3734

AN ABSTRACT OF THE THESIS OF

Etsuko Nonaka for the degree of Master of Science in Forest Science presented on November 25, 2003.
Title: Disturbance and Landscape History as a Reference for Evaluating Forest Management Effects at a Regional Scale: Examples from the Coast Range of Oregon, USA.

Abstract approved: ______________________________

Thomas A. Spies

History is an invaluable source of information to understand and evaluate management influences on contemporary ecosystems and landscapes. The first two chapters (Chapters 2 and 3) explored the concept of historical range of variability (HRV) in landscape structure and stand structure using a stochastic fire simulation model to simulate presettlement (before 1850) landscapes of the Oregon Coast Range. HRV has been defined as the bounded variability of a system within constraints imposed by larger-scale phenomena (e.g. climate, topography) and without significant modern human influence. HRV of landscapes has been proposed as a guide for biodiversity conservation in the past decade.

In Chapter 2, I estimated HRV of a regional landscape and evaluated the similarity of current and alternative future landscapes under two land management scenarios to the conditions within the HRV. The simulation results indicated that historical landscapes of the region were dynamic, composed of patches of various sizes and age classes ranging from 0 to > 800 years as well as numerous small unburned island patches. The current landscape was outside the HRV. The landscape did not return to the HRV in 100 years under either scenario largely because of lack of old-growth forests and overabundance of young forests. This study showed that the HRV can provide a reference condition for concrete, quantitative evaluations of landscape conditions and alternative management scenarios if sufficient data exist for estimating HRV. Departure from HRV can serve as an indicator of landscape conditions, but results depend on scale and quantification of landscape heterogeneity.

In Chapter 3, I investigated the HRV in live and dead biomass and examined variability in disturbance history and forest stand development. I calculated biomass as a function of disturbance history. The HRV of live and dead wood biomass distributions revealed that the majority of the landscape historically contained > 500 Mg/ha of live wood and 50-200 Mg/ha of dead wood. The current dead wood condition is outside HRV. There was a wide variation in dead wood biomass because of variations in disturbance history. This study suggests that natural disturbance regimes and stand development are characterized by much larger variation than is typically portrayed or appreciated. The HRV approaches to evaluating landscape conditions need to include both landscape and stand characteristics to better represent ecological differences between managed and unmanaged landscapes.

In Chapter 4, I used remotely sensed data and historical vegetation data in a GIS to examine changes occurred in vegetation cover since settlement in two major valleys, the Coquille and Tillamook, in the region. I used existing historical vegetation maps of the two valleys and collected historical vegetation data from the General Land Office (GLO) survey records. I characterized current vegetation conditions using an unsupervised classification of satellite images. Historically, the Coquille Valley was dominated by hardwood trees and the Tillamook was by conifers. Valley bottoms in both areas differed in vegetation from nearby uplands. Tree-covered areas have declined substantially in both valleys as a result of agriculture and development. The historical data offered reference conditions for assessment of changes in biodiversity that have occurred in these unique habitats.

This thesis illustrates the benefit of using historical landscape information for better understanding of human influence on the landscape. Historical data often have many assumptions and limitations, but ecological impacts of landscape changes on native biota can be better understood by comparisons with historical conditions.

Disturbance and Landscape History as a Reference for Evaluating Forest Management Effects at a Regional Scale: Examples from the Coast Range of Oregon, USA.

by

Etsuko Nonaka

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Master of Science

Presented November 25, 2003
Commencement June 2004

Master of Science thesis of Etsuko Nonaka presented on November 25, 2003.

APPROVED:

Major Professor, representing Forest Science

Head of the Department of Forest Science

Dean of the Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Etsuko Nonaka, Author

ACKNOWLEDGEMENTS

I have received numerous assistance and encouragement from many people over the three years of working on this project. Thank you very much. First of all, I would like to express my sincerest gratitude to my major advisor, Dr. Tom Spies. He provided valuable advice and creative ideas that led this thesis to the completion. I learned a lot of ideas and critical thinking from numerous discussions with him throughout the three years, which I cannot get from just taking classes. On the very first conversation I had with him (when I was looking for a graduate program), I remember that I told him I wanted to "operationalize the idea of ecological integrity and create Ecological Integrity Index!" I think I was naïve not to think about all the difficulties that would be waiting for me on the way. It was a lot more difficult than what I expected. I think I kind of achieved that goal in some way, and it would not have realized without his help and patience. Thank you very much.

I would also like to thank my other committee members, Drs. Bill Ripple, Fred Swanson, and John Hayes, who were always willing to help me out. I also would like to thank Dr. Paul Doescher for taking time to participate in my project.

I was very fortunate to be surrounded by many great forest ecologists and researchers in the Corvallis Forest Sciences Lab. It is an incredible place to learn about forests. Also, I was fortunate to be involved in the Coastal Landscape Analysis and Modeling Study (CLAMS) and to see how scientists interact to advance science and achieve their research goals. I highly appreciate the generous funding from the project.

I would like to express my deep gratitude to Dr. Mike Wimberly for being cooperative on my project and giving me wonderful guidance. I would like to thank Rob Pabst for being such a great officemate and willing to spend his time for helping me at any time. I would not be able to survive in the office without him there. I am very grateful to Dr. Janet Ohmann for giving me advice, especially for good writing skills. I am very appreciative of the time that Matt Gregory and Keith Olsen spent providing lots of GIS help and answering my questions. I also highly appreciate the time Dr. Ken Pierce spent to give me lots of statistical advice on my project. His daughter's smiles on his computer screen cheered me up so many times. I would like to thank Dr. Kevin McGarigal and his students for answering my numerous questions about FRAGSTATS and statistics over email. I would like to also thank Dr. Bruce McCune for his generous help for multivariate techniques. I am grateful to Dr. Steven Garman for teaching me modelers' mentality ("Thick heads") and to Dr. Mark Harmon for answering my numerous questions about dead wood.

Most of all, I would like to thank all of my friends and family for their encouragement and support throughout this process. I am sincerely grateful to my parents for their support and understanding of my studying abroad for more than 9 yrs. I will try my best to meet their expectations as much as I can for the rest of my life.

I am very thankful to the Saubert and the Bacon families for their generous contribution of the Saubert and the Bacon Scholarships.

CONTRIBUTION OF AUTHORS

I am grateful to Drs. William Ripple, Frederick Swanson, and John Hayes for their review of the chapters included in this thesis document. I am also grateful to Drs. Janet Ohmann, Lisa Schulte, Robert Pabst, and Bob Zybach for their generous review of Chapter 4.

TABLE OF CONTENTS

Page

CHAPTER 1: GENERAL INTRODUCTION 1

CHAPTER 2: HISTORICAL RANGE OF VARIABILITY (HRV) IN LANDSCAPE STRUCTURE: A SIMULATION STUDY IN THE COAST RANGE OF OREGON, USA 8

Abstract 8

Introduction 9

Methods 12

Study area 12
Disturbance regimes 14
Human influence 14
Model simulations 16
Landscape metrics 20
Data analysis 24

Results 25

The HRV in landscape structure and comparison with current conditions 25
Future scenarios: current policy (CPS) and wildfire (WFS) 37
Time to reach HRV under the WFS 39

Discussion 39

Historical landscape dynamics of very young and very old forests in the Oregon Coast Range 39
Management effects on current landscape structure and dynamics 43
Comparisons between the two scenarios: current policy (CPS) vs. wildfire regimes (WFS) 45
Challenges in estimating HRV 47
Management implications 50

Conclusions 51

CHAPTER 3: HISTORICAL RANGE OF VARIABILITY (HRV) IN LIVE AND DEAD WOOD BIOMASS: A SIMULATION STUDY IN THE COAST RANGE OF OREGON, USA 53

Abstract 53

Page

Introduction 54

Methods 57

Study area 57
Disturbance regimes 59
Conceptual model of biomass dynamics 60
The simulation model of fire and biomass dynamics 63
Data creation and analysis 69
Sensitivity analysis 72

Results 72

HRV in live and dead wood biomass in the landscape of the Oregon Coast Range 72
Characterizing variation in disturbance history and stand development 80
Sensitivity analysis 82

Discussion 84

Model limitations and assumptions 84
Spatial patterns of live and dead wood biomass in the Oregon Coast Range 85
The HRV and current distribution of live and dead wood biomass 87
Variability in disturbance history 88
Management implications 90

Conclusions 92

CHAPTER 4: VEGETATION CHANGE IN BOTTOMLANDS OF MAJOR COASTAL RIVER VALLEYS OF THE OREGON COAST RANGE FROM CA. 1850 TO PRESENT 93

Abstract 93

Introduction 94

Methods 96

Study area 96
Developing the valley map for the Oregon Coast Range 99
Historical vegetation 100
Current vegetation 101
Quantifying vegetation cover change in the valleys 102

TABLE OF CONTENTS (Continued)

Page

Results 106

Valley bottomlands in the Oregon Coast Range 106
Historical vegetation cover 108
Current vegetation cover and change since settlement 115

Discussion 117

Changes in the valleys 117
Historical landscapes in the two valleys 117
Implications of landscape change in the valleys for regional biodiversity 119
Uncertainty related to the data and density calculation 120
Limitations and constraints in this study 121

Conclusions 122

CHAPTER 5: GENERAL CONCLUSIONS 124

Key findings 128

BIBLIOGRAPHY 130

LIST OF FIGURES

Figure		Page
2.1	The vegetation maps of the Oregon Coast Range Province	13
2.2	The ownership map in the Oregon Coast Range	15
2.3	The two climatic zones used in the LADS model	18
2.4	PCA ordination results for the landscape-level and class-level analyses	31
3.1	The two climatic zones used in the LADS model	58
3.2	Dynamics of live and dead wood biomass in response to different fire severities and frequencies	61
3.3	Disturbance history types based on initial live biomass and fire severity	62
3.4	The conceptual diagram of the deadwood model in LADS.	65
3.5	Examples of live and dead wood biomass patterns from simulation runs of LADS for the Oregon Coast Range, USA	73
3.6	The historical ranges of variability in percent of landscape covered by different deadwood biomass classes based on 100 simulated landscapes	77
3.7	The historical ranges of variability by age class in percent of landscape covered by different deadwood biomass classes	79
3.8	A schematic diagram showing comparison between the HRV of biomass under the historical fire regime and typical biomass found in plantations in the Oregon Coast Range.	89
4.1	The map of the Oregon Coast Range and the valleys identified in this study	97
4.2	The allocation of ownerships in the valleys, uplands, and the Oregon Coast Range.	107
4.3	The historical (a) and current (b) vegetation maps of the Coquille Valley	109
4.4	Relative frequency, relative basal area (m^2), and importance value (IV: average of relative frequency and relative basal area) of species recorded by the surveyors in the Coquille Valley	110

LIST OF FIGURES (Continued)

Figure | Page

4.5 Relative frequency of trees by diameter class for a) the Coquille Valley and b) the Tillamook Valley 111

4.6 The historical (a) and current (b) vegetation maps of the Tillamook Valley 113

4.7 Relative frequency, relative basal area (m^2), and importance value (IV: average of relative frequency and relative basal area) of species recorded by the surveyors in the Tillamook Valley 114

4.8 Change in percent valley area covered with trees or no trees in the Coquille and Tillamook Valleys 116

LIST OF TABLES

Table Page

2.1 The descriptions of forest structure by age classes based on forest dynamics in temperate wet coniferous forests in Pacific Northwest described in literature 21

2.2 The landscape metrics used to quantify the landscape structure in the modeled landscapes of Oregon Coast Range, Oregon, USA 22

2.3 Eigenvalues of the principal components and Pearson correlations of the original variables with the PC axes for the ordinations of the simulated historical landscapes of the Oregon Coast Range, Oregon, USA 26

2.4 The mean, minimum, maximum, and standardized range ((range/max)*100) of the metrics from the simulated historical landscapes that fell in the 90% HRV likelihood of the age classes and values of the metrics measured on the current and potential future landscapes of the Oregon Coast Range, Oregon, USA. 28

2.5 PC scores and linear combinations for the current, yr 50, and yr 100 landscapes for the Oregon Coast Range, Oregon, USA 33

2.6 The number of years for each metric to reach the HRV 40

3.1 The model parameters used in this study showing the baseline values and values used for the sensitivity runs 67

3.2 The mean variance of live and dead wood biomass from the texture analysis by analysis window size (in number of pixels) 75

3.3 The mean percent coverage of live and dead wood biomass classes in the Oregon Coast Range 76

3.4 The mean percent coverage of live and dead biomass classes for the current landscape of the Oregon Coast Range. 78

3.5 The frequency (%) of 8 disturbance history types by the climate zones. 81

3.6 The summary statistics for 1000-yr disturbance histories 81

3.7 Results from sensitivity analysis of LADS model simulations of historical Coast Range landscapes 83

LIST OF TABLES (Continued)

Table		Page
4.1	Descriptions of the vegetation types on the historical maps and calculated vegetation characteristics using the witness tree data for the Coquille and Tillamook Valleys	103
4.2	The most probable species for the plant names mentioned in the surveyors' field journals.	105
4.3	The percent of valley area covered by vegetation types on the current landscapes of the Coquille and Tillamook Valley based on an unsupervised classification of satellite imageries taken in mid 1990's	105

DISTURBANCE AND LANDSCAPE HISTORY AS A REFERENCE FOR EVALUATING FOREST MANAGEMENT EFFECTS AT A REGIONAL SCALE: EXAMPLES FROM THE COAST RANGE OF OREGON, USA

CHAPTER 1: GENERAL INTRODUCTION

Forested landscapes are mosaics of various forest types and ages, which are maintained by ecological processes and environmental heterogeneity at multiple scales of time and space (Spies and Turner 1999, Turner et al. 2001). The principles of landscape ecology address that landscape patterns or structure (i.e. composition and configuration) influence maintenance of biodiversity and ecological processes (Forman and Godron 1986, Hunter 1990, 1999, Turner et al. 2001). Disturbance and succession are important components of ecological processes that changes landscape structure over time and space within the constraints of climate and landforms (Urban et al. 1987, Delcourt and Delcourt 1988, Swanson et al. 1988, Baker 1992a, 1995, Spies and Turner 1999, Turner et al. 2001, Frelich 2002). Humans have altered natural disturbance regimes and landscape patterns for a long time (Boyd 1999, Vale 2002), and the consequence of landscape changes caused by humans on biota and ecological processes is a subject of considerable interests in landscape ecology (Burgess and Sharpe 1981, Godron and Forman 1983, Forman and Godron 1986, Franklin and Forman 1987, Turner 1987, Zonneveld and Forman 1990, Forman 1995, Turner et al. 2001).

