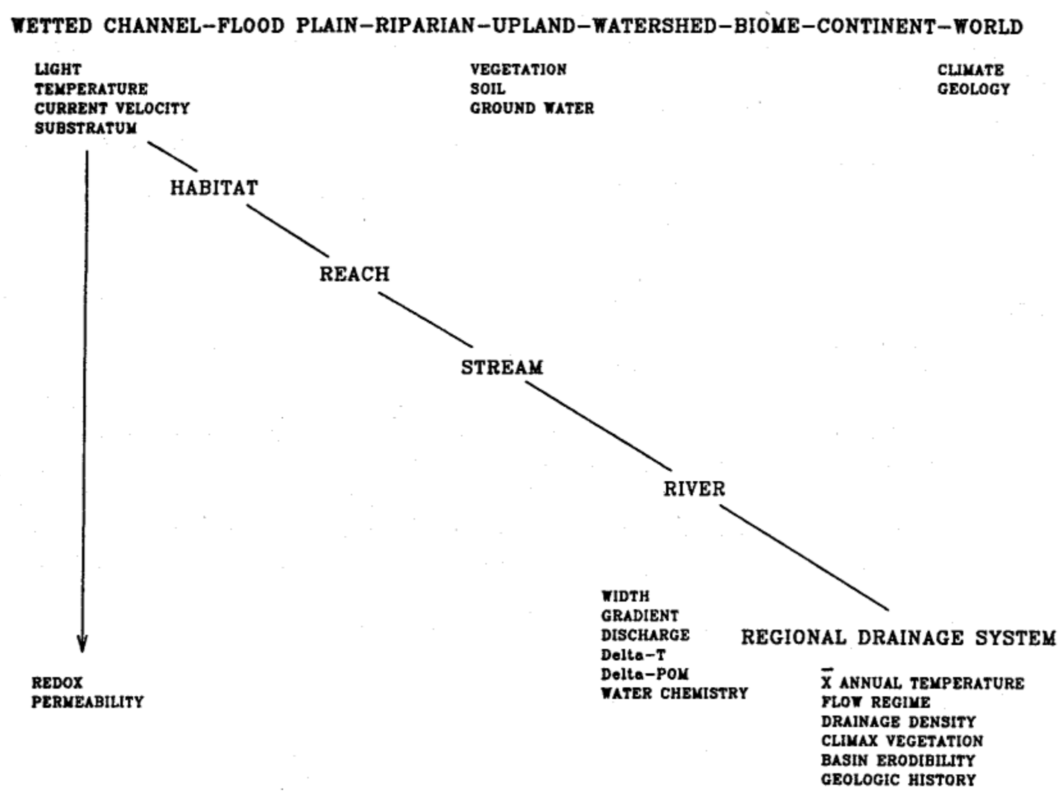

Figure 2--Illustration of how the principal factors responsible for the structure and behavior of stream ecosystems may differ with scale of resolution (from Minshall 1988).

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.

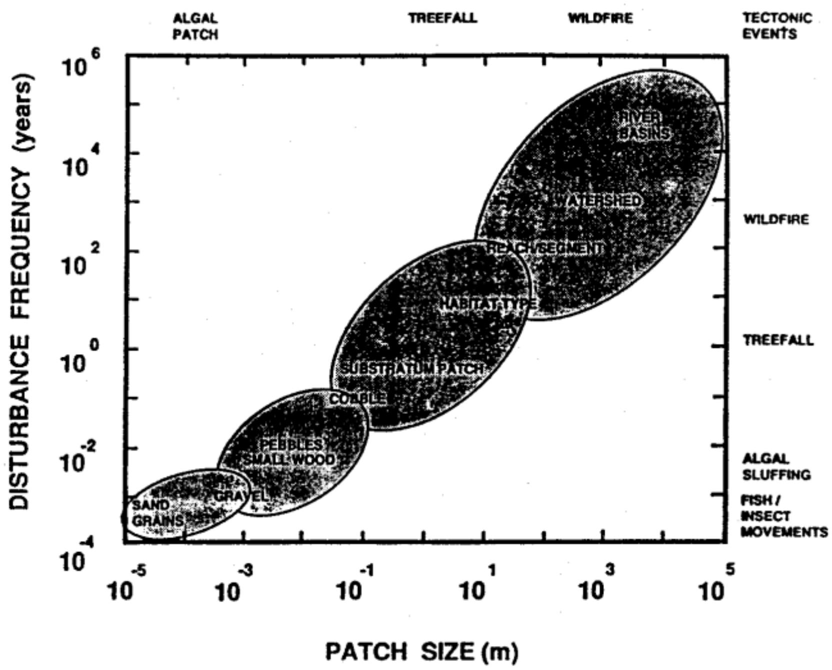

Figure 3--Time-space continuum for stream ecosystems (based on values given in Minshall 1988, tables 3 and 4).

