LANDSCAPE BIODIVERSITY MODELING AS A TOOL FOR APPLYING ECOLOGICAL THEORY TO LAND USE PLANNING IN THE LITTLE APPLEGATE WATERSHED

by

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A THESIS

Presented to the Environmental Studies Program
and the Graduate School of the University of Oregon
in partial fulfillment of the requirements
for the degree of
Master of Science

June 1999

"Landscape Biodiversity Modeling as a Tool for Applying Ecological Theory to Land Use Planning in the Little Applegate Watershed," a thesis prepared by Christina E. Whitman in partial fulfillment of the requirements for the Master of Science degree in the Environmental Studies Program. This thesis has been approved and accepted by:

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June 1, 1999

Date

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Vice Provost and Dean of the Graduate School

An Abstract of the Thesis of

Christina E. Whitman for the degree of Master of Science

in the Environmental Studies Program to be taken March 1999

Title: LANDSCAPE BIODIVERSITY MODELING AS A TOOL FOR APPLYING ECOLOGICAL THEORY TO LAND USE PLANNING IN THE LITTLE APPLEGATE WATERSHED

Approved [redacted]

Dr. Bart Johnson

Alteration of habitat from human land use is the primary threat to biodiversity in the Little Applegate Watershed. Landscape scale modeling provides a method for assessing impacts to biodiversity from future landscape change, increases understanding of significant variables, informs land use policy, and helps bridge the gap between science and land management. I examine the potential for the application of ecological research to land use planning and policy using biodiversity modeling in the Little Applegate Watershed, including a review of: existing projects in the watershed, the ecological basis for landscape biodiversity modeling, landscape scale biodiversity risk assessment case studies, and a survey of Little Applegate land managers. Results indicate that the application of landscape scale biodiversity assessment to land use decision making in the Little Applegate will be largely dependent on social and institutional factors but that improved scientific understanding, additional data, and increased modeling capability will also be required.

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ACKNOWLEDGMENTS

The author expresses sincere appreciation to Professor Bart Johnson for his assistance in the development of this research project, and to Professors Bart Johnson and John Baldwin in the review and preparation of this document. In addition, I wish to thank the land managers and residents of the Little Applegate Watershed for their insightful comments on this research topic. I also thank my friends in the University community and beyond who have supported me in the completion of this project.

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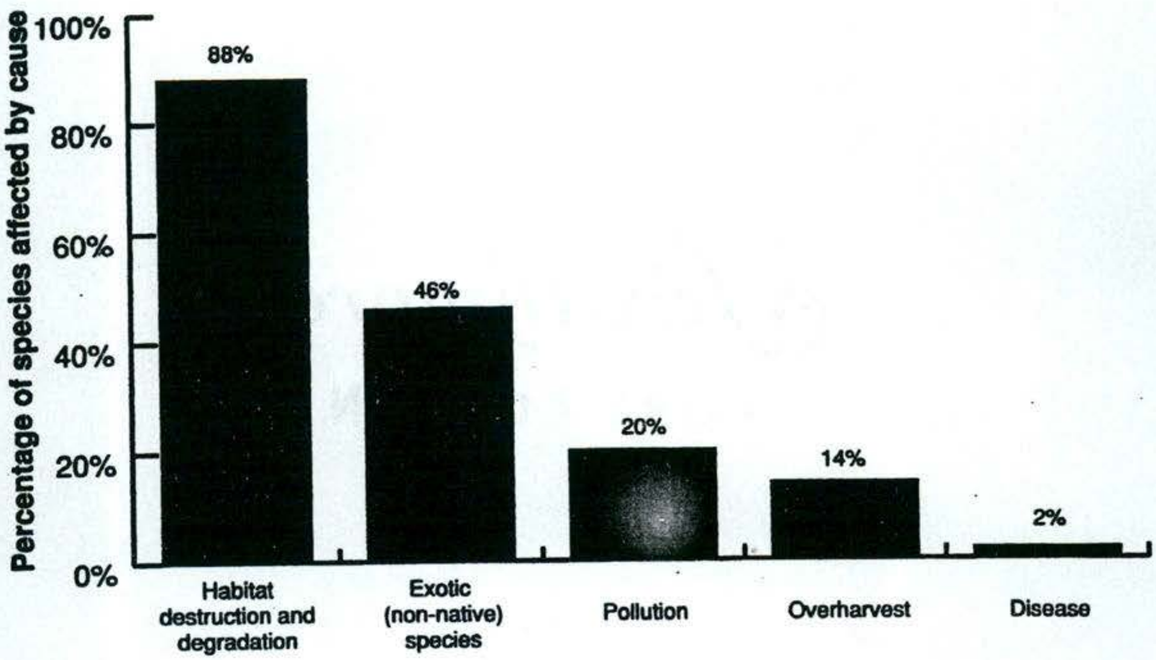
CHAPTER I

INTRODUCTION

Changes in habitat from human land use, including the diversity of habitat types, habitat area, and spatial configuration of habitat, has been identified as the primary threat to biodiversity in the Little Applegate Watershed (AMA 1995a). The Little Applegate is located in the Siskiyou region of southwestern Oregon, an area of high environmental and biological diversity. Substantial research and planning efforts on a variety ecological and social variables that influence the maintenance of natural resources in the area have been initiated in the watershed. Efforts are needed to integrate these existing projects, expand the level of information regarding biodiversity in the watershed, and aid in the application of project results to land use policy. Landscape scale biodiversity modeling provides a method for assessing the potential impacts to biodiversity from future landscape change and can inform the process of land use decision making. Spatially explicit modeling can bridge the gap between science and land management as it includes social and economic land use variables as well as ecological variables. In this research project I examine the potential for the application of ecological research to land use planning and policy using biodiversity modeling in the Little Applegate Watershed through a review of the ecological basis for landscape scale biodiversity protection and modeling, examination of case studies of applied biodiversity risk assessments, and interviews with land managers and scientists working within the watershed.

The alteration of habitat is the largest threat to biodiversity that species, communities, and ecosystems face today (Soule 1987, Grumbine 1990, Franklin 1993, Pienkowski 1995, Noss, O'Connell, and Murphy 1997) (Figure 1). Habitat alteration includes direct habitat destruction through conversion to another land use; habitat fragmentation, the breaking of large contiguous patches of habitat into smaller, often isolated patches; and habitat degradation, which results in changes in the composition, structure or function of an ecosystem (Franklin 1993, Pienkowski 1995, Schumaker 1996, Noss, O'Connell, and Murphy 1997). While habitat destruction has been widely recognized as a major threat to biodiversity for some time, fragmentation is now regarded as an equally critical conservation problem. Patterns of land use and habitat fragmentation can have as great an effect on species persistence as an overall change in habitat area (Turner 1987, Dale et al. 1994).

Figure 1. Causes of Endangerment of Threatened and Endangered Species in the United States. From Noss, O'Connell, and Murphy (1997)



Biodiversity Protection at the Landscape Scale

The field of conservation biology is responding to increased understanding of the role of landscape pattern and process on species persistence with a shift away from the traditional species by species approach to management to a broader, ecosystem or regional perspective. Biodiversity can represent a variety of things, the most basic of which is species diversity. To many conservation biologists the term biological diversity, or biodiversity, represents the ecological environments where species occur, as well as the species themselves. This wider definition involves more than species numbers and includes the attributes of ecosystems that "determine, and in fact constitute, the biodiversity of an area" (Noss 1990, 356). The transition from species to regional habitats as units of management that is occurring stems from the realization that biodiversity includes more than species richness or abundance, and that human related disturbances are so extensive that their effect must be evaluated at the landscape or larger scale to incorporate the cumulative effects of land use activities and efficiently address the issue of conservation (Flather and King 1992, Thompson and Welsh 1993, Franklin 1993). The sheer numbers of species, many unidentified or poorly understood, makes the problem of managing for biodiversity on an individual species basis intractable in terms of "time, resources, societal patience, and scientific knowledge" (Franklin 1993, 202). Moreover, the growing field of landscape ecology is focusing much needed attention on the matrix lands that dominate the landscape and illustrating the importance for biodiversity of broad scale habitat patterns and processes across both matrix and reserve lands.

The matrix, or complex of semi-natural and domesticated lands outside of reserve systems, is the dominant land type in most inhabited areas, occupies the most productive regions, and likely contains the majority of biodiversity (Franklin 1993, Noss, O'Connell, and Murphy 1997). Matrix lands supplement the role of protected areas by providing habitat, influencing the effectiveness of protected areas, and controlling connectivity in the landscape (Franklin 1993, Pienkowski et al. 1995, Everett and Lehmkul 1996, Noss, O'Connell, and Murphy 1997). Management of the wider countryside to meet conservation objectives is necessary because protected areas are too small for the maintenance of viable populations of most species, the distribution of protected areas is not representative of all habitats, these protected areas are influenced by the context in which they are embedded, and geographical continuity of habitat must be retained for species to potentially shift range in the face of climate or other broad scale environmental changes (Soule 1987, Grumbine 1990, Spellerberg, Goldsmith, and Morris 1991, Franklin 1993, Pienkowski et al. 1995, Noss, O'Connell, and Murphy 1997). In most regions, the land that supports the most productive and diverse habitats are private, and these lands are an essential component of any large scale biodiversity plan (Franklin 1993, Noss, O'Connell, and Murphy 1997). Conservation of biodiversity over the long term will involve land held predominantly in private ownership. Extending biodiversity concerns from reserves to matrix lands should enhance the conservation of biodiversity and result in a system of reserves better integrated with the surrounding land uses and ecological principles.

In landscape scale conservation, the context of individual sites and the spatial configuration of habitat becomes as important as site content and habitat area. A current focus of landscape scale conservation has been on providing corridors for species dispersal. As the role of corridors in achieving functional connectivity is largely unproved, increased attention to the matrix lands may offer alternatives for maintaining connectivity in the landscape, by minimizing the contrast between reserves and other land use types (Franklin 1993, Schumaker 1996, Dellasala et al. 1996). Ecosystem and landscape scale research into the effects of forest cutting patterns, species habitat relationships, and species distributions in relation to forest pattern have been pivotal in bringing the concepts of landscape ecology to the management arena; in the Pacific Northwest this transition has largely resulted from research and management efforts concerning the northern spotted owl and the Northwest Forest Plan (Swanson and Franklin 1992, Pojar et al. 1994).

In addition to maintaining natural patterns of biodiversity at a variety of scales across the landscape, a coarse filter approach to land management seeks to maintain the ecological processes important to biodiversity (O'Connell and Noss 1992). While species richness and diversity are classic concepts in ecology and remain important, the new emphasis in ecological research and management is on the functional components and processes of ecosystems. "In general, ecological and evolutionary processes, including interactions between communities and species, are at least as important to the maintenance of biodiversity as conservation of species and communities themselves" (O'Connell and Noss 1992, 437). A land use planning approach based on ecological function is hindered at this point by a lack of understanding of the significant processes and their interactions. Landscape ecological planning in rural areas must also recognize that the spatial pattern of the landscape is a reflection of both social and ecological processes (Hulse and Melnick 1990). A comprehensive approach to the protection of biodiversity at the landscape scale will include research and management attention to underlying cultural and biological conditions.

Protection of biodiversity over the long term may require a management strategy that prioritizes the role of regional landscape patterns and processes over local concerns (Noss 1983, Turner 1989). Land managers have traditionally worked toward the goal of optimizing local site diversity but this strategy may operate at the expense of species and ecosystems in need of protection at the regional scale (Noss 1983). Because site based management has tended to focus on localized investigations of environmental and economic impacts, the cumulative effects of these management decisions may be incompatible with broader-scale management objectives (Montgomery, Grant and Sullivan 1995). The traditional focus of ecology and land management must be broadened to incorporate the heterogeneity of landscapes and regions. One method for this is the maintenance of characteristic regional features and management that aims to retain more natural levels of ecosystem complexity (Swanson and Franklin 1992, Pienkowski et al. 1995, Dellasala et al. 1996).

Motivation for extending the geographic scope of conservation measures has come from planning and management fields as much as from the disciplines of ecology and conservation biology (Flather, Brady, and Inkley 1992, Swanson and Franklin 1992). There is wide recognition that successful conservation of biodiversity requires the integration of ecological principles into land use planning and policies over broad regions. As biodiversity is not distributed randomly or uniformly across the landscape an ecosystem approach to conservation is generally more effective and efficient than a species by species approach because it will include a broader range of habitat types and ecological processes significant to biodiversity (Noss, O'Connell, and Murphy 1997). Large tracts of land and the economic and social support systems necessary for their protection will generally not be available to significantly increase the amount of land allocated to biodiversity, the practical alternative is an attempt to improve the quality of protection in areas managed for multiple objectives. Improving the quality of protection across the landscape will require extensive communication and coordination among land managers, landowners, and the scientific community. The ecosystem management paradigm is an attempt to integrate multiple scales and disciplines in the process of ecological research as well as land use planning and management (Table 1).

Table 1. Dominant Themes of Ecosystem Management
Adapted from Grumbine (1994)

Ecosystem Management Theme	Theme Description
Hierarchical Context	When working on a problem at any one level or scale, managers must seek the connections between all levels
Ecological Boundaries	Management requires working across administrative/political boundaries and defining ecological boundaries at appropriate scales
Ecological Integrity	Protecting total native diversity and the ecological patterns and processes that maintain that diversity
Data Collection	Ecosystem management requires more research and data collection, as well as better management and use of existing data
Monitoring	Managers must track the results of their actions so that success or failure may be evaluated quantitatively
Adaptive Management	Management is a learning process or continuous experiment where incorporating the results of previous actions allows managers to remain flexible and adapt to uncertainty
Organizational Change	Implementing ecosystem management requires changes in the structure of land management agencies and the way they operate
Humans Embedded in Nature	Humans are fundamental influences on ecological patterns and processes and are in turn affected by them
Values	Regardless of the role of scientific knowledge, human values play a dominant role in ecosystem management goals
Interagency Cooperation	Using ecological boundaries requires cooperation among federal, state, and local agencies as well as private parties

It is imperative that any regional plan involve the local and regional human population in the planning and decision making process (Rookwood 1995). Realization of the limitations of protected areas to adequately protect biodiversity has led to increased attention on the potential role rural landscape can play in providing habitat (Pierce 1996). Although the resource commodity function, or its legacy, still dominates most rural areas, these regions are now recognized for the tremendous range of biodiversity functions they can provide. Conservation strategies must account for social environments and issues as well as scientific principles and acknowledge the attitudes of local individuals, organizations, and communities. The effects of regional planning and management for biodiversity on the surrounding communities must also be investigated (O'Connell and Noss 1992). Federal resource management efforts need to acknowledge and plan for the fact that communities evolve and change over time in response to a variety of influences, and adaptive management provides an administrative structure designed to operate within changing conditions (Haynes et al. 1996).

Linking Science And Land Use Planning

The success of biodiversity conservation programs will depend in large part on land use policies and thus these policies need a strong connection to conservation biology (Noss

1983, Pienkowski et al. 1995, Noss, O'Connell, and Murphy 1997). An ecological framework for planning should be scientifically based, address the role of humans as members of regional landscapes, and include public and private members in the decision making process (Grumbine 1990). While the shift to a larger temporal and spatial extent of management dictated by ecosystem management is a necessary and positive transition, much remains to be learned about the functioning of natural systems at the ecosystem, landscape, and regional scale. The human dominated landscape has disrupted the processes and patterns of nature at a multitude of scales and our ability to understand these systems and ultimately predict the consequences of management practices is unsubstantiated (Dellasala et al. 1996). "Attempts to understand ecosystems has generally taken one of two main approaches- understanding them by focusing on ecosystem population and community, or understanding them by focusing on ecosystem process and function" (Hulse and Melnick 1990, 246). A landscape approach to the protection of biodiversity requires a combination of these approaches and understanding of the relationships between populations, communities and processes.

Further research into the role of landscape pattern and process on biodiversity is needed as the concept of designing landscapes for biodiversity objectives is relatively new and is likely to change as knowledge of ecology at the landscape and regional scale increases (Swanson and Franklin 1992). Biodiversity research focused on large scale trends is needed to answer questions and supplement knowledge needed for planning and management decisions (Noss, O'Connell, and Murphy 1997). Mechanisms are also needed that increase the applicability of ecological research to land use planning and policy. To fully realize conservation biology principles, a research approach is needed that emphasizes a prescriptive capacity, and enables application to management decisions (Grumbine 1990, Flather and King 1992, Swanson and Franklin 1992). Landscape scale habitat conservation planning is interdisciplinary and includes the following essential components:

1. Scientific evaluation of alternatives
2. Spatially explicit assessment of population status or viability
3. Reserve selection and design
4. Sustenance of evolutionary and ecological processes
5. Adaptive management
6. Linkage of disparate planning efforts into a unified approach

(Noss, O'Connell, and Murphy 1997, 135)

As only a fraction of the available scientific information is ever applied to the decision making processes that guide landscape change, the transfer of information from scientist, to planner, and on to politicians and the general public, must be given a high priority (Rookwood 1995). The landscape planning process can provide this link between science and policy by creating a set of guidelines or objectives for planning which are based in sound scientific theory (Rookwood 1995). In addition to providing an ecological basis for the management of ecosystems and landscapes, scientists also have a role in the social processes that determine the future course of management (Swanson and Franklin 1992). A framework for addressing questions of how management practices can promote ecosystem management and the protection of biodiversity will need to consider the social and economic causes of land use change as well as the social, economic and policy implications of ecological research (Turner 1987, Dale et al. 1998).

Conceptual and simulation models provide a tool for investigating the relationships among and between ecological and social factors, and can help inform and guide the process of land management. The use of models to link science and land use planning is increasing with the development of spatially explicit models that can be used to evaluate change over time at landscape or regional scales. Continued research and management attention to the potential for models to aid in the integration of ecological theory and land use planning is needed, but the use of modeling in ecology is well established and application is rapidly expanding in investigations of biodiversity and landscapes.

Management Decisions

> Scientists owe it to the rest of society to provide rules of thumb, even when they know that sometimes the rules will be misunderstood and misused
> (Soule 1987, 175)

The enormous complexity of ecosystems forces land managers, planners, and policy makers to make decisions based on a limited number of key components of the system or process of interest. The identification of significant structural and functional features of ecosystems is a major role of ecologists in the conservation process (Swanson and Franklin 1992). Given the fact that science is better at describing than predicting, the responsibility of conservation biology to forecast the outcomes of alternative management options is formidable (Noss, O'Connell, and Murphy 1997). However, a basic understanding of ecosystems, sufficient to predict their trajectories reasonably well, is possible. Understanding of the dominant variables that control pattern and process in ecosystems, coupled with life history knowledge of indicator, rare, or sensitive species, will provide enough confidence to move forward with planning (Noss, O'Connell, and Murphy 1997). If needed, adaptive management can be used to realign management actions with conservation objectives. The conflict between the scientific admission of uncertainty and unpredictability and the desire of managers and landowners for certainty and solutions can be resolved through the use of adaptive management.

Adaptive management is defined as an iterative process which includes planning, inventory, implementing actions, monitoring, and evaluation, and emphasizes the role of feedback and adjustment between all of these stages (Bormann et al. 1994). Because of its focus on reformulating responses to changing circumstances, the adaptive management paradigm has been adopted for all federal lands covered in the Northwest Forest Plan, and is being used by other organizations and agencies as well (Bormann et al. 1994). Landscape scale planning, with its need for linked actions that integrate science and management and an emphasis on possible futures, is well suited to adaptive management. Many of the underlying tenets of adaptive management, such as responding quickly to social and ecological change, learning from alternative management actions, increasing collaboration between stakeholders, and improving decision making, are all components of successful landscape ecological research. Monitoring is a critical component of the adaptive management process and is also an important link between science and management. Monitoring should be incorporated as an essential part of the scientific research process and used to test specific questions that are relevant to improving ecological knowledge as well as the natural resource decision making process (Noss 1990). The expectation that plans will be modified as knowledge increases should be clearly outlined to all participants and explained as a key component of adaptive management (Noss, O'Connell, and Murphy 1997).

Land managers need a broad range of tools, both social and technological, to address issues of uncertainty in the management of natural systems (Swanson and Franklin 1992). The quality of these methods will depend in large part on the integration of land management and science. While land managers have a long history of attempting to weigh the potential consequences of alternative management actions, there are few examples of comprehensive guidelines that direct the application of biological data to the design of reserve systems, ecosystems, or landscapes for the protection of biodiversity (Murphy and Noon 1992, Hansen et al. 1993). Noss, O'Connell, and Murphy provide one example of guidelines, or ecological principles for land managers in their 1997 book The Science of Conservation Planning (Table 2). For guidelines and measures of biodiversity to be useful to managers "they must be meaningful (biologically and socially), measurable (quantitative and obtainable from available data), and manageable (subject to change based on human decision)" (Silbaugh and Betters 1997, 245). A well structured approach to the application of science to management should facilitate data interpretation and clearly outline the analysis process, its underlying assumptions, and any additional uncertainties (Everett and Lehmkuhl 1996). An ecological approach to uncertainty dictates that the fewer data or the higher the uncertainty involved, the more conservative a management plan should be (Noss, O'Connell, and Murphy 1997). Using the same logic, a plan with a minimal predicted impact on a particular species or ecosystem would receive less scientific scrutiny.

Table 2. Ecological Principles for Biodiversity Planning
Adapted from Noss, O'Connell, and Murphy (1997)

Philosophical Principles

- Ecosystems not only are more complex than we think, but more complex than we can think
- Nature is full of surprises
- The fewer data or more uncertainty, the more conservative management should be
- The less the predicted impact of a project on species or ecosystems, the less scientific scrutiny is needed
- Conservation goals must be clearly stated and correspond to the best available scientific information
- Biodiversity conservation must be concerned with many different spatial and temporal scales
- Conservation biology is interdisciplinary but biology must provide the foundation

Principles for Species Conservation and Reserve Design

- Species well distributed across their native range are less susceptible to extinction than species confined to small portions of their range
- Blocks of habitat close together are better than blocks far apart
- Habitat in contiguous blocks is better than fragmented habitat
- Interconnected blocks of habitat are better than isolated blocks
- Blocks of habitat that are roadless or otherwise inaccessible are better than roaded and accessible blocks
- Populations that fluctuate widely are more vulnerable than populations that are more stable
- Disjunct or peripheral populations are likely to be more genetically impoverished and vulnerable to extinction, but also more genetically distinct than central populations

Principles for Community and Ecosystem Conservation

- Maintaining viable ecosystems is usually more efficient, economical, and effective than a species-by-species approach
- Biodiversity is not distributed randomly or uniformly across the landscape
- Ecosystem boundaries should ideally be determined by reference to ecology, not politics
- Because conservation value varies across a landscape, zoning is a useful approach to land-use planning and design

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Futures Planning

The need to predict the effect of alternative management scenarios on ecosystems and landscapes has sparked much of the interest land use planners have in the applications of landscape ecology and geographic information systems to land management (Hulse and Melnick 1990). Applied science attempts to evaluate the potential consequences of alternative management options for a specific setting, and to learn more about the significant variables and the relationships between them in the analytical process (Jorgensen 1994). Spatially explicit data on environmental conditions and models of the interactions between these environmental conditions can be used as a strategy for assessing the effects of land use impacts on natural systems (Dale et al. 1998). For land use planning to make better informed decisions, methods are needed that predict, within some reasonable range of accuracy, the potential effects of human activities on biodiversity at the site, ecosystem, landscape, and regional scale (White et al. 1997). Habitat based ecological models can play an important role in the research and management of biodiversity at the landscape scale, particularly if they are used to explore and understand complex relationships rather than as predictive wizards in natural resource decision making. The basic premise is that modeling real world landscapes will increase the ability of managers to compare alternative management scenarios and, optimally, make better decisions (Dunning et al. 1995, Freemark et al. 1996, Hansen et al. 1996).

"A key challenge in sustaining biodiversity is to understand the effects of human activities on species distributions and to use this knowledge to predict likely patterns of biodiversity under future change scenarios" (Hansen et al. 1995, 556).

As important as ecological research into the evaluation of alternative futures is the need for a comprehensive social vision of the desired future state of the landscape or region and the roles that biodiversity protection might take (Rookwood 1995). "Landscape planning and design are activities fundamentally concerned with making value based decisions about how to physically structure a landscape, decisions which are effected by a broad spectrum of human values" (Hulse and Melnick 1990, 244). Landscape planning for biodiversity protection must incorporate the concerns and desires of rural communities, and include social and economic factors with ecological constraints and opportunities into the planning process. The integration of multiple objectives that would likely result from the involvement of local and regional constituents and agencies in the futures planning process may greatly increase the chances of implementing a regional biodiversity protection plan (Rookwood 1995).

Landscape Scale Biodiversity in the Little Applegate Watershed

The variety of landforms and vegetation in the Little Applegate Watershed results in a great diversity of habitat types and biodiversity; with 272 terrestrial species known or suspected to occur in the watershed (AMA 1995a). Forty-eight of these species have special status with either state or federal agencies (AMA 1995c). Terrestrial special status species include amphibians, reptiles, birds, and mammals, and at least four fish species are proposed or candidates for listing under the federal Endangered Species Act (AMA 1995a). A watershed analysis completed in 1995 documented that the primary issue regarding the persistence of terrestrial biodiversity in the Little Applegate Watershed is the diversity of habitats, of sufficient acreage, distributed across the landscape through time (AMA 1995a). Management that simultaneously meets the habitat needs of these diverse species will require a landscape approach to biodiversity protection.