On managed landscapes, some natural disturbances have been effectively locally suppressed (e.g. fire, flood), while some still affect it at various scales (e.g. wind throw, landslides, volcanic eruptions). Timber harvesting and land conversion may dominate disturbance regimes on managed landscapes, and plantation and silvicultural treatments may alter successional and stand developmental pathways, compared to those that occurred under the natural disturbance regime. Shifts in disturbance regimes due to management have altered landscape pattern and composition (Forman and Godron 1986), hydrologic regimes (Swanson et al. 1990), habitat quantities and qualities (Forman 1995, Bissionette 1999), and susceptibility to other disturbances (Franklin and Forman 1987). However, because landscapes are intrinsically dynamic, the influence of forest management on landscape patterns and biodiversity cannot be truly evaluated without a reference to the inherent variability.

The historical range of variability (HRV) in landscape structure created by natural disturbances can provide a reference for evaluating managed landscapes. The HRV is the bounded variability of a system (i.e. composition, structure, function) within limits imposed by larger-scale constraints such as climate and topography (Morgan et al. 1994, Aplet and Keeton 1999). There are many phrases for HRV and akin concepts such as "natural range of variability," "natural variability," and "reference variability" (Aplet and Keeton 1999, Landres et al. 1999), but in this thesis I use "historical" to conceptually include the influence of Native Americans on the fire regime before the arrival of Euro-Americans.

The concept of HRV has been proposed as a guide for biodiversity conservation in the past decade (Morgan et al. 1994, Swanson et al. 1994, Aplet and Keeton 1999, Landres et al. 1999). The premise of using this concept as a management guide is that landscape dynamics driven by natural disturbance regimes shaped native communities we wish to protect and maintain (Swanson et al. 1994, Cissel et al. 1994, Landres et al. 1999). Therefore, maintaining landscape conditions within the range will theoretically ensure a high probability of producing conditions suitable for most native species including ones poorly known. The use of HRV of landscape patterns for biodiversity conservation is a "coarse-filter" strategy, in which a representative array of habitat communities and ecosystems are protected, and the strategy can operate with relatively little information about individual species (Hunter 1988, 1990, Franklin 1993). In the Pacific Northwest, Cissel et al. (1998, 1999) developed forest management plans that utilized the historical fire regimes inferred from field data to allocate harvest blocks of different harvest frequency and overstory-removal levels in a watershed. The Montreal Process, an international effort for conservation and sustainable management of boreal and temperate forests, recognizes HRV in disturbed forest area as benchmark for an indicator of forest health (Montreal Process Working Group 1998). Research on disturbance-based forest management, which attempts to emulate landscape heterogeneity created by wildfires through harvesting patterns and silvicultural treatment, has been investigating the effectiveness of the strategy for sustainability and biodiversity conservation throughout North America and Scandinavia (e.g. Schieck et al. 2000, Bergeron et al. 2002, Harvey et al. 2002, Kuuluvainen 2002, Seymour et al. 2002, Simon et al. 2002).

A number of papers and government reports have evaluated the HRV concept and its application to management (Morgan et al. 1994, Swanson et al. 1994, Manley et al. 1995, Quigley et al. 1996, Cissel et al. 1998, 1999, Landres et al. 1999, Aplet and Keeton 1999, Swetnam et al. 1999, Committee of Scientists 1999, Swanson et al. 2003). The major values of the concept are 1) to help understand effects of disturbances and landscape change over time (i.e. landscape dynamics), 2) provide reference conditions for evaluating managed landscapes and alternative management scenarios, and 3) provide a guide for developing management regimes. Several recent regional projects have utilized historical information as base data for assessment of ecosystem conditions (e.g., Sierra Nevada Ecosystem Project, Interior Columbia Basin Ecosystem Management Project). The limitations of HRV approaches are that 1) past conditions may not be relevant to current and future, especially with climate change, 2) reference time periods (e.g., presettlement) can be arbitrary, 3) magnitude of ranges depends on scale of assessment, 4) historical data are scarce and our ability to interpret the data is incomplete, and 5) it is difficult to concisely characterize HRV of a variety of landscape characteristics.

Morgan et al. (1994) summarized the main methods for describing HRV: 1) tree ring analysis, 2) paleoecological methods, 3) written and photographic records, 4) GIS data layers, 5) unaltered landscapes (e.g., wilderness areas), and 6) modeling (see Morgan et al. for examples of each method). Swetnam et al. (1996) provide an overview of sources of historical information and their temporal and spatial scale domains. The first part of this study utilized the spatially-explicit modeling approach to establish the HRV for the Oregon Coast Range. The main advantages of modeling approaches are 1) ability to estimate HRV by simulating many possible landscapes under a historical regime that incorporates temporal variability and stochasticity in disturbances (Lertzman et al. 1998), 2) ability to evaluate possible future scenarios

(Landres et al. 1999), and 3) ability to facilitate understanding of temporal and spatial dynamics of landscapes.

Quantitative, modeling studies on the HRV concept are not limited, but approaches and objectives differ among studies (Baker 1992, 1995, Wallin et al. 1996, Wimberly et al. 2000, in press, Wimberly 2002, Keane et al. 2002, Roworth 2001, Hemstrom et al. 2001, Swanson et al. 2003, Tinker et al. 2003). Most of the studies focus on landscapes with disturbance regimes that are characterized by large-scale fires. In the Oregon Coast Range, previous studies (Wimberly et al. 2000, in press, Wimberly 2002) used a spatially-explicit stochastic fire simulation model, the Landscape Age-class Demographics Simulator (LADS), to examine the effects of spatial scales in HRV estimation (Wimberly et al. 2000) and to quantify the HRV in landscape patterns using simple age classes and metrics (Wimberly 2002, Wimberly et al. in press). The main conclusions from these studies are that 1) the regional scale (i.e. the entire Coast Range) is the appropriate spatial scale to establish HRV in landscape patterns of old forests, 2) old-growth forests were abundant, but younger forests were also important on the historical landscape, and 3) the current landscape is outside the HRV. The landscape currently has less and more isolated mature and old growth forests than it did historically. The disturbance rates in the 150 years since settlement, mainly from timber harvesting, were higher than under the historical regime (Cohen et al. 2002), resulting in an abundance of young forests in the current landscape.

HRV can be used to evaluate future landscapes that might result from alternative management scenarios. This approach allows us to compare and contrast HRV with alternative future landscapes to detect deviations from the historical conditions. More strategically, with the approach we can compare alternative management scenarios and contrast their possible effects on landscape pattern for more informed decision-making.

The overall goals of this study are to explore the use of historical information to understand and evaluate the ecological condition of managed landscapes of the Oregon Coast Range. In Chapters 2 and 3, I explore the concept of HRV using a simulation model. I use the spatially-explicit computer model, LADS, to simulate landscape dynamics under the historical fire regime and compile information from numerous landscape maps to develop the HRV of landscape structure. I set the reference time period for the simulations as 1000 yrs prior to Euro-American settlement. Fire regimes in the Coast Range in the last 500 yrs have been relatively well studied by fire patch mapping, fire scar studies, and paleoecological studies (Teensma 1991, Ripple 1994, Impara 1997, Long et al. 1998, Long and Whitlock 2002). The climate and vegetation in the region has changed little in the last millennium (Long et al. 1998).

The overall objective of Chapter 2 is to characterize the HRV of landscape structure of the Oregon Coast Range and to compare the current and potential future landscapes of the region with the HRV. I am aware of a relatively small number of studies that used the HRV concept to evaluate future management scenarios (Wallin et al. 1996, Andison and Marshall 1999, Cissel et al. 1999, Hemstrom et al. 2001, Swanson et al. 2003). The possible future landscapes of the Oregon Coast Ranges have not been examined by previous studies. In this chapter, I quantify the HRV of landscape structure using multivariate analysis of landscape metrics. Landscape metrics are known to be highly correlated (Hargis et al. 1998), but multiple metrics are necessary to describe complex landscape patterns (Gustafson 1998). Past HRV studies used only a handful of metrics to characterize HRV. The previous HRV studies for the Oregon Coast Range examined landscape structure that was characterized by a limited number of age classes, and thus the diversity and dynamics of the forests were not fully described. This study uses age classes which recognized extreme age classes (e.g. < 10 yrs, > 800 yrs) that were not examined previously for the region.

I evaluate the conditions of the current and potential future landscapes under two alternative scenarios: the current policy and the wildfire scenarios. I then compare the projected future landscapes with the HRV. The forest policies currently implemented in the region have different management emphasis on biodiversity conservation, and the collective effects of the policies on landscape structure at the regional scale have not been rigorously assessed. The wildfire scenario is a hypothetical case. However, since forest managers and scientists have been discussing emulation of disturbance regimes by forest management for biodiversity conservation, it is important to examine the effectiveness of the strategy on managed landscapes with footprints of intensive human utilization of forests already in place.

The overall objective of Chapter 3 is to examine the variability in forest stand structure and disturbance history in the Oregon Coast Range under the historical fire regime. Forest stand structure was characterized by live and dead wood biomass. Previous HRV studies examined only live tree structure implied by age or forest structure type classes and did not include dead wood components. Amounts of dead wood structure in stands are not well correlated with age or live wood structure because of the effects of stand history (Spies et al. 1988). Therefore, there is a need to examine both live and dead wood dynamics in order to characterize HRV of stand structure and understand cumulative effects of disturbance on stand structure. Stand biomass is an important measure of forest structure, and the dynamics relates to other characteristics of forests such as biogeochemistry and wildlife habitats (Bormann and Likens 1979). LADS was modified to simulate live and dead wood biomass change over time. I conceptualize the biomass dynamics in 2-dimansional, live and dead wood biomass space, in which stands change its state over time due to disturbance and stand development. I characterize the HRV in live and dead wood biomass under the historical fire regime and compare the current level of dead wood with the HRV. I also contrast the historical spatial distribution of live and dead wood at the regional scale. Stochasticity in spread and heterogeneity in severity of disturbances can produce a variety of stand disturbance histories and thus forest structure. This chapter also characterizes variation in fire disturbance history and stand development. Using the 2-dimensional live-dead biomass scheme, I categorize stand disturbance types and quantify their relative frequencies.

In the Oregon Coast Range, as in other parts of the world, prominent landscape changes have occurred in productive, low elevation areas, especially in valley bottomlands. The simulation model used for Chapters 2 and 3 largely focuses on landscape dynamics in upland forests and do not show finer-scale patterns in valley bottomlands. It is, however, essential to characterize landscape changes occurred in bottomlands to assess the overall condition of the region because of the disproportional importance of wetlands and riparian areas as wildlife and plant habitats for the regional biodiversity (Kauffman et al. 2000, Mitsch and Gosselink 2000). Because simulation models are not available for this purpose, I use another approach to using historical information for evaluating valley landscapes in Chapter 4.

The overall objective of Chapter 4 is to quantify vegetation changes that have occurred since Euro-American settlement in two major valleys in the Oregon Coast Range. I use an approach that is a combination of remotely sensed data and historical information stored in a GIS. I chose the Coquille and Tillamook Valleys because they are the major coastal valleys in the region and historical vegetation inferred primarily from General Land Office (GLO) survey data have been compiled in a map format by previous studies by Benner (1991) and Coulton et al. (1996). I describe the historical and current landscapes of the two valleys and quantify the difference in a spatially non-rigorous manner.

CHAPTER 2: HISTORICAL RANGE OF VARIABILITY (HRV) IN LANDSCAPE STRUCTURE: A SIMULATION STUDY IN THE COAST RANGE OF OREGON, USA

ABSTRACT

Historical range of variability (HRV) of landscapes under natural disturbance regimes can be used as a reference condition to evaluate managed landscapes and alternative management scenarios. I estimated HRV of a regional landscape and evaluated the similarity of current and alternative future landscapes to the conditions within the HRV. I used a stochastic fire simulation model to simulate presettlement (before 1850) landscapes of the Oregon Coast Range and quantified the HRV of landscape structure using multivariate analysis of landscape metrics. I examined two alternative policy scenarios using two spatially-explicit simulation models at the regional scale. In the first, currently implemented policies were simulated for 100 years into the future using current management regimes in the region: reserve-based policies on the Federal lands and multiple use and timber production dominated on the State and private lands. In the second scenario, I simulated a wildfire regime with no active management for 1500 years and quantified the time the managed landscape took to return to a condition within the HRV.

The simulation results indicated that historical landscapes of the region were dynamic, composed of patches of various sizes and age classes ranging from 0 to > 800 years including numerous, small unburned island forests. The current landscape was outside the HRV. The landscape did not return to the HRV in the 100 years under either scenario largely because of lack of old-growth forests and the abundance of young forests. Under the current policy scenario, the highly contrasting management regimes among ownerships and ownership pattern constrained development of landscape structure, and the vegetation pattern after 100 years reflected the ownership boundaries. Surprisingly, the wildfire scenario initially moved the landscape away from the HRV during the first 100 years, after which it moved toward HRV, but

required 800 years to reach it. This study showed that the HRV approach can provide a reference condition in relation to the historical landscapes for quantitative evaluations of landscape conditions and alternative management scenarios if sufficient data exist for estimating HRV. Extensive forest management in the last few decades has left legacies on the landscape that could take centuries to be obliterated by wildfire. Departure from HRV can serve as an indicator of landscape conditions, but results depend on scale and quantification of landscape heterogeneity.

INTRODUCTION

The historical range of variability (HRV) in landscape and forest structure created by natural disturbances has been proposed as a guide for biodiversity conservation in the past decade (e.g., Attiwill 1994, Cissel et al. 1994, 1999, Morgan et al. 1994, Swanson et al. 1994, Reeves et al. 1995, Aplet and Keeton 1999, Engstrom et al. 1999, Landres et al. 1999, Committee of Scientists 1999, Davis et al. 2001). The Montreal Process, an international effort for forest conservation and sustainable management, recognizes HRV in disturbed forest area as a benchmark for an indicator of forest health (Montreal Process Working Group 1998). At the national level, HRV was considered as an important concept for defining landscape conditions such as ecosystem diversity for sustainable forest management under the National Forest Management Act of 1976 (Committee of Scientists 2000, *unpublished report*). In the Pacific Northwest, Cissel et al. (1998, 1999) developed forest management plans that utilized the historical fire regime to allocate harvest blocks of different harvest frequency and overstory-removal levels in a watershed. Despite the attention and theoretical appeal, studies of effectively quantifying HRV for assessing conditions of managed landscapes are rather limited (but see Baker 1992a, Wallin et al. 1996, Tinker et al. 2003, Wimberly et al. in press).