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.

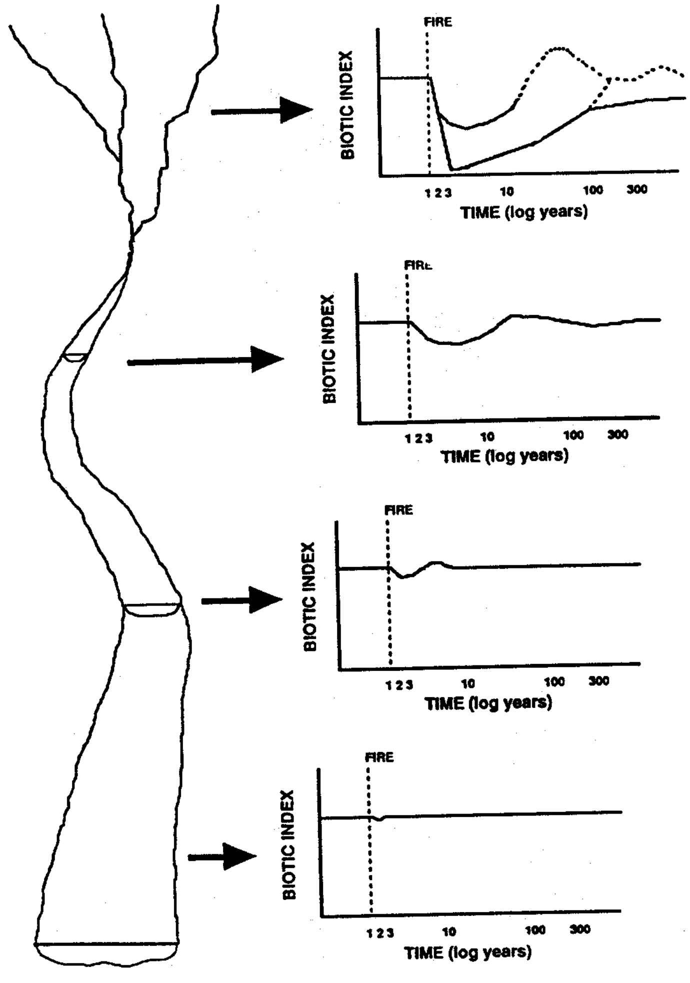

Figure 4--Postulated temporal response of stream communities, at different locations along a river drainage net, to a disbursed (heterogeneous) landscape-scale disturbance.

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 Regional geology Regional topography Regional terrestrial vegetation Flow regime			Region; State; Forest; District	Topographic maps (15') Geologic maps (15') Landsat photos Annual discharge records
Stream system (10^3 - 10^4 m)	Local climate Local geology Local topography Local terrestrial vegetation 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') Geologic maps Vegetation maps Aerial photos Annual temperature recorc
Segment system (10^2 - 10^3m)	Tributary junctions Major geologic discontinuities	Tributary junctions major falls; bedrock lithologic or strucutural discontinuities	Valley sideslopes or bedrock out-crops controlling lateral migration	Paired watersheds Segment classes (e.g. uplands vs lowlands)	Topographic maps (7.5') Ground reconnaisance Low level aerial photos
Reach system (10^1 - 10^2m)	Channel slope Valley form Bed material 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 - 10^1m)	Bed form and material Origin Persistence 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 - 10^0m)	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 (ΔT)
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 Hardness pH Specific conductance Turbidity	Grab samples analyzed using standard methods	General water quality
Biotic Factors:		
Macroinvertebrates	Rapid bioassessment protocol III	Biotic condition indicators Community structure indices
Fish community structure	Rapid bioassessment protocol V	Biotic condition indicator 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 Magnesium Nitrate-nitrogen Phosphorus (ortho) Sulfate	Filtered sample Colormetric field procedure	Delineation of main cations Principal plant nutrients 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 Total biomass	Improved indicators Estimates of 2^{o} 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

**M. Jurgensen, P. Hessburg, J. Lehmkuhl, W. Megahan, J. Nesser,
R. Ottmar, and E. Reinhardt**

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 overcrowding, 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 rangeland 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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