While over seventy percent of the Little Applegate Watershed (52,158 acres) is owned and managed by the United States Forest Service and The Bureau of Land Management, there are no lands managed with biodiversity protection as the top priority (wilderness areas, etc.). The majority of public lands are managed for multiple uses, and successful integration of biodiversity protection with a variety of land uses will require a broad based perspective in terms of the area covered and the people and agencies involved. The location of the Little Applegate Watershed within the geographic scope of the Northwest Forest Plan, and the designation of the federal lands within the watershed as an Adaptive Management Area, has helped foster an ecosystem or landscape approach to management in the watershed. In addition, active citizen participation in land use management, through organizations such as the Applegate Partnership and the Rogue Institute for Ecology and Economy, have broadened the traditional scope of natural resource management to include private lands along with economic and social issues, and helped locate the watershed at the forefront of landscape planning and management. As such, the Little Applegate Watershed has begun to connect ecology and land use planning, particularly in the areas of river systems and vegetation communities. An environmental history was compiled for the watershed (LaLande 1995) as a component of the watershed analysis; together these documents provide a good picture of current conditions and the actions that led to them. Guild based habitat modeling was also conducted as a component of the 1995 watershed analysis; this effort has begun to investigate the effects of landscape pattern on terrestrial species in the Little Applegate Watershed. An additional landscape scale project in the region, Enhancing Community Capacity to Use Spatial Information, was conducted for the entire Applegate Watershed by Interrain Pacific and the U.S. Forest Service's Pacific Northwest Research Station, and included extensive geographic information systems mapping (http://www.interrain.org/applegate/html). Application of the Forest Landscape Analysis and Design Process (Diaz and Apostol 1992) in the Little Applegate Watershed, a methodology for determining landscape pattern objectives and creating a spatial landscape design has been used to investigate possible futures for the watershed.

Landscape scale biodiversity modeling provides a method for assessing the potential impacts to biodiversity from future landscape change. Substantial work at the landscape scale has been conducted or is underway in the Little Applegate Watershed. These existing projects provide much of the information needed as model input in a landscape scale biodiversity assessment. In addition, the range of analysis that has been conducted can guide research direction and refine model development. Next steps in the process of regional planning for biodiversity in the Little Applegate Watershed include more detailed investigations of wildlife and habitats at the landscape scale, as well as additional attention to futures planning and the integration of social variables. Research attention is also needed into the specific management actions that will guide both social and ecological trajectories in the direction of desired landscape pattern goals once they have been identified through landscape scale assessment and evaluation of alternative land use scenarios.

Research Description

This research explores the potential for the application of ecological theory to land use planning in the Little Applegate Watershed using landscape biodiversity models. While the term biodiversity has been defined earlier in this document to include species and the ecosystems in which they exist, for the purpose of this research, I am primarily using the term biodiversity to describe the richness and abundance of terrestrial wildlife. In the next section, chapter two, I describe the social and environmental context of the Little Applegate Watershed, and provide background information on existing landscape scale projects that have been completed in the watershed. Chapter three is based on a review of the literature and outlines the ecological basis for landscape biodiversity modeling. This chapter is divided into three sections, including discussion of wildlife-habitat relationships, landscape ecology, and ecological modeling, all presented in the context of landscape scale biodiversity modeling. In chapter four I introduce three case studies of the application of landscape scale biodiversity modeling to land use planning; these include projects for the Camp Pendleton region of southern California; Monroe County, Pennsylvania; and the Muddy Creek Watershed in western Oregon. The frameworks used in each of these examples are then examined in terms of project scale, project objectives, the biodiversity assessment methodology employed, alternative futures, and the partners involved in the project. The suitability of these frameworks for application in the Little Applegate Watershed context is also explored. In order to gain insight into potential applications of biodiversity assessment at the landscape scale from the local perspective, select land managers and scientists with responsibilities for biodiversity protection in the Little Applegate Watershed were surveyed. Survey participants included representatives from both government and non-governmental organizations. Chapter five outlines the survey process, questions, and results. The final chapter of this paper consists of the conclusions and recommendations that resulted from all aspects of research involved. It presents a general framework, or set of guidelines, for applying landscape scale biodiversity modeling to the land use planning process, as well as opportunities and constraints specific to the Little Applegate Watershed.

CHAPTER II

LITTLE APPLEGATE WATERSHED

This chapter provides background information on the Little Applegate Watershed, including a description of its biophysical and social context and an outline of landscape scale projects completed or underway in the watershed. The relation of environmental and social factors, and the potential application or expansion of existing landscape scale research to landscape scale biodiversity modeling is also explored.

Environmental Context

Applegate River Watershed

The Applegate Valley is a 500,000 acre watershed located in the Siskiyou region of southwest Oregon within the Rogue Basin (Figure 2). Over seventy five percent of the land within the Applegate Watershed is in federal ownership. A wide range of geologic parent materials, precipitation, elevation, and associated soils and vegetation types characterize the Applegate River Watershed. The Siskiyou Mountains traverse the Applegate and represent an important biological connection between the coast range and the Klamath Mountains. Resulting regional biodiversity is among the highest in the United States (Yaffee et al. 1996). Over fifty percent of the Applegate River Watershed is managed as the Applegate Adaptive Management area by the Rogue River National Forest, The Medford District Bureau of Land Management, and the Siskiyou National Forest (PNW 1998).

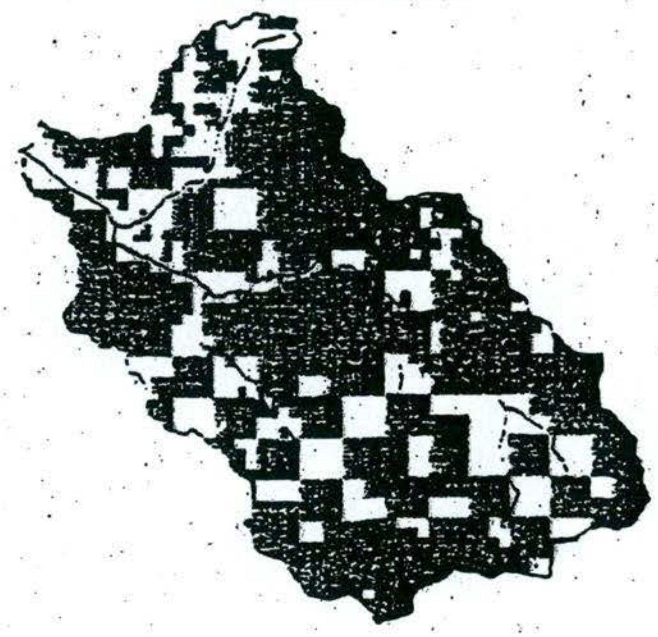
Figure 2. Applegate and Little Applegate Watershed Context

Little Applegate River Watershed

The Little Applegate Watershed is located in the southeastern corner of the Applegate River Watershed (Figure 2). At 72,262 acres, it comprise nearly fifteen percent of the Applegate River Watershed and two percent of the Rogue River Watershed (AMA 1995a). Over seventy percent of lands in the Little Applegate Watershed are federally owned and managed, with the United States Forest Service (USFS) controlling 23,219 acres, thirty two percent, and the Bureau of Land Management (BLM) controlling 28,939 acres, or forty percent (AMA 1995a). Private individuals and corporations control 19,784 acres, or twenty seven percent of land ownership in the watershed, and the State of Oregon owns 320 acres, or just under one half of one percent (AMA 1995a). Current land uses in the watershed are primarily forest lands managed for timber in the upper half of the watershed and agriculture and private residences in the lower portions of the watershed. The dominant human activities that affect wildlife habitat and the ecological function of the watershed include farming, grazing, and rural development in the lower elevation zones, public and private forest management in the mid to upper watershed, and fire suppression across the entire watershed.

Fire suppression, hydraulic mining, livestock grazing, timber harvest, agriculture, and residential development have all had major effects on habitat composition and pattern at both the stand and landscape scale. In addition to past and current management practices, land ownership patterns also illustrate the need for management at the landscape scale (Figure 3). Private land ownership in the watershed is concentrated in the lower elevation areas, with most private land occurring along stream corridors on the valley floor. Private land is also scattered throughout the upper elevation, although to a lesser extent, and these lands are primarily owned by private timber interests. "Land ownership patterns within the Little Applegate Watershed dictates that no one landowner acting independently of the others can effectively work to restore or enhance critical ecological functions " (AMA 1995a, 3).

Figure 3. Public and Private Ownership: Little Applegate Watershed



Federal lands of the Little Applegate Watershed are contained within the Applegate Adaptive Management Area, one of ten AMA's designated by the Northwest Forest Plan. The Applegate Adaptive Management Area was created to "develop and test variations on established management practices including partial cutting, prescribed burning, and low impact approaches to forest harvest so as to provide for a broad range of forest values, including late successional forest and high quality riparian habitat" (PNW 1998). A variety of factors, including the Northwest Forest Plan, the designation of the Applegate Watershed as an adaptive management area, and the active role of community members and private landowners in land use issues through the Applegate Partnership, the Applegate Watershed Council, and the Rogue Institute for Ecology and Economy, has resulted in increased capacity of federal agencies to incorporate community concerns and practice collaborative land use planning in the Little Applegate Watershed.

Vegetation

An wide range of physical conditions influence the environmental features of the watershed. These factors include elevation, which ranges from 1466 to 7418 feet, annual precipitation, which ranges from twenty five to fifty five inches per year and falls primarily in winter and spring, and geologic conditions, which are characterized by diverse soil characteristics due to the wide variety of parent materials (AMA 1995a). As a consequence, vegetation zones in the Little Applegate Watershed are also very diverse and range from low elevation grass forb communities through mountain shrublands, deciduous hardwood forests, mixed conifer-deciduous forests, high temperate conifer forests and sub-alpine forest parks (AMA 1995a). Grassy dry hillsides and mountain shrub communities occur primarily at lower elevations with southern exposures, but these plant communities can occur at any elevation if the environmental factors of a site make it inhospitable for the successful establishment of tree species. Deciduous hardwood and mixed conifer-hardwood forest plant communities in the Little Applegate Watershed occur primarily on low elevation, poor quality sites, and are often not capable of sustaining commercial timber products. Mixed conifer and high elevation forests provide almost all of commercial timber in the Little Applegate Watershed (AMA 1995b). Historic and current land use practices have all had major effects on the composition and pattern of vegetation communities at both the stand and landscape scale. The most prominent changes have been reductions in late successional forest stands, open pine dominated stands, open grasslands, and riparian forests (AMA 1995b).

Wildlife

The high diversity of elevation, precipitation, soils, and vegetation support a variety of habitat types and high wildlife species diversity within the Little Applegate Watershed. One hundred and thirty eight species of terrestrial vertebrates are known to occur in the Little Applegate Watershed, and another 134 are suspected to occur based on the presence of suitable habitat and location within their geographic range. Species that warrant special notice for their ecological or cultural significance or rare status include large mammals: Blacktailed Deer, Elk, Black Bear, and Cougar; large birds: Peregrine Falcon, Bald Eagle, Northern Spotted Owl, and Northern Goshawk; and a variety of other organisms including the Siskiyou Mountain Salamander, Western Pond Turtle, Tailed Frog, Townsend's Big-eared Bat and a second species of bat, Yuma Myotis. The upper elevations of the watershed are part of a land bridge between the Siskiyou and Coast ranges and this linkage has been identified as an ecological diverse area that is important for maintaining the genetic connectivity of the region. High habitat diversity and significant regional connections make species dispersal and the spatial configuration of the landscape important variables in landscape planning for biodiversity in the Little Applegate Watershed (AMA 1995a).

Social Context

The Northwest Forest Plan of 1993, established in response to both public and industry pressure to resolve contentious timber management issues on federal forest lands, shifted direction of federal land management away from a site and commodity based approach to a more holistic approach of ecosystem planning for public lands (Marcot and Thomas 1997). Extensive research into the effects of forest structure and spatial configuration on northern spotted owl populations spurred new attention, particularly in the fields of applied and landscape ecology, to connections between wildlife distribution and abundance and landscape scale habitat conditions. Intense scrutiny of scientific decisions regarding the ecological condition and future needs of the northern spotted owl helped speed the development of new approaches in scientific research and opened debate on the use of uncertain and disparate sources of data in making land management decisions (Marcot and Thomas 1997). The focus on large scale effects of habitat fragmentation and alternative management decisions on northern spotted owls brought the use of ecological models into realm of decision making and highlighted the potential for ecological modeling, landscape ecology, and geographic information systems to be applied to ecological planning and assessment (Marcot and Thomas 1997).

In response to the need for large scale planning that integrates science and management, ten adaptive management areas were designated throughout the area covered in the Northwest Forest Plan, including four in western Washington, four in western Oregon, and two in northwestern California (PNW 1998). Locations of adaptive management areas were selected based on current or potential future benefit to goals of the Northwest Forest Plan in terms of ecological and social capacity, and most exist in sub-regions that have been heavily impacted, economically and socially, by reduced timber harvest on federal lands. Adaptive management areas are intended as areas where technical and social experimentation should occur in the area of progressive forest management. Use of the adaptive management framework for natural resource planning is intended to promote learning about how to manage in a responsive and ever improving manner. In addition to the designation of specific adaptive management areas, the Northwest Forest Plan required adoption of the adaptive management paradigm for all federal lands covered in the plan (Bormann et al. 1994). The Northwest Forest Plan also emphasizes landscape scale planning and coordination among federal and private lands, in response to growing awareness of the fact that federal forests in the Pacific Northwest cannot be managed in isolation (Bormann et al. 1994).

The Applegate Valley, like resource dependent rural communities throughout the Pacific Northwest, has been experiencing significant environmental, economic, and social changes, motivated primarily by declines in the forest products industry and population growth in rural areas and across the region. There are numerous small communities in the Applegate; Wilderville, Murphy, Provolt, Williams, Applegate, and Ruch, which total nearly 12,000 in population and are all un-incorporated (Yaffee et al. 1996). Larger communities closely tied to the Applegate Watershed include Grants Pass, Medford and Jacksonville. Community response to social and environmental change has taken many forms, including the formation of citizen based approaches to long term, land use planning and management for the region that emphasizes the integration of social, economic, and ecological factors.

Applegate Partnership

The Applegate Partnership was created in 1992 to develop a collaborative approach to land use management involving private land owners and natural resource agencies. Partnership members include environmentalists, timber industry representatives, agricultural interests, and a range of community members from the watershed. Partnership members work on community approaches to land use planning through the interconnected perspectives of ecology, economy, and society. The vision of the Applegate Partnership reads as follows:

> The Applegate Partnership is a community-based project involving industry, conservation groups, natural resource agencies, and residents cooperating to encourage and facilitate the use of natural resource principles that promote ecosystem health and diversity. Through community involvement and education, this partnership supports management of all land within the watershed in a manner that sustains natural resources and that will, in turn, contribute to economic and community stability within the Applegate Valley.
> (Applegate Partnership 1997, 1)

The Applegate Partnership's action plan emphasizes restoration activities that create jobs for watershed residents. Priority is also given to landscape scale projects which will improve overall watershed conditions. In 1994, federal agencies left the Partnership as full participants, but clarification of the relationship between agencies and the Partnership resulted in improved communication and collaboration with federal agencies. At this point, the Applegate Partnership adopted the new role of Applegate River Watershed Council and began publishing the Applegator, the only valley-wide newspaper (Applegate Partnership 1997).

Rogue Institute for Ecology and Economy

The Rogue Institute for Ecology and Economy (Rogue Institute) was founded to bring together ecology and economy in ways that strengthen communities (Priester 1994). Major areas of work at the Rogue Institute include community forestry, job training, landscape analysis, market and social research, and strategic and futures planning

(Priester 1994). As the Institute is located in southern Oregon, the majority of projects have been focused on communities and resource issues in the Klamath Province which incorporates the western portions of southern Oregon and northern California. The Institute conducted a social assessment, Words into Action: a Community Assessment of the Applegate Valley, for the Applegate Partnership, and has assisted with a strategic planning process for the Applegate River Watershed (Priester 1994).

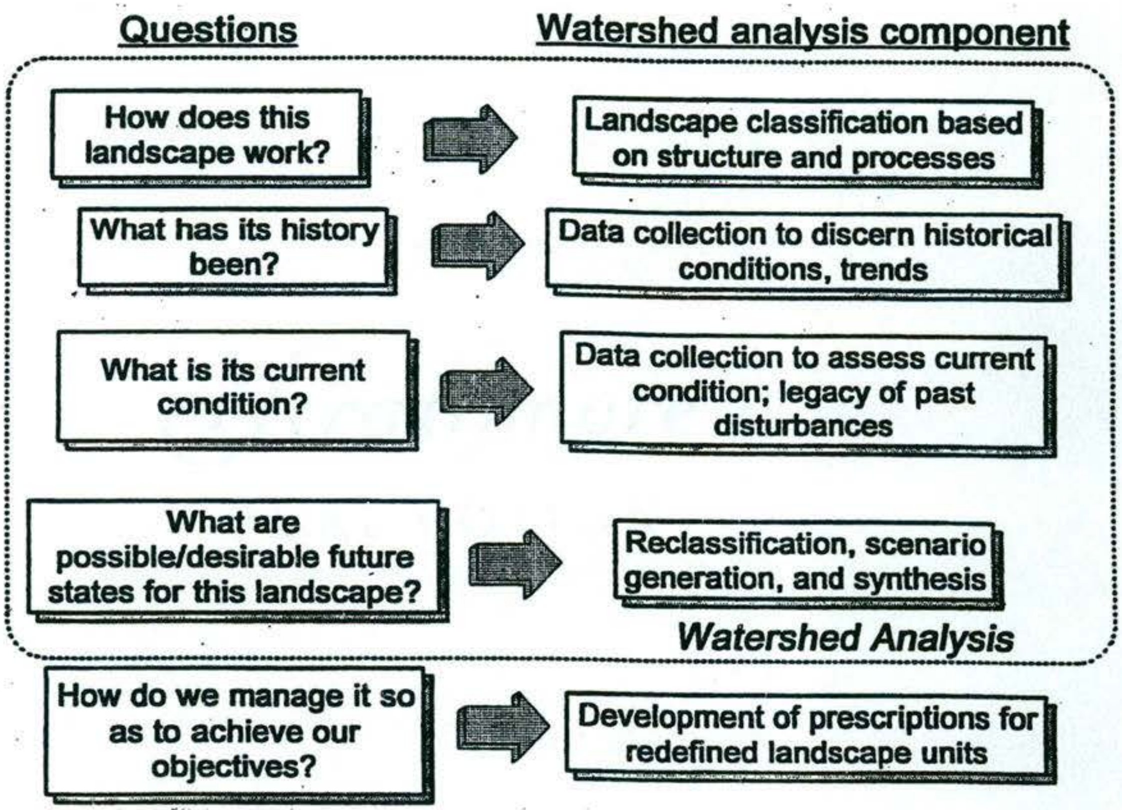
Existing Landscape Scale Projects

Numerous landscape scale projects have been conducted in the Little Applegate Watershed. These include a watershed analysis, guild based wildlife-habitat modeling, and the creation of an alternative future landscape, developed through the Forest Landscape Analysis and Design Process. These projects have completed much of the ground work needed to implement a landscape assessment of risk to biodiversity from future landscape change, and have developed many of the partnerships and cooperators that will be required to successfully apply this information to land use decision making processes.

Little Applegate River Watershed Assessment

As a component of the Northwest Forest Plan, the newly designated Forest Ecosystem Management team (FEMAT) identified a system of key watersheds throughout the northwest (AMA 1995a). The Little Applegate Watershed was named a pilot watershed due to its potential to contribute to implementation of the Northwest Forest Plan in the areas of erosion reduction, fish habitat, pre-commercial thinning, prescribed fire, and commercial timber sales (AMA 1995a). Other significant factors in the selection of the Little Applegate Watershed as a focal watershed are its location within the Applegate Adaptive Management Area and the presence of active community organizations, most notably the Applegate Partnership. The Record of Decision for Amendments to Forest Service and Bureau of Land Management Planning Documents Within the Range of the Northern Spotted Owl and the Standard and Guidelines for Management of Habitat for Late-Successional and Old-Growth Forest Related Species Within the Range of the Northern Spotted Owl identify watershed analysis as an important component of the implementation process of the Northwest Forest Plan. As part of a national effort to develop consistent procedures for watershed analysis, the Little Applegate Watershed was selected as a pilot watershed, and analysis procedures followed those established by federal guidelines. Analysis was conducted as a watershed scale assessment of ecosystem health, and included attention to ecological processes and functions as well as the role and concerns of human communities (AMA 1995a) (Figure 4). In addition to a description of current and historic conditions in the watershed, the Little Applegate River Watershed Analysis identified eight major areas of ecological and social concern, including: water quality and quantity, fish populations, terrestrial biodiversity, commodity production, fire risk, riparian zones, social/economic, and site productivity (AMA 1995a).

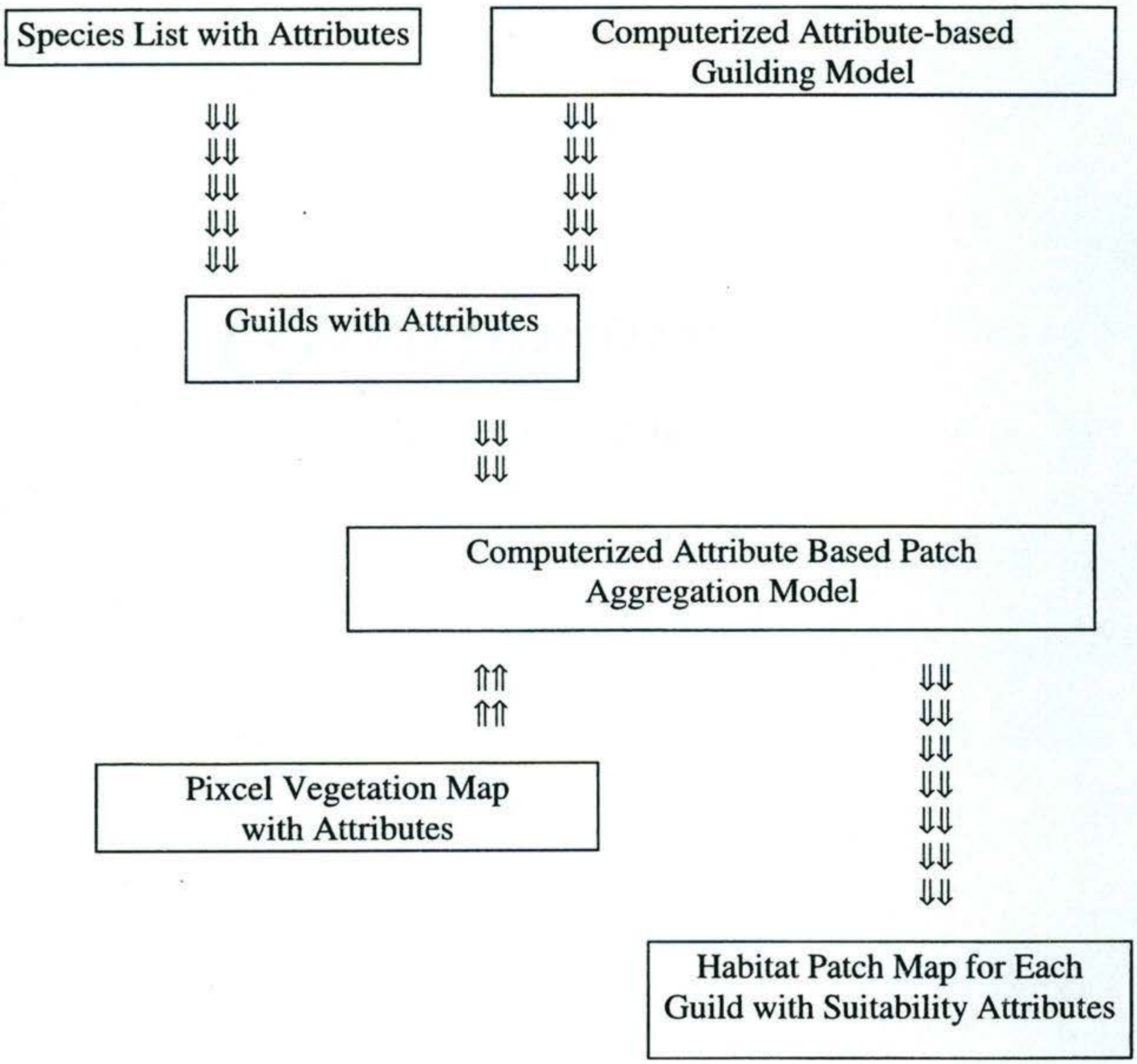
Figure 4. Watershed Analysis Framework



Multiple Species Habitat Suitability Analysis

The Little Applegate Watershed Assessment identified landscape scale changes in habitat area and pattern, and the effects of these changes on terrestrial biodiversity, as one of eight primary issues for the Little Applegate River Watershed. A landscape approach was then employed to look at habitat in the Little Applegate Watershed across all land ownerships in an attempt to investigate questions concerning the effects of human activities on landscape scale habitats and biodiversity. As previous analysis of wildlife-habitat associations had been conducted at the stand scale, a new process was required for investigating habitat suitability across the entire landscape. The methodology selected was based on the *Habscapes* model, originally developed to assess habitat suitability at the landscape scale for the Mt. Hood National Forest, Oregon (Figure 5). The *Habscapes* process involves using a common vegetation and habitat classification to link wildlife-habitat association and life history databases to a spatially referenced vegetation database (AMA 1995c). Computerized attribute based models are used to create guilds and then to define the availability of habitat for these guilds, including amount and spatial configuration. Model input requires attribute lists for species, guilds, and vegetation, as well as life history information on area and pattern requirements of guilds.