Previous HRV analyses were based on simulation models (e.g., Wallin et al. 1996), dendrochronological data (Cissel et al. 1999), historical remotely-sensed imageries (e.g., Spies et al. 1994, Hessburg et al. 1999), and contemporary natural reserves (e.g., Mladenoff et al. 1993, Tinker et al. 2003). The historical dynamics inferred from fire scars and historical records is only one realization of landscape dynamics out of many other possible behaviors under the historical disturbance regime (Lertzman et al. 1998). Stochastic simulation models have the advantage of characterizing a range of possible landscape dynamics for a long period of time and large areas.

HRV has been defined as the bounded variability of a system within constraints imposed by larger-scale phenomena (e.g., climate, topography) and without significant modern human influence (Morgan et al. 1994, Aplet and Keeton 1999). The HRV in landscape structure characterizes the dynamic behavior of landscapes and provides reference ranges of conditions to evaluate landscapes for habitat diversity and arrangement (Aplet and Keeton 1999, Landres et al. 1999). HRV can be used as a "coarse-filter" approach to biodiversity conservation by providing information about habitat amounts, arrangement, and processes that sustain the represented ecosystems (Aplet and Keeton 1999). The implications of recent landscape changes and shift from natural- to anthropogenic-disturbance dominance for native biodiversity are difficult to assess without understanding long-term history of landscape dynamics (Baker 1992a).

Wimberly (2002) and Wimberly et al. (in press) investigated the HRV in amounts of major seral stages and basic landscape patterns at the regional scale in the Oregon Coast Range. They concluded that old-growth forest (> 200 years) was the dominant forest type prior to Euro-American settlement, usually occupying at least 40 % of the landscape on average in large patches (> 2000 ha). The amount of old-growth forests in the region has been considerably reduced and fragmented, while young forest dramatically increased and now comprises the matrix of the landscape with scattered small old-growth patches. They also concluded that the current landscape of the region is outside the HRV for the major seral stages and basic spatial patterns. These previous studies, however, did not examine the full diversity of stand development, especially very young (< 30 years) and very old stages (> 450 years).

Stand conditions may rapidly change after stand-replacing fire, and very young stands may vary substantially in types of habitat they provide, depending on dead wood or legacy structure. Douglas-fir forests over 200 years of age continue to develop structurally and compositionally (Spies and Franklin 1991, Franklin et al. 2002), so that characterizing the HRV of very old forests is needed for a full assessment of biodiversity.

Although quite a few studies examined the potential of the HRV approach to quantitatively assess landscape conditions (e.g., Hessburg et al. 1999, Wimberly et al. 2000, in press, Keane et al. 2002, Wimberly 2002, Tinker et al. 2003), a relatively small number of studies have used HRV to evaluate alternative future scenarios (Wallin et al. 1996, Andison and Marshall 1999, Cissel et al. 1999, Hemstrom et al. 2001, Swanson et al. 2003). Wallin et al. (1996) examined how alternative management scenarios differed in their potentials to return relatively small landscapes to the HRV in the central Oregon Cascade Range. Hemstrom et al. (2001) examined the amounts of major forest types and assessed "landscape health" of historical landscape and landscapes under the 3 alternative management scenarios in the interior Columbia River Basin. These studies found that some management approaches can bring landscapes back toward HRV at regional scales. No studies, however, have examined how currently implemented policies and wildfires would change intensively managed landscapes relative to HRV at regional scales.

Previous work on HRV in the Oregon Coast Range was also limited because it examined only a limited number of landscape metrics and did not explore the correlation structure among the metrics. Numerous landscape metrics are available, and many of them are known to be correlated at various degrees (Li 1989, Hargis et al. 1998, Riitters et al. 1995, Gustafson 1998). Use of multiple metrics may collectively better describe complex landscape structure and changes (O'Neill et al. 1996, Li and Reynolds 1994), but it is important to be certain that the metrics measure the major characteristics of landscape structure (Gustafson 1998). Variability in landscape structure is sensitive to patch classifications (e.g., composition and structural types) and types of metric (Keane et al. 2002). The sensitivity of metrics to regime change differs among metrics, which underlines the importance of using a variety of metrics (Baker 1992a). Use of comprehensive sets of stand development classes and landscape metrics would provide a more complete evaluation of HRV than has been done previously.

The main objective of this study was to evaluate the use of the HRV approach to assess effects of forest management at a regional scale. The specific objectives were to 1) establish the HRV of landscape structure using a wide array of age classes and landscape metrics, 2) compare the current landscape condition with the HRV, and 3) evaluate the similarity of alternative future landscapes to HRV.

METHODS

Study area

The Oregon Coast Range is a 2-million-ha physiographic province in Oregon, USA (see the inset map in Fig. 2.1). The climate is characterized by mild wet winter and dry cool summer (Franklin and Dyrness 1988). As a result of the geographic setting, the western half of the region has a moister climate than the eastern half. The topography is characterized by highly dissected mountains, steep slopes, and a high density of streams. The soils are deep to moderately deep and fine to medium texture, derived from sand stone, shale, or basalt (Franklin and Dyrness 1988). Two major vegetation types, *Picea sitchensis* (sitka spruce) Zone and *Tsuga heterophylla* (western hemlock) Zone, cover the region, juxtaposed with Willamette Valley foothills along the eastern margin (Franklin and Dyrness 1988). The forests are dominated by relatively few species and are highly productive. The modern vegetation composition started to form about 5000 years ago (Whitlock 1992, Worona and Whitlock 1995).

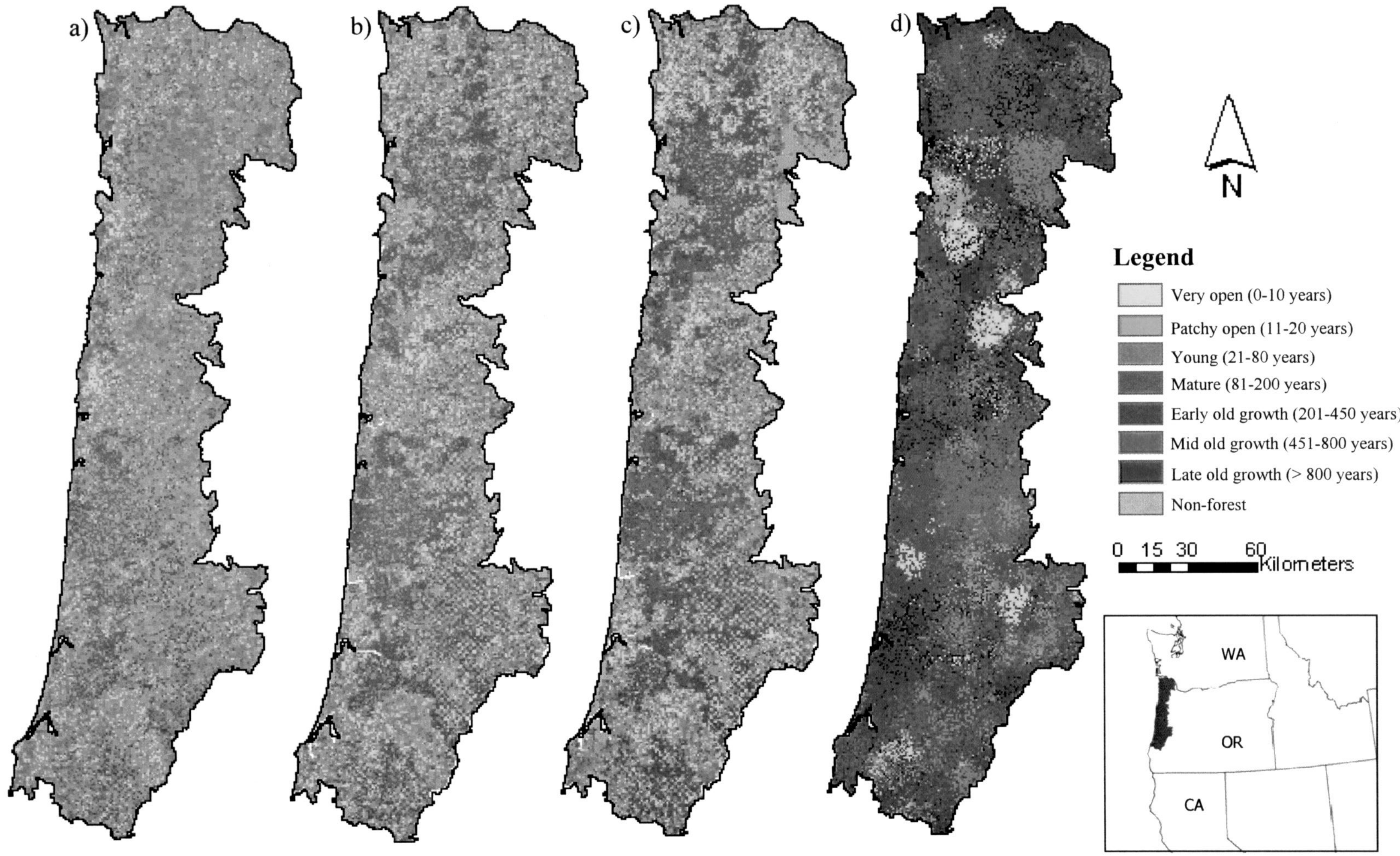

Figure 2.1: The vegetation maps of the Oregon Coast Range Province: a) the current vegetation as of 1996, b) the future projection at year 50, c) year 100, and d) an example of simulated historical vegetation maps from LADS.

Disturbance regimes

Large-scale wildfire was the most important disturbance that shaped forests of the Oregon Coast Range (Agee 1993, Impara 1997, Long et al. 1998, Long and Whitlock 2002). The fire regime was relatively stable for 1000 years prior to Euro-American settlement (Long et al. 1998). In presettlement time, the estimated mean fire-return interval ranges from 150 to 350 years for high-severity fires in this region (Fahnestock and Agee 1983, Agee 1993, Teensma et al. 1991, Ripple 1994, Long et al. 1998, Long and Whitlock 2002). Moderate-severity fires occurred often in mixture with high-severity fires (Impara 1997). High-severity fires often led to stand replacement, while moderate-severity fires left unburned forest patches and single trees (Agee 1993, Impara 1997), which influenced subsequent stand development (Goslin 1997, Weisberg in press). Fires were set by Native Americans in the coastal valleys and adjacent Willamette Valley for agriculture and hunting (Boyd 1996), some of which would have occasionally burned into the foothills of coastal mountains, but the evidence of this is not strong (Agee 1993, Whitlock and Knox 2002). The region experienced more extensive fire occurrences following Euro-American settlement in mid 1800s (Impara 1998, Weisberg and Swanson 2003), and high-severity fires were prevalent in mid 1800 to mid 1900 (Morris 1934, Arnst 1983). Effective fire suppression efforts began in the 1940s in western Oregon (Weisberg and Swanson 2003).

Human influence

The Oregon Coast Range is a mosaic of five major land ownership classes; USDA Forest Service (USFS), USDI Bureau of Land Management (BLM), the State of Oregon, private industrial, and private non-industrial (Fig. 2.2). The two Federal agencies collectively own about 21% of the study area and operate under the Northwest Forest Plan (FEMAT 1993). Current management goals on the Federal lands emphasize protection of late-successional forest and aquatic habitat.

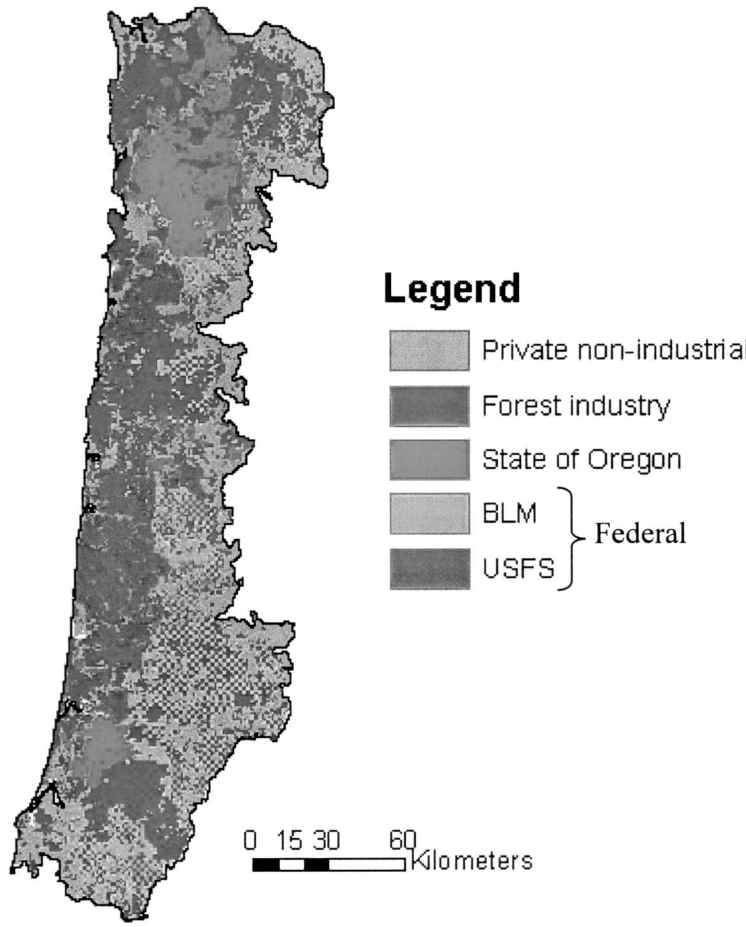

Figure 2.2: The ownership map for the Oregon Coast Range.

Consequently, most of these lands are in late-successional and riparian reserves where timber production is prohibited except thinning to promote late-successional habitat structure in < 80 year-old stands (USDA Forest Service and USDI Bureau of Land Management 1994). Where most of timber harvesting occurs ("matrix" lands), longer rotations (~ 80 years) with green-tree and deadwood retentions are used (USDA Forest Service and USDI Bureau of Land Management 1994).

The State of Oregon lands, about 10% of the Coast Range, are managed under the State Forest Plans. Management standards for the Plans are set by the Oregon Forest Practices Act (Oregon Department of Forestry 1996), but their practices often exceed the standards. For example, the Forest Plan developed for the State forests in northwestern Oregon aims at maintaining diversity in forest structure and patch structure in the landscape context (Bordelon et al. 2000). The management goals are to sustain healthy forests producing abundant timber supply and to maintain productivity, fish and wildlife habitat, air and water quality, and other forest uses. Private industrial landowners own about 33% of the region, and private non-industrial landowners own the remaining 36%. Both private landowners also comply with the Oregon Forest Practices Act. Timber production is the high priority of management on private-industrial lands, and the protection of environment for fish and wildlife required by the Act may constrain their management options. Private-industrial landowners often use clear-cutting and timber rotations of 40 to 50 years. Private non-industrial landowners also manage their lands for timber but use more partial cutting and somewhat longer rotations (Spies et al. 2002b).