Figure 5. Habscapes Conceptual Framework



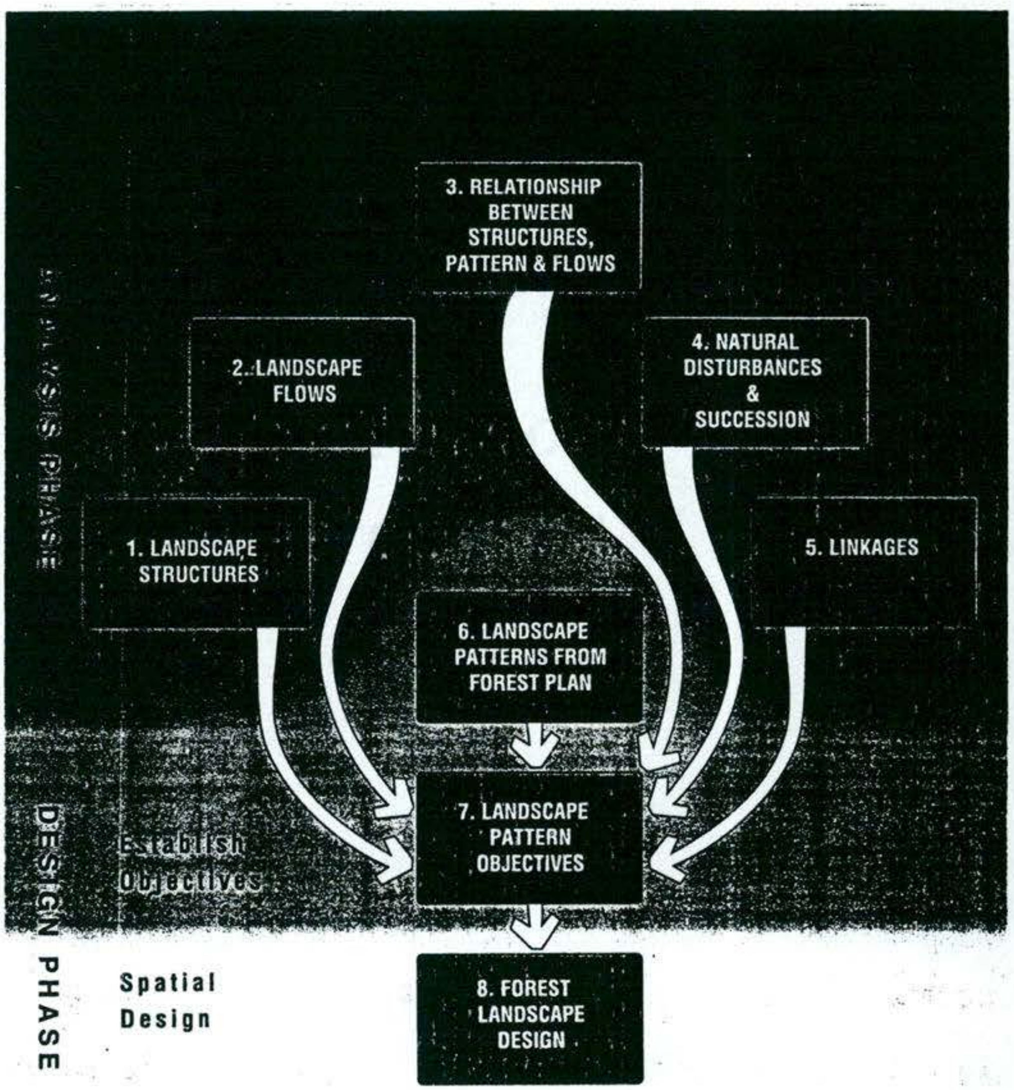
The high number of species involved and the large geographic area being covered precluded a species by species approach, and terrestrial vertebrate species were grouped into three principle groups including terrestrial, riparian, and special habitats for the analysis. Guilds were based on life history attributes including habitat preference, home range size, habitat use patterns, and affinity to special habitats. The 272 terrestrial species known or suspected to occur in the Little Applegate Watershed were grouped into eighteen guilds (AMA 1995a). Once the groupings were complete, patch aggregation models were used to analyze habitat suitability for guilds at the landscape scale. These models constructed patches of suitable habitat from the base vegetation map based on the variables of habitat use pattern, home range size, and seral stage preference of each guild. Three patch models were used to include the habitat use patterns 'patch', 'mosaic' and 'contrast' to better describe the patterns of use by individual guilds (AMA 1995c).

Analysis of the landscape using the *Habscapes* model indicated that less than fifteen percent of all watershed acreage met the habitat requirements for nine of the eighteen guilds. These nine guilds are composed primarily of species dependent on riparian habitat (n=4), late-successional habitat (n=3), and edge habitat between early and late successional habitat (n=2) (AMA 1995a). This modeling approach, although very coarse in its representation of individual species, provided a landscape scale assessment of habitat available for biodiversity and indicated habitat types and guilds that may warrant further investigation. Guilds and habitats identified as 'at risk' by the *Habscapes* modeling process could be used as a starting point for further modeling of risks to biodiversity in the Little Applegate Watershed. Modeling of individual species from these guilds, or selected indicator species would refine habitat evaluation as many factors are averaged in the guilding process used by *Habscapes*, and would also aid in the evaluation of the *Habscapes* modeling process itself.

Forest Landscape Analysis and Design

Because the technical watershed analysis and *Habscapes* modeling efforts did not adequately provide direction regarding future management of the landscape, the *Forest Landscape Analysis and Design* process was applied to the Little Applegate Watershed. *Forest Landscape Analysis and Design* was selected as it provided a framework for futures planning at the landscape scale that was based on ecological, economic, and social variables, and resulted in a spatially explicit design, not merely a set of written recommendations or guidelines (Figure 6). To apply this process to the Little Applegate Watershed ecological information from the watershed analysis, policy guidelines from national and regional management plans, and citizen participation in the development of a long-term vision for the watershed was incorporated.

Figure 6. Forest Landscape Analysis and Design Framework



The *Forest Landscape Analysis and Design* process was developed by Diaz and Apostol (1992) at Mt. Hood National Forest in Northwestern Oregon to provide a methodology for describing, understanding, and better managing forests as ecological systems at the landscape scale (Pojar et al. 1994). Through the forest landscape design process, which attempts to connect principles of ecology and land management, landscape structures, flows, disturbances and succession, and their relationships and linkages are applied to management objectives to produce a spatial design for the landscape (Diaz and Apostol 1992). A key component of the landscape analysis and design process is the discussion and evaluation of potential futures, in terms of landscape elements, that would result from alternative management scenarios (Diaz and Apostol 1992, Pojar et al. 1994).

Little Applegate Forest Landscape Design

Dean Apostol, one of the original creators of the *Forest Landscape Analysis and Design* framework, was contracted to lead the process for the Little Applegate Watershed. The first step involved defining a set of key issues and common objectives to guide the effort; these were selected by an interdisciplinary team of citizen and agency representatives and stakeholders. The remainder of the process was completed through iterative stages involving three teams of students in a University of Oregon planning studio, a citizen/agency team from the Little Applegate, public meetings to solicit opinions from community members, a technical team, and final design work by Dean Apostol and Bart Johnson, an ecologist and landscape architect in the Department of Landscape Architecture at the University of Oregon. The students worked through the framework to develop landscape designs and received feedback from the community and the citizen/agency and technical teams throughout the process. Designs were reviewed again following their completion by the citizen/agency and technical teams as well as the community to assess how well they met initial objectives and evaluate their ecological, economic, and social suitability. Under the guidance of Apostol and Johnson, a single design was synthesized from the three initial designs based on subsequent reviews, new analysis, and ground truthing. It retained the best characteristics of the three initial proposals, while addressing further issues raised in the design review process. The resulting design exists in map and text form and is intended to provide a guide for landscape scale management in the Little Applegate Watershed. The design is intended to be usable by all involved in land management in the watershed, as it is based on existing land use policies and regulation, ecological appropriateness and potential, and social and economic visions of a sustainable landscape. Moreover, the design is expected to be modified over time, as new understandings of the watershed develop. The Little Applegate Forest Landscape Design is an attempt to chart a course for land management, not necessarily define an endpoint.

The landscape scale work that has already been done in the Little Applegate Watershed, including the watershed analysis and its modeling efforts, and the Little Applegate Forest Landscape Design process, sets the stage for the meaningful application of modeling to land use planning through the evaluation of future scenarios. As the results of the *Forest Landscape Analysis and Design* are primarily based on the potential and desired spatial arrangement of the landscape in terms of vegetation structure and communities, the next steps for long-term biodiversity planning in the Little Applegate Watershed include a more detailed analysis of the design in terms of biodiversity indicators. The wildlife-habitat modeling that was conducted as a component of the watershed analysis in the Little Applegate provides a baseline for comparison of the designed future landscape(s). Through application of the guild-based *Habscapes* process to the spatial configuration and composition of vegetation defined by the Little Applegate Forest Landscape Design, quantitative comparisons can be made between existing and potential future conditions in terms of potential risk to biodiversity through wildlife habitat relationships. A challenge for comparative modeling will be reconciling the different classification systems used in the watershed analysis and the landscape design; a common system for describing vegetation that works effectively within the constraints of *Habscapes*, or another selected model will be required. In addition, a biodiversity modeling approach that includes all species or indicator species may be needed to aid in the assessment of the guild based approach and provide more detailed information for species of particular social or ecological significance. Use of a variety of modeling approaches can strengthen confidence in particular model results and improve understanding of the habitat and landscape variables most significant to terrestrial biodiversity. Another major challenge is the interpretation and incorporation of model results. Efforts to involve and inform all stakeholders early in the modeling process, and to increase comprehension of the ways that modeling can help all stakeholders better understand the potential effects of alternate management practices, must be an essential component of any modeling effort.

CHAPTER III

THE ECOLOGICAL BASIS FOR LANDSCAPE SCALE BIODIVERSITY MODELING

Relationships between habitat and species presence have been studied extensively, and a strong empirical and theoretical basis for using habitat as an indicator of abundance has been established (Flather, Brady, and Inkley 1992, Hansen et al. 1993, Edwards et al. 1996). The analysis of wildlife habitat is inherently spatial as species and habitats do not exist randomly across ecosystems, landscapes, or regions (Garcia and Ambruster 1997). In addition to the variables of habitat type and size outlined by traditional wildlife-habitat associations, applied and theoretical research in the field of landscape ecology has determined a relationship between species persistence and the spatial configuration and context of habitat (Farhig and Merriam 1985, Baker 1989). Ecological modeling provides a means to investigate these complex interactions between habitat and landscape variables and animal populations. As the ability to conduct broad scale, replicated manipulations of landscape variables is severely limited, and even where feasible will take many years to evaluate, biodiversity assessment at large temporal or spatial scales must employ modeling instead of the more traditional empirical methods. Ecological modeling itself has a long history of use in science and mathematics, but predictive models based on wildlife-habitat relationships and the tenants of landscape ecology are relatively new. The most useful applications of spatially explicit models to land management today are in making qualitative statements about the likely relationship between landscape change and population dynamics (Pulliam 1995, Dunning et al. 1995). In these situations, comparative information regarding the direction and scale of species response to habitat alteration, not predictions, are intended model outputs (Dunning et al. 1995).

Wildlife-Habitat Associations

Habitat has long been used as a predictor of animal abundance (Hansen and Urban 1992). Wildlife habitat associations generally emphasize vegetation composition, structure, and/or spatial patterning and represent a scientifically credible and logical approach to conservation. "It has long been commonplace in ecology that plant community structure influences species' distributions and abundance, and more generally the composition of communities" (Holt et al. 1995, 21). Wildlife-habitat associations provide a conceptual link between the research and management concepts of carrying capacity and populations and as a result have a higher potential for application to land use and natural resource decisions than conceptual models based on demography (Schamberger and O'Neil 1986).

Basic to any wildlife habitat association is the assumption that species distributions reflect the physical variables of the habitat and vegetation (Bayer and Porter 1988). Because wildlife distribution and abundance is assumed to be closely tied to habitat, habitat characteristics can then be used to make inferences regarding wildlife populations. This approach further assumes that habitat conditions can be accurately described with a relatively small set of variables. Current debate centers on the relative importance of composition versus structure in determining the distribution of wildlife, as well as the role that scale plays in this relationship (O'Neil et al. 1995). Another complication is the varied response to habitat variables observed in different taxonomic groups; for example, it has been suggested that birds respond more to vegetation structure than to composition (Edwards et al. 1996). While the use of wildlife habitat associations in applied and theoretical ecology is extensive, the strength of the association is strongly influenced by the habitat attributes considered, the scales at which they are measured, and the quality of demographic data used (Hansen et al. 1995).

Life history characteristics and vegetation are the primary variables employed in wildlife habitat associations (Hansen and Urban 1992, Edwards et al. 1996). Habitat approaches to biodiversity conservation are common because of the simplicity of measuring habitat characteristics when compared to the detailed data required for demographic investigations of populations (Hansen et al. 1995). Demographic data is expensive and time consuming to collect and as a result is generally not available for a majority of species. A major limitation of wildlife-habitat associations that lack demographic data is the inherent assumption that the habitat with the most species or the largest area is also the most suitable. Investigations of source-sink dynamics documented that small high quality habitat patches can provide a significant source of individuals which may disperse and support larger habitat areas where mortality may actually exceed the birth rate (Pulliam and Danielson 1991). The role of habitat quality and dispersal are currently underrepresented or absent in most biodiversity modeling efforts; although the significance of these factors is widely accepted, detailed understanding at the level for incorporation into modeling efforts does not currently exist for the majority of species. Spatially explicit population models provide a method for integrating demographic and habitat information into one analysis process; widespread application of these models is limited by the high data requirements and a lack of understanding of the most important variables for inclusion. Further use and research of these models can aid the development of scientific understanding and lead to improved models. Despite these limitations, population viability is often strongly related to area of habitat and to population size, which is also often a function of habitat area (Hansen et al. 1993). Most population dynamic models assume that habitat and resources are homogeneous across landscapes and ecosystems but spatially explicit population models are beginning to investigate the relationship between species persistence and configuration of habitat as well as habitat area (Farhig and Merriam 1994). Until understanding of demographic factors and the availability of demographic data increases, biodiversity modeling approaches based on habitat seem to offer the best alternative for the broad scale application and management of multiple species intended by the landscape approach to biodiversity protection.

Landscape Approach

The conceptual model that underlies landscape scale biodiversity analysis and management is that the biological response to landscape conditions can be explained by an understanding of the species life history characteristics and the character of the landscape (Hansen and Urban 1992, Hansen et al. 1993). Further, it is projected that knowledge of life history attributes and landscape change patterns will enable prediction of future dynamics of biodiversity (Hansen and Urban 1992). These models are fundamentally based on the assumptions of wildlife-habitat associations as it is difficult to establish landscape variables without explicit reference to the habitat requirements of component wildlife (Hansen et al. 1993). The move to model landscape change adds additional uncertainty to the level already associated with wildlife habitat associations. In addition to assumptions regarding species' response to and need for various habitat types and seral stages, spatially explicit models assume a good understanding of the details of the changes in structure, function, and composition of the habitat over time and the interactions between these and other variables (Verner 1986).

Analysis of the life history characteristics of species and the patterning of habitat across the landscape can be used to identify those species that are likely to be sensitive to landscape change and require additional research attention (Hansen et al. 1993). While local studies have begun to model the relationships between biodiversity and landscape attributes reasonable well, broad based applicability of these models is not yet possible. The life histories of species from different regions often differ markedly and as a result species from different geographical regions may also respond differently to a given landscape change (Hansen and Urban 1992). Until there is a basic understanding of the processes that control observed patterns of biodiversity, predictability will remain low and the relationships will remain site specific (VanHorne 1986). This implies that conservation strategies should be designed to fit each region based on the local life history characteristics (Hansen and Urban 1992, Hansen et al. 1995).

As researchers begin to explore the potential of landscape scale models for application to biodiversity protection and management, the need to expand wildlife-habitat associations to include spatial and temporal variables has become apparent. The static approach to habitat suitability assumes that all locations of a given vegetation class or habitat type have equal value for a given species (Lancia, Adams, and Lunk 1986). This approach ignores the potential roles of succession and disturbance through time, and the associations between cover types that result from landscape pattern (Lancia, Adams, and Lunk 1986, Freemark 1995). Resources are not randomly or uniformly distributed across the landscape, thus, species must move through the landscape and are influenced by the spatial configuration of habitat patches and their dispersal capabilities (Pulliam, Dunning, and Liu 1992, Freemark 1995, Hof and Flather 1996, Garcia and Ambruster 1997). While factors such as landscape pattern and connectivity are believed to critical to the maintenance of biodiversity, understanding of the specific role these variables play is not well understood. Research is needed on the role of these spatial and temporal variables as landscape ecologists and planners must identify measurable components of landscapes that have significance to biodiversity.

Defining Biodiversity

The creation of broad classification systems for the purposes of biodiversity protection are relatively recent and their implementation at the landscape scale is limited by a lack of data. The development of a common classification system that is usable by the range of partners involved in landscape scale management is a necessary step toward the implementation of comprehensive biodiversity planning (Haufler, Mehl, and Roloff

1996). Resource managers require accurate and efficient means of assessing habitat suitability for wildlife (Block et al. 1994). While coarse filter strategies are proposed as offering greater hope in the preservation of multiple species, maintaining biodiversity at large spatial scales depends on our ability to accurately classify systems and map species distributions (O'Neill et al. 1995). There are several landscape scale variables that have been empirically shown to explain patterns of biodiversity, these include: "habitat size, isolation, boundary characteristics, patch juxtaposition, and patch diversity" (Hansen and Urban 1992, 163).

Classification systems for biodiversity have relied almost exclusively on the use of maps of dominant vegetation community associations (Hulse et al. 1995). Creating a classification system of grouped vegetation types that are perceived similarly by organisms is a necessary step in expanding the reach of wildlife habitat associations from single species to the level of ecosystem or landscape diversity (O'Neil et al. 1995). As our ability to map and describe vegetation and habitat classes usually exceeds the level of information on species life history attributes, vegetation types must be grouped in a manner that accurately represents known information about species distribution and abundance, as each identified vegetation type probably does not represent a unique wildlife habitat (O'Neil et al. 1995, Edwards et al. 1996).

Structural characteristics of vegetation are standard indices used to associate animal populations with habitat. While these factors have been shown to play a major role in habitat associations for many species, particularly forest birds, the reliance on structure as a surrogate for all factors that may influence habitat selection, especially for non-bird species in non-forest habitats, is questionable (Kellner, Brawn, and Karr 1992). Because species differ widely in their response to habitat variables, the identification of significant variables for the species of interest is critical. Some species are generalists in their use of habitat while others are restricted to a specific habitat type. The scale of investigation also plays a major role in the accuracy of wildlife habitat relationships. If a habitat type is below the resolution of the vegetation map, is difficult to identify with remote sensing, or is combined with another vegetation type in the classification system used, the predicted range of species dependent on that habitat type will not be accurate (Edwards et al. 1996). Amphibians and bats, with their use of small scale habitat features such as caves or ponds, are examples of organisms whose ranges are often over-predicted by traditional species-habitat associations using a broad based vegetation classification (Edwards et al. 1996).

A guild is a group of species that utilize the same habitat in similar ways and respond similarly to changes in habitat conditions (Bayer and Porter 1988). Guilds of species are created based on the definition of shared functional relationships to habitat types through analysis of life history attributes (Hansen and Urban 1992). The validity of guild approaches in wildlife habitat associations is a topic of much debate as the level of uncertainty rises dramatically with the combining of individual species (and their associated uncertainty) and assumptions that guild members are appropriately classified (Bayer and Porter 1988). One of the primary problems with a guild based approach is species that use the same habitat but differ in other life history attributes such as home range size or dispersal characteristics. Guild approaches are thus most useful as coarse filters for identifying potentially sensitive habitats or species, but more refined guilds or the use of single species modeling may be required for additional levels of analysis. Despite uncertainty regarding effects on individual species, guild approaches offer a methodology for a broad-based approach to wildlife habitat and the potential for a better fit with vegetation classification systems. Detailed information for a wide range of species and their habitat requirements are needed for successful implementation of the guide concept to landscape scale management of biodiversity. For most investigations, the adoption of a guild approach will require additional data or the use of a simplistic conceptual model with lower data needs. Indicator species or assemblages of species can also be used in area where detailed species information is lacking, but methods are needed to accurately select and classify indicators (Kremen 1992). Most work with indicator species has relied on the use of one or a few indicator species; an indicator approach to the analysis of biodiversity at the landscape scale will require the use of a greater variety of indicator species.

Uncertainty In Wildlife-Habitat Associations

As the use of large scale and spatially explicit approaches to modeling biodiversity increases, current methods of wildlife habitat evaluation may be inadequate. Most wildlife habitat associations are highly simplified representations of complex relationships and ignore the role of inter or intra specific competition, dispersal, and habitat quality. In addition, the temporal and spatial scales at which they are evaluated are often too small, and most have not been validated with predictions and field studies that link habitat suitability to occupancy (Kellner, Brawn, and Karr 1992, Nelson and Buech 1996). Also, while wildlife habitat associations may imply knowledge regarding species persistence, the variables used in the relationship are not necessarily those most significant to species persistence in an area (Kellner, Brawn, and Karr 1992). Approaches that integrate spatially explicit data with demographic models may offer a solution to this problem, although these models are extremely complex, data intensive, and as a result are often associated with high levels of uncertainty.

Wildlife habitat associations are fundamentally based on correlation and thus their application to cause and effect style predictions is essentially unjustified (Wolff 1995). More information is needed on the processes and relationships that underlie wildlife habitat associations, rather than the current reliance on presence /absence observations with unverified conclusions (Wolff 1995). Field validation of wildlife-habitat associations should be an essential part of any investigation as the accuracy of any model predictions is limited by the accuracy of the wildlife-habitat associations and the relationships used (Lancia, Adams, and Lunk 1986). Despite the numerous assumptions and complications that underlie wildlife-habitat associations, they have been used extensively in ecological research and are effective in evaluating coarse scale trends. The assessment of potential risk to biodiversity from future landscape change represents a research shift to a larger spatial scale, the number of species and habitats examined; in these cases, even general information regarding trends can be extremely useful. In addition, expanded use of biodiversity modeling based on wildlife-habitat relationships should result in improved understanding, and better models, of these relationships.

Landscape Ecology

A landscape can be defined as a heterogeneous area composed of two or more ecosystems, communities, or land uses, and the characteristic patterns determined by these elements. The terms patch, matrix and corridor are typically used to describe landscapes and refer to relatively homogenous units, the most contiguous portion of the landscape, and the connections between similar patches through a dissimilar matrix, respectively (Diaz and Apostol 1992, Noss, O'Connell, and Murphy 1997). Landscape ecology is the investigation of the ecological patterns, processes, and relationships that result from heterogeneity in a landscape (Kotliar and Wiens 1990). Both anthropogenic and natural processes create heterogeneity in landscapes, and one of the most conspicuous spatial phenomena of landscape development is habitat fragmentation, a major threat to global biodiversity (Forman and Godron 1986, Kotliar and Wiens 1990, Noss 1990, Wu and Levin 1994).

While the importance of the spatial pattern of habitats within regions is widely recognized, the underlying processes are poorly understood (Tucker et al. 1997). Thus, investigation of the landscape, its spatial patterns, how they develop, and the relationships between pattern and process are central to the field of landscape ecology (Urban, O'Neill, and Shugart 1987, Flather, Brady, and Inkley 1992, Haines-Young 1993, Wu and Levin 1994). Essentially, landscape ecology hopes to understand the ecological consequences of environmental patchiness (Kotliar and Wiens 1990, With 1997). The extension of ecosystem analysis to include the investigation of spatial relationships in landscapes brings important interactions among physical, biological, and social processes to the field of ecology (Turner et al. 1985). Through an integration of scales and concepts, landscape ecology identifies patterns traditionally ignored in ecological research (Noss 1990).

Landscape structure, or pattern, refers to the number, size, and spatial configuration of landscape patches (Dunn et al. 1992). Landscape composition represents the presence, absence or relative proportions of landscape components (Dunning, Danielson, and Pulliam 1992). Pattern, generated by process at various scales, is the hallmark of the landscape and refers to the identity, distribution, richness and proportions of patch types (Urban, O'Neill, and Shugart 1987, Noss 1990). Landscape metrics that are useful to describing landscape structure in terms of its influence on biodiversity include: the number of patches, patch size, amount of edge, and patch configuration (With, Gardner, and Turner 1997). In pattern studies, data on the presence or absence of species is correlated with spatial characteristics of the landscape (Opdam et al. 1994). While population models investigate demographic processes and estimate persistence rates for species, these studies currently lack an explicit spatial relationship and thus are inapplicable to the landscape design process (Opdam et al. 1994). Many population models, by assuming that patches are equal in size and quality and are equally accessible by dispersers, ignore the significance of spatial data (Farhig and Merriam 1994). Efforts are needed that integrate demographic and population models with those investigating spatial effects at the landscape scale. The primary features needed to describe a landscape in the context of biodiversity protection are the size and composition of habitat types and the spatial arrangement of these habitats (Noss 1990, Dunning, Danielson, and Pulliam 1992, Arnold 1995). Additional factors that may influence populations include: variation in habitat quality, boundary characteristics of patches, the condition of habitat area surrounding the patch, and the overall connectivity of the landscape (With 1997).

The structure of a landscape must be defined in ecologically meaningful ways before the complex interactions between patterns and processes can be understood (Turner 1989). Landscape ecology attempts to discern ecological process through investigations at multiple scales. The relative importance of ecological variables appear to vary significantly with spatial scale and studies have suggested that a relatively low number of variables, when considered at appropriate scales, may be enough to predict landscape pattern (Turner 1989). In landscapes, component patches and disturbances generally occur at characteristic scales, management actions that match these scales may provide a link between landscape ecological theory and application (Urban, O'Neill, and Shugart 1987). To effectively prescribe the scale of management actions based on ecological objectives, research is needed to identify characteristic scales in a wide range of ecosystems and landscapes, investigating structure and function. Researchers must also bear in mind that the scale at which landscape ecological investigations are conducted may dramatically effect study conclusions as patterns and processes critical at one scale may not be significant at another scale (Turner 1989).

A major impact of human caused change at the level of landscapes and regions is the re-scaling of patterns in time and space (Urban, O'Neill, and Shugart 1987). The general rule regarding change events of small/fast and large/slow, does not stand for human induced disturbances which tend to be cumulative and occur over large temporal and spatial extent (Urban, O'Neill, and Shugart 1992). Disturbance can be defined as any relatively discrete event that disrupts population, community or ecosystem structure and causes changes in the physical environment (Turner 1989). Landscape heterogeneity can act to promote or inhibit the spread of disturbance (Turner 1989). The relationship between landscape heterogeneity and natural or human disturbance is difficult to predict, particularly for cumulative effects, and should be investigated further (Turner 1989). In fragmented landscapes, disturbances and lack of disturbances can pose threats to the integrity of the system (Noss 1983). The best approach for managing natural disturbances is to maintain large areas with natural disturbance and recovery patterns intact (Noss 1983). Unfortunately, this is not usually possible in fragmented landscapes. The specific effects of human re-scaling of natural systems may alter the natural set of constraints that govern lower level biological processes, change the degree of interaction among patches , create new boundaries, and render natural systems that incorporate disturbances less effective (Urban, O'Neill, and Shugart 1987). In general, resource management decisions that impact landscape pattern should be scaled to mimic natural patch dynamics to best fit the adaptive capabilities of the local species' (Urban, O'Neill, and Shugart 1987). Landscape planning and exploration of management futures is primarily a question of alternative disturbance trajectories; while the most attention has been paid to the effects of human impacts in altering the direction and quality of landscape change, further research is also needed on corrective and maintenance trajectories to investigate landscape response to restoration activities, and conservation or preservation plans, respectively (Hulse and Melnick 1990).

Ecological Landscape Planning

Loss of biodiversity is a direct result of decreases in habitat areas and changing landscape patterns. Traditional conservation programs, generally initiated for endangered or game species, were not designed at the landscape scale and thus may not provide the habitat heterogeneity that is required by species or ecosystems for long term persistence (Arnold 1995). In addition, biodiversity protection measures often occur in areas with high human disturbance or fragmentation, areas where the natural pattern of the landscape has been replaced with one that may differ significantly in spatial and temporal scale (Arnold 1995). Attention to the tenets of landscape ecology, including the multiple scales of diversity and the preservation of the ecological processes that promote diversity, must be incorporated into biodiversity planning. Interactions between events at different scales, and the relationships between ecological function and structure must also be explored and included in the land use planning and management process.

Extensive research on habitat fragmentation has resulted in widespread acceptance of the significant relationship that exists between structural features of the landscape and biodiversity (Noss 1990). It is now understood that elements of the landscape such as patch size, boundary, homogeneity, and inter-patch connectivity can all play a role in determining the distribution, abundance and persistence of some species (Noss 1990).

Despite knowledge of the variables that control species' responses to habitat fragmentation, landscape scale attempts to model population response to fragmentation in heterogeneous landscapes is limited by the fact that individuals may be utilizing multiple habitats to varying degrees across the landscape (With et al. 1997). In addition, difficulties in defining how organisms perceive patch structure and configuration further complicate the description of relationships between environmental complexity and biodiversity (With, Gardner, and Turner 1997). An organism centered approach to landscape pattern is likely the best way to link species interactions with complex spatial patterns, although it means that data requirements will be high and the modeling effort required will be extensive (With, Gardner, and Turner 1997).

Habitat fragmentation can include: habitat loss that results in an increase in the patchiness of habitat, a loss in area of the remaining habitat patches, or an increase in the distance between patches (Opdam et al. 1994). Fragmentation and connectivity are dependent on the scale at which the species of interest perceive the patchiness of the landscape and connectivity can be quite important for species persistence (With 1997). Functional connectivity is defined using the spatial relationship among habitat patches, the condition of the matrix lands between patches of suitable habitat, and the response of organisms to these characteristics (With, Gardner, and Turner 1997). Species that are good dispersers or generalist habitat users should perceive landscapes as functionally connected across a greater range of fragmentation than species with limited dispersal capabilities or specialized habitat requirements (With, Gardner, and Turner 1997, Dale et al. 1998). Lord and Norton (1990) investigated the effects of scale on the impacts of fragmentation and offered the following general guidelines: ecosystem function is more likely to be disrupted at finer scales of fragmentation; complex systems are more likely to be disrupted at a given scale of fragmentation than ecological simpler systems; the finer the scale of fragmentation the smaller the organism adversely affected; habitat specialists are more likely to be disturbed at finer levels of fragmentation; coarser scales of fragmentation are likely to be used as continuous habitat by organisms with high mobility only (Lord and Norton 1990).

The use of corridors of suitable habitat to connect otherwise isolated patches has received much attention in conservation biology but the success of this approach has not been empirically proven. While ecological theory has demonstrated the necessity of functional linkages between patches, there is a need to empirically evaluate the potential of corridors as functional connections in actual landscapes. In a simulation of possible network configuration, Turner (1989), found that one linkage between habitat patches was responsible for the majority of variance in biodiversity response and that two or more linkages had no significant effect, regardless of the spatial configuration (Turner 1989). A more integrated approach to connectivity, which focuses on functional connectivity via the matrix lands, if possible, may make more sense then the notion of corridors as discrete structures (With 1997).

Analysis of the relationship between fragmentation and species distribution indicates that fragmentation is not a linear function of habitat loss; instead, it is a threshold phenomena (Noss 1983, With, Gardner, and Turner 1997). "Critical thresholds are transition ranges across which small changes in spatial pattern produce abrupt shifts in ecological response" (With and Crist 1995, 2446). Interest in landscape scale thresholds is relatively new and in most cases these thresholds have not been tested empirically (Turner 1989, With and Crist 1995). The identification of critical thresholds in habitat fragmentation and the functional connectivity of habitats would provide significant information for managers regarding the design of landscapes for biodiversity. "Above the threshold, the primary consequence is a reduction in the area of habitat, below the threshold the landscape is effectively disconnected or fragmented, owing to the prevalence of small, isolated patches of habitat" (With, Gardner, and Turner 1997, 168). Research is needed to further investigate the role of critical thresholds in landscape ecology and biodiversity (Turner 1989, With and Crist 1995). Simulation models can be used in addition to field studies to explore the concept of thresholds in landscape ecology. Investigations of ecological thresholds should also occur at scales both smaller and larger than landscapes, as critical thresholds may exist at a variety of scales, or at different scales for different variables.

General guidelines for the protection of biodiversity at the landscape scale have resulted from ecosystem research, population ecology, and landscape ecology. While species, site, or regionally specific adjustments may be necessary, and debate regarding optimal configuration of habitat continues, guidelines do offer a scientifically based method for the application of ecology to land management (Table 3). By focusing on the practical messages needed by landscape planners, emerging principles of landscape ecological research can be used as broad guidelines for the application of ecology to land use management (Karieva and Wennergren 1995).

Table 3. Landscape Ecological Principles for Biodiversity Planning Adapted from Karieva and Wennergren (1995)

Principles from a Review of Spatially Explicit Ecological Models	Management Implications
Species that act as metapopulations live within a threshold requirement for habitat, below which they face inevitable extinction.	Species response to habitat loss is difficult to anticipate with standard census data. Extinction can occur long before all the habitat has been removed.
Destruction of habitat can cause dramatic loss in biodiversity that is long delayed, nonlinear, and conspicuous only after substantial habitat disappearance with surprisingly negligible effects.	Monitoring programs and trend analysis may offer a false sense of security that hides the risks of continues habitat loss.
The details of how habitats are arranged and how organisms move between them can mitigate the above risks.	Management options that emphasize dispersal rates or the deployment of habitat can help to maximize the benefits of conservation efforts.
The mere fact that species interact in a spatial world makes possible an extraordinary richness of possible distributional patterns and population dynamics.	Exactly what patterns or dynamics we observe will depend on how long a time period and over how large an area we collect our data. Local census programs of limited duration may misrepresent the true population dynamics at play and mislead management efforts.
The spread of human populations fractures habitats into relatively isolated parcels and simplifies natural systems.	One of the impacts of landscape change may be the elimination of opportunities for species coexistence through spatial complexity.

Research Needs

A barrier to progress in landscape ecology is the complexity and uncertainty that arise when data collection and analysis are applied to the landscape or regional scale (Haines-Young 1993). Landscape ecological investigations have a higher level of uncertainty resulting from the multiple scales and broad range of inferences that are incorporated. The combination of data from a variety of sources, essentially a given in any landscape scale analysis, raises concerns about the accuracy and compatibility of the data in terms of methods used in collection, the classification system employed, and the scale selected (Dunn et al. 1992). The risk of invalid conclusions can potentially be reduced through careful consideration of the resolution and classification systems used in each study (Cullian and Thomas 1993).

Quantitative methods to define landscape structure are needed to enable comparison of different landscapes, identify changes through time, and relate observed landscape pattern to ecological process and function (Turner 1989). The investigation of change in landscape patterns will require a common classification system that can be applied uniformly over time series (Dunn et al. 1992). The value of any measurement is a function of how landscape units were classified and the spatial scale of the analysis as landscape pattern does not respond in the same way to changes in grain and extent

(Turner 1989). The influence of spatial pattern on ecological processes at broad scales must be studied to better apply landscape ecological research to management decisions (Turner 1989, Pojar et al. 1994). While most land management and use activities have an impact on landscape pattern, the relationships between different elements in a landscape have been traditionally ignored or treated independently by planners, ecologists and land managers (Turner 1989).

There are a variety of technological tools well suited to exploration of the relationship between landscapes and biodiversity, and the number and quality is increasing rapidly, particularly with the advent of geographic information systems (GIS) and the ever increasing capacity of desktop computers. Remote sensing data is also extremely useful in studies of landscape ecology as it enables periodic inventories of the availability of habitats over broad regions, a monitoring step that is often missing due to the high time and resource commitments extensive surveys require (Noss 1990). Quantitative measures of habitat suitability, based on wildlife-habitat associations and applied to remotely sensed data, would permit a broad scale monitoring of temporal changes in landscape biodiversity (Turner 1989). The abilities of geographic information systems and other advances in spatial information systems are currently ahead of conceptual models capable of describing the function of ecological processes at the landscape scale (Haines-Young 1993). Further development of landscape models and future scenarios may provide landscape ecologists with an instrument to evaluate alternative management plans

(Opdam et al. 1994). The continual process of landscape change dictates that a major direction of landscape ecology be focused on the interaction between landscapes and species persistence (Dunn et al. 1994, Barbault 1995).

The use of models is an area of rapidly growing interest in the field of landscape ecology as models can be used to quantitatively describe spatial phenomena at the landscape level, predict change over time in landscapes, and integrate multiple scales (Sklar and Costanza 1992). Conceptual models have meaningful connections to land management and planning as well as the potential to map flows of matter and energy, designate sink and source habitats, predict effects of succession, and determine cumulative thresholds for a variety of human impacts (Sklar and Costanza 1992). While reductionist science is excellent at establishing cause and effect relationships and testing alternative hypothesis in many instances, the complexity of landscape scale issues and ecological processes will require the use of conceptual and simulation models to describe correlation among variables (McCullough 1992).

Ecological Modeling

As simplified representations of real systems, all models incorporate the competing demands of simplicity and realism (Wennergren et al. 1995). Each representation requires that assumptions be made about components and relationships that exist in the actual system. The theories underlying the development of models each contain simplifying assumptions, and while these assumptions are necessary for analysis, model results can be sensitive to changes in them (Soule 1987). Model simplifications are balanced by achieving as good a fit as possible to observed data, while model accuracy depends on the underlying assumptions and the accuracy with which model parameters are selected and estimated (Spellerberg, Goldsmith, and Morris 1991, Jorgensen 1994). Identification of significant factors and correct incorporation of interactions among important parameters is a major challenge to model development but models can also be used to explore understanding of these components (Verner 1986, Jorgensen 1994). Models can provide an opportunity for the synthesis of knowledge from a variety of disciplines or sources within a common framework. The first step of all modeling approaches is the development of a conceptual model, where significant parameters and their relationships are identified based on model objectives and the highest level of data and understanding. The next stage involves translating the conceptual model into a quantitative model to be used in model simulations (Grant 1998). The final stages of model development and application includes testing the validity of model results, conducted with sensitivity analysis. Sensitivity analysis involves varying the range of one variable at a time within the model, to decide if the value employed for that variable is critical to model output (Grant 1998).

Simple models are needed to be useful given the current level of ecological understanding and available data, while increased realism is required for applicability to management (Wennergren, Ruckelshaus, and Karieva 1995). Study objectives must be used to determine whether precision and realism or broad applicability for a wide range of systems is desired, as the goals of simplicity and realism can not be maximized simultaneously (Costanza, Sklar, and White 1990). Model development is a useful exercise as it requires the investigator to explicitly define ecological processes and relationships (Spellerberg, Goldsmith, and Morris 1991). Models are also commonly used in a variety of fields, including increased applications in ecology and conservation planning, as evaluators of specific management decisions. In general terms, models are conceptual tools that assist in the definition of problems, the organization of ideas, the understanding of data, the testing and communication of that understanding, and the making of predictions based upon knowledge learned in the earlier steps of model development (Starfield and Bleloch 1986, Verner 1986, Jorgensen 1994).

Landscape Models

A variety of models are used in efforts to predict changes in landscape pattern and development of landscape models has proceeded from the analysis of single variables in a landscape, to complex models that attempt to simulate multiple and interactive relationships between landscape pattern and ecological processes (Table 4). All models of landscape change include information or data layers describing initial conditions, transition or change functions, and output of model parameters or landscape configuration. Whole landscape models are used to express the value of some variable, such as land use or cover type, across the landscape. Distributional landscape models begin to incorporate spatially explicit information by modeling the distribution of land among a landscape variable, such as land use or cover type. These models have been the most widely applied of landscape models, particularly in the area of land use planning and management, as they output the changing percentages of area in particular land use types. Spatially explicit landscape models include the fate of landscape elements and their configuration, using a set of whole or distributional landscape models as sub-models (Baker 1989).

Table 4. Three General Types of Landscape Models
Adapted from Baker (1989)

Model Type	Type Description
Whole Landscape Models	Models of landscape phenomena, in aggregate, for the landscape as a whole
Distributional Landscape Models	Models of the distribution of land area among classified landscape types
Spatial Landscape Models	Models of the spatial location and configuration of classified landscape types

Investigations of biodiversity at the landscape scale are approached with habitat suitability models, which infer presence or absence of species from habitat area, individual based models of landscape change which focus on the response of species to the spatial configuration of habitat based on life history attributes of that organism, and move up in complexity to process-based models, which attempt to simulate the relationships between landscape pattern and ecological processes significant to species abundance and distribution (Table 5). Individual-based models have the advantage of being relatively simple to validate because the attributes of species and their interactions with the environment can be measured (Huston, DeAngelis, and Post 1988, Baker 1989). Process or population-based models may be more difficult to verify than habitat association models, but if they match well with observed data, they provide more extensive information than individual based models (Huston, DeAngelis, and Post 1988).

Table 5. Three Kinds of Applied Biodiversity Assessment Models

Modeling Approach	Application	Benefits and Limitations
Habitat Suitability Models	Used to infer presence or absence of organisms based on habitat area. Used to determine what habitats are available or needed.	Data needs are low, requiring a habitat association matrix that describes the range of habitat conditions for a particular species. Does not include the spatial variables that may influence species-habitat relationships or demographic data.
Spatially Explicit Habitat Suitability Models	Used to infer presence or absence of organisms based on the amount and spatial configuration of available habitat. Used to determine what habitats and habitat arrangements are available or needed.	Data needs are moderate, requiring a habitat association matrix as well as information on habitat configuration. More realistic than suitability models as it includes the spatial dynamics of species-habitat relationships. Does not include demographic variables that influence, and are influenced by, species-habitat relationships.
Spatially Explicit Population Models	Consideration of species-habitat relationships, spatial configuration of habitat, and population demographics. Used to investigate the response of individuals or populations to landscape change.	Data needs are high as information on species-habitat associations, habitat configuration, and demographic variables are included. Model interpretation is limited as understanding of the relationships between spatial and population variables is uncertain.

Landscape biodiversity models serve a variety of purposes with a range of reliability; while models are extremely useful in guiding empirical efforts, defining future research directions, and exploring relationships, they also have a role in exploring management alternatives (Akcakaya, McCarthy, and Pearce 1995, Turner et al. 1995). Because of the inherent assumptions and uncertainty involved with all models, relative results should be emphasized over absolute results. Models can be used to indicate species or habitats that may potentially be sensitive to future landscape change, or to offer general trend information useful in comparing management alternatives. To be useful to managers, the assumptions, uncertainties, and appropriate uses of models should all be clearly stated (Turner et al. 1995). The development of large scale models is limited by the availability of data needed to analyze landscape and regional scale ecological phenomena accurately, and by a disregard of management application in the planning and design stages. Too often, models are developed and the potential applications are only explored once the model is complete (Flather, Brady, and Inkley 1992).

Spatially Explicit Models

Landscape scale planning for biodiversity requires understanding of the response of actual populations to landscape change; thus, the ability of models to analyze spatial land use and habitat data in relation to species dynamics may ultimately have great utility to the conservation planning process (Murphy and Noon 1992, With, Gardner, and Turner 1997, Noss, O'Connell, and Murphy 1997). Ecological response to landscape change is perhaps the most common question investigated with spatially explicit models. One of the most important advances in landscape ecological research is the linking of geographic information systems with simulation models. Two general types of spatial models are employed to investigate relationships between wildlife populations and habitat conditions. The first describes the ecosystem or landscape in terms of suitable or unsuitable habitat and provides a qualitative look at overall population trends through an inferred wildlife-habitat association, while a second approach describes habitat suitability and also incorporates population dynamics within habitats (Karieva and Wennergren 1995). Of particular use to the management of biodiversity at broad scales are spatially explicit population models which combine demographic models of species with the spatial distribution of habitats and other significant elements in the landscape (Turner et al. 1995, Noss, O'Connell, and Murphy 1997).

Spatial models are useful tools for investigating species response to management at the landscape scale as comprehensive experimental studies of species response to habitat change, especially at the landscape scale, are usually not feasible (Turner et al. 1995). In addition to providing managers with information on the amount and type of habitat needed, spatially explicit population models also address the arrangement of habitats across the landscape (Turner et al. 1995). Unlike habitat association and suitability models, spatially explicit models describe the distribution of species across the landscape and how this distribution responds to changes in the spatial and temporal distribution of habitat (Dale et al. 1998). Like all models, they are only as good as the data upon which they are based, and increased empirical research is also needed to increase understanding of the relationships between landscape pattern and species persistence. Retrospective analysis of species and landscape response to different management scenarios and land use patterns, conducted by comparing historic and current land use and habitat patterns, can provide data on the probable range of possibilities for future management and design of the landscape (Freemark 1995). Demonstrating the ability to model past behavior of a system lends credence to its application to future management scenarios (Costanza, Sklar, and White 1990).

Spatially explicit population models enable exploring the effect of alternative land use strategies on population persistence because they simultaneously consider species habitat relationships, demographics, and the configuration of habitat in the landscape (Turner et al. 1995, Dunning et al. 1995, Noss, O'Connell, and Murphy 1997). For species that use more than one habitat or cover type, simulation of the effects of alternative management scenarios requires incorporation of the spatial relationships among habitat types. Spatially explicit population models allow researchers to simultaneously understand the processes that influence the abundance and distribution of organisms and to predict the effects of management alternatives on these patterns of abundance and disturbance, creating a common framework useful to land managers and ecologists (Turner et al. 1995, Conroy et al. 1995, Noss, O'Connell, and Murphy 1997). Despite clear benefits, the current use of spatially explicit population models has two major limitations in their application to conservation biology; the first is that they are often interpreted too literally, and secondly, in the interest of increasing realism, models are made so complex that data is unavailable and they are impractical to use (Wennergren, Ruckelshaus, and Karieva 1995, Schumaker 1996).

The data requirements of spatially explicit population models are extremely high and assume knowledge of the features that define habitat, accurate life history and demographic data, availability of data at the appropriate spatial, temporal and quality resolutions, and the capability to link species to habitat attributes accurately (Dale et al. 1994). Landscape ecological research often requires combining data from multiple sources, collected for different objectives, and at varying levels of reliability (Turner 1989). Each of these factors introduces assumptions and uncertainty to the model. Because errors are propagated into model predictions, it has been suggested that simplified models would improve the match between model complexity and the quality of available data (Ruckelshaus, Hartway, and Karieva 1997). Conceptually simple models may also have greater potential for incorporation into land use planning. Successful application of models to management requires that decision makers fully understand the analysis that is underlying the model as a decision support system. Overly complex models which are difficult to interpret run the risk of being ignored, or used incorrectly to make absolute predictions.

The appropriate resolution for spatially explicit models changes in response to the life history attributes of the species of interest and the scale of the landscape changes being modeled (Dale et al. 1998). Models have a limited capacity to capture patterns and processes at multiple scales as they generally utilize a single resolution. While a smaller grain size can provide a more accurate representation, it also increases the data handling requirements (Baker 1989). Selection of a large grain size, however, can lead to substantial error if significant variables occur at a smaller scale and are missed by the coarse level of analysis (Baker 1989). Another scale related issue is the type of model used. Landscape modeling requires the definition of landscape features and boundaries. While raster, or cell based models are commonly used, they are not accurate in their treatment of interior landscape elements and linear features, as whole cells are classified according to the presence of small linear features such as roads or streams(Turner 1989, Jin and Wu 1997). As with models themselves, cell based models face the balancing act between increasing the number of cells to improve realism and decreasing the number of cells for computational and conceptual simplicity (Turner 1989, Jin and Wu 1997). Vector based models are better at simulating interior landscape features but are more data intensive and complex to interpret.

The grouping of ecosystem data is often required in landscape and regional models, and a system of aggregating fine scale components to enable prediction of coarse scale properties is often built into the model (Rastetter et al. 1992). While data in landscape models typically comes from a wide range of sources and is often collected with different objectives and at multiple scales, a characteristic scale must be selected for the model. As elements in the landscape have been shown to occur at characteristic scales, selection of model parameters and grain and extent should be connected, so that the model achieves the best 'fit' to the scale of interest (Urban, O'Neill, and Shugart 1987, Baker 1989). In addition to characteristic temporal and spatial scales for a variety of landscape patterns and processes, landscape ecology has also identified a hierarchy of organization among many landscape variables (Urban, O'Neill, and Shugart 1987, Miller 1994). In response, the collection and analysis of landscape and biodiversity data should be guided by the scale and organization of spatial variables (Miller 1994). Interpretation of model results should occur at a scale appropriate to the scale of data, the model, and the hypothesis being investigated (Turner 1989, Baker 1989). A common cause of failure in the application of ecological models to land use planning decisions is the misalignment of scales between model development and the scale of application.

Multiple species modeling further complicates the selection of model resolution; grouping of species into guilds based on functional habitat area offers one potential solution to this issue. Grouping species by life history attributes and scale relationships may enable broader application of model results to conservation planning (Conroy and Noon 1992). While guild classification is often simplistic, it does provide a methodology for the modeling of broad scale variables (Acevedo, Urban, and Ablan 1995). More realistic guild or indicator species models will depend on increased information on individual species behavior and their relationships to habitat at the patch and landscape scale, and increased resources for modeling efforts. While the use of guilds or indicator species are useful to managers in that they simplify models, a high level of certainty is required in the selection of guilds or indicators to avoid huge errors in model predictions (Silbaugh and Betters 1997). Single species models can be used to provide more detailed information where input data is available and to test the accuracy of guild based models.

Wildlife Data

A limiting factor in the application of spatially explicit population models is the large number of model parameters and assumptions that are a direct cost of increasing realism (Dunning et al. 1995, Noss, O'Connell, and Murphy 1997). As with all models, accuracy of results depends on the quality of the data available (Noss, O'Connell, and Murphy 1997). While broad scale data on vegetation and habitat types is usually available, species data, including demographics, dispersal characteristics, and life history information, is often incomplete and difficult to obtain and verify (Noss, O'Connell, and

Murphy 1997, Ruckelshaus, Hartway, and Karieva 1997). Projection of the effect of changes in habitat at the landscape scale requires that the model include an explicit relationship between population demographics and landscape structure (Dunning et al. 1995). Definition of these interactions demands considerable data and assumes a good understanding of species-habitat relationships at multiple scales. Data on effects of habitat alteration and fragmentation on amphibians, reptiles, and even mammals is needed, as most empirical investigations of the relationship between habitat structure and populations have involved birds (Verner 1986). Large gaps in species specific data, for all but a few well studied organisms, is the major limitation to the current utility of spatially explicit population models (Schumaker 1996, Noss, O'Connell, and Murphy 1997).

Definitions of habitat connectivity in spatially explicit population models should ideally be based on species dispersal characteristics as the relationships between species life history attributes and landscape pattern are largely defined by dispersal (Ostfeld 1991, Schumaker 1996). Most models ignore dispersal due to the lack of reliable data and the complexity required in obtaining it. Sensitivity analysis of model simulations has indicated that small errors in species data, particularly regarding dispersal capabilities, resulted in much larger reductions in predictive capabilities than errors in habitat and vegetation classification (Pulliam, Dunning, and Liu 1992, Wennergren, Ruckelshaus, and Karieva 1995, Ruckelshaus, Hartway, and Karieva 1997). In general, modeling of species interactions with habitat dynamics through dispersal and habitat selection and quality variables has proven tenuous at best (Walters 1992). A major factor in the lack of understanding of significant dispersal attributes is the fact that species interact with habitat and the landscape across a variety of spatial and temporal scales. However, the continued creation of models that include dispersal variables may be one method to help direct research towards a better understanding of the underlying 'rules' that guide species response to landscape change (Walters 1992). In addition to improved understanding and data on dispersal, other variables related to fragmentation and connectivity such as specialized habitat requirements, source-sink habitats, and edge effect must become a research priority and be incorporated into spatially explicit population models (Dale et al. 1994).

Vegetation Data

Compounding the uncertainty created by assumptions regarding animal life history attributes, spatially explicit models also embody assumptions about the structure and dynamics of vegetation (Holt et al. 1995). Because the structure of plant communities has been shown to be important for many species, the linkage of population and vegetation models offers a fairly straightforward method for the development of complex landscape scale models (Farhig and Merriam 1985, Holt et al. 1995). Attention to the spatial and temporal scales needed for accurate analysis incorporates both vegetation and wildlife populations. Grain size, which defines the smallest resolution of the investigation, and extent, determining the largest area covered in the study, typically the landscape or region, should be selected with both animal and plant attributes in mind (Dunning et al. 1995, Holt et al. 1995). Vegetation types are often classified based on remote sensing data. While this approach offers the opportunity to cover broad areas accurately and dynamically, input data for all variables should be defined with similar precision. Wildlife-habitat association data often is uncertain, or operates at a different scale of resolution, than the remote sensing vegetation data used in GIS (Akcakaya, McCarthy, and Pearce 1995, Dale et al. 1998). Animal distributions are generally not known with the accuracy of plant community distributions, and data on change over time in animal communities lags far behind vegetation data in quality and quantity. The number of functional habitats may not directly correspond to the number of vegetation or cover types identifiable through remote sensing, as many species occupy habitats at scales smaller or larger than common vegetation classification systems. In addition to the selection of habitat types at scales appropriate to the resolution of the wildlife and vegetation data and the functional habitat requirements of the species of interest, spatially explicit population models must develop a method (conceptual and/or computational) for the linkage of habitat suitability to landscape characteristics (Garcia and Ambruster 1997). This step is critically important, as problems with linkage may be the overriding factor in determining model reliability (Mayer 1986, Holt et al. 1995). The linkage of vegetation models with animal population models requires corresponding classification systems for the input and output variables of both systems, as well as attention to the scales of significant parameters in each model (Holt et al. 1995).