Model simulations

Historical landscapes

Historical landscapes were simulated using the Landscape Age-Class Dynamics Simulator (LADS), Version 3.1 (Wimberly 2002). LADS is a spatially-explicit, stochastic computer simulation model designed to simulate forest landscape dynamics under fire regimes specified by the user. The Oregon Coast Range was represented as a grid of 9-ha cells (300m x 300m). LADS was parameterized to the historical fire regimes prior to Euro-American settlement around mid-1800 using reconstructed fire boundary maps, dendrochronological and paleoecological studies in the region (for details, see Wimberly 2002). Fire frequency, severity, and size were all modeled as random variables drawn from specific distributions to reflect variability in fire and uncertainty in fire data. Fire frequency was calculated as a function of natural fire rotation (NFR), and the temporal pattern of decadal fire occurrence was modeled as a Poisson random variable. Fires occurred in a mixture of moderate- and high-severity disturbance, and the proportion of high-severity disturbance within a fire is modeled as a uniform random variable. The proportions of high-severity fire were weighted by susceptibility of topography and vegetation to fire, and the minimum and maximum were specified for various fire size classes. Topographic susceptibility increases with elevation so that it reflects drier conditions in uplands and the tendency of fires to burn uphill. Vegetation susceptibility reflects changes in fuel loads with time since last fire (Agee and Huff 1987). The lognormal probability distributions parameterized by using historical fire maps (Teensma et al. 1991) were used to determine the size of fire as randomly drown from the distributions. The fire shapes were calibrated to match the boundaries of fire events on historical fire maps and satellite imageries. The probability of fire ignition in randomly selected initiation cells is computed also as a function of topographic and vegetation susceptibility to fire.

The region was subdivided into two climate zones, coastal and interior (Fig. 2.3). The climate of the coastal zone is moist and characterized with a greater NFR, while that of the interior zone is dryer and historically more frequently burned (Impara 1997). Fires were more likely to be severe and larger in the coastal zone than in the interior. The model simulates both high- and moderate-severity fires, and fires leave unburned or partially burned forest "islands" within larger burns.

For analysis, I used outputs from 200 model simulations for 1000 years with 10-year intervals. Numerous model runs were necessary to obtain outputs that represent the full range of possible landscape patterns from stochastic models (Keane et al. 2002, Wimberly 2002). Forest stand development was indexed by the time since

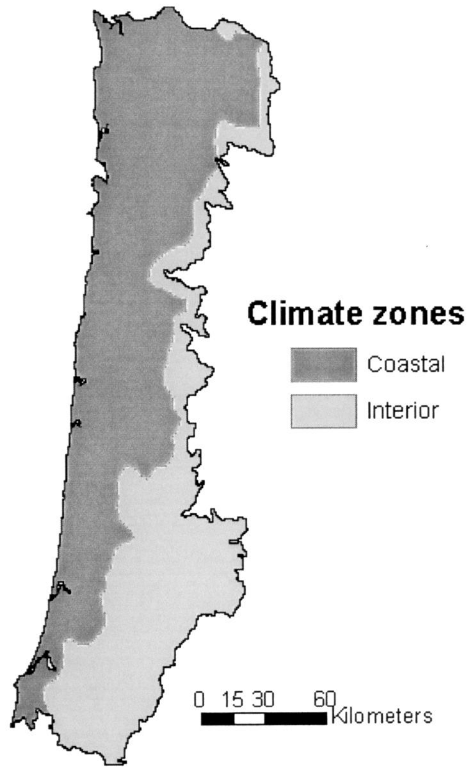

Figure 2.3: The two climatic zones used in the LADS model. The coastal zone is more moist and characterized with more infrequent, catastrophic fires than the interior zone. The NFR was set at 200 years for the coastal zone and 100 years for the interior zone.

the last stand-replacing fire. Conceptually, disturbed stands recover deterministically through stand development over time (see Age Class section). I randomly selected one time step from each simulation for estimating HRV to ensure independence among maps (a total of 200 maps).

Current and alternative future landscapes

The Coastal Landscape Analysis and Modeling Study (CLAMS) provided the current (1996) and alternative future landscape vegetation maps for this study. The current landscape was derived from a statistical model based on satellite imagery, inventory plots, and GIS layers (for details, see Ohmann and Gregory 2002). For comparison with the outputs from LADS, I resampled the map from 25-m to 300-m (9 ha) cell size using ARC/Info RESAMPLE command in the grid module with the nearest-neighbor assignment (ESRI 1995). The accuracy of the map at the 9-ha resolution was 69% with 7 classes (Ohmann and Gregory, *unpublished data*).

Two alternative future management scenarios were a current policy scenario (CPS) and a wildfire scenario (WFS). The CPS was simulated using the Landscape Management and Policy Simulator (LAMPS; Bettinger and Lennette 2002, Bettinger et al. *unpublished manuscript*), a spatially-explicit, grid-based model that projects alternative management scenarios in the Oregon Coast Range into the future. The CPS simulated forest management for 100 years into the future under the policies currently implemented in the region. The CPS assumed that the Federal landowners comply with the Northwest Forest Plan and non-Federal landowners with the Oregon Forest Practices Act. The details of ownership behaviors were varied both among and within the ownership groups to reflect the differences in forest management goals and practices (for details, see Bettinger et al. *unpublished manuscript*). I used outputs for year 50 and 100 from the model simulation for analysis.

LADS was used to simulate the WFS using the current landscape as the initial condition. This scenario was to demonstrate a hypothetical implementation of a natural-disturbance management policy. Forestry practices that emulate natural disturbances are often advocated as a strategy for maintaining landscape conditions within the historical range (e.g. Hunter 1993), but no studies explicitly examined the effectiveness of the strategies on already highly altered landscapes. I ran the model 10 times for 1500 years and analyzed the output maps at every 50 simulation years.

Age classes

Forests in the Pacific Northwest are often described by several developmental stages (Oliver and Larson 1996, Spies and Franklin 1996, Franklin et al. 2002). I grouped the decadal age classes from LADS into 7 age classes based on structural development and ecological function (Table 2.1). I used the age of overstory-cohort, as represented by time since the last high-severity fire (AGE), for the historical landscapes and the age of dominant trees for the current and future landscapes to assign an age class to each pixel.

Landscape metrics

I measured both landscape-level and class-level metrics (Table 2.2). The landscape-level metrics describe overall landscape structure with all classes together, and the class-level metrics describe landscape structure by class (McGarigal et al. 2002). Using FRAGSTATS v3.0 (McGarigal et al. 2002), I measured 16 landscape-level and class-level metrics that are commonly used in ecological literature or identified as important in parsimonious sets of landscape metrics (Li 1989, McGarigal and McComb 1995, Riitters et al. 1995, Gustafson 1998, Cushman et al. *unpublished manuscript*). I used the 8-neighbor rule to define connectivity of adjacent pixels. The major characteristics of landscapes measured by the metrics were 1) amount of class, 2) patch size, variation, and density, 3) edge density and contrast, 4) patch shape, 5) patch type diversity, 6) isolation and connectivity, and 7) contagion and interspersion (Table 2.2).

Table 2.1: The descriptions of forest structure by age classes based on forest dynamics in temperate wet coniferous forests in Pacific Northwest described in literature. AGE is the time (year) since the last stand-replacing fire in years.

AGE (year)	Age class	Description
0-10	Very open	The canopy is very open. The stand can contain very high amount of down wood and snags created by the fire and carried over from previous stand. A new cohort may be establishing at various rates among locations. There may be a few large residual trees that survived the fire.
11-20	Patchy open	The stand can contain very high amount of downwood and snags created by the last fire and carried over from previous stand. The canopy is not yet closed, and small trees may be patchily distributed. Trees may be establishing, but the canopy is mostly single story. There may be few large residual trees.
21-80	Young	The canopy is closed and single story. Tree regeneration is low. The trees are small to medium size. The stand structure is homogeneous. There can be accumulations of well-decayed deadwood from previous stands.
81-200	Mature	The canopy is dominated by shade-intolerant species with low to moderate amounts of shade-tolerants. Some canopy gaps may form to create heterogeneity in the canopy, but stand structure is largely homogeneous. The trees are medium to large size. The amount of deadwood can be low.
201-450	Early old growth	The canopy is dominated by shade-intolerants but heterogeneous with gaps, where shade-tolerant species are regenerating and adding vertical structural diversity. There are more very large (> 100 cm diameter at breast height) trees than the younger stands. Downwood and snags are often abundant. Old-growth characteristics become well developed during this stage.
451-800	Mid old growth	The canopy is a mixture of shade-intolerants and shade-tolerants and is increasingly heterogeneous with gaps, adding both vertical and spatial diversity. There are many very large trees, while the original cohort gradually disappears and is replaced by shade-tolerant species. Downwood and snags are often abundant. Structural development slow down in this age class compared to the younger classes.
>801	Late old growth	The canopy is composed mainly of shade-tolerant and is heterogeneous with shade-tolerant trees at various ages. There may be a few very large survivors of the original cohorts, but they are often senescing. Downwood and snags are abundant. The initial cohort of shade-intolerant species is largely lost from the stand and replaced by shade-tolerant species.

Table 2.2: The landscape metrics used to quantify the landscape structure in the modeled landscapes of Oregon Coast Range, Oregon, USA. The 7 categories represent the important characteristics of landscapes.

Categories	Metric	Abbreviations*	Landscape-level	Class-level
Amount	Percent of Landscape	PLAND	Not used for this level.	Total area of the landscape occupied by patches of the focal class.
	Total core area	TCA	Total area of the landscape occupied by core area (=area within the patch that is further than the specified depth-of-edge distance from the patch perimeter) of patches. The depth-of-edge distance was specified as 300m.	Total area of the landscape occupied by core area (=area within the patch that is further than the specified depth-of-edge distance from the patch perimeter) of patches of the focal class. The depth-of-edge distance was specified as 300m.
Patch size/abundance	Mean patch area	MPA	Mean of patch size distribution of all patches.	Mean of patch size distribution of patches of the focal class.
	CV of patch area	PACV	Coefficient of variation of patch area distribution of all the patches as expressed as percentage of mean.	Coefficient of variation of patch area distribution of the focal class as expressed as percentage of mean.
	Largest patch index	LPI	Percent of the landscape occupied by the largest patch on the landscape.	Percent of the landscape occupied by the largest patch of the focal class.
	Patch density	PD	Number of patches on the landscape per 100 ha.	Number of patches of the focal class per 100 ha.
Edge	Edge density	ED	Edge length of patches of all types on per hectare.	Edge length of patches of the focal class per
	Total edge contrast index	TECI**	Degree of contrast along the patch perimeter of all patches.	Degree of contrast along the patch perimeter of the focal class.
Shape	Perimeter-area fractal dimension	PAFRAC	Degree of patch shape complexity of all the patches.	Degree of patch shape complexity of the focal class.

Table 2.2: continued.

Diversity	Simpson's evenness index	SIEI	Evenness in proportional abundance of classes.	Not used for this level.
Isolation/connectivity	Mean nearest neighbor distance	MNN	Mean of Euclidean nearest neighbor distance distribution between a focal patch and the nearest patch for all patches.	Mean of Euclidean nearest neighbor distance distribution between a focal patch and the nearest patch of the same type.
	CV of nearest neighbor distance	NNCV	Coefficient of variation of Euclidean nearest neighbor distance distribution of all patches as expressed as percentage of mean.	Coefficient of variation of Euclidean nearest neighbor distance distribution of the focal class as expressed as percentage of mean.
	Patch cohesion index	COHESION	Physical connected of the patches of all classes.	Physical connectedness of patches of the focal class.
	Mean proximity index	PROX	Mean isolation of all patches based on proximity to and size of patches of the same class with in the search window. The search radius was specified as 1000m.	Mean isolation of patches of the focal class based on proximity to and size of patches of the same class within the search window. The search radius was specified as 1000m.
	Mean similarity index	SIMI	Mean isolation of all patches based on proximity to and size of patches of the same class within the search window weighted by similarity between classes. The search radius was specified as 1000m.	Mean isolation of patches of the focal class based on proximity to and size of patches of the same class within the search window weighted by similarity between classes. The search radius was specified as 1000m.
Contagion/interspersion	Interspersion and juxtaposition index	IJI	Interspersion or intermixing of patches of all classes.	Juxtapositioning of patches of the focal class with other classes.
	Aggregation index	AI	Cell aggregation of all the class.	Cell aggregation of the focal class.

Notes: McGarigal et al. 2002 for a complete description and definition of each metric.
* The metrics are referred by their abbreviations in the text. The abbreviations follow McGarigal et al. (2002).
** TECI requires an edge contrast matrix, which contains edge contrast weights. I selected values so that the weights reflected the degree of structural contrasts between age classes. For example, the edges between old-growth forests and open stands were assigned the highest weight.

Data Analysis

Principle component analysis of the landscape metrics

I used principle component analysis (PCA) to reduce the number of metrics into the major components of landscape structure and to facilitate visualizing HRV and positions of managed landscapes in relation to HRV (McGarigal et al. 2000, McCune and Grace 2002). PCA was used for similar purposes in previous studies (Milne 1992, Cushman and Wallin 2000, Roworth 2001). I used only the first two principle components (PCs) for ease of interpretation. The analysis was conducted in PC-ORD (McCune and Mefford 1999).

I conducted PCA separately for a set of the landscape-level metrics and 7 sets of class-level metrics (i.e. 7 age classes) for the 200 simulated historical landscapes. I checked the scatter plots of all metric combinations and transformed them by log, square root, or arcsine to linearize intervariable relationships and reduce skewness in the distributions, if necessary. I successfully reduced the skewness to ≤ 1 in the absolute value for most of the metrics. Since these analyses were descriptive and not inferential, the assumptions of linearity and normality were relaxed (McGarigal et al. 2000, McCune and Grace 2002). Because aggregation index (AI) had near perfect correlation with edge density (ED) at the landscape level, the former was excluded from the analysis. The resultant ordinations were rotated by a multiple of 90° without changing the amounts of variation explained by the component axes to facilitate interpretation among different ordinations.