Model Validation

Model validation is conducted through the comparison of independent observations and model output (Conroy et al. 1995). Estimates of species abundance and distribution predicted by a model can be tested against data on these same variables taken from field surveys, expert opinion, and the literature (Block et al. 1994). The most rigorous test of model accuracy is field validation, which applies an independent data set to the model and measures the degree of correspondence (Hurley 1986, Verbyla and Litvaitis 1989). The usefulness of a model depends, in part, on the extent to which the required parameters can be measured in the field (Pulliam and Danielson 1991). Model design should include quantifiable output to allow for validation of the model (Schroeder and Haire 1993). In addition, testing models should be considered an essential step in model development. One of the most important areas of model improvement needed is in the area of developing methods and standards for comparing landscape change models to data, and in the process learning more about both the model and the data (Walters 1992).

Models are only as good as the data and understanding they are based upon, and better information on long term, large scale relationships between population and spatial dynamics are needed (Soule 1987). While the most desirable test of any model would be the application of independent data with known outcomes, empirical testing is usually not possible for spatially explicit models due to the large scales and complexity involved (Flather and King 1992). Landscape scale experiments and replication are difficult or even impossible, and studies of landscape processes are often based on a specific landscape with limited application to other areas (With and King 1997). Because compatible large-scale data sets are rare, randomly generated patterns can also be used as null models to test model predictions (Baker 1989). Null, or neutral landscape models, are simulation models based on theoretical distributions that are used as baselines in comparisons with observed landscape patterns and processes and other landscape models (With and King 1997). Divergence between model prediction and empirical observation is evidence that other processes not in the model, or represented inaccurately, are operating. When model predictions are not consistent with observations, model assumptions and parameters can be reexamined and revised to improve the model (With and King 1997). If model predictions are consistent with observation, it must be remembered that simulation models do not carry the same weight as empirical evidence, and that model fit does not necessarily mean that the model is correct (McCullough 1992). The use of null models can speed the identification of indicators important to biodiversity at the landscape scale by allowing numerous simulations to be conducted in a relatively short period of time.

Sensitivity analysis attempts to weigh the relative importance of model parameters through the variation of parameter values and observations of model output (Stoms, Davis, and Cogan 1992, Conroy et al. 1995). By running multiple simulations and varying parameter values in isolation and combination, it can be determined if the model is sensitive to the exact values used for certain parameters (Dunning et al. 1995). For example, the accuracy of input data on species home range sizes can be tested with a sensitivity analysis that defines a reasonable range of area requirements and runs simulations with values spanning the estimated ranges (Pulliam, Dunning, and Liu 1992). This information is useful as it identifies potential constants to be used in further analysis and guides the identification of gaps in data or understanding (Dunning et al. 1995). Sensitivity analysis has been used to identify the significance and lack of ecological understanding of animal behavior in the areas of dispersal and habitat selection (Dunning et al. 1995). When applied in the development phase, sensitivity analysis can also be used to identify the resolution of the data needed to achieve model objectives (Stoms, Davis, and Cogan 1992). While sensitivity analysis does not compare the model with an exact measure of realism, it is useful in determining whether or not model output appears valid at a specified level of certainty (Stoms, Davis, and Cogan 1992).

Wildlife-Habitat Associations

Increasing application of spatially explicit population models to management decisions warrants further attention to the fundamental assumptions that underlie all of these approaches. Wildlife distribution and abundance is assumed to be intrinsically tied to habitat, and this implies that habitat characteristics can be used to make inferences regarding wildlife population levels (Flather and King 1992). Habitat is commonly related to populations with models in a deterministic manner, assuming presence of habitat implies the presence of species (Dale et al. 1998). Sensitivity analysis or model validation to assess the accuracy of the wildlife habitat association is required for either approach.

Not all species are habitat limited or impacted primarily by factors related to habitat features; variables such as competition or dispersal may instead play a primary role (Dale et al. 1998). In addition, wildlife-habitat association data are difficult to obtain at any scale and the application of this data to spatially explicit population models often involves the extension of site specific data to a landscape or regional context (Flather and King 1992). Sensitivity analysis can be used to test the affect of variables such as dispersal on model output. Consideration must be given to the fact that significant landscape characteristics, as well as species-habitat characteristics, vary according to species and geographic region (Garcia and Ambruster 1997). Landscape scale models that incorporate large numbers of species may also be limited by reliable habitat-association data for certain species, linkages between species, or between species and habitat, in grouped approaches. While wildlife-habitat relationships offer a useful means for investigating the effects of spatial factors on biodiversity, model output should clearly note the species for which data are weak (Block et al. 1994). The impact of spatially explicit landscape models on the management of biodiversity would be improved by increased quantity and quality of data for a larger variety of taxa over a wider range of biological organization, with attention to regional, not site specific, patterns (Freemark 1995). Application of spatially explicit population models to ecological research or land use planning is limited primarily by the quality and quantity of biological data needed to meet the input requirements of the model (Schumaker 1996).

Model Application

Current utility of spatially explicit population models is not in forecasting, but in the potential for increasing general ecological knowledge of landscape biodiversity through the testing of hypothesis and determination of the types of data required to improve the predictive capabilities of the models (Noss, O'Connell, and Murphy 1997). As landscape change operates at scales larger than ecological research has typically considered, landscape models offer an alternative to conventional population models and field investigations, and provide opportunities for understanding the response of organisms to habitat change at a variety of spatial and temporal scales (Pulliam, Dunning, and Liu 1992, Dunning et al. 1995). In these situations, comparative information regarding the direction and scale of species response to habitat alteration, not predictions, are intended model outputs (Dunning et al. 1995). It is important to keep in mind that models are abstractions of the real world and that while they can help us understand, in general terms, the relationships between spatial complexity and biodiversity, they are most useful as one tool in a suite of strategies used to make decisions (With 1997). Although models are not capable of absolute predictions for management to rely on, it is also true that management decisions must still be made despite uncertainty, and models offer another layer of information on which to base those decisions (Chalk 1986, Walters 1992). Application of ecological research, such as that produced by models, to land management and planning should recognize that the level of uncertainty acceptable to managers is often higher than that required by statistical methods in science (Hurley 1986).

Better integration of science and planning is needed to overcome current limitations to the use of simulation models in land management. Economic and social variables effect landscape patterns through their influence on land use decisions and should be included in landscape models (Turner 1987). Incorporation of the social and economic, as well as ecological, mechanisms driving land use change is needed for models to be useful in a more realistic capacity. Many landscape change models mask the strong relationship between transition probabilities in land use patterns and other non-ecological variables by leaving them out of the analysis (Turner 1987). Successful management of landscapes for biodiversity protection will be as dependent on comprehension of trajectories of change in social conditions as it is on ecological understanding.

Increasing human expansion and economic exploitation in rural areas has created immense pressures on the capacity of landscapes to adequately support biodiversity (Mayer 1986, Soule 1987, Noss, O'Connell, and Murphy 1997). Better understanding of the integrated and long term effects of human and natural disturbances on landscapes is needed, and spatially explicit models offer a method for improving ecological understanding and exploring the results of alternative management scenarios. Attention to the relationship between management and science is also needed to effectively apply new information to management decisions that effect biodiversity. Models have a number of roles in the conservation process, including: applied synthesis of the state of the knowledge, as expressions of the linkages between causes and effects, and as explicit rules for making decisions (Salwasser 1986). Future effort must focus on the potential of ecological research and science to use the insights generated by spatially explicit models to solve practical problems in natural resource management (Karieva and Wennergren 1995).

CHAPTER IV

FRAMEWORKS FOR THE APPLICATION OF LANDSCAPE BIODIVERSITY MODELS TO LAND USE PLANNING

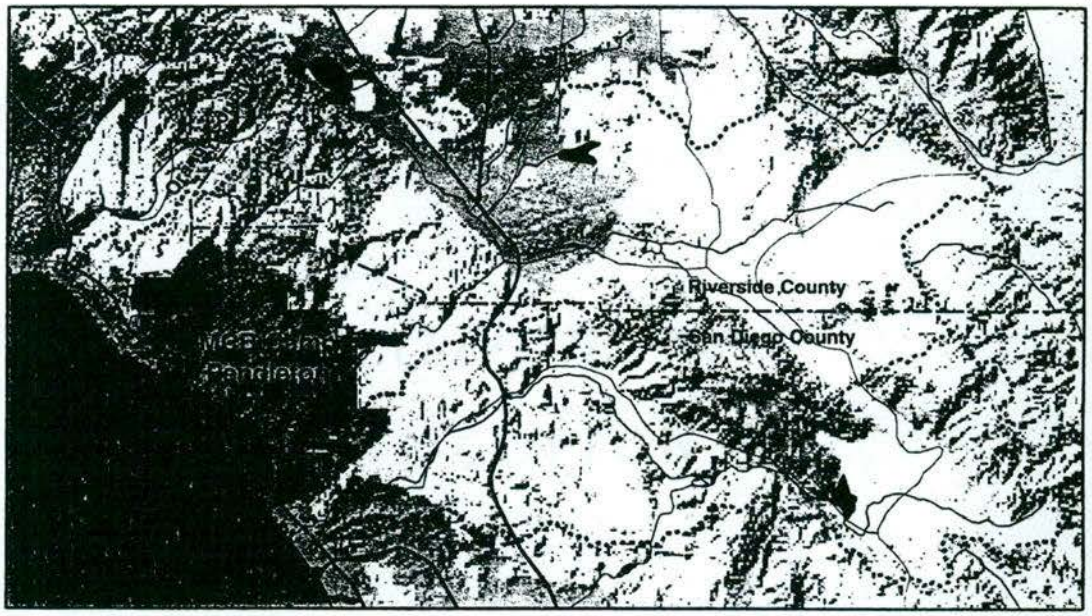
In this chapter I describe three major case studies of the application of landscape scale biodiversity modeling to land use planning; including projects for the Camp Pendleton region of southern California; Monroe County, Pennsylvania; and the Muddy Creek Watershed in western Oregon. All three projects utilized landscape scale ecological models based on wildlife-habitat relationships and attempted to analyze the potential impacts to biodiversity from future landscape change. While each project was conducted with site specific variations in project objectives and assumptions, scale, the biodiversity assessment methodology employed, and the level of citizen participation solicited, all offer insight into the benefits and limitations of using landscape scale biodiversity modeling to assess alternative future landscapes and make decisions regarding land management. Much can be learned through investigation of these major research efforts about the potential framework that will be needed to successfully apply ecological research on risks to biodiversity to land use planning in the Little Applegate Watershed.

Camp Pendleton, California

Introduction

A two-year research program, "Biodiversity and landscape planning: alternative futures for the region of Camp Pendleton, California" was conducted by The Harvard Graduate School of Design, Utah State University, the United States Department of Agriculture Forest Service, the National Biological Service, and the Nature Conservancy; with cooperation from two regional government entities, the San Diego Association of Governments and the Southern Californian Association of Governments (Steinitz 1996) (Figure 7). Concern over land management of the military base after transfer to non-federal ownership was the primary impetus for this project, which focused on the effects of urban, suburban, and rural development on regional biodiversity. Potential land use impacts to biodiversity included direct habitat loss due to conversion to residential homes, streets, and non-native landscapes, and indirect changes in vegetation resulting from changes in the hydrologic and fire regimes (Steinitz 1996). Models employed in the study included soils, hydrology, fire, visual conditions, and biodiversity. Biodiversity was assessed with three separate models which included investigations of the landscape ecological pattern, habitat potential models for select indicator and species of special concern, and regional species richness (Steinitz 1996). Evaluation of alternative futures projected in the research effort were intended to be used by stakeholders in assessing the current land use policies and trends and to provide information with which to devise and compare additional development and conservation scenarios (Steinitz 1996). The multiple approaches used in this project to analyze potential impacts to biodiversity from future landscape change offer insight into the application of landscape scale biodiversity models to land use planning.

Figure 7. Camp Pendleton Region, CA

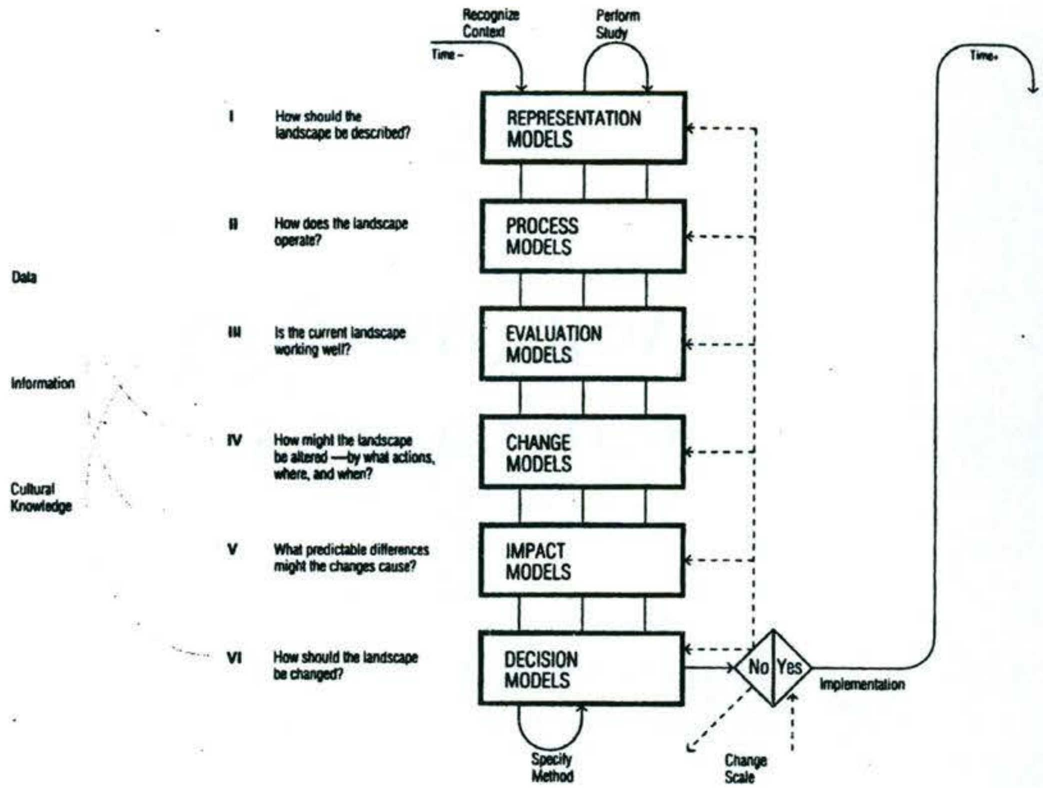


The Region of Camp Pendleton

0 1 3 5 kilometers
0 1 3 5 miles

The Camp Pendleton biodiversity project followed the framework for planning outlined by Steinitz (1990) and included six primary questions, to be investigated in order and then revisited in an iterative manner; in the Camp Pendleton project each question was visited a minimum of three times (Figure 8). The first question in the framework, "How should the landscape be described?" involves the creation of models to represent the landscape. In this study the focus was on available spatially explicit data and representation models were created for soils, vegetation, elevation, annual rainfall, hydrology, roads, land use and public land ownership (Steinitz 1996). The second step of the framework is to define how the landscape operates, and is achieved with process models that describe the relationships between landscape components (Steinitz 1996). The next question, "Is the current landscape working well?" requires the development of evaluation models which are based on an analysis of the current or baseline conditions described in the first two steps.

Figure 8. Steinitz Framework



Once the landscape and its major processes have been described in terms of existing conditions, the questions in Steinitz's framework begin to involve alternative management actions and evaluation of the potential future conditions created by implementation of these different management strategies. The process requires investigators to look at how the landscape might be altered, including elaboration of the potential actions and their spatial and temporal locations. The forth step includes at least two change models in order to be useful to land use planning and management, the first reflects a continuation of current trends and a minimum of one additional change model is developed based on the projected purposeful implementation of an alternative course of action, for example a conservation plan. Researchers then examine what predictable differences might be brought about by the various changes in each of the alternative futures. This involves the use of impact models, which in essence run the landscape process models developed in steps one and two on the proposed futures developed in step four. The final question in the framework is typically the most difficult as it deals with making decisions regarding alternative courses of action. It is important to note that while Steinitz's framework enables a structured look at alternatives, actual decisions regarding the significance of the variety of information gleaned will be the responsibility of those that will influence and be influenced by the direction of future actions; the community and the governments that represent them locally and regionally (Steinitz 1996).

Biodiversity Assessment

The Camp Pendleton alternative futures project was based on the assumption that changes in the landscape ecological pattern of a region can cause a change in the biodiversity of the area, and that land use planning that maintains the landscape pattern may preserve biodiversity (Steinitz, 1996). A variety of modeling approaches were utilized to investigate the connection between habitat at the landscape scale and biodiversity. Habitat potential models were used for individual species identified as state or federal special status species (threatened, endangered, or special concern), those species particularly sensitive to environmental change, and species with adequate data to support such detailed modeling efforts (Steinitz 1996). Species richness was investigated in terms of potential habitat for all terrestrial vertebrates in the study area to describe the spatial implications of biodiversity for the region. Life history attributes and habitat use patterns were linked to the vegetation maps through creation of a common classification system which identified the spatial extent of potential habitat. This approach included many more species than those included in the single species modeling efforts, but the information gained is compiled by habitat types, not specific species and is less detailed in nature (Steinitz 1996). Biodiversity was also investigated in terms of the landscape ecological pattern and its impact on regional biodiversity. This landscape pattern model emphasized the spatial aspects of functional habitat and was based on the fundamental components of landscape ecology: patch, corridor and matrix.

Single Species Potential Habitat Models

Single species habitat models were created based on available information from the literature and personal communication with wildlife experts. All wildlife-habitat relationships used in the models were also reviewed by Camp Pendleton biologists (Steinitz 1996). Species for modeling were selected based on their status as state or federally listed species (threatened, endangered, or special concern), as species particularly sensitive to environmental change, or species with adequate data to support such detailed modeling efforts (typically also sensitive or special status species). In addition, species were selected so that each major vegetation community in the region was represented by at least one single species model and a range of terrestrial taxa were included (Steinitz 1996). The eight species selected for individual potential habitat modeling in this research effort were the arroyo southwest toad, the orange-throated whiptail lizard, the coastal cactus wren, least bell's vireo, the brown headed cowbird, gray fox, mule deer, and the California cougar (Steinitz 1996). Outcomes of the single species models included potential habitat maps by species for current conditions as well as the estimated changes in potential habitat based on the alternative development and conservation scenarios. Because the models were created using available life history and habitat data, they vary in precision by species in relation to the precision of the complied data.

Regional Species Richness

Like the single species models, the species richness model also used life history attributes and habitat use patterns linked to vegetation maps to identify the spatial extent of potential habitat and to compare scenarios. Because the species richness model examined all resident terrestrial vertebrates, the abundance of species was also mapped by habitat type to illustrate the spatial implications of biodiversity throughout the entire region (Steinitz 1996). Instead of describing the species in detail, the regional species richness model is a coarse indicator of the properties of all species associated with a particular habitat type (Steinitz, 1996). This multi-species approach to assessing biodiversity is based on the premise that it is possible to use the large sample size of all vertebrate species to provide general patterns of species richness and response to change, even though there is not a great amount of information for the majority of species included (Steinitz 1996).

Three hundred and forty five species were considered to be potentially present in the regional species richness model for the Camp Pendleton region, which included 234 birds, 62 mammals, 36 reptiles and 13 amphibians (Steinitz 1996). Wildlife habitat relationship models were created to link species to vegetation through a common classification system that grouped the vegetation classes into habitat classes. This step was necessary to justify the quality and quantity of the data used in the models as the information regarding vegetation was of a much finer resolution than that available for the majority of the large number of species included in the richness model. The amount of potential habitat was then mapped for each species based on the relational database that linked species to vegetation communities. This map was also used as the basis for further analysis estimating relative changes in species richness under different development scenarios. Although species richness is a simple measure of biodiversity that ignores factors such as habitat quality or configuration, it can provide a basis by which management decisions can be made at the landscape or regional scale.

Landscape Ecological Pattern

The landscape ecological pattern model for the Camp Pendleton region described the major landscape components in terms of patches, corridors and matrix, and the relationships between these components in terms of their impact on regional biodiversity. The landscape ecological pattern model developed for the Camp Pendleton project required that the landscape contain the following patterns to adequately provide for biodiversity at the landscape scale: 1. large patches of natural vegetation to protect the species richness of the area, 2. connectivity between large patches in the form of corridors or clusters or stepping stones of small patches of vegetation to allow species movement, 3. vegetated stream channels and rivers to enable species movement and provide a number of indirect benefits to terrestrial biodiversity including erosion control, nutrients, and aquatic habitat, and 4. patches distributed throughout the less suitable matrix to provide stepping stone corridors and special habitats (Steinitz 1996). Specific landscape components identified for the Camp Pendleton landscape ecological pattern model were as follows: the current matrix was defined as contiguous areas larger than 500 hectares, patches were defined as areas of natural vegetation of less than 500 hectares, edges were defined as bands between contiguous vegetation patches and disturbed areas, stream corridors and other linear features that may act as ecological connectors were described as corridors, agricultural and military lands were defined as disturbed landscape, all urban lands and roads were classified as built landscapes, and water areas received their own delineation (Steinitz 1996). Evaluation of the landscape patterns of alternative development and conservation scenarios are based on the hypothesis that there are spatial landscape patterns that will conserve the majority of natural processes in any landscape, and that while these patterns will not be all that is needed to protect all species, preserving the spatial pattern will maintain the most important attributes of the system for biodiversity (Steinitz 1996).

Project Results and Management Implications

The direct consequences to biodiversity for all scenarios in the Camp Pendleton alternative futures project were negative. In all cases, although to varying degrees, the landscape ecological pattern was shown to be increasingly fragmented, with natural areas reduced in size and increasingly isolated (Steinitz 1996). Despite the negative trend common to all alternatives, biodiversity assessment of the variety of future scenarios did indicate potential guidelines or general strategies for management of biodiversity at the landscape and regional scale; indicating select species and habitats that may be sensitive to landscape change. Recognition of the ecological importance of landscape and regional habitat conditions to biodiversity, and incorporation of these principles into the land use planning and management process can have a significant impact on biodiversity in the Camp Pendleton region.

The variance in effect on regional species richness that resulted from the particular pattern and type of rural development indicated that the land management practices of individual landowners can impact landscape and regional biodiversity over the long-term (Steinitz 1996). Results from all three biodiversity assessment processes, single-species modeling, regional species richness, and landscape ecological pattern, indicated that major divergence in alternative management scenarios in terms of their impact on biodiversity will occur in the immediate future (Steinitz 1996). This clear short-term impact of the different management scenarios illustrates the need for land use planning to incorporate issues of biodiversity as soon as possible. This will require institutional changes in the local and regional land use planning and management agencies, as well as education of the general population as to the importance of biodiversity (Steinitz 1996). The Camp Pendleton biodiversity assessment makes a strong case for more regional coordination of management and planning in development, as well as conservation. Some of the most significant issues were shown to cross jurisdictional boundaries and indicated that even the large military base cannot be considered in isolation as it affects, and is affected by, conditions of regional habitat and wildlife populations.

Project Evaluation

The relatively high levels of biodiversity, available species habitat information, and increasing development pressures in the Camp Pendleton region made the area a good place to conduct a regional analysis of land use impacts to biodiversity. The creation of a common classification system for all landscape representations made it possible to compare and evaluate current and potential future conditions, and the multiple approaches to biodiversity assessment allowed a relatively high level of confidence given the reliance on existing data (Steinitz 1996). The project was limited by its reliance on available data; while comparatively good for this region, adequate data for biodiversity assessments at the landscape and regional scale is still lacking and influences the detail and accuracy of project models and results. Because project design was based on the existence of available information, significant issues or relationships may have been overlooked or excluded.

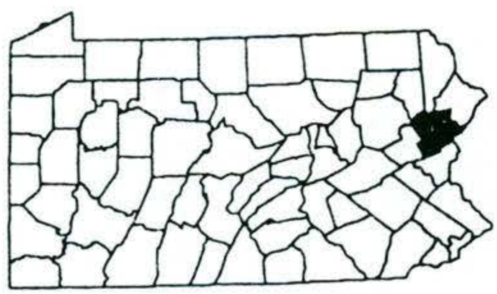
High federal ownership and involvement due to the location of the Camp Pendleton military base provided the level of commitment and support needed to complete a project of this magnitude. Increasing development pressures and growing public awareness of the potential impacts of continued trends also aided the project in securing support from regional government. Despite this, the Camp Pendleton alternative futures project was facilitated primarily by agencies and organizations based outside of the region. The lack of involvement of citizens and local government in the design and execution of the research project will make adoption of project recommendations more difficult as these entities lack both the background information and ownership of the project and its results. In addition to assessing the potential of alternative measures of regional biodiversity and the impacts of a variety of land management strategies on biodiversity, the project was intended to evaluate the role that a modeling approach to landscape planning can play as a basis for planning and negotiation among area stakeholders. The lack of participation of private interests in the project greatly limits the projects ability to comment on its third objective, the potential of landscape modeling to act as a common framework for federal agencies, state and local government, tribes and local interests. Continuation of this project, with attempts to incorporate it into the decision making process of local government and citizens may provide insight into this important question. The large scale of this project may have been a limiting factor in the level of public involvement solicited for this project but regionalization of land use management will require new models for soliciting public participation at the regional scale. Better integration of science, planning, and management, at all phases of land use and biodiversity projects, was recommended in the final stages of this project but not explored fully in the project itself.

Monroe County, Pennsylvania

Introduction

A project to investigate the risks to biodiversity from future landscape change was undertaken for Monroe County, Pennsylvania, an area that forms the core of the Poconos region (White et al. 1997) (Figure 9). Projections of high growth in both seasonal and year round populations led to the implementation of a modeling to analyze the potential effects of changes in landscape pattern on biodiversity.

Figure 9. Monroe County, PA



The objective of the project was to develop quantitative methods for assessing potential impacts to biodiversity from alternative land use management scenarios at the landscape and regional scale. The project included the development of alternative future land use scenarios and the evaluation of species richness and habitat abundance for current conditions and each of the six alternative futures (White et al. 1997).

> The essential components of the approach are: 1. a large, representative sample or enumeration of species; 2. natural history characteristics of the species, specifically their habitat and area requirements; 3. a map of habitats; 4. future landscape alternatives that can be mapped as changes in habitat; and 5. a method to assess potential risk posed by future landscapes compared to the present using summary statistics of changes in species richness and habitat abundance.
> (White et al. 1997, 350)

Carl Steinitz, of the Harvard Graduate School of Design, and the Monroe County Planning Department worked collaboratively to develop the alternative futures used in the project and to compile the species-habitat association and land and habitat class information (White et al. 1997). The alternative futures developed for the project differed in degree and spatial distribution of impact, and represented a variety of ways in which the projected growth in the human population could be manifested on the landscape (White et al. 1997). Two of the alternative futures were based on a continuation of existing land use trends in the County and the implementation of the County's comprehensive plan which was created in 1981 (White et al. 1997). A buildout alternative was created as the most extreme development scenario, representing full development based on current zoning (White et al. 1997). Four other alternatives were created, one to represent a high conservation future, the 'park plan', and three others of intermediate change intended to represent a range of alternatives between development and conservation (White et al. 1997).

Biodiversity Assessment

A habitat classification system of thirteen habitat classes was created for current conditions and the alternative futures, enabling comparison of biodiversity indicators between baseline and future land use scenarios. The study included terrestrial vertebrates, which were grouped by taxa into birds, mammals, and herpetofauna. A total of 231 species were included, representing all species except 8 non-native species and 9 species for which area requirement data was lacking (White et al. 1997). Species and habitat association data were compiled from existing sources and a minimum and maximum range for a breeding or reproductive unit area requirement was assigned to each species based on information in the literature on home ranges, territory sizes, sampled population densities, or dispersal distances (White et al. 1997). Species at risk were inferred from shortages of suitable habitat area and assessed using two indicators: species richness and habitat abundance (White et al. 1997). Both species richness and habitat abundance were assessed in two ways, first by using total habitat area only and then again including breeding area requirements, which included only those habitat areas which met the minimum area requirements, for a total of four methods in the biodiversity analysis (White et al. 1997).

Habitat Abundance

The assessment of habitat abundance for current and future alternatives in the Monroe County project included the total habitat area for each species in each landscape scenario as well a the total number of breeding habitat units, calculated with species area requirements, for each species (White et al. 1997). The ratios of habitat abundance (in terms of both total area and units) were compared between each future landscape and the current conditions (White et al. 1997). Modeled changes to habitat abundance were significant and followed the expected pattern of the greatest losses occurring in the highest development alternatives and the smallest losses occurring in the high conservation alternatives (White et al. 1997). In addition, risks of loss of habitat to amphibians and reptiles were generally greater than risks to either mammals or birds (White et al. 1997).

Species Richness

Species richness in this study was based on the presence or absence of suitable habitat patches. A species was described as extinct if no habitat area or habitat units were present. The ratios of species richness in the six alternative futures compared to current conditions was also computed for each species and summarized by taxa (White et al. 1997). There were little or no changes observed in species richness between the six alternative land use scenarios and current conditions, and changes in species richness among the six future landscapes were also slight (White et al. 1997). This is due, in part, to the analysis method used. For example, the assessment of species richness for total area requires no habitat for a species in order for a species to be recorded as extinct; as each alternative future had at least one pixel of habitat from each class, the risks using this method were zero for all species (White et al. 1997). This method is likely to underestimate extinction as it requires only one breeding area unit of habitat for any species, and one unit is unlikely to sustain any population.

Project Results and Management Implications

Changes between the current and alternative future landscape scenarios indicated substantial changes in the proportions of habitat classes, with the largest changes occurring through increases in residential area and decreases in forest area (White et al. 1997). As anticipated, the buildout alternative future reflected the greatest changes from present conditions, while the park alternative reflected the least amount of change (White et al. 1997). In addition, the designed alternative future landscapes had a lower impact on habitat abundance than any of the alternatives based on existing trends or management plans (White et al. 1997). While there was little change in species richness for any of the alternative futures, habitat abundance did change significantly. The magnitude of changes in habitat abundance varied by alternative scenario as well as by taxonomic group, with the herpetofauna being the most negatively impacted (White et al. 1997).

Project Evaluation

The investigation of systematic and quantitative approaches to assessing risks to biodiversity from future landscape change in Monroe County, Pennsylvania was an excellent follow-up to the habitat classification and alternative futures work that had occurred there previously, enabling a detailed look at biodiversity indicators without increasing the project to an unrealistic scope. The creation of a common classification system for current and alternative futures made it possible to compare and evaluate current and potential future conditions in terms of potential impacts to biodiversity. The general indicators and processes used in this approach required relatively few parameters and are applicable to other biodiversity definitions as well as projects in other regions and at different scales. General limitations in the biodiversity models result from the simplifying assumptions that are needed in order to analyze a large set of species at a large spatial scale. These include the limited number of habitat classes used, species-habitat association information that provides presence or absence data only, area requirements which are constant for species across all habitat types used, and no consideration of context or spatial configuration of the habitat patch (White et al. 1997).

Connections with related projects in the region, which is experiencing high human population growth, helped facilitate the involvement of citizens and representatives from local government and interest organizations in the process of planning and land management. Involvement of local interest groups as well as government in the development of alternative futures, and adaptation of existing plans into alternative futures should have a large impact as local decision makers will be able to evaluate these plans with the same criteria as the other alternative futures. This strategy of participation from the onset may also foster improved local decisions and a higher likelihood of implementation of favored plans.

Biodiversity assessment was based on species-habitat associations, a standard process, especially for landscape and regional scale projects. Attempts to increase the level of refinement in the indicators of biodiversity, beyond simple inference based on habitat presence, illustrates many of the difficulties with assessing biodiversity for high numbers of species over large areas. In addition to potential errors due to the classification of habitat and the assignment of species to habitat classes, habitat is evaluated in terms of presence or absence only, with no indicator of quality. The inclusion of area requirements to both the species richness and habitat abundance biodiversity indicators was a way to increase the accuracy of species-habitat associations and include the spatial configuration of habitat, albeit in a limited way. Habitat area requirements for the majority of species are not well understood and incorporation of them into a modeling effort with such large numbers of species may take away as much accuracy as it is intended to supply. This was mitigated in this case as sensitivity analysis of the area requirements was conducted, which increased the level of confidence in the area requirements used.

As the richness indicator used in the Monroe County biodiversity assessment was derived totally from the habitat abundance information, the assessment of species richness was not qualitatively different than the other primary indicator used, habitat abundance. Species richness using habitat area only (without the area requirement) was meaningless in this effort as the criteria for extinction was zero pixels of habitat and each alternative future and the current conditions included a minimum of one pixel for each habitat class. This extinction criteria may skew interpretation and understanding of results by those not closely involved in the development of the model as it likely underestimates the extinction probabilities by requiring breeding habitat for just one reproductive unit and ignoring additional factors such as fragmentation, isolation, or habitat patch quality. Sensitivity analysis of the number of habitat units required to actually sustain populations would improve the realism of this effort. For models assessing impacts to biodiversity at the landscape scale to be broadly applicable to land use planning and management, the underlying assumptions, and their resulting implications must be clearly understood as well as stated in model documentation. Despite these limitations, which are shared by most multi-species modeling efforts, the approach used in this project allows researchers to begin to assess the risks to biodiversity at the landscape scale with a quantitative analysis of species and their habitat requirements. Further application of quantitative methods to biodiversity analysis of future landscape changes also increases understanding of the benefits and limitations of landscape scale modeling and should result in improved models in the future.

Muddy Creek Watershed, Oregon

Introduction

The Muddy Creek Project was developed to provide an example framework for local communities to use in the creation of alternative land use scenarios for the future of their communities. The Muddy Creek project was funded by the Western Ecology Division Environmental Effects Research Lab of the US Environmental Protection Agency and primary objectives of the project included:

> 1. selecting a representative case study within the Willamette River Basin; 2. characterizing the study area and representing shared community concerns for its future and the natural and cultural factors affecting possible futures; 3. working with people who live in and make their living from the study area to create a spectrum of possible futures that depict conservation and development activities in varying intensities and locations, and; 4. evaluating each possible future for its effect on biodiversity and water quality.
> (Muddy Creek Project Web Site 1997, 2)

The alternative futures developed for the Muddy Creek Project included a high development future, a moderate development future, a future representing the expression of current plans and trends, a moderate conservation future, and a high conservation future (Freemark, Hummon, White, and Hulse 1996). In addition to comparisons of the alternative land use scenarios with current conditions, an alternative scenario was also created to represent the 1850's landscape, based on data from the Oregon Natural

Heritage Program (Freemark, Hummon, White, and Hulse 1996). The Muddy Creek Project included researchers from the University of Oregon, Oregon State University, the Canadian Wildlife Service, and E&S Chemistry Consulting Company, as well as numerous watershed residents and stakeholders (Muddy Creek Project Web Site 1997). The project was conducted at the watershed scale for the 320 square kilometer Muddy Creek Watershed in the Willamette River Basin of Western Oregon (Figure 10).

Figure 10. Muddy Creek Watershed, OR

A major component of the Muddy Creek Project was citizen involvement and one of the first steps was the articulation of the goals and mandates of land managers and community members in the watershed. The objective of this participation piece was to increase understanding of the various existing policies and plans operating in the watershed for both watershed stakeholders and the Muddy Creek research team. Involvement by community members was solicited in a variety of ways. Initially, the research team and the team of students who would be developing the alternative futures held a series of public meetings to gather as much information as they could on the concerns and visions of watershed residents. A second series of public meetings were held to present the summary results from the first meetings to the residents and check with them regarding the accuracy of the team's interpretation of their sentiments at the initial meetings. Through this process, a core team of stakeholders was created from a representative group of interested watershed residents. This group met twice monthly to examine the assumptions underlying the alternative futures being designed by the researchers.

Biodiversity Assessment

The objective of the biodiversity assessment in the Muddy Creek Project was to measure changes in biodiversity from land use changes and the method used compared species habitat area between the present and alternative futures and the pre-settlement past. This modeling approach was based on that developed by Denis White for the Monroe County, Pennsylvania project and has essentially the same data requirements: a common habitat classification system, maps for each scenario (past, present, and futures), a list of resident species, and species-habitat association data. Breeding terrestrial species were used in the assessment and species lists were created with data from the literature, local and regional agencies with wildlife data, and input from watershed stakeholders and local experts. A total of 236 species was used in the Muddy Creek biodiversity assessment, including 204 native species, 14 introduced species, 10 rare species, and 8 extirpated species; representing 14 amphibians, 16 reptiles, 135 birds, and 71 mammals. Habitat classes were created from a cross-referenced system based on modified land cover maps for the current, future and pre-settlement alternatives and wildlife habitat classification systems from the literature. The project used a total of 26 habitat classes, 38 land cover classes, and 22 pre-settlement vegetation classes. Average risk was calculated by species, taxonomic group, native and introduced species, native habitat specialists, and rare and special status species, calculated as the percent change in total habitat area from current conditions (Freemark, White, Hummon, Hulse 1996).

The number of native species at risk varied for each alternative future land use scenario but was highest for the high development future, with lower risk in the moderate development and planned trend futures. Habitat area improved in the moderate and high conservation futures, as well as in the pre-settlement past. This same trend seen for species taken collectively was also observed in the risk trends for birds, mammals, and amphibians, but reptiles showed very little change in any of the alternative futures and the largest improvement in amount of habitat in the pre-settlement past. Average risks for the rare and special status species indicated a correlation between the listed status and the degree of risk. Native habitat specialists showed the same trend of increasing risk with increased development, while native habitat generalists showed little change between any alternative land cover scenario. Introduced non-native species were at risk in all alternative futures, and showed a trend of higher risk in the conservation futures (Freemark, White, Hummon, Hulse 1996).

Project Results and Management Implications

One finding with major implications for land use management was that in these simulations, habitat changes from the pre-settlement to the present are different in magnitude and pattern from those between the current conditions and all alternative futures, with more species habitat change from the present to the future than from the past to the present (Freemark, White, Hummon, Hulse 1996). Another interesting finding was that introduced non-native species, which responded negatively to all alternative futures, showed the highest risk from the high conservation future (Freemark, White, Hummon, Hulse 1996). A total of 41 high risk species were identified through the biodiversity assessment. High risk was defined for this project as those species at risk of losing greater than 50% of their habitat in one or more of the alternative future land use scenarios (Freemark, White, Hummon, Hulse 1996). Results also indicated that rare or vulnerable species were more likely to be high risk in the future than the native species list as a whole; these species also lost the most habitat from the pre-settlement past to present conditions, possibly leading to their current status as rare or vulnerable (Freemark, White, Hummon, Hulse 1996). These impact trends indicated by biodiversity risk assessment in the Muddy Creek Project could aid in the ranking of species of special concern, based on potential future loss of habitat area.

Project Evaluation

This large scale, multiple species approach to biodiversity assessment allowed researchers a closer look at a variety of species groupings; in this case rare and threatened species, native and non-native species, and habitat specialists and generalists. This type of analysis can provide additional information on general biodiversity trends, and may illustrate certain species, groups of species, or habitats in need of additional research or management attention. It can be used to discriminate the effects of land use change on specific types of biodiversity as well as regional biodiversity, and can help inform the decision making process by providing quantitative data for comparison of management alternatives. The inclusion of pre-settlement conditions and an alternative land use scenario enabled quantitative comparisons between habitat area trends for biodiversity from past and future development and land use patterns. Strong correlation between high habitat loss from pre-settlement conditions and species status or rare species supported the accuracy of the pre-settlement mapping.

Species risk was determined by the percentage change in habitat area, and in the Muddy Creek Project, habitat was defined by the sum of all habitat areas with no regard to the spatial configuration of habitat, habitat context, or the area requirements of species. While this approach is useful for projects with high numbers of species and habitat types, like the Muddy Creek Project, further refinement of the habitat criteria in the model, including habitat quality and size data where data is available to support it, will lead to much improved results. Validation of species-habitat models and habitat classes would also increase project accuracy, as would the inclusion of habitat area and demographic data into the extinction criteria, which in this case required a total loss of habitat. This situation underestimated extinction, as illustrated by the inclusion of 8 known extirpated species, which were not identified in the comparisons between pre-settlement and current conditions. Other factors that may have affected this discrepancy in the model include errors in species habitat associations and extinction caused by factors other than loss of habitat area (Freemark, White, Hummon, Hulse 1996). While the evaluation of habitat area could add refinement to the project, population viability analysis would greatly improve the assessment by including persistence probabilities for each species. At this point, most landscape scale biodiversity assessment projects using large numbers of species are unable to attain this level of detail due to severe data limitations for most species including additional life history data and regional context information for each population. As projects exploring the risks to biodiversity from land use changes increase their application to land use planning, users may find the administrative support to expand research and improve input data on species-habitat relationships.

Other project limitations, shared by essentially all landscape scale biodiversity assessments in the compilation of input data, result from the creation of a common classification system, a limited set of habitat classes, and species-habitat association data that is unverified by regional field studies. The use of averages to represent changes in habitat area for species across selected grouping may minimize the errors from input data (Freemark, Hummon, White, and Hulse 1996). Despite these limitations, the biodiversity assessment conducted for the Muddy Creek Project enables a quantitative examination of the risks to biodiversity associated with land use changes and should help facilitate better land use management discussions and ultimately lead to more informed decisions regarding regional biodiversity. Because the project required relatively few parameters despite the inclusion of all terrestrial vertebrate species, it should have value in developing local support for a more comprehensive approach to land use planning that includes consideration of potential risks to biodiversity (Freemark, Hummon, White, and Hulse 1996). In addition, the project is general enough to be applicable to other places and at a variety of spatial scales; additional use and modification of landscape scale biodiversity assessment projects should help refine data input needs and model design, leading to improved biodiversity indicators for determining risks from land use change.

Evaluation of Existing Frameworks

Examination of the three applications of biodiversity risk assessment to the land use planning process provide insight into general conditions and recommendations for a related project in the Little Applegate Watershed. Major themes and outcomes of these three projects are addressed in the following section, including project objectives and assumptions, project scale, biodiversity assessment methodology employed, and the level of citizen involvement. A summary table comparing major components of the projects is provided in Table 6, followed by a more detailed discussion and implications for the Little Applegate Watershed.

Table 6. Framework Comparison: Case Studies of Landscape Scale Biodiversity Risk Assessment

	Camp Pendleton	Monroe County	Muddy Creek
Project Objectives	Impacts of alternative land use scenarios on biodiversity, soils, hydrology, fire, and visual conditions. Alternative futures developed by project team.	Impacts of alternative land use scenarios on biodiversity. Development of alternative futures with local government and project team.	Impacts of alternative land use scenarios on biodiversity and water quality. Development of alternative futures with stakeholders and project team.
Project Scale	10,720 square kilometers	1,580 square kilometers	320 square kilometers
Biodiversity Assessment Methodology	Based on wildlife-habitat associations and existing data. Single species habitat potential models for indicator and special status species, regional species richness for terrestrial vertebrates, landscape ecological pattern assessment. Risk calculated by indicator and special status species, species, and taxonomic group.	Based on wildlife-habitat associations and existing data. Species richness and habitat abundance with habitat area and species area requirements. Species richness and habitat abundance with habitat area only. Risk calculated by species and taxonomic group.	Based on wildlife-habitat associations and existing data. Species richness and habitat abundance with habitat area. Risk calculated by species, taxonomic group, native and introduced species, native habitat specialists, and rare and special status species.
Participation	Participation of stakeholders beyond the project team was low.	Participation of stakeholders beyond the project team was moderate; with local government and plans of community groups included.	Participation of stakeholders beyond the project team was high, and included local residents, government, and a stakeholder project team.

Project Objectives and Underlying Assumptions

The three projects described above were each designed as pilot studies intended to explore the potential to assess risks to biodiversity from future land use change at the landscape scale. Of paramount interest in all projects was the robustness of indicators developed to assess biodiversity for multiple species at large scales. The development of alternative futures was common to all projects and was used for a variety of purposes including citizen involvement and to provide a range of land use alternatives for the biodiversity analysis. The Camp Pendleton and Muddy Creek projects investigated the impacts of alternative land use scenarios on ecological indicators in addition to biodiversity, these included hydrology, soils, fire, and visual conditions in the Camp Pendleton project and water quality in the Muddy Creek project. While the Camp Pendleton and Monroe County projects were developed with biodiversity assessment as the primary objectives, earlier projects in Monroe County and the Muddy Creek project also focused on the development of alternative futures using local involvement. All projects used potential habitat area to reflect biodiversity and biodiversity assessment models were based on the assumption that species distribution and abundance reflect the physical variables of habitat and vegetation.

Project Scale

The scales of the case studies investigating risks to biodiversity from future land use change ranged from the watershed to the regional scale with all projects large enough to be defined either as landscape or regional projects using the standard definitions used in the field of landscape ecology; with a landscape defined as a spatially heterogeneous area, at a scale of 10,000's of hectares and regions encompassing 100,000's of hectares (Pojar et al. 1992). The Camp Pendleton project encompassed the largest area, with five river basins and 10,720 square kilometers included. The Monroe County, Pennsylvania project was of intermediate scale, at 1,580 square kilometers, and the Muddy Creek project included the entire Muddy Creek Watershed, 320 square kilometers. While the size varied considerably in these projects, all were large enough to include a minimum of thirteen habitat classes and between 231 and 345 species of terrestrial vertebrates, allowing investigation of landscape scale ecological patterns and their associated biodiversity.

Biodiversity Assessment Methodology

All three case studies of biodiversity assessment relied on the relationship between habitat and species populations for their analysis and were based on the underlying assumption that changes in habitat area will impact biodiversity at the landscape or regional scale. These projects relied heavily on the literature for development of species lists and species-habitat relationship matrices, and all were conducted for terrestrial vertebrates only. None of the projects included population viability analysis. Although inclusion of demographic information would increase model accuracy, detailed data is generally unavailable for the majority of species in landscape scale projects. While all three projects included multiple indicators of risks to biodiversity, the methodology employed in the Camp Pendleton project provided the most information regarding regional biodiversity as it included detailed models for indicator species and an assessment of the potential impacts of both spatial extent and configuration of habitat to biodiversity. The Monroe County and Muddy Creek projects investigated habitat area but neither project examined the additional effects landscape pattern may have on biodiversity. The major differences in the approaches used in the Monroe County and Muddy Creek projects were the inclusion of a pre-settlement land cover alternative in the Muddy Creek and the use of breeding species area requirements in the Monroe County assessment.

Project Participants

The Muddy Creek project had the highest level of citizen participation in all stages of the project, with watershed residents involved in the development of alternative futures, species lists, and species-habitat relationships through a series of public meetings and the creation of a stakeholder group. The Monroe County project also included area residents in the development of alternative futures, but in a slightly less interactive manner. Instead of involving community members in the development phase of the project, the Monroe County project incorporated information from existing land use documents and plans created by both the County and area interest groups. Camp Pendleton had the lowest level of participation from regional residents, but two regional governmental organizations were involved, albeit in a minor way. The range of citizen participation seen in the three projects is likely strongly linked to feasibility given the scale of the projects, with the smallest project attaining the highest level of community involvement. The levels of citizen participation are also a result of the different overall objectives of the three projects, with Muddy Creek objectives including citizen involvement as well as biodiversity assessment.

Alternative Futures

The Camp Pendleton, Monroe County and Muddy Creek projects all included multiple alternative land use scenarios. Each project included an alternative representing current conditions, as well as a future land use scenario that represented the continuation of current trends and management plans. The projects also each included a version of the extremes, in terms of both high development and high conservation futures. Inclusion of this range of alternatives provides land managers and community members with both quantitative and visual aids in evaluating the impacts of land use change on biodiversity as well as other factors. The Monroe County and Muddy Creek projects involved area residents in the development of alternative futures. The connection to the goals and objectives of area residents in the development of alternative land use scenarios, while accomplished with different methods, should provide both the Monroe County and Muddy Creek projects with a higher likelihood of land use decisions that are based on information gained in the biodiversity assessment than projects that are not based on some level of community input.

Management Implications

Results of all three landscape scale biodiversity assessments indicate a high level of risk to biodiversity from the continuation of existing planning and management trends, and negative trends were evidenced for all alternative futures in all projects, except the high conservation plan in the Muddy Creek project (Freemark, Hummon, White and Hulse 1996, Steinitz 1996, White et al. 1997). Results also indicate that the impacts to biodiversity in the short term future will be significantly different than those that have occurred in the past or will occur in later years of alternative futures, indicating a need to act now with regard to the incorporation of biodiversity concerns into regional and landscape scale land use management (Freemark, Hummon, White and Hulse 1996, Steinitz 1996). Biodiversity assessment results also indicated a general trend of higher risk for herpetofauna and species currently classified as special status or rare (Freemark, Hummon, White and Hulse 1996, Steinitz 1996). The wide range of alternative futures examined in the project also provided insight into the significance of alternative landscape patterns on biodiversity, indicating the need for a regional or landscape scale approach to land management for biodiversity as well as illustrating the impact that the type and spatial configuration of rural development may have on biodiversity (Freemark, Hummon, White and Hulse 1996, Steinitz 1996, White et al. 1997). Overall project results highlighted the importance of including a biodiversity assessment process to land use planning and the value of comparing and creating alternative futures, as designed future alternatives that included any conservation considerations at all proved better for terrestrial wildlife habitat than the implementation of current plans or trends.

Application to the Little Applegate Watershed

The Forest Landscape Design project, watershed analysis, and guild based modeling conducted in the Little Applegate Watershed all share elements with the frameworks used for developing alternative futures in the three case studies examined above. Each of these projects included steps to characterize the landscape elements and the relationships between them and the Forest Landscape Design process also designed an alternative future for the watershed. Instead of creating a range of futures based on widely varying management objectives and examining the potential impacts to biodiversity, the Forest Landscape Design project incorporated multiple concerns into the design of one future land use alternative, created from qualitative technical team review and evaluation of three designs generated by teams of students. The eight steps in the Forest Landscape Analysis and Design process cover many of the same processes as the framework developed by Steinitz (1990), and applied in the Camp Pendleton and Monroe County projects, as well as those used in the Muddy Creek project. Once existing landscape conditions have been defined and mapped with a common classification system for the project area, the next steps of both the Steinitz framework (1990) and the Forest Landscape Analysis and Design process involve exploration of potential alternative futures. In the Forest Landscape Analysis and Design process the landscape pattern objectives from the Northwest Forest Plan, existing policies guiding land use on private lands, ecological objectives determined through analysis of existing landscape characteristics, and citizen participation were used to design an alternative future land use scenario (Pojar et al. 1992).

While similar to the three landscape scale biodiversity assessment projects outlined above, the Little Applegate Forest Landscape Design process did not conduct biodiversity assessments at the level of detail of these projects, and quantitative methods were not employed. Because the process attempted to incorporate all major issues of social and ecological significance, the refinement in analysis for each issue was coarse. What the process did create however, were characterizations for current landscape conditions and an alternative future land use scenario that could be used in the application of a landscape scale biodiversity assessment in the Applegate. In addition, habitat based multiple species modeling using guilds of terrestrial species was conducted for current conditions in the watershed analysis process; results of this process were incorporated into the Forest Landscape Design process. As a result, species lists and species-habitat association matrices are already developed for the Little Applegate Watershed. The watershed analysis also included a coarse level investigation of pre-settlement vegetation conditions, this information could potentially be used with the present and future landscape alternatives to analyze the changes in habitat area between past, present, and future alternatives, as in the Muddy Creek project. Additional work in this area would be needed to achieve the level of information needed to compare historic conditions to future or current scenarios, as historic vegetation was described in a general way and not mapped.