Quantifying the HRV

I defined 90% HRV likelihood for each ordination on the 2-dimensional space of the first 2 PC axes. I used the kernel density estimation method (Seaman and Powell 1996) in ArcView 3.2 with the extension program, Animal Movement SA v. 2.04 beta (Hooge et al. 1999). The least square cross validation option was applied for the smoothing parameter (Seaman and Powell 1996). This method estimates the density surface from the spatial distribution of data points and encloses the specified density within the range. Lundquist et al. (2001) and Roworth (2001) used this method to delineate the range of variability in landscape structure on their ordinations. The HRV likelihood can be considered as a 90% confidence range of historical landscape conditions that could have happened under the historical fire regime (Roworth 2001, Wimberly 2002). I calculated mean, minimum, maximum, and range standardized by maximum (range/maximum*100) for individual metrics measured on the landscapes that fell within the 90% HRV likelihood for each analysis.

Comparing the current and future landscapes with HRV

I projected the current and future landscapes under the two scenarios onto the multivariate ordination spaces. For the WFS, I calculated the statistics at year 50 and 100 from 10 replicate runs and also quantified the amount of time each metric took to reach and stabilize within corresponding HRV.

RESULTS

The HRV in landscape structure and comparison with current conditions

Landscape-level analysis

The first PC (PC1) explained 63% of the variation and was highly correlated with many of the metrics that are related to patch size (LPI, MPA, PACV, TCA), connectivity of classes (COHESION, PROX, SIMI), proximity of patches (MNN, NNCV), and abundance of edge (ED) and patches (PD) (Table 2.3). This axis represented class aggregation and large patch dominance. The second PC (PC2) explained an additional 14% of the variation and was moderately correlated with edge contrast (TECI) and patch juxtaposition (IJI) (Table 2.3). This axis suggested a gradient of intermixing and contrasts among patches of different classes. The eigenvalues of the first two axes were > 1, indicating that these axes individually summarized more information than any single original variable (Table 2.3). The

Table 2.3: Eigenvalues of the principal components and Pearson correlations of the original variables with the PC axes for the ordinations of the simulated historical landscapes of the Oregon Coast Range, Oregon, USA. Correlations > ±0.6 are in boldface type.

	Landscape		Class-level													
			Very Open		Patchy open		Young		Mature		Early old-growth		Mid old-growth		Late old-growth	
	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2	PC1	PC2
Eigenvalue**	9.39	2.05	8.04	2.50	8.37	2.10	8.24	2.07	8.54	2.03	8.42	2.77	9.48	2.22	9.82	2.29
% of variance explained	62.5	13.6	50.9	15.7	52.3	13.1	51.5	12.9	53.4	12.7	52.6	17.3	59.3	13.9	61.4	14.3
Correlations with the PC axes																
PLAND*	NA	NA	**0.96**	-0.11	**0.97**	-0.08	**0.97**	-0.11	**0.97**	-0.11	**0.97**	-0.10	**0.95**	-0.03	**0.95**	-0.21
TCA	**0.87**	-0.32	**0.97**	0.12	**0.98**	0.06	**0.96**	0.05	**0.95**	0.11	**0.94**	0.15	**0.93**	0.24	**0.94**	-0.15
MPA	**0.86**	-0.16	**0.94**	0.22	**0.95**	0.18	**0.96**	0.05	**0.95**	-0.08	**0.96**	0.03	**0.97**	0.14	**0.96**	0.15
PACV	**0.86**	0.21	**0.91**	0.12	**0.93**	0.07	**0.88**	0.05	**0.79**	0.33	**0.77**	0.44	**0.79**	0.51	**0.80**	0.46
LPI	**0.80**	0.27	**0.97**	0.12	**0.98**	0.08	**0.95**	0.03	**0.93**	0.18	**0.88**	0.30	**0.86**	0.42	**0.87**	0.35
PD	**-0.86**	0.17	0.06	**-0.82**	0.09	**-0.76**	-0.50	-0.43	**-0.71**	-0.07	**-0.67**	-0.27	0.43	**-0.75**	**0.71**	**-0.61**
ED	**-0.92**	-0.27	**0.87**	-0.36	**0.91**	-0.25	**0.80**	-0.38	**0.74**	-0.47	**0.77**	-0.53	**0.92**	-0.29	**0.91**	-0.39
TECI	0.34	**-0.65**	0.18	-0.21	0.15	0.16	0.07	0.24	0.13	0.40	0.13	0.44	-0.20	0.04	-0.01	0.28
PAFRAC	**-0.65**	0.44	0.24	-0.50	0.23	-0.41	-0.06	-0.59	-0.23	-0.58	-0.17	**-0.71**	0.26	-0.50	0.36	-0.56
SIEI*	**-0.78**	-0.47	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
MNN	**0.91**	-0.31	-0.23	**0.86**	-0.29	**0.85**	-0.28	**0.77**	-0.47	**0.78**	-0.56	**0.74**	**-0.83**	0.48	**-0.83**	-0.45
NNCV	**0.68**	-0.31	-0.23	**0.67**	-0.15	**0.64**	-0.16	0.59	-0.43	**0.62**	-0.56	**0.62**	**-0.74**	0.53	**-0.73**	0.57
COHESION	**0.91**	0.14	**0.95**	0.13	**0.96**	0.10	**0.98**	0.04	**0.91**	0.15	**0.80**	0.04	**0.90**	0.16	**0.91**	0.29
PROX	**0.81**	0.39	**0.94**	0.07	**0.94**	0.11	**0.94**	0.03	**0.95**	0.03	**0.90**	0.23	**0.95**	0.27	**0.94**	0.27
SIMI	**0.87**	0.34	-0.01	0.11	-0.15	-0.12	-0.08	-0.35	-0.25	0.14	-0.48	0.59	-0.60	0.32	-0.58	-0.13
IJI	-0.48	**-0.62**	0.00	-0.14	0.20	0.10	0.13	0.46	0.30	0.28	0.28	-0.11	0.32	-0.14	0.38	0.40
AI*	NA	NA	**0.93**	0.27	**0.94**	0.20	**0.95**	0.15	**0.94**	0.18	**0.93**	0.22	**0.94**	0.27	**0.92**	0.31

* PLAND was not available for the landscape-level analysis. AI was not used for the landscape-level analysis because of a near perfect correlation with ED. SIEI was not available for the class-level analysis.
** The eigenvalue of > 1 indicates that these axes summarized more information than any single variable. The eigenvalue for the third PC axes were all ≤ 2. The percentage of variance explained by PC3 ranged from 9.2 to 12.3%.

standardized ranges of metrics suggested that historical variability was much higher for largest patch index (LPI) and mean similarity index (SIMI) than any other variables (Table 2.4).

The current landscape was outside the HRV in terms of PC1 but not PC2 (Fig. 2.4a). The current landscape had more aggregated patch configuration than what would be expected under the historical disturbance regime. Patches were more simply shaped (PAFRAC) and distributed at distances of greater variation from the nearest patch of the same class (NNCV), compared to the simulated historical landscapes. Of 16 individual landscape metrics, 15 were outside of the corresponding HRV for the current landscape (Table 2.4). The relatively high contribution of a perimeter-area fractal dimension (PAFRAC) to the departure indicated that the shape of the patches on the current landscape was considerably simpler than that on the historical landscapes (Table 2.5).

Class-level analyses

Very open and patchy open: PC1 explained 51% of the variation for very open and 52% for patchy open and was highly correlated with many metrics that were associated with the amount of class area (PLAND), patch size (LPI, MPA, PACV, TCA), edge density (ED), and connectivity of classes (COHESION, AI, PROX) (Table 2.3). This axis represented a class amount and aggregation gradient. PC2 explained additional 16% of the variation for very open and 13% for patchy open and strongly correlated with mean nearest neighbor distance (MNN) and patch density (PD) (Table 2.3). This axis suggested a gradient of patch proximity and density. The standardized ranges of metrics suggested that historical variability was higher for total core area (TCA), largest patch index (LPI), similarity index (SIMI), and mean proximity index (PROX) than any other variables (Table 2.4).

The current landscape was outside the HRV in terms of PC2 only (Fig. 2.4b, c). There were more patches and shorter mean nearest neighbor distances on the current landscape than would be expected within HRV. The ordination suggested that the current landscape had very high patch density (PD) of these two classes. High

Table 2.4: The mean, minimum, maximum, and standardized range ((range/max)*100) of the metrics from the simulated historical landscapes that fell in the 90% HRV likelihood of the age classes and values of the metrics measured on the current and potential future landscapes of the Oregon Coast Range, Oregon, USA. The time steps for the future landscapes are 50 and 100 year after the initiation of the simulation. See Table 2.2 for the abbreviations and definitions of the metrics.

	Historical						
Metric	Mean	Min	Max	Standardized range*	Current	Yr 50	Yr 100
Landscape							
TCA	450741	345555	622773	44.5	302976	211626	311544
MPA	72	65	84	22.1	61	62	59
PACV	2998	1964	6210	68.4	6651	1595	2635
LPI	11.27	4.07	33.19	87.7	33.60	4.64	9.33
PD	1.38	1.19	1.53	22.1	1.64	1.61	1.68
ED	31.3	28.0	33.7	17.0	34.0	35.8	33.8
TECI	24.3	19.7	30.1	34.5	31.0	34.6	36.5
PAFRAC	1.684	1.676	1.691	0.9	1.632	1.616	1.605
SIEI	0.92	0.83	0.96	13.9	0.81	0.89	0.90
MNN	758	726	797	8.9	792	786	784
NNCV	48.3	39.7	66.4	40.3	87.6	61.8	60.0
COHESION	97.4	96.3	98.7	2.5	98.8	94.9	96.0
PROX	152	71	357	80.2	709	81	125
SIMI	7329	3056	21841	86.0	33610	1946	4278
IJI	83.3	74.6	89.9	16.9	70.9	74.2	73.2
Very open							
PLAND	2.7	0.8	10.4	92.2	8.6	14.5	13.9
TCA	9389	234	77247	99.7	5931	9387	9090
MPA	74	26	289	91.0	26	40	42
PACV	952	176	2546	93.1	409	294	403
LPI	1.00	0.03	9.12	99.6	0.31	0.26	0.46
PD	0.04	0.02	0.06	75.2	0.33	0.36	0.33
ED	1.8	0.8	5.0	84.8	8.0	12.3	11.7
TECI	81.1	59.1	89.6	34.0	34.9	43.2	44.4
PAFRAC	1.691	1.663	1.719	3.3	1.578	1.602	1.609
MNN	763	694	885	21.6	813	724	740
NNCV	86.2	29.3	261.5	88.8	42.1	33.9	36.2
COHESION	89.4	63.2	99.4	36.3	68.9	74.8	79.6
PROX	22	2	264	99.1	3	5	7
SIMI	1938	177	36746	99.5	24407	1346	1814
IJI	87.7	51.3	97.0	47.1	67.3	68.2	68.1
AI	43.9	25.3	75.9	66.7	30.5	36.1	36.9
Patchy open							
PLAND	2.7	0.8	12.1	93.4	18.4	10.7	9.4
TCA	9980	234	123552	99.8	18009	5949	1773
MPA	77	24	419	94.3	36	33	27
PACV	1015	176	2544	93.1	765	260	143
LPI	1.16	0.03	10.97	99.8	0.90	0.12	0.04
PD	0.04	0.02	0.06	62.8	0.51	0.32	0.35
ED	1.8	0.8	4.4	81.9	16.2	9.4	9.0
TECI	54.8	38.1	65.0	41.5	21.7	19.8	21.2
PAFRAC	1.691	1.652	1.720	4.0	1.656	1.591	1.590
MNN	773	702	863	18.6	689	785	762
NNCV	89.6	34.1	191.6	82.2	25.2	40.0	38.3
COHESION	89.9	59.5	99.4	40.1	85.3	70.9	57.5
PROX	29	2	264	99.4	13	3	2
SIMI	5132	707	112282	99.4	45293	2361	3917
IJI	87.7	54.2	97.5	44.4	60.0	67.5	68.5
AI	44.3	23.5	77.5	69.7	34.1	34.0	28.4

* standardized value = ((max – min /max)*100)

Table 2.4: continued.

Metric	Historical Mean	Historical Min	Historical Max	Historical Standardized range	Current	Yr 50	Yr 100
Young							
PLAND	15.3	8.1	27.0	69.9	48.5	35.8	25.0
TCA	77717	13527	249588	94.6	218205	65232	43371
MPA	107	55	243	77.4	297	129	79
PACV	1996	826	3906	78.8	4318	1171	1090
LPI	4.84	0.81	15.57	94.8	33.60	3.03	2.47
PD	0.15	0.11	0.19	43.7	0.16	0.28	0.32
ED	8.7	6.0	11.4	47.6	25.3	24.5	17.8
TECI	39.7	28.6	43.5	34.3	30.7	32.8	33.6
PAFRAC	1.693	1.674	1.706	1.9	1.671	1.668	1.635
MNN	716	693	748	7.5	632	636	678
NNCV	41.9	29.1	71.8	59.5	14.4	15.5	29.4
COHESION	96.8	92.3	99.1	6.9	99.7	96.6	94.2
PROX	115	19	626	97.0	7010	262	77
SIMI	6275	1555	61182	97.5	7578	2651	7276
IJI	83.4	64.6	94.5	31.7	80.5	79.9	74.0
AI	55.6	40.6	71.7	43.3	60.7	48.5	46.5
Mature							
PLAND	23.0	14.5	34.4	57.9	10.8	24.4	31.7
TCA	118249	32688	294804	88.9	12159	72162	151299
MPA	110	64	213	70.1	34	84	116
PACV	2333	1304	4193	68.9	984	2091	3191
LPI	6.64	1.67	19.59	91.5	0.62	4.64	9.33
PD	0.21	0.15	0.28	46.3	0.32	0.29	0.27
ED	13.0	9.9	16.6	40.3	9.3	15.5	16.9
TECI	22.5	20.9	26.7	21.9	32.0	34.8	39.2
PAFRAC	1.693	1.677	1.707	1.8	1.643	1.611	1.594
MNN	710	683	743	8.1	817	752	722
NNCV	35.7	27.5	48.8	43.5	47.7	33.7	30.9
COHESION	97.6	94.7	99.3	4.6	88.7	97.3	98.7
PROX	236	39	1175	96.7	15	178	656
SIMI	7172	2020	52933	96.2	44879	1646	1540
IJI	82.5	56.7	91.6	38.1	66.9	72.1	81.5
AI	56.4	44.6	70.6	36.8	35.5	52.1	59.9
Early old-growth							
PLAND	28.3	18.1	49.6	63.5	2.0	2.9	5.9
TCA	143905	37134	486216	92.4	369	477	3537
MPA	97	46	236	80.6	17	20	31
PACV	2741	1348	5270	74.4	168	166	416
LPI	8.11	1.61	32.87	95.1	0.04	0.03	0.24
PD	0.30	0.21	0.40	46.9	0.12	0.15	0.19
ED	16.5	12.5	19.9	37.3	2.2	3.1	5.4
TECI	21.2	15.5	35.0	55.7	49.6	38.2	32.6
PAFRAC	1.688	1.673	1.701	1.6	1.600	1.624	1.633
MNN	710	675	754	10.4	987	1013	879
NNCV	32.6	22.2	44.6	50.2	90.8	92.9	77.2
COHESION	97.8	94.7	99.5	4.8	45.4	51.8	75.4
PROX	365	59	2320	97.4	1	1	4
SIMI	6496	2038	31627	93.6	28589	2595	9539
IJI	85.1	72.8	92.6	21.4	71.8	62.5	50.6
AI	55.0	40.0	71.5	44.0	18.9	21.6	31.3

Table 2.4: continued.

Metric	Historical				Current	Yr 50	Yr 100
	Mean	Min	Max	Standardized range			
Mid old-growth							
PLAND	16.2	7.9	29.5	73.3	0.2	0.2	0.4
TCA	48282	6786	140571	95.2	18	9	81
MPA	47	27	105	74.5	14	14	17
PACV	2002	848	5166	83.6	112	90	163
LPI	3.10	0.65	13.82	95.3	0.01	0.00	0.01
PD	0.35	0.28	0.42	32.9	0.01	0.01	0.02
ED	12.1	7.5	17.6	57.6	0.2	0.2	0.4
TECI	15.6	10.4	25.7	59.6	44.9	30.7	22.0
PAFRAC	1.682	1.670	1.695	1.4	1.616	1.574	1.611
MNN	770	712	854	16.6	1907	1743	1454
NNCV	42.7	31.8	61.3	48.1	317.9	155.9	109.2
COHESION	94.0	84.7	98.8	14.2	34.2	30.4	47.5
PROX	77	9	571	98.4	0	0	1
SIMI	8880	3103	30503	89.8	25846	5730	10674
IJI	81.6	70.9	90.9	22.0	71.5	64.6	56.3
AI	42.4	28.6	56.3	49.1	14.3	14.8	20.0
Late old-growth**							
PLAND	10.5	3.4	16.2	79.3	0.0	0.0	0.05
TCA	29715	1440	74988	98.1	0	0	0
MPA	34	17	49	65.7	NA	NA	10
PACV	1609	285	3530	91.9	0	0	37
LPI	1.71	0.10	5.27	98.1	0.00	0.00	0.00
PD	0.31	0.20	0.40	50.4	0.00	0.00	0.00
ED	8.4	3.5	12.5	71.8	0.0	0.0	0.1
TECI	13.4	7.8	23.8	67.1	NA	NA	17.6
PAFRAC	1.661	1.639	1.677	2.3	NA	NA	1.350
MNN	848	763	1017	25.0	NA	NA	2538
NNCV	52.1	38.8	78.5	50.6	0.0	0.0	141.2
COHESION	89.1	54.2	96.6	43.9	NA	NA	9.0
PROX	30	1	124	99.1	NA	NA	0
SIMI	9977	3118	61059	94.9	NA	NA	15931
IJI	79.9	71.2	92.3	22.9	NA	NA	53.0
AI	37.6	18.5	49.6	62.7	NA	NA	6.6

** Late old growth did not appear until year 100.

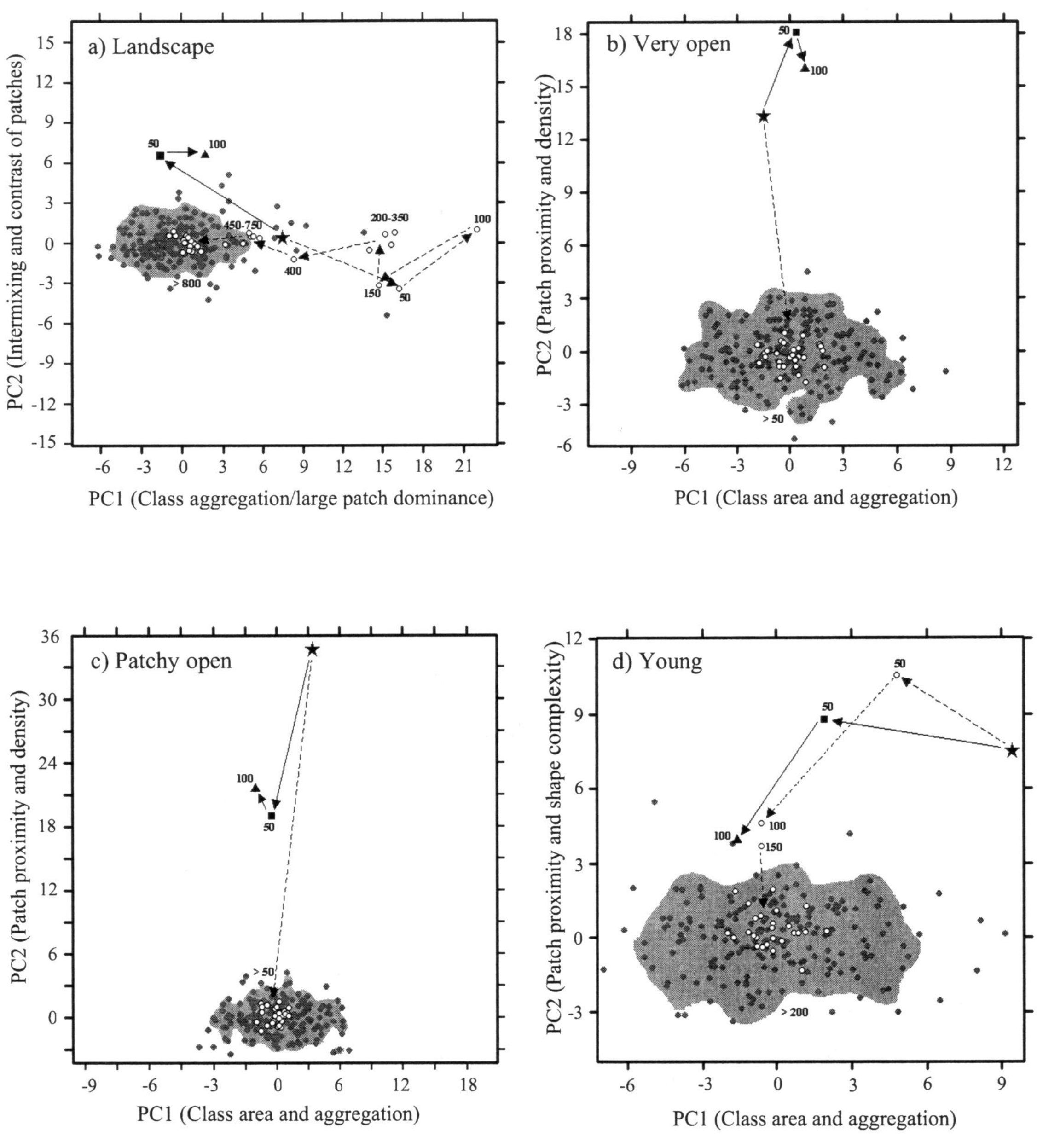

Figure 2.4: PCA ordination results for the landscape-level and class-level analyses. The shaded areas are the 90% HRV likelihood delineated by the density estimation method: ★ = current, ■ = year 50, ▲= year 100, • = simulated historical landscapes, ○ = landscapes from the wildfire scenario simulation. The trajectories of management (solid arrow) and wildfire scenarios (dashed arrows) are also shown on the ordination. The numbers indicate the time steps in the simulations. The numbers with > indicate the time for landscapes to reach and stabilize HRV. Note the difference in scale among the figures.

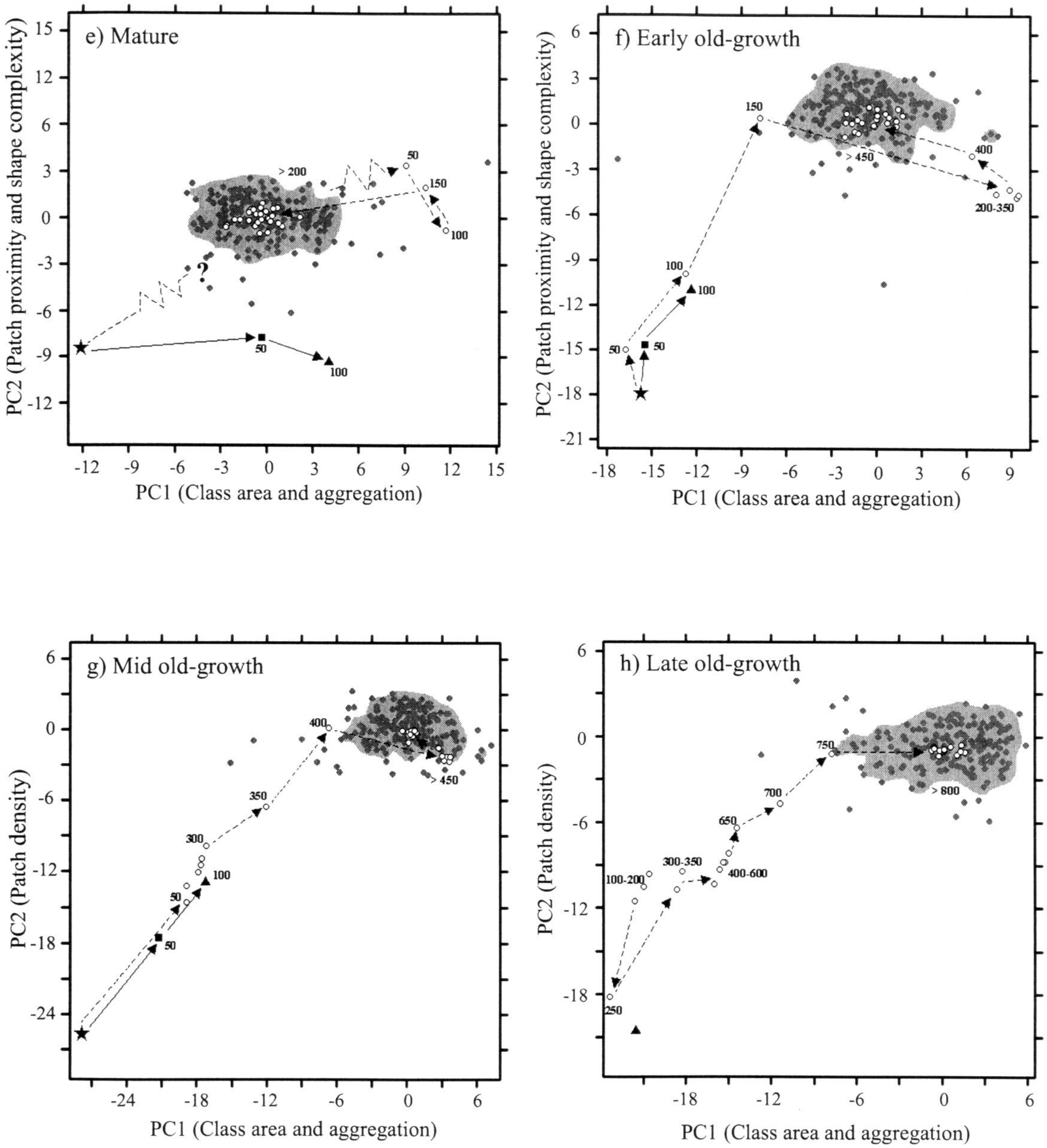

Figure 2.4: continued.

Table 2.5: PC scores and linear combinations for the current, year 50, and year 100 landscapes for the Oregon Coast Range, Oregon, USA. The values for the metrics are eigenvector x standardized value* of the metrics. The sum of those values makes the PC score. See Table 2.2 for the abbreviations of the metrics.

	Current		Yr 50		Yr 100	
	PC1	PC2	PC1	PC2	PC1	PC2
Landscape						
TCA	-0.686	-0.539	-1.283	-1.008	-0.640	-0.503
MPA	-0.705	-0.277	-0.632	-0.248	-0.814	-0.320
PACV	0.822	-0.423	-0.694	0.357	-0.161	0.083
LPI	0.623	-0.458	-0.505	0.371	-0.107	0.079
PD	-0.806	-0.330	-0.713	-0.292	-0.944	-0.387
ED	-0.460	-0.286	-0.735	-0.457	-0.442	-0.275
TECI	0.271	1.105	0.419	1.711	0.499	2.035
PAFRAC	2.768	3.987	3.596	5.180	4.176	6.015
SIEI	0.636	-0.825	0.176	-0.229	0.138	-0.179
MNN	0.466	0.344	0.373	0.275	0.338	0.250
NNCV	1.554	1.505	0.519	0.503	0.449	0.434
COHESION	0.677	-0.230	-1.292	0.439	-0.770	0.262
PROX	1.002	-1.021	-0.419	0.427	-0.132	0.134
SIMI	0.822	-0.687	-0.723	0.605	-0.296	0.248
IJI	0.507	-1.398	0.381	-1.051	0.420	-1.158
PC score	7.490	0.464	-1.532	6.582	1.714	6.717
Very open						
PLAND	0.712	0.149	1.004	0.210	0.982	0.206
TCA	0.077	-0.016	0.187	-0.040	0.179	-0.038
MPA	-0.512	0.213	-0.281	0.116	-0.254	0.105
PACV	-0.371	0.085	-0.543	0.124	-0.378	0.086
LPI	-0.164	0.037	-0.206	0.047	-0.058	0.013
PD	0.652	16.659	0.729	18.631	0.661	16.888
ED	1.201	0.886	1.541	1.137	1.501	1.107
TECI	-0.458	-0.964	-0.389	-0.819	-0.378	-0.797
PAFRAC	-0.938	-3.537	-0.739	-2.786	-0.678	-2.557
MNN	-0.107	-0.715	0.085	0.565	0.051	0.342
NNCV	0.120	0.628	0.162	0.847	0.149	0.779
COHESION	-0.703	0.169	-0.549	0.132	-0.409	0.098
PROX	-0.587	0.076	-0.392	0.050	-0.271	0.035
SIMI	-0.007	-0.222	0.000	-0.012	-0.001	-0.033
IJI	0.003	-0.277	0.003	-0.267	0.003	-0.269
AI	-0.361	0.191	-0.225	0.119	-0.207	0.110
PC score	-1.441	13.361	0.388	18.057	0.892	16.075
Patchy open						
PLAND	1.115	0.184	0.821	0.135	0.754	0.124
TCA	0.346	-0.041	0.097	-0.012	-0.174	0.021
MPA	-0.299	0.115	-0.344	0.133	-0.460	0.177
PACV	-0.047	0.007	-0.584	0.084	-0.882	0.127
LPI	0.123	-0.019	-0.400	0.062	-0.675	0.104
PD	1.781	30.782	1.072	18.520	1.192	20.596
ED	1.746	0.973	1.326	0.739	1.293	0.720
TECI	-0.321	0.713	-0.339	0.753	-0.326	0.724
PAFRAC	-0.226	-0.819	-0.651	-2.354	-0.655	-2.368
MNN	0.216	1.267	-0.029	-0.172	0.030	0.175
NNCV	0.087	0.744	0.067	0.573	0.070	0.593
COHESION	-0.207	0.042	-0.607	0.123	-0.909	0.184
PROX	-0.064	0.014	-0.478	0.107	-0.639	0.143
SIMI	-0.187	0.283	0.029	-0.044	-0.008	0.012
IJI	-0.278	0.276	-0.216	0.214	-0.207	0.205
AI	-0.258	0.108	-0.259	0.108	-0.391	0.164
PC score	3.525	34.628	-0.496	18.970	-1.988	21.702

Table 2.5: continued.