The scale of the Little Applegate Watershed, 292 square kilometers, the number of habitat classes created in the Forest Landscape Analysis Design process and the watershed analysis (eighteen and thirteen, respectively), and the total number of terrestrial species used in the analysis (n=272), all align well with the biodiversity assessment methodology employed in the case studies examined above. Through the Little Applegate watershed analysis and Forest Landscape Analysis and Design process the majority of information fields needed for landscape scale biodiversity modeling to assess risks from landscape change have been compiled. The framework outlined by Steinitz, while typically applied with multiple future land use alternatives, only requires that there be a minimum of two to allow comparisons, representing current conditions and some alternative land use pattern. One current limitation to the application of landscape scale biodiversity modeling in the Little Applegate Watershed is the lack of a common classification system for the past, current, and future landscape alternatives. Another difference was the use of a guild approach in the biodiversity modeling conducted for the watershed assessment in the Little Applegate Watershed. Given the large amount of information already accumulated, the previous modeling effort and geographic information systems mapping of the Applegate Watershed, and the high level of sustained participation by watershed residents and area land managers, it should be feasible to gain support for the development of a justified vegetation and habitat classification system and exploration of an expanded range of biodiversity indicators.

CHAPTER V

SURVEY OF LITTLE APPLEGATE LAND MANAGERS

To gain insight into what local factors could influence the application of biodiversity modeling to land use planning, interviews were conducted with land managers working in the Little Applegate Watershed. The objective of the interview process was to discern if there are issues or trends related to the Little Applegate context that differ markedly from, or are in support of, the general guidelines that arose through other components of this research project. The interview process was also used to help define context specific opportunities and constraints to applying landscape scale biodiversity modeling to land use decision making in the watershed.

Phone interviews were conducted with professionals working on natural resource issues that related to biodiversity protection in the Little Applegate Watershed. Interviewees included representatives of public land management agencies and citizen organizations working on land management issues in the watershed. Interviewees were selected to represent a range of professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed, and respondents included individuals with policy, research, mapping, analysis, planning, and management responsibilities. Interview participants had a wide range of knowledge regarding landscape scale biodiversity modeling and its potential for application to land use planning and management, but all respondents had experience in natural resource science and policy and all had been involved at some level in one or more of the existing landscape scale projects in the watershed. Interview questions covered the potential applications of landscape scale biodiversity modeling, desired model output and confidence, and perceived changes necessary to improve decision making in the Little Applegate Watershed. Seven participants were solicited, and five completed the interview process. Participants included three representatives from federal land management agencies, as well as representatives of two non-profit organizations. Survey results and their implications for research and management are described below. Summarized results are provided below in Tables 7 through 10; see Appendix A: Interview Questionnaire and Responses for specific interview results.

Survey Results

Participant Descriptions

Interview respondents were initially asked to describe their responsibilities related to biodiversity or habitat protection in the watershed, and to answer additional questions in the context of these responsibilities (Table 7). Professional responsibilities of all respondents included a landscape or watershed scale approach to land management issues, and all involved some relationship to biodiversity protection. The shared ecosystem management approach among the range of participants reflects the shift to large scale, holistic planning and management that has occurred throughout the many fields involved with natural resource issues.

The interview results indicate a wide range of perspectives regarding landscape scale biodiversity modeling, as anticipated from the diversity of responsibilities and backgrounds of participants solicited (Table 7). The largest variance among participants was in their level of modeling experience, with representatives of federal land management agencies having the most experience with modeling or the application of model results to land management decisions. All respondents had knowledge of the Little Applegate Watershed Analysis and the Forest Landscape Design, but only those working for the federal land management agencies were participants in either of these processes. A major area of commonality among participants was in the perceived need for improved understanding by a wide range of stakeholders of the limitations and potential applicability of models in the land use planning and management process, and the related need for social and institutional changes to support this broad based understanding as well as implementation of biodiversity modeling efforts at large spatial and temporal scales. The lack of direct participation in modeling efforts by interview respondents, who all share professional responsibilities in natural resource management at large scales, indicates that a major effort will be needed to increase proficiency with modeling and applying model results among resource experts, as well as with a larger stakeholder group.

Table 7. Professional Positions of Interview Participants and Experience with Biodiversity Modeling

Participant Number	Interviewee Experience with Biodiversity Modeling
1	Coordinator for the Applegate Watershed Council. Agency staff from region serve the council as technical advisors and modeling has come up peripherally in discussion but the council has not been directly involved.
2	Silviculturalist for the USFS. Have worked with the Little Applegate Landscape Design.
3	Field administrator for Jobs in the Woods Crew. Rogue Institute involved in social assessment components of watershed analysis and forest landscape analysis and design project. No direct involvement in modeling piece.
4	Project mapping, planning, and analysis for the United States Forest Service. Over the past ten years all analysis work at USFS has incorporated computer assisted GIS; before GIS used overlay maps and looked for patterns. Involved in coarse level modeling done for the LA forest landscape design project. Queries and analysis for habitats for special status species. Through analysis of habitat attempted to be predictive 'What are the habitats that might be required?'
5	Project initiator for Forest Landscape Design. Project design and analysis for the USFS. Over the past ten years the forest service has taken an integrated approach to projects, investigating multiple, not single resources for projects. No direct modeling but inclusion of biodiversity and habitat in wider analysis of project impacts and benefits, bigger picture analysis.

Need for Biodiversity Modeling

Participants were asked to comment on the potential usefulness or applicability of landscape scale biodiversity risk assessment to their work. This included a discussion of general areas where they perceived that landscape scale biodiversity modeling would be most useful, as well as specific model output that they could apply in the context of their professional responsibilities within the Little Applegate Watershed. Four of the five respondents focused on the value of additional information that emphasized large spatial and temporal scales and included multiple species and habitats in the analysis. Rationale given to support additional large scale information included the need to move from a species and site based management paradigm for special status species, improved ability to involve private landowners, need for additional information to evaluate alternative management actions related to the implementation of the Little Applegate Forest Landscape Design, the need to move management and policy to a large scale, long time frame perspective, and the ability to view small scale forest and watershed health projects within the big picture. Biodiversity modeling was seen as one method for achieving these improvements, but the need for improved understanding of significant ecological variables and improved models was also noted as a current limitation to applicability of model results. Noted limitations of landscape scale biodiversity modeling included the lack of high resolution data for select species, information on habitat quality, and integration with other management objectives in addition to biodiversity. Biodiversity modeling was perceived as a one tool for increasing ecological understanding of species-habitat relationships, as significant factors can be explored more rapidly with models than field investigations, although additional empirical research efforts into these variables will also be required. Better integration of existing efforts in the watershed will also facilitate improved information as agencies and organizations can learn from each other and collaborate on large, data intensive projects.

Table 8. Desired Model Output and Potential Application: Interview Responses

Participant	Desired Model Output and Potential Application
1	Not familiar enough with potential outcomes/possibilities to say how models could be used but can make general comments. Would be most useful at larger scale, holistic information more useful, could help council work with private landowners. Any additional information on large scale biodiversity or habitat would be useful.
2	Need some measure of biodiversity, an index which tells us how well we are meeting the needs of a drainage. May need more than a single index; perhaps an index for each group of species, since requirements shift from species to species. Don't know if overall risk assessment is useful as risks are species specific. Would rather have an index for various groups of species. You could then average the results but that may not really be useful. Maybe we have to decide which species we need to manage for.
3	No direct benefit as modeling is long term and large scale and jobs in the woods projects are small and numerous, not necessarily all related to big picture, just go where funding/work is. Need to see efficiency of connecting small projects to the overall landscape design. Plan can be used as a template for work, increase project success through showing connectivity through relation to overall plan. No direct benefit as modeling is long term and large scale and jobs in the woods projects are small and numerous, not necessarily all related to big picture, just go where funding/work is. If managers buy off on the process the use would move down to the on-the-ground level. Plan can be used as a template for work, increase project success through showing connectivity through relation to overall plan.
4	Habitat quality needs to be looked at more, modeling could aid this effort. Habitat needs to be managed for over a larger scale and longer time frame than is currently being practiced, modeling could be applied to this as well. Long term, large scale habitat quality and quantity information. For select species, what are the optimum and lesser quality habitats, distribution, pattern. What are the range of habitats for certain species.
5	Now that we have the Forest Landscape Design, need to know what the impacts to species would be for a variety of implementation scenarios. Many ways to reach designed landscape, effects to biodiversity and habitat could differ largely depending on the approach. Need information on the risks to species from different management trajectories. Also need information on how these management scenarios would impact key flows identified in forest landscape design process. Most useful output would be maps illustrating risk assessment to species, habitats, and flows for a variety of management scenarios that could be used to move vegetation towards the desired future state. How do vegetation patches change over time in the landscape. Transitional views of impacts of alternative management on biodiversity and vegetation.

Model Output

Comments on specific model output closely mirrored those provided above, focusing on watershed, landscape, or regional scale habitat information and improved data on habitat quality and specific groups of species. Specific model output recommended by representatives of the federal land management agencies included an index of biodiversity by groups or guilds of species, more detailed information on habitat quality and the range of habitat variability for various species, and specific information on risks to biodiversity for a variety of potential management trajectories specifically related to implementation of the Little Applegate Forest Landscape Design. Comments on model output from representatives of the non-profit sector included the need for increased simplicity and clarity of output to promote wider understanding among all watershed stakeholders, and inclusion of private lands in modeling efforts. Multiple biodiversity indicators and model structures, such as those outlined in the previous chapter, could be applied in the Little Applegate Watershed to achieve some of the model output goals outlined above. Improved integration of existing efforts and information, particularly the guild based *Habscapes* model, pre-settlement vegetation mapping, and the Little Applegate Forest Landscape Design, can provide much of the input data that will be required for this level of analysis, although increased efforts will be needed to include private lands and a wider range of stakeholders in the process. Specific attention will need to be paid to the character of model output and how it is presented, making sure it is capable of reaching a wider audience.

Model Confidence

The level of confidence required for the application of information from biodiversity modeling to land use decisions varied broadly among respondents, but was directly related to the kind of application perceived by that interviewee. As the use and application of landscape scale biodiversity modeling was perceived by the majority of respondents as just one component of many informing the decision making process, predictability was not a major factor. However, while respondents themselves felt comfortable weighing model confidence when considering how to use model results, most emphasized the need for improved understanding of the limitations of models among all stakeholders. Respondents with more direct modeling experience also noted that the question behind the specific modeling effort drives the entire modeling effort, including the level of confidence needed in the results. Two respondents also noted that the level of risk to specific animal or vegetation species or communities would affect the level of confidence required; with species perceived to require immediate management action for protection requiring less model certainty. All respondents perceived models as useful tools, with the caveat that model results must be interpreted with clear understanding of the underlying data and model limitations. Overall, concerns with model applicability had more to do with the interpretation of results within the broader planning context than with the models themselves.

Table 9. Model Confidence Levels: Interview Responses

Participant Number	Certainty Required in Model Results
1	If general and less accurate, the information would likely be used as a first pass assessment. If necessary, could build more detailed information later. Good understanding, by all involved, of the limitations of models is needed.
2	Generally a confidence of two standard deviations is good in natural resource work.
3	Not as much a question of certainty as of improved mechanisms for education and involving more people in a way that they better understand the process. Increase confidence by including more residents, local organizations in the process.
4	Research/analysis question drives data and output; need to assess the sensitivity of the model to provide the output you are looking for. Certainty or confidence dependent on best available data so acceptable/optimal levels of confidence are not formally defined. Know if the data going in is at a coarse level not to imply too much truth in the output. Need to remember not to be seduced by the maps; not the answer but a tool to improve understanding and ask more and better directed questions. Modeling brings more clarity to the process and gets us to think more analytically, it does not provide specific answers.
5	A Bayesian decision making modeling effort, which includes attempts to integrate smaller models used in the Little Applegate, is helping to evaluate the confidence of modeling efforts in this area. The level of certainty required would be risk based, changing depending on how at risk a species or community is right now in the landscape.

Application of Biodiversity Modeling

On the question of what changes, if any, are needed to improve the application of information from landscape scale biodiversity modeling to land use management decisions, interviewees suggested a broad range of improvements (Table 10). Four out of the five respondents stressed the significance that social factors will play in affecting the successful usefulness of biodiversity modeling for decision making. The influence of social variables on acceptance of, and actions towards, a proposed landscape design were noted by three of five respondents. The two respondents working most closely with private landowners and watershed residents stressed the need for involvement of all stakeholders early in the process, as well as increased education to improve understanding of model strengths and weakness to this same group of stakeholders. From those currently working most directly with modeling and the application of results to management decisions, changes to improve the applicability of model results also focused on the level and kind of information available from models. Three of the five respondents cited the need for clear management objectives with regard to potential future landscapes and the relationship of management activities to this desired future, indicating a need for a comprehensive institutional approach to landscape scale planning for biodiversity in the watershed. Specific barriers noted were the lack of widespread understanding and acceptance of management at such large spatial and temporal scales and the complex patchwork system of land ownership and management policies that exist within the watershed. Despite the shift to a larger scale perspective shared by agencies and organizations working within the watershed, and cooperation on specific projects, substantive integration of policies and management directions has not occurred. Implementation of a desired management trajectory for the watershed will require a higher level of institutional cohesiveness among existing agencies and organizations. This will require the justification of local, regional and even national goals, as federal land agencies have a large impact on management in the watershed; expansion of collaborative landscape scale projects can aid in the development of a united approach to land management in the Little Applegate Watershed.

Table 10. Changes Needed for Successful Application of Biodiversity Modeling to Land Planning and Policy in the Little Applegate Watershed: Interview Responses

Participant Number	Changes Needed to Apply Landscape Biodiversity Modeling
1	A private land component is needed; landowners need to be involved. Understanding of how models work by all affected/involved needs to be improved. Need to be careful with decisions made based on models, limitations need to be recognized and accounted for.
2	I don't know that any changes need to be made. It may be important to have the same level of information concerning stand development across the watershed so that areas could be assessed on an equal basis.
3	Need a concerted education and outreach effort on landscape scale futures planning. Need manager level acceptance to connect goals of small scale, on-the-ground projects to overall design. Could create a system of incentives for conducting work that supports the landscape design to foster adherence at the local level.
4	Need better information of the range of habitats used by species, better background understanding of the natural range of variability. Need clear management goals for the landscape. The primary question in defining management goals is 'what are the range of values you want to manage for?' and this is essentially a social question.
5	Largest changes needed are societal, including the private land ownership system, and land use patterns, as well as changes in attitude by all involved. Difficult to get commitment from agencies, organizations, and individuals over the longer time frame. Need systems in place to promote management over a longer time frame.

Research and Management Implications

Social Factors

Four out of five interview respondents noted the significance that social factors will play in the application of landscape biodiversity modeling in the Little Applegate Watershed. Although the sample size was small, this consistent response by both representatives of federal agencies and local non-profit organizations indicates that limitations to landscape scale biodiversity risk assessment may be more closely aligned with social and institutional conditions than to scientific capability or data limitations. Changes mentioned in the realm of social and institutional conditions in the watershed included the need for:

1. Widespread participation among stakeholders;
2. Coordinated and consistent management objectives;
3. Improved collaboration among and between existing agencies, organizations, and individuals;
4. Improved understanding among all stakeholders of model process and potential application;
5. Inclusion of social and economic factors that affect biodiversity;
6. Better integration with private lands and landowners; and
7. Shift in policy and management to larger spatial and temporal scales.

Improved Models

While most attention on broadening the application of landscape scale biodiversity modeling in the Little Applegate Watershed emphasized social factors, comments were also made regarding the models themselves. Modeling efforts are limited by the low level of understanding of significant ecological variables and relationships needed for model development and the amount, quality, and kind of data that is available. Additional research and data collection will be needed to improve landscape scale biodiversity models; the following data gaps were specifically mentioned in the interviews:

- Improved wildlife-habitat association data for a majority of species;
- Improved information on vegetation, habitat, and wildlife populations on private lands;
- Increased information on factors other than habitat that affect biodiversity; and
- Development and testing of a variety of models and indicators for biodiversity assessment.

Implementation

Throughout the project interviews, respondents commented on the potential roles that the various existing land management agencies and organizations could play in fostering the changes that will be required for successful application of biodiversity modeling information to land use decisions in the Little Applegate Watershed. The Applegate Adaptive Management Area, which is already tackling many of the social and institutional changes outlined above, was identified as having initiated the changes in administrative structure and increased involvement of stakeholders that will be needed to spearhead such an effort. Although the focus of the current processes occurring under the auspices of the Applegate Adaptive Management Area are focused heavily on forest management, many of the overall objectives match those listed above, and the work that is already underway to initiate these social and institutional changes could be applied to a landscape scale biodiversity risk assessment process. In addition, participants in the Applegate Adaptive Management Area process play a major role in the selection and development of research in the watershed, and could promote the development of landscape scale biodiversity modeling as one component of this research.

Roles specific to the different kinds of land management agencies and organizations working within the watershed were also noted in the interview process. Non-profit organizations were identified as potential lead organizations for the expansion of participation by watershed residents and stakeholders and increased inclusion of private landowners and lands. The non-profit organizations were also identified as potential intermediaries between federal agencies and the general public, assisting in improving understanding of modeling by all participants. Federal land management agencies were identified as potential lead organizations for the more technical aspects of improved applicability of biodiversity modeling, including data collection, research on wildlife and habitat, and expanded development of ecological modeling efforts. The federal agencies were also identified as having a primary role in establishing and maintaining a management directive for the watershed that encompassed a longer time frame than most contemporary land management; while the role of local organizations and individuals in helping design and evaluate this management directive was also stressed. Improved cooperation and collaboration between the federal agencies and the non-profit organizations working within the watershed was also identified as a primary need; implementation of a majority of the social, institutional, and scientific capacity changes will be dependent on the level of integration that is achieved between existing institutions. All interview respondents felt that better integration was a possible and positive outcome of landscape scale modeling efforts in the watershed, and that individual policy directives of the various existing organizations had already begun to work towards this goal.

Table 11. Recommendations of Organizational Roles for the Application of Biodiversity Modeling to Land Management in the Little Applegate Watershed

Non-Profit Organizations	Federal Agencies	Non-Profit Organizations & Federal Agencies
Involve more stakeholders	Increase quantity and quality of data	Increase understanding of model process and potential application by all stakeholders
Include private lands and landowners	Develop and test models	Develop and implement a long term management strategy
Improve understanding of social factors significant to biodiversity	Facilitate long term management	
	Improve understanding of ecological factors significant to biodiversity	

Overall, interview respondents identified a need for biodiversity assessment at the landscape scale in the watershed, and recognized the value that modeling could play in this effort. Despite the many improvements that were noted as necessary for the successful implementation of model results to land use planning and management in the watershed, respondents felt that the institutional capacity needed to bring about these changes is already present in the watershed, and that the work of many existing landscape scale projects has begun the process of change. One area noted as particularly significant that has not received much attention at this point is the education of a wide range of residents and stakeholders to improve understanding of the process and potential of modeling. Again, respondents felt that current organizations could achieve this goal, if a common management direction to conduct this work was reached. Interview results indicate that the application of biodiversity modeling to the land use planning process in the Little Applegate Watershed is a desired and necessary next step in the progression of landscape scale planning projects that are underway, but will require a substantial commitment from agencies, organizations, and individuals to implement. The most important factor may be the collaborative development of general policy objectives for the watershed that incorporate past and existing landscape scale projects but also help chart the direction of future research and management towards a landscape approach that would include biodiversity modeling as one component.

CHAPTER VI

RECOMMENDATIONS AND CONCLUSIONS

In this research project I have examined the potential for the application of ecological research to land use planning in the Little Applegate Watershed using landscape scale biodiversity modeling. Review of existing landscape scale projects in the watershed, the ecological basis for landscape scale biodiversity modeling, case studies of applied landscape scale biodiversity risk assessment, and a survey of land managers in the watershed result in a few major categories of recommendations. General guidelines for the application of landscape scale biodiversity risk assessment to the land use planning and management process within these categories are outlined below, followed by a discussion of opportunities and constraints specific to the Little Applegate Watershed context.

Framework for Application

A general framework for the application of landscape scale biodiversity modeling to land use planning and management includes recommendations for ecological researchers as well as those involved with policy decisions regarding land use. In support of the conclusions reached through the interview process with Little Applegate land managers, many of the components of a successful landscape scale biodiversity modeling effort relate more directly to social factors than the level of scientific certainty or model confidence. While improvements in the underlying level of ecological understanding and model accuracy are needed, changes in science with the greatest potential for influencing the application of biodiversity modeling to land use decisions involve the relationship between science and policy, not the science itself. If biodiversity modeling is to become an effective tool in the land use planning process, ecological researchers will need to expand their efforts to better incorporate a wider range of participants throughout the process, to include social variables in their analysis, to offer information in a format useful to managers, and to broaden the potential applications of models. In addition, planners and policy makers, as well as watershed residents, will need to become better informed and involved in the process of quantitatively analyzing landscapes. I have identified six categories of recommendations, which together outline a general approach:

1. Adaptive Management
2. Ecological Guidelines
3. Citizen Involvement
4. Integrating Science and Management
5. Improved Models
6. Additional Research

Adaptive Management

Application of an adaptive management paradigm is supported by a review of existing conditions in the Little Applegate Watershed, as well as by the literature survey, the three biodiversity risk assessment case studies, and the survey of Little Applegate land managers. An adaptive management paradigm is in the process of being applied for approximately seventy percent of the Little Applegate Watershed; those areas falling directly in the Applegate Adaptive Management Area and additional lands managed by federal agencies under the direction of the Northwest Forest Plan. To date, the primary management issue actually receiving applied adaptive management is focused on improving forest health; and a variety of cutting and burning practices are being tested for their effect on moving forest structure and composition towards the natural range of variability for the area. Still in the analysis stage, explorations of alternative management prescriptions that could be used towards implementation of the Little Applegate Forest Landscape Design, are also operating under the adaptive management paradigm. As illustrated in earlier discussions of adaptive management, the framework is well suited to landscape scale modeling efforts as it works well with large scale projects and projects with varying levels of uncertainty. Adaptive management allows for, and even requires, adjustment in research and management direction as understanding increases. This management paradigm may facilitate the resolution of the conflict that typically exists between the level of certainty desired or required by land managers and the uncertainty inherent in science, particularly in the realm of large scale, multiple species, alternative futures modeling. An adaptive management framework can be used to help bridge this gap between science and policy and better integrate social, economic, and ecological factors that affect habitat at the landscape or regional scale. The underlying tenets of adaptive management, which include a rapid response mechanism to changes in social and ecological factors, learning from management actions, and increased communication and cooperation among stakeholders, are all essential components of successful landscape ecological research. Adaptive management, and its capacity for change, may be required in the implementation of projects that occur over large spatial and temporal scales; as new ecological and social knowledge and constraints are almost certain to emerge related to projects of this magnitude.

Ecological Guidelines

Ecological guidelines or principles for biodiversity protection, such as those outlined by Noss, O'Connell and Murphy, and Karieva and Wennergren in Tables 2 and 3, respectively, are needed to guide the direct application of scientific research to the design of landscapes for biodiversity protection. Scientists must define those relationships and landscape elements that are understood as significant to landscape or regional biodiversity and habitat variables, in a manner that is useful to land managers, and participate in their interpretation. Common classification systems for both vegetation and habitat could be developed that would enable better comparison of projects occurring in different regions, for different overall objectives, or at different scales, and could also be designed to help facilitate improved application of information and model output across disciplines. As the success of biodiversity conservation is largely dependent on land use decisions, these decisions need to be based on principles of ecology and conservation biology, and then applied within social constraints. To achieve this end, scientists must play a role in improving communication with a range of land use decision makers as well as with the general public. Land managers have a long history of weighing options in the decision making process, but few examples exist of comprehensive ecological guidelines to direct the application of biodiversity data to the planning process for ecosystems, landscapes, or regions. An ecologically based framework for land use planning and management is needed to include biodiversity and habitat protection consideration in the design of future communities and regions. Interviews with Little Applegate land managers stressed the importance of improving communication and understanding among all stakeholders; the Applegate Adaptive Management Area and existing non-profit organizations working in the watershed have the potential to accomplish this task.

Citizen Involvement

The level of citizen participation needs to be high for successful implementation of desired future alternatives in terms of biodiversity protection. Efforts should be made in each project to have adequate representation from residents and members of government, non-governmental organizations, and agencies, as well as a cross-section of disciplines. A framework for planning is needed that includes both socio-economic and ecological conditions, visions, and capacity. Landscape scale habitat planning must be conducted in an interdisciplinary manner, as land use decisions are inherently based on value decisions which are effected by a broad range of human values throughout communities and regions. In addition, the inclusion of multiple stakeholders and objectives from the onset of a project should result in an integrated project with increased potential of successful implementation.

Results and recommendations taken from the three case studies of applied landscape scale biodiversity risk assessment, as well as the Little Applegate land manager interview process indicate that early involvement of stakeholders and improved understanding of models by all involved are needed for successful implementation of land management designed to achieve desired future landscapes. Improved integration of social and ecological factors will be required to create the social climate needed for acceptance and application of broad scale projects to protect biodiversity and habitats across landscapes or regions.

Integrating Science and Management

Improved communication between scientists and land managers will be required for the application of ecological research to land use decisions. Land managers must be included early in the development and conceptual design phase of projects. Early involvement of land managers can improve their investment in project outcomes and applications, and will also inform the model building process as to the information needs of policy makers. Land managers and other stakeholders can aid scientists in the development of alternative future landscapes, enabling a more realistic comparison of alternative actions. Additional research in modeling is needed that emphasizes the prescriptive capacity of landscape biodiversity models as improved model confidence will enhance the applicability of model output to land use decisions. Landscape scale habitat conservation planning is interdisciplinary and efforts should be made to unify disparate planning efforts into a more comprehensive, efficient, and effective approach. Management level acceptance by agencies and organizations working in the Little Applegate Watershed will be required to apply the results of landscape scale biodiversity risk assessment to land use decisions at broad and project specific scales, as indicated by the Little Applegate interview process, as well as the general literature review and case studies.