	Current		Yr 50		Yr 100	
	PC1	PC2	PC1	PC2	PC1	PC2
Young						
PLAND	1.240	0.289	0.919	0.214	0.540	0.126
TCA	0.572	-0.065	-0.006	0.001	-0.202	0.023
MPA	0.886	-0.087	0.178	-0.017	-0.239	0.024
PACV	0.706	-0.083	-0.416	0.049	-0.478	0.057
LPI	0.896	-0.054	-0.112	0.007	-0.198	0.012
PD	-0.187	0.316	-1.289	2.182	-1.665	2.819
ED	1.757	1.664	1.703	1.613	1.179	1.116
TECI	-0.092	0.643	-0.071	0.492	-0.062	0.433
PAFRAC	0.073	-1.524	0.082	-1.710	0.190	-3.969
MNN	0.671	3.656	0.634	3.453	0.301	1.642
NNCV	0.364	2.673	0.340	2.496	0.120	0.881
COHESION	0.770	-0.067	-0.084	0.007	-0.478	0.042
PROX	1.635	-0.092	0.394	-0.022	-0.068	0.004
SIMI	-0.016	0.145	0.033	-0.296	-0.014	0.128
IJI	-0.026	0.180	-0.031	0.213	-0.075	0.523
AI	0.210	-0.065	-0.323	0.100	-0.408	0.127
PC score	9.457	7.528	1.950	8.783	-1.560	3.987
Mature						
PLAND	-1.005	-0.237	0.067	0.016	0.415	0.098
TCA	-1.320	0.309	-0.269	0.063	0.168	-0.039
MPA	-0.990	-0.163	-0.234	-0.039	0.032	0.005
PACV	-0.514	0.434	-0.115	0.097	0.281	-0.237
LPI	-1.070	0.424	-0.125	0.049	0.203	-0.081
PD	-0.941	0.185	-0.688	0.135	-0.551	0.108
ED	-0.588	-0.764	0.402	0.522	0.624	0.811
TECI	0.338	-2.161	0.419	-2.683	0.534	-3.418
PAFRAC	0.647	-3.326	1.069	-5.493	1.303	-6.692
MNN	-1.059	-3.600	-0.412	-1.399	-0.114	-0.388
NNCV	-0.359	-1.068	0.058	0.172	0.140	0.417
COHESION	-2.540	0.841	-0.111	0.037	0.298	-0.099
PROX	-0.973	0.067	-0.044	0.003	0.445	-0.031
SIMI	-0.294	-0.332	0.204	0.230	0.214	0.241
IJI	-0.310	0.582	-0.217	0.408	-0.029	0.054
AI	-1.074	0.432	-0.258	0.104	0.127	-0.051
PC score	-12.052	-8.376	-0.253	-7.776	4.092	-9.301
Early old-growth						
PLAND	-1.437	-0.246	-1.386	-0.237	-1.221	-0.209
TCA	-1.239	0.348	-1.230	0.345	-1.096	0.308
MPA	-1.255	0.079	-1.178	0.074	-0.943	0.059
PACV	-0.800	0.795	-0.800	0.796	-0.723	0.720
LPI	-0.901	0.540	-0.905	0.542	-0.796	0.477
PD	1.278	-0.897	1.074	-0.754	0.734	-0.516
ED	-1.966	-2.367	-1.842	-2.217	-1.512	-1.820
TECI	0.258	-1.539	0.179	-1.070	0.132	-0.789
PAFRAC	0.879	-6.281	0.633	-4.524	0.546	-3.905
MNN	-2.150	-4.924	-2.349	-5.379	-1.305	-2.988
NNCV	-2.190	-4.271	-2.272	-4.430	-1.674	-3.264
COHESION	-2.649	0.204	-2.429	0.187	-1.508	0.116
PROX	-0.662	0.298	-0.656	0.295	-0.619	0.279
SIMI	-0.420	-0.911	0.227	0.493	-0.124	-0.269
IJI	-0.267	-0.186	-0.423	-0.295	-0.603	-0.421
AI	-1.671	0.679	-1.554	0.632	-1.128	0.458
PC score	-15.192	-18.679	-14.909	-15.543	-11.839	-11.764

Table 2.5: continued.

	Current		Yr 50		Yr 100	
	PC1	PC2	PC1	PC2	PC1	PC2
Mid old-growth						
PLAND	1.018	-0.056	1.019	-0.056	1.004	-0.055
TCA	0.870	0.458	0.875	0.461	0.849	0.447
MPA	1.171	0.346	1.170	0.345	0.966	0.285
PACV	0.591	0.785	0.599	0.795	0.576	0.764
LPI	0.703	0.701	0.712	0.711	0.689	0.688
PD	1.259	-4.607	1.261	-4.615	1.220	-4.463
ED	1.389	-0.889	1.390	-0.890	1.362	-0.873
TECI	0.757	-0.343	0.390	-0.177	0.163	-0.074
PAFRAC	0.799	-3.181	1.303	-5.187	0.858	-3.417
MNN	4.404	-5.286	3.966	-4.760	3.079	-3.695
NNCV	10.045	-15.007	4.116	-6.149	2.405	-3.593
COHESION	1.718	0.645	1.796	0.674	1.427	0.536
PROX	1.364	0.814	1.382	0.825	1.157	0.691
SIMI	0.374	-0.411	-0.134	0.147	0.076	-0.083
IJI	0.218	-0.201	0.351	-0.323	0.499	-0.460
AI	1.095	0.665	1.075	0.652	0.874	0.530
PC score	27.775	-25.568	21.272	-17.548	17.205	-12.772
Late old-growth						
PLAND	NA	NA	NA	NA	-0.936	-0.419
TCA	NA	NA	NA	NA	-0.842	0.283
MPA	NA	NA	NA	NA	-1.284	0.422
PACV	NA	NA	NA	NA	-0.583	0.693
LPI	NA	NA	NA	NA	-0.738	0.612
PD	NA	NA	NA	NA	-1.388	-2.470
ED	NA	NA	NA	NA	-1.114	-0.981
TECI	NA	NA	NA	NA	-0.004	-0.279
PAFRAC	NA	NA	NA	NA	-4.412	-14.350
MNN	NA	NA	NA	NA	-4.077	-4.602
NNCV	NA	NA	NA	NA	-1.642	-2.693
COHESION	NA	NA	NA	NA	-1.552	1.008
PROX	NA	NA	NA	NA	-1.492	0.874
SIMI	NA	NA	NA	NA	-0.205	0.095
IJI	NA	NA	NA	NA	-0.551	1.219
AI	NA	NA	NA	NA	-1.164	0.798
PC score	NA	NA	NA	NA	-21.984	-19.790

* standardized value = (raw value – mean of HRV)/(standard deviation of HRV) (McCune and Grace 2002)

patch density was the major factor that put the current landscape outside the HRV along PC2 for both the classes (Table 2.5). Of 16 individual metrics, 4 metrics for very open and 6 for patchy open were outside the corresponding HRV (Table 2.4).

Young, mature, and early old growth: PC1 explained 52 % of the variation for young and 53% for both mature and early old growth and had a set of correlated variables similar to the youngest two classes, representing a class amount and aggregation gradient (Table 2.3). PC2 explained additional 13% of the variation for young and mature and 17% for early old growth (Table 2.3). PC2 was moderately correlated with mean nearest neighbor distance (MNN), CV (coefficient of variation) of nearest neighbor distance (NNCV), and patch shape (PAFRAC) (Table 2.3). This axis represented a gradient of patch proximity and patch shape complexity. The standardized ranges of metrics suggested that historical variability was higher for mean proximity index (PROX), mean similarity index (SIMI), largest patch index (LPI), and total core area (TCA) than any other variables (Table 2.4).

The current landscape was outside the HRV in terms of both PC1 and PC2 (Fig. 2.4 d, e, f). In the current landscape, young forests were better connected and had larger patch areas and simpler patch shapes than in the historical landscapes. The mature and early old-growth forests were less abundant and occurred in fewer and smaller patches that were more isolated and simpler in shape than what would be expected under the historical fire regime. Many individual metrics were outside the HRV (Table 2.4). Mean nearest neighbor distance (MNN) and CV of nearest neighbor distance (NNCV) had relatively high contribution to the deviation for young class (Table 2.5). For mature and early old-growth classes, many metrics evenly contributed to the deviation on the ordination along PC1, and mean nearest neighbor distance (MNN) and patch shape (PAFRAC) mainly contributed to the deviation along PC2 (Table 2.5).

Mid and late old growth: PC1 explained 53% of the variation for mid old growth and 61% for late old growth. PC1 had a set of correlated variables similar to the previous classes and represented a class amount and aggregation gradient (Table 2.3). PC2 explained an additional 14% of the variation for both classes. PC2 was moderately correlated with patch density (PD) and represented a patch density gradient (Table 2.3). The standardized ranges of metrics suggested that historical variability was higher for mean proximity index (PROX), largest patch index (LPI), total core area (TCA), and mean similarity index (SIMI) than other variables.

The current landscape was outside the HRV in terms of both PC1 and PC2 for mid old growth (Fig. 2.4g). Late old growth did not occur on the current landscape (Table 2.4). Mid old-growth forests were less common and occurred in patches that were fewer, smaller, more isolated, and simpler in shape than occurred in the HRV simulations. Most of the metrics were outside HRV for this class (Table 2.4). Many metrics evenly contributed to the deviation on the ordination along PC1, and CV of nearest neighbor distance (NNCV) contributed highly to the deviation along PC2 (Table 2.5).

Future scenarios: current policy (CPS) and wildfire (WFS)

<u>Landscape-level analysis</u>

Under the CPS, the landscapes did not return to the HRV in 100 years (Fig. 2.4a). The CPS, however, brought the landscape condition within HRV in terms of class aggregation and large patch dominance (PC1) but not patch contrast and intermixing (PC2). Edge contrast (TECI), patch juxtaposition (IJI), and patch shape (PAFRAC), which were important variables for the ordination, either did not move much toward HRV (IJI) or moved away (TECI and PAFRAC) from the HRV (Table 2.4).

The WFS continuously moved the landscape away from the HRV in terms of class aggregation and large patch dominance in the first 100 years (Fig. 2.4a). After 100 years, the landscape gradually moved back toward HRV and stabilized within HRV after 800 years. The simulation sequence showed that, under the WFS, the landscapes became more homogeneous and occupied by large patches of mature forests in the first 100 years followed by breaking up of the large mature patch into smaller patches of various ages (data not shown).

Class-level analyses

Under the CPS, none of the age classes returned to the HRV in the 100 years. With the exception of very open, however, landscape condition moved toward the HRV (Fig. 2.4b-h). The very open class moved further from the HRV because it increased in patch density (PD) along with area (PLAND). The patchy open class approached the HRV mainly due to considerable decrease in patch density (Table 2.4). The landscape pattern of the young class moved considerably toward the HRV because many metrics, with the exception of patch density and patch shape (PAFRAC), approached their HRV. After 100 years, young patch density was higher and shape was simpler than at year 0. For the mature class, most metrics, with the exception of patch shape, moved substantially toward HRV. As a result, the landscape noticeably approached the HRV in terms of class area and aggregation, but not in the direction of patch proximity and shape complexity. Patch shape of mature forests consistently became simpler over time. All old-growth classes were very rare on the current landscape so that change in landscape pattern was more sensitive to increase in area than to change in configuration. The late old-growth class did not appear for the first 100 years of the simulation.

Under the WFS, there were three general trajectories. The very open and patchy open classes returned to the HRV by year 50 (Fig. 2.4b, c). The young, mid, and late old-growth classes more or less moved consistently toward the HRV and reached it by year 200, 450, and 800, respectively (Fig. 2.4d, g, h). The mature and early old-growth classes overshot the HRV at year 50 and 200 and then returned to the HRV by year 200 and 450, respectively (Fig. 2.4e, f). The different trajectories reflected the changes associated with the development of existing young forests into older forests during simulation time. The large patches of young forests became large patches of mature and early old-growth forests in the first few centuries of the simulation. A couple of centuries of the wildfire regime were needed to break these large patches into smaller patches of various age classes that were characteristic of the simulated historical landscapes.

Time to reach HRV under the WFS

The amount of time it took for the landscape metrics to reach and stabilize in HRV varied by metric and age class from 50 to 800 years (Table 2.6). For the landscape-level analysis, PD, MPA, IJI, and NNCV took > 700 years to return to the HRV, but MNN and PAFRAC took only 100 and 150 years. For the class-level analysis, in general, older age classes took longer than younger forests to reach the HRV, and the amount of time more or less matched the age of the forests although mid and late old growth showed erratic responses probably because of their small area. The interspersion and juxtaposition index (IJI), a measure of intermixing of patch types based on patch adjacency, generally took longest time of all metrics to reach HRV. The only exception for IJI was the late old-growth class probably because this class area was typically so small that it was easier for this class to have a variety of neighbors in relatively equal amounts.