Improved Models

Better models are needed before widespread application of landscape biodiversity modeling efforts can impact land use decision making on a broad scale. Confidence in model results is one limiting factor in the application of biodiversity risk assessment to land use planning and management. Improvements are needed in understanding of habitat-wildlife associations and the relationships among landscape scale components to aid in the development of models with higher confidence levels. Currently wildlife data is generally of a lower resolution and confidence than vegetation data, and as landscape ecology is a relatively new field, improved understanding and confidence in significant factors at the landscape scale is needed. Specific data gaps noted in the Little Applegate interviews included habitat quality, improved data for groups or guilds of species, complimentary information on the development of vegetative and habitat conditions, and improved understanding of the natural range of variability in species habitat associations. Also mentioned was the need for information on the effects to species and habitats from a variety of management strategies that might be used to move vegetation communities towards the landscape proposed by the Forest Landscape Design. Increased research and application of landscape scale biodiversity modeling can improve understanding of model components and their interrelationships for future efforts, can provide projects to build stakeholder involvement and education around, and can also be used to explore the range of potential applications for information generated by modeling.

Research Needs

Further research is needed to facilitate model improvement and adoption by land managers, and land managers should be included early in design of models, not just at the output stage as is currently common. Research is needed to further explore the role of demographics, dispersal, and population viability as they relate to landscape scale land use patterns. The Northwest Forest Plan and connections seen in ecological research between wildlife distribution and abundance and landscape scale habitat conditions has effectively shifted land management in the Little Applegate Watershed over the past ten years to a larger scale, ecosystem or regional approach, and additional research at these larger scales is needed to improve management at this scale. Biodiversity research focused on large scale and long term trends is needed to increase the knowledge needed for planning and management decisions. More research into improving confidence and identifying those relationships and variables most significant to biodiversity at the ecosystem or landscape scale are needed for the development of planning guidelines founded in conservation biology. Exploration of significant variables should include research on social and economic, as well as ecological factors. Further research into alternative futures is also needed, particularly in the understanding of the actions necessary for achieving desired future states. Monitoring, an essential component of the adaptive management process, will also be needed to evaluate management actions and improve understanding of significant variables and to enable the adaptive management paradigm to operate effectively.

Application in the Little Applegate Watershed

Like the guidelines outlined above in the general framework for application, attention to the context of the Little Applegate Watershed indicates that the role of social and institutional systems in the watershed will play a large role in the application of biodiversity modeling to land use planning in the Little Applegate. Model interpretation and understanding of model uncertainty by stakeholders was noted as a primary limitation by interview respondents; four out of five land managers interviewed cited a need for changes in social systems to support increased understanding and utility of modeling. While scientific understanding must be increased to improve model realism and utility, it is largely the social systems that will direct broad-scale research and management objectives in the watershed. Specific opportunities and constraints that exist within the Little Applegate Watershed related to the application of biodiversity modeling are discussed below (Table 12).

Table 12. Opportunities and Constraints Related to the Application of Biodiversity Modeling to Land Use Planning in the Little Applegate Watershed.

Opportunities	Constraints
Quantitative biodiversity modeling can provide a common framework for the assessment of alternative futures, needed because of the area's high regional biodiversity, human population pressures, and changing landscape pattern.	Lack of data on potentially significant factors such as habitat quality, the natural range of variability of species habitat associations, and the impacts of alternative management scenarios. Consensus by stakeholders to long term planning that emphasizes biodiversity protection does not currently exist. Overall understanding of model interpretation is low.
Existing ecological and socio-economic assessments and landscape scale projects. Extensive data has been compiled at the landscape scale.	Lack of common classification systems and overall objectives of existing projects. Social assessments and ecological analysis have been largely separate efforts.
High federal land ownership which can provide continuity of management objectives, technical expertise in modeling, and ecological data.	Institutional climate committed to fostering and maintaining momentum for long term, large-scale biodiversity projects not yet in place.
Presence of non-profit organizations working within the watershed on issues of ecological and social significance. Citizens and stakeholders have become involved in landscape scale natural resource issues.	Poor understanding of potential applications and limitations of ecological models by individuals and organizations outside of the federal agencies.
Presence of the Applegate Adaptive Management Area which has fostered the development of partnerships and collaborative projects under the adaptive management paradigm.	Direction of the Applegate Adaptive Management Area is largely focused on forest health in terms of reduced fire risk and improved production; cooperators not currently working on regional biodiversity issues.
Wildlife-habitat association data has been compiled and current habitat availability has been modeled with a guild-based patch model.	The forest landscape design alternative future scenario uses a different habitat classification system than the existing guild based model. Historical data on vegetation communities is not in a spatial format. Data is not available to support individual-based modeling efforts.
Spatially explicit models can help bridge the gap between ecological research and land use decisions.	Demographic data needed for spatially explicit population models does not exist for most species. Land use planners not currently involved with ecological projects.

Three key factors: high regional biodiversity, the identification of landscape scale habitat change as a major threat to wildlife in the watershed, and increasing human population pressures in rural areas, all support the utility that landscape scale biodiversity modeling can play in evaluating future land management activities in the Little Applegate Watershed. The landscape scale work that has already been conducted sets the stage for the meaningful application of biodiversity modeling to land use planning through the evaluation of future land use scenarios. Institutions exist within the watershed that have the capacity to collect data, conduct a landscape scale biodiversity risk assessment, and aid in the application of results to land use decisions. These agencies and organizations also have the capability to promote increased understanding and participation by watershed residents, scientists, and land managers of the potential utility of large scale biodiversity modeling as a tool to inform land use planning. This capacity is largely contingent on whether or not the existing institutions can work collaboratively; landscape scale modeling can provide a basis for this cooperation. Together, these organizations have expertise and experience in adaptive management, ecological modeling, large scale ecological analysis, social and economic assessments, and the implementation and monitoring of education and restoration projects at a variety of scales and levels of public involvement.

While the presence of non-profit organizations and federal agencies working on large scale watershed management has resulted in a high number of social and ecological assessments and projects within the Little Applegate, better integration of the various stakeholder organizations and agencies, as well as their projects, is needed. Barriers to achieving improved integration include the wide range of organizational objectives, financial constraints and capabilities. While efforts have been made to better align the federal land management agencies under the adaptive management area designation, many institutional barriers still persist among and between agencies and organizations. Better communication and collaboration within and between agencies and organizations working within the watershed can increase project compatibility, avoid duplication of efforts, and provide complimentary information. The application of biodiversity risk assessment to the land use decision making process will require improved integration between the socio-economic and ecological variables, and cooperation among existing organizations and agencies is a necessary first step to achieving this integration.

At this point, it is the federal land agencies that have undertaken modeling efforts related to landscape scale biodiversity and habitat issues, and capacity for extensive ecological modeling efforts resides almost exclusively within these agencies. Overall understanding of the potential application and limitations of models in the Little Applegate Watershed is low. The variety of organizations and stakeholders with understanding of modeling, and involvement in modeling, will need to be broadened before results can be applied to the land use planning and management process throughout the watershed. Support of landscape scale evaluation of future land use alternatives will require involvement by more than the federal agencies. Because people, not models, will be implementing landscape change over large areas and long time frames, social and political will, as well as institutional capability, will be needed to maintain management direction. The importance of building societal and institutional support for implementing biodiversity protection projects at spatial and temporal scales larger than historic land management practices can not be overstated. Recent efforts by the Applegate Adaptive Management Area related to improving forest health and reducing fire risk throughout the area have initiated a process and institutional structure for improving participation of watershed residents and stakeholders. This same system could be expanded and applied to the landscape scale biodiversity risk assessment process.

A scientific barrier to the application of landscape scale biodiversity risk assessment is the poor understanding of the ecological variables and relationships significant to biodiversity at the ecosystem, landscape, and regional scale. In particular, the detailed demographic data needed for spatially explicit population modeling does not exist for the majority of species within the Little Applegate Watershed. While this is a constraint to the application of model results to the land use decision making process, modeling itself can aid in improving understanding of these variables and relationships. From conceptual model diagrams that have long been used to describe ecological relationships, to the more quantitative approaches described in this paper, modeling can be used to increase understanding of significant variables and the relationships among them. Modeling efforts developed to increase understanding of large scale biodiversity distribution and abundance might also be used to include non-governmental agencies in the modeling process as well as develop stakeholder involvement and understanding of model potential and limitations. Spatially explicit biodiversity models allow for the inclusion of land use and social factors as well as ecological variables and can help bridge the gap between science and land use policy. Collaborative or cooperative projects developed with numerous organizations and agencies, as well as relational classification systems and databases, may increase the speed at which widespread understanding is increased. The federal agencies are currently in the process of exploring the potential to integrate existing efforts and models into a larger approach that could be used to analyze the risks to biodiversity and habitats from alternative management trajectories; with the intention that this information be applied to the development, evaluation, and implementation of a designed future landscape.

A variety of spatially explicit modeling approaches can be used to investigate the risk to biodiversity from future landscape change in the Little Applegate Watershed (Table 13). As a first step, the guild-based approach used in the application of the *Habscapes* model to assess current habitat conditions as a component of the watershed analysis could be expanded and applied to the Little Applegate Forest Landscape Design. This would enable a detailed evaluation of this alternative future in terms of terrestrial biodiversity. One habitat classification system will need to be created that meaningfully relates the classes used to describe current conditions with that used in the Forest Landscape Design. An extension of this project would be the development of a spatially mapped version of the historic vegetation information, also justified with current and future habitat classifications, to enable analysis of changes in habitat between the past and current and future conditions. Inclusion of historic landscape conditions in the Muddy Creek Project enabled characterization of the kinds of change projected in the future, compared to past changes and also helped to validate the model as information on wildlife trends was available.

Additional spatially explicit modeling approaches will likely be needed to better capture the potential impacts of changes in landscape pattern on individual species, as the guild approach can mask individual effects by grouping large numbers of species into single habitats and averaging life history characteristics such as home range size. Since data is not available for most species, individual-based habitat models, or spatially explicit population models, could be initiated with select indicator, or special status species, for which the level of data is currently high enough to support this approach. One risk of this approach is that species not currently identified as special status or indicators but that are sensitive to future landscape change would be missed in the analysis.

Spatially explicit modeling for individual species, but still based on wildlife-habitat association data, could be developed with minimal additional data collection and compilation, although model development and execution will be time and resource intensive. To conduct spatially explicit population modeling, detailed demographic data will be required, in addition to the wildlife-habitat association data. This approach would involve a high level of data collection and compilation, as well as substantial efforts to develop and validate the models. While spatially explicit population models are data and resource intensive, realism and model accuracy can be greatly improved with the inclusion of additional variables such as survival and dispersal to the habitat area and landscape configuration information.

Another method that increases the accuracy of results, without necessarily requiring extensive data collection, is the use of multiple approaches to biodiversity assessment. For example, in the Camp Pendleton Project, risks to biodiversity were determined in three ways: single species potential habitat, regional species richness, and landscape ecological pattern. The use of multiple indicators and models can improve ecological understanding of significant variables and assist in the evaluation and improvement of specific models, while also providing a more realistic assessment of risks to biodiversity. If alternatives are shown to be robust across multiple models, or potential outcomes, confidence for planning and management decisions is increased (Hilborn and Ludwig 1993). A variety of modeling approaches are available and applicable to the Little Applegate Watershed, particularly in light of the extensive landscape scale assessment and futures planning work that has been conducted already; but collaborative decisions will need to be made regarding the integration of existing projects, and the management direction for future efforts. In addition to influencing the application of biodiversity modeling to land use planning and management, the level of social and institutional support for landscape scale biodiversity risk assessment in the Little Applegate Watershed will also play a large role in determining the methodology or modeling approach employed. Numerous options exist for modeling approaches; as increased model realism is generally associated with greater data, time, and resource needs, a broad spectrum of social and institutional support will be required.

Table 13. Evaluation of Potential Approaches to Biodiversity Modeling in the Little Applegate Watershed

Model Approach	Strengths	Limitations
Expanded guild-based modeling (*Habscapes)*	Guilds have been created. Data is available to support this approach. Provides a broad-based assessment of habitat conditions for all terrestrial wildlife. Current conditions and an alternative future have been described.	Alternative future landscape and current conditions are described with different classification systems. Guild-based modeling may not provide an accurate assessment of risks to individual species as life history variables are averaged across guilds and the role of multiple habitat types and demographic variables are ignored.
Spatially explicit habitat-based modeling of select indicator or special status wildlife species	Data is available for most species. Wildlife-habitat associations are defined for each species.	Alternative future landscape and current conditions are described with different classification systems. Demographic factors are ignored. Potentially sensitive species may be left out of the analysis.
Spatially explicit habitat-based modeling of all terrestrial wildlife species	Wildlife-habitat associations are defined for each species. This approach enables assessment by species or any other indicator such as native/non-native, special status species, or select taxa.	Alternative future landscape and current conditions are described with different classification systems. Demographic factors are ignored. Data is not available for most species.
Spatially explicit population modeling of select indicator or special status wildlife species	Data is available for many species. This approach includes wildlife-habitat and demographic variables, which increases model realism. Supports multiple indicator assessments of the risk to select species from future landscape change.	Alternative future landscape and current conditions are described with different classification systems. Approach is data and resource intensive. Ecological understanding of significant factors may be insufficient to support model development. Potentially sensitive species may be left out of the analysis. Uncertainty in model interpretation is increased with model complexity; relationships between spatial and population factors are hard to define.
Spatially explicit population modeling of all terrestrial wildlife species	Includes wildlife-habitat and demographic variables, which increases model realism. Supports multiple indicator assessments of the risk to biodiversity from future landscape change.	Alternative future landscape and current conditions are described with different classification systems. Data is not available for most species. Approach is data and resource intensive. Ecological understanding of significant factors may be insufficient to support model development. Uncertainty in interpretation increases with model complexity; relationships between spatial and pop. factors hard to define.

Landscape scale biodiversity modeling can be used as a tool for developing desired social and ecological visions of the Little Applegate Watershed, increasing understanding of significant variables, and for evaluating the ecological impacts of alternative land use patterns. While uncertainty is a limiting factor in using models as predictors of future outcomes, models can be used to test hypothesis, direct future research activities, and enhance planning and policy decisions by providing a visual means of assessing potential impacts of alternative actions (Johnson 1999). Despite the many barriers that face implementation of large scale, multiple ownership projects over the long term, land management in general, and particularly within the Little Applegate Watershed, is moving towards this broad scale approach. The level of ecological and social analysis and assessment of current and potential future conditions that are already underway in the Little Applegate Watershed form a solid basis for expansion of landscape scale modeling and management activity. As attempts are made to integrate and expand modeling efforts, additional efforts will be needed in model interpretation and understanding as increased model realism is coupled with increased uncertainty. Social agreement on the desired role and benefits of models will become increasingly important as the evaluation of alternative futures, and the management trajectories employed to achieve those states, is undertaken. Education must be an integral component of any landscape scale biodiversity modeling efforts. The application of landscape scale biodiversity risk assessment to land use decisions in the Little Applegate Watershed has been initiated; by integrating knowledge and understanding from similar projects in other places, as well as the wealth of information that exists in past and current projects within the watershed, meaningful application of landscape scale biodiversity risk assessment to the land use decision making and management process can be applied throughout the Little Applegate Watershed.

APPENDIX

SURVEY QUESTIONAIRE AND RESULTS

Appendix A. Survey Questions and Results: Respondent One

Question	Response
Do you have professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed?	Coordinator for the Applegate Watershed Council. Focus is larger than the Little Applegate but right now the Little Applegate is a focus in the first updating of information for the watershed assessment/analysis.
Can you describe these responsibilities?	Take a ridgetop to ridgetop approach, more involved with the whole picture (uplands, wildlife) in addition to aquatic habitat and fish than a lot of other watershed councils. Have a wildlife biologist on staff; do wildlife habitat surveys along with aquatic and riparian surveys.
Have you had experience with biodiversity or habitat assessment that included modeling?	Agency staff from region serve the council as technical advisors and modeling has come up peripherally in discussion but the council has not been directly involved.
Or other ecological modeling?	Starting to use hydrologic models, have a consultant out of Colorado working on a project with the council and have recently hired a person with hydrologic modeling capabilities as council staff.
Based on your needs as a land manager, in what ways might modeling of the risks to biodiversity from future landscape change be useful?	Not really familiar enough with potential outcomes/possibilities to say how it could be used but can make general comments. Would be most useful at larger scale, holistic information more useful, could help council work with private landowners, industrial forests, etc. Although mostly federal lands in the Little Applegate the council is still interested in addressing connectivity/big picture issues. Any additional information on large scale biodiversity or habitat would be useful.

Appendix A. Survey Questions and Results: Respondent One Continued

Question	Response
What information, or model output, would be most useful to you?	Would be most useful at larger scale, holistic information more useful. Although mostly federal lands in the Little Applegate the council is still interested in addressing connectivity/big picture issues. Any additional information on large scale biodiversity or habitat would be useful.
How much certainty or confidence would you require in model results for it to be useful to you in the context of your responsibilities?	If general and less accurate the information would likely be used as a first pass assessment. If necessary, they could build more detailed information later. Good understanding, by all involved, of the limitations of models is also needed.
What changes, if any, do you think are needed to successfully apply the results of biodiversity modeling to land use decisions in the Little Applegate Watershed?	A private land component is needed; landowners need to be involved somehow. The council could play a role in this education process. Understanding of how models work by all affected/involved needs to be improved. Need to be careful with decisions made based on models, limitations need to be recognized and accounted for.

Appendix A. Survey Questions and Results: Respondent Two

Question	Response
Do you have professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed?	Silviculturalist for USFS
Can you describe these responsibilities?	Responsible for carrying out proper timber stand treatments as they relate to the landscape design, which relates to both biodiversity and habitat protection.
Have you had experience with biodiversity or habitat assessment that included modeling?	Have worked with the Little Applegate Landscape Design process
Or other ecological modeling?	No.
Based on your needs as a land manager, in what ways might modeling of the risks to biodiversity from future landscape change be useful?	Need some measure of biodiversity, an index which tells us how well we are meeting the needs of a drainage. May need more than a single index; perhaps an index for each group of species, since requirements shift from species to species. There may be great diversity for one set of species but it may spell disaster for another. For example, good conditions for deer and elk may be poor conditions for spotted owls since the habitat needs are so different. Could try to have some for each but as the diversity of habitat shifts it will have a positive or negative effect on owls.
What information, or model output, would be most useful to you?	Don't know if overall risk assessment is useful as risks are species specific. Would rather have an index for various groups of species. You could then average the results but that may not really be useful. Maybe we have to decide which species we really need to manage the watershed for.

Appendix A. Survey Questions and Results: Respondent Two Continued

Question	Response
How much certainty or confidence would you require in model results for it to be useful to you in the context of your responsibilities?	Generally a confidence of two standard deviations is plenty good in natural resource work. I might be swayed by information at the one standard deviation level but I don't know if I would make a long term commitment to it.
What changes, if any, do you think are needed to successfully apply the results of biodiversity modeling to land use decisions in the Little Applegate Watershed?	I don't know that any changes need to be made. It may be important to have the same level of information concerning stand development across the watershed so that an area could be assessed on an equal basis.

Appendix A. Survey Questions and Results: Respondent Three

Question	Response
Do you have professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed?	Field administrator for Jobs in the Woods Crew.
Can you describe these responsibilities?	Current project on BLM and eight adjacent private landowners property. Fuel reduction, attempting to recreate pine oak savannah, also wildlife ponds, riparian tree plantings.
Have you had experience with biodiversity or habitat assessment that included modeling?	Rogue Institute involved in social assessment components of watershed analysis and forest landscape analysis and design project. No direct involvement in modeling piece.
Or other ecological modeling?	Not directly, specifications for jobs in the woods work do sometimes result from agency management decisions based on modeling.
Based on your needs as a land manager, in what ways might modeling of the risks to biodiversity from future landscape change be useful?	No direct benefit as modeling is long term and large scale and jobs in the woods projects are small and numerous, not necessarily all related to big picture, just go where funding/work is. If managers buy off on the process the use would move down to the on-the-ground level. Need to see efficiency of connecting small projects to the overall landscape design. Plan can be used as a template for work, increase project success through showing connectivity through relation to overall plan.

Appendix A. Survey Questions and Results: Respondent Three Continued

Question	Response
What information, or model output, would be most useful to you?	More than specific information is the way it is presented. Need to simplify process, vocabulary, and maps so that more people can be reached by the process. (e.g.: huge, hard to reproduce 20 color maps- need to be simplified to achieve more widespread acceptance and applicability)
How much certainty or confidence would you require in model results for it to be useful to you in the context of your responsibilities?	Same as above, not as much a question of certainty as of improved mechanisms for education and involving more people in a way that they better understand the process. Increase confidence by including more residents, local organizations in the process.
What changes, if any, do you think are needed to successfully apply the results of biodiversity modeling to land use decisions in the Little Applegate Watershed?	Need a concerted education and outreach effort on landscape scale futures planning. Need manager level acceptance to connect goals of small scale, on-the-ground projects to overall design. Could create a system of incentives for conducting work that supports the landscape design to foster adherence at the local level. Adaptive Management Area could facilitate this as they are experienced in working with local residents and have the administrative structure that would be required to keep the momentum over time.

Appendix A. Survey Questions and Results: Respondent Four

Question	Response
Do you have professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed?	Project mapping, planning, and analysis for the United States Forest Service (USFS)
Can you describe these responsibilities?	Provide mapping and GIS analysis for Applegate Watershed and USFS. Supervise planning of projects and NEPA compliance for the USFS.
Have you had experience with biodiversity or habitat assessment that included modeling?	Over the past ten years all analysis work at USFS has incorporated computer assisted GIS; before GIS used overlay maps and looked for patterns. Involved in coarse level modeling done for the LA forest landscape design project. Queries and analysis for habitats for special status species. Through analysis of habitat attempted to be predictive 'what are the habitats that might be required?'
Or other ecological modeling?	Project to use satellite imagery to understand plant communities. Modeling under Northwest Forest Plan, Fire modeling.
Based on your needs as a land manager, in what ways might modeling of the risks to biodiversity from future landscape change be useful?	I am concerned about current management practices that provide circles of habitat around existing species sites and then take a hands-off approach. Not managing for habitat but habitat conditions are different today than in the past, primarily due to the exclusion of fire as a natural disturbance process. Habitat quality needs to be looked at more, modeling could aid this effort. Habitat needs to be managed for over a larger scale and longer time frame than is currently being practiced, modeling could be applied to this as well.

Appendix A. Survey Questions and Results: Respondent Four Continued

Question	Response
What information, or model output, would be most useful to you?	Long term, large scale habitat quality and quantity information. For select species, what are the optimum and lesser quality habitats, distribution, pattern. What are the range of habitats for certain species.
How much certainty or confidence would you require in model results for it to be useful to you in the context of your responsibilities?	Research/analysis question drives data and output; need to assess the sensitivity of the model to provide the output you are looking for (e.g.: Can you measure it and does it matter at that sensitivity?). Certainty or confidence dependent on best available data so acceptable/optimal levels of confidence are not formally defined. Know if the data going in is at a coarse level not to imply to much truth in the output. Need to remember not to be seduced by the maps; not the answer but a tool to improve understanding and ask more and better directed questions. Modeling brings more clarity to the process and gets us to think more analytically, it does not provide specific answers.
What changes, if any, do you think are needed to successfully apply the results of biodiversity modeling to land use decisions in the Little Applegate Watershed?	Need better information of the range of habitats used by species, better background understanding of the natural range of variability. Hard to define natural, generally thought of as pre-settlement but native Americans also managed the landscape. Need clear management goals for the landscape. The primary question in defining management goals is 'what are the range of values you want to manage for?' and this is essentially a social question.

Appendix A. Survey Questions and Results: Respondent Five

Question	Response
Do you have professional responsibilities related to biodiversity or habitat protection in the Little Applegate Watershed?	Project initiator for Forest Landscape Design. Project design and analysis.
Can you describe these responsibilities?	Integrated resource analysis for projects. Project team makes recommendations to forest rangers. Pilot project in upper glade area of little Applegate to investigate overall forest improvement, not just harvest.
Have you had experience with biodiversity or habitat assessment that included modeling?	Over past ten years forest service has taken an integrated approach to projects, investigating multiple, not single resources for projects. No direct modeling but inclusion of biodiversity and habitat in wider analysis of project impacts and benefits, bigger picture analysis.
Or other ecological modeling?	Worked on assessments and watershed analysis that included some elements of modeling. Currently working with modeler's to develop a bayesian decision making model to guide decision making. Attempt to integrate smaller models that have been used (e.g.: habscapes) to help build probability pathways into decision models
Based on your needs as a land manager, in what ways might modeling of the risks to biodiversity from future landscape change be useful?	Now that we have the Forest Landscape Design, need to know what the impacts to species would be for a variety of implementation scenarios. Many ways to reach designed landscape, effects to biodiversity and habitat could differ largely depending on the approach. need information on the risks to species from different management trajectories. Also need information on how these management scenarios would impact key flows identified in forest landscape design process.

Appendix A. Survey Questions and Results: Respondent Five Continued

Question	Response
What information, or model output, would be most useful to you?	Most useful would be maps illustrating risk assessment to species, habitats, and flows for a variety of management scenarios that could be used to move vegetation towards the desired future state. How do vegetation patches change over time in the landscape. Transitional views of impacts of alternative management on biodiversity and vegetation.
How much certainty or confidence would you require in model results for it to be useful to you in the context of your responsibilities?	Bayesian decision making modeling effort, which includes attempts to integrate smaller models used in the Little Applegate are helping to evaluate the confidence of modeling efforts in this area. Level of certainty required would be risk based, changing depending on how at risk a species or vegetation community is right now in the landscape, is action needed right away or can it wait until information is improved.
What changes, if any, do you think are needed to successfully apply the results of biodiversity modeling to land use decisions in the Little Applegate Watershed?	Largest changes needed are societal, including the private land ownership system, and land use patterns, as well as changes in attitude by all involved. It took a long time to create current landscape conditions, but we seem to be looking or requiring a rapid solution. Difficult to get commitment from agencies, organizations, and individuals over the longer time frame. Need systems in place to promote systems over a much longer time frame.

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