DISCUSSION

Historical landscape dynamics of very young and very old forests in the Oregon Coast Range

The simulations indicated that the historical disturbance regime during 1000 years prior to Euro-American settlement was characterized by a variety of structurally diverse landscapes. The proportions of the 7 age classes fluctuated from uneven to relatively even distributions as indicated by Simpson's evenness index. Patch shape was quite complex, and class area was highly aggregated. The high values for physical connectivity (COHESION) and patch juxtaposition (IJI) and the output maps suggest that the historical landscapes had many large patches (coarse-grained) and intermixing of patch types. Patchiness and juxtaposition of different habitat types are important characteristics of landscapes for regional biodiversity (Forman and Godron

Table 2.6: The number of years for each metric to reach the HRV. See Table 2.2 for the full names and definitions of the metrics. PLAND and AI were not included in the landscape-level analysis, and SIEI was not included in the class-level analyses.

Metric	Landscape-level	Class-level: Very open	Patchy open	Young	Mature	Early OG	Mid OG	Late OG
PLAND	NA	50	50	100	200	450	750	800
TCA	400	0	0	50	200	450	750	800
MPA	750	50	0	50	200	200	450	800
PACV	400	0	0	50	200	450	450	800
LPI	400	0	0	50	200	450	350	800
PD	700	50	50	100	200	400	400	800
ED	450	50	50	100	200	450	400	800
TECI	400	50	200	200	150	400	650	400
SIEI	450	NA	NA	NA	NA	NA	NA	NA
PAFRAC	150	50	50	100	200	450	800	750
MNN	100	0	50	100	200	150	400	800
NNCV	700	0	50	100	200	400	400	800
COHESION	400	50	0	50	200	450	450	800
PROX	400	50	0	100	200	450	450	800
SIMI	400	50	400	450	450	200	450	450
IJI	750	50	450	650	750	800	800	400
AI	NA	0	0	0	200	450	750	800

1986, Angelstam 1997, Pickett and Rogers 1997), and create different types of ecotones and edge habitats by various combinations of edges (Hunter 1990, McGarigal and McComb 1995). Many studies showed that a mixture of forest conditions is associated with higher species diversity in this region (e.g., McGarigal and McComb 1995, Martin and McComb 2002, Cushman and McGarigal 2003).

The very open and patchy open classes tended to be infrequent components of the landscape, each occupying about 3% of the region on average. These two extreme age classes had not been separately quantified by the previous studies. The previous studies (Wimberly 2002, Wimberly et al. in press) reported a median of 17% for early-successional (< 30 years). The difference arose because they not only used different age class definitions but also considered that both high-severity and moderate-severity fires reset the stand development. In this study, moderate-severity fire did not affect age. These two young classes were ephemeral on the landscape and blinked on and off as fires burn and vegetation filled in.

The dominant cover type on the landscape was early old-growth forests (201-450 years), occupying 28% of the Oregon Coast Range on average. Together with mid and late old growth, old-growth forests ($\geq$ 200 years) historically comprised 54% of the region on average. The previous studies (Wimberly 2002, Wimberly et al. in press) found that old-growth forest was the most dominant cover type and covered 42% of the region on average (29% - 52% for 90% interquartile range) using different age class definitions as mentioned above. Other studies showed that old-growth forests covered about 40% of the region around 1850 (Teensma et al. 1991, Ripple 1994) and approximately 61% before the widespread fires in 1840s (Ripple 1994).

Mid and late old-growth forests were consistently present on the historical landscapes although the fire rotation periods were shorter than their age. Differentiating old-growth forests revealed that early old-growth forests were about two to three times more abundant than the two oldest classes, but those two oldest were not uncommon under the historical fire regime. The patches of mid and late old-growth forests occurred where fires failed to completely burn for a long time, relative to the simulated fire rotation period. These types, which can be considered "remnant patches" (Forman 1995) collectively occupied 26% of the region on average. Remnant patches can be created either by chance or because they occur in environments that are not fire-prone (Zackrisson 1977, Angelstam 1997). Angelstam (1997) suggested that landscapes with infrequent fire regime (< 1 fire per century) may have both types of remnant patches. In LADS, the parameters used did not simulate strict fire-refugia, but the wetter climate zone and lower slope positions had lower probability of fire. Even without fire-refugia, the landscape still had a considerable amount of remnant patches sometimes in large size.

The patch characteristics of mid and late old-growth classes were somewhat different from other classes. The two oldest classes had high patch density and mean nearest neighbor distance and low core area. These metrics collectively imply that these two classes were often more isolated and occurred in smaller patches than other classes. Also, patch shape tend to be simpler than other classes on average. The difference in fractal dimension (PAFRAC) indicates that pattern generating processes for these two oldest classes may be different from other classes (Krummel et al. 1987). The pattern-generating process for the two classes includes aging of patches and fragmentation of large patches caused by fires. In contrast, younger patches are formed by the footprints of more or less single fire events. Scattered remnant patches can be important habitat refugia on landscapes characterized with infrequent, high-severity fires (Delong and Kessler 2000) and may provide critical source habitat from which individuals survived fires can disseminate to colonize younger patches around them (Peterken and Game 1984, Matlack 1994, Sillett et al. 2000). Wimberly and Spies (2002) showed in their simulation study that the post-fire recruitments of fire-sensitive, western hemlock was sensitive to the abundance of remnant patches in a small watershed in the Oregon Coast Range.

Although the extent of the study area was large relative to the mean fire size, occasional large fires created unbalanced age class distributions leading to non-equilibrium behavior. These relatively extreme conditions increased the range of variability in landscape structure. According to the models by Shugart (1984) and Turner et al. (1993), my study area falls in between "equilibrium" and "non-equilibrium' landscape types. Shugart (1984) suggested that if a landscape is 50 times larger than disturbance area, the landscape is often in quasi-equilibrium, where forest age distribution in the landscape is stable. The extent of my study area is 320 times larger than the mean fire size (7300 ha) but only 11 times larger than the largest fire size (9% of the landscape) observed in the 1000-year simulation. The degree of equilibrium of landscapes depended on the mean size of fires as well as the size of largest fires.

Baker (1989) observed "mosaic of different non-steady-state mosaics" in the Boundary Waters Canoe Area, Minnesota. He attributed the non-equilibrium to "mismatch between disturbance grain and environmental grain" on the fire-prone landscape. On such landscapes, environment (e.g. topography, fuel conditions, ignition probability) is heterogeneous, and some parts of a landscape simply do not burn. On the other hand, even a heterogeneous environment may not hinder the spread of disturbance under extreme conditions, resulting in relatively homogeneous vegetation cover. He stated that the possibility of equilibrium in fire-prone landscapes may be low. Even larger variation in landscape structure can be expected on the rugged terrain of Oregon than the northern Minnesota landscapes (Baker 1989, Wallin et al. 1996). Although the historical landscape conditions of the Oregon Coast Range were reasonably bounded, there were variations within the range, and occasional extreme conditions were also characteristics of the landscape dynamics.

Management effects on current landscape structure and dynamics

Forest harvesting dominates the disturbance regime in the current landscape, and extensive and intensive forest management in the last century created the landscape that little resembles the ones shaped by historical wildfires. The multivariate ordinations indicated that the current landscape is outside the HRV. On the simulated historical landscapes, most of the patches were well aggregated and had complex shapes because of the presence of internal unburned patches, or islands, and convoluted edges. In contrast, the current landscape has highly simplified patch shapes, skewed forest age distribution to young forests, and reduced amount of older forests. The class-level analyses indicated departures in patch shape, patch density, aggregation of cells, and nearest neighbor distance on the current landscape.

In the Pacific Northwest, extensive logging occurred on private lands in the first half of 1900s, and dispersed patch cutting or checkerboard pattern of clear-cutting (30-50 acres per patch) was common on the Federal lands after WWII until early 1990s (Franklin and Forman 1987, Swanson and Franklin 1992). Prevalence of dispersed patch cut altered the landscape structure by increasing edge and decreasing interior forests in Pacific Northwest forests (Franklin and Forman 1987). Since early 1990s, timber harvest has nearly ceased on the Federal lands after the implementation of the Northwest Forest Plan (FEMAT 1993). Larger clear-cuts are more common on private lands than on public lands.

Between 1972 and 1995, cut units were larger and more spatially aggregated on private industrial lands (mean 22 ha, SD 220 ha) than public and private non-industrial ownerships (mean 8–13 ha, SD 8–14 ha), but patch size of cut units on any ownership was much smaller than fire-burned patches (Cohen et al. 2002). According to the simulation results, the mean area of burned patches (the mean patch area of the very open class) was 74 ha (the size was quite smaller than the mean fire size specified in the model because of many burned single pixels), and the mean CV of patch area was 952% during the 1000-year simulation. The mean largest patch size of fire was about 23,000 ha on the historical landscapes in the simulation, while Cohen et al. (2002) reported that the largest harvest unit size was about 9,000 ha, aggregated over the 23 years of harvesting on private industrial lands.

Disturbance frequency also differs between the current and historical landscapes. Disturbance return interval sets the basic template of landscape structure because it influences the patch dynamics on a landscape through patch creation and destruction (Baker 1992a). For the historical landscapes, the natural fire rotations used in the LADS simulations were 100 years (interior) and 200 years (coastal). For the current landscape, based on the data from Cohen et al. (2002), the clear-cut rotation between 1972 and 1995 was 51 years for the private industrial land, 100 years for the private non-industrial land, and 140 to 189 years for the public land. Together with large young forests on the State lands after wildfires in 1930s and 40s, the short rotation on the extensive private land (> 1/3 of the region) has created the current matrix of young forests with scattered older forests.

Temporal variability in the distributions of patch size, shapes, and fire return intervals are also important for structuring landscape patterns (Baker 1992b, Lertzman et al. 1998). In managed landscapes, temporal variability of those disturbance attributes are narrow, and return intervals are more regularly spaced (Lertzman and Fall 1998). Loss of temporal variability in attributes of disturbances leads to reduced dynamics of landscape structure (Baker 1992b). Habitat patches can have higher connectivity over time in dynamic landscapes than less dynamics landscapes when habitat abundance is limited. Wimberly (*public communication*) demonstrated that species occupancy of habitat was higher in dynamic landscapes when habitat area is near the critical threshold.

Comparisons between the two scenarios: current policy (CPS) vs. wildfire regimes (WFS)

The simulations indicated that 100-year management did not fully return the overall conditions of the landscape to the HRV under either scenario. First, 100-year period was too short for old forests to reach to the HRV. On the current landscape, the amounts of forests older than 80 years of age are well below the historical level, and especially, old growth forests are very rare. Second, patch shape moved away from the HRV consistently over the 100 year period offsetting changes toward the HRV in other attributes of landscape structure. Patch shape became simpler over time, and the vegetation map after 100 years resembled the ownership pattern.

Under the CPS, ownership pattern indirectly constrained development of landscape pattern as a consequence of contrasting forest management regimes by ownership types. The three general ownership groups (Federal, State, and private) have unique sets of management goals and regulatory constraints, and the ranges of forest conditions that can be produced within a particular ownership may be limited (Wimberly et al. in press). For example, young forests will be more abundant in the private lands because of the short rotation periods, and the Federal lands in reserves and wilderness areas will be the main source of old-growth forests in the future.

Many studies found the strong influence of ownership patterns on the vegetation and disturbance patterns in western Oregon (Spies et al. 1994, Garman et al. 1999, Cohen et al. 2002, Spies et al. 2002a, 2002b, Stanfield et al. 2002, Black et al. 2003). This study indicated that ownership boundaries can affect patch shape. Patch shape of the ownership tracts were considerably simpler than that of fire patches. The decreasing trend in fractal dimension in patch shape from current to the future may reflect in part the constraints on pattern imposed by the underlying ownership pattern. For example, the checker-board pattern of forest industry and BLM ownerships in the southeastern part of the study area became more evident on the vegetation maps over time (Fig. 2.1). Although there is heterogeneity (e.g., stand age, structure, uncut stands) within any ownership, given the high contrasts in management regimes among ownerships, ownership boundaries may lead to simpler patch shapes at the regional scale.

Ownership boundaries may also control the location and characteristics of edge types. Because overall patch types (e.g., age) are likely to be fixed by ownership, certain combinations of edge types can be reduced or increased at the regional scale. For example, edge between old growth and open, very young stands may only be found in or around reserves on the Federal lands abutting on the private lands. Also, the ownership pattern may reduce intermixing of forest types compared with the historical wildfire landscapes. Based on the landscape-level analysis, patch juxtaposition (IJI) did not reach HRV over the 100-year management. Altered patch adjacency may lead to disruption in source-sink processes of movement of organisms and materials across different forest types (Forman 1995).

For the landscape-level analysis, the WFS took the landscape away from the HRV almost in an opposite direction from the trajectory of the management scenario. This result was somewhat surprising because emulating natural disturbance regimes is hypothesized as a way of conserving biodiversity and maintaining landscape conditions within HRV (e.g., Hunter 1990, 1993, Bergeron et al. 2002, Kuuluvainen 2002). In the simulations, the large young forest patches on the current landscape developed into massive, highly connected patches of mature forests in several decades. It took a couple of centuries in the simulation for these patches to be broken up by wildfire and develop into various age classes. These observations illustrated legacy effects of past management on the landscape (Baker 1992a, 1993, 1995, Wallin et al. 1994, Landres et al. 1999). Wallin et al. (1994) demonstrated that landscape dynamics may show inertia in response to change in disturbance regimes due to the legacy effects of altered landscape structure by dispersed patch cuts in the Pacific Northwest. The legacy of past management affects the potential of a landscape to return to the HRV in the future.

Challenges in estimating HRV

Quantitatively estimating HRV imposes many challenges because available data are often insufficient and the methodology has not been well established. Existing literature on HRV indicates a wide variety of approaches and suggests that analysis methods are case-specific for available information and study objectives. Estimating HRV of landscape structure using simulation models requires considerable information on fire regimes. Data were needed for disturbance frequency, size, shape, severity, pattern of spread, and effects of topography and vegetation on forest susceptibility to fire to parameterize the model. Empirical data are sparse for these variables, and the simulation model used in this study was calibrated using a few studies conducted in the Oregon Coast Range. The dendrochronological study (Impara 1997) which provided input data for fire return intervals examined only the middle portion of the region, for example.

Despite that, the ecology and history of the Oregon Coast Range are relatively amenable to simulation modeling. In other regions, historical information may no longer exist where disturbance was rare or very frequent, agents that can hold records (e.g., fire scars in trees) are short-lived, and recent disturbances have erased the records (Swanson et al. 1994, Morgan et al. 1994). The characteristics of Douglas-fir (fire-tolerant, shade-intolerant, and long-lived) are exceptionally suited to stand-history studies, and having the species as a dominant is the advantage of my study region. At the continental scale, the western United States is relatively new to Euro-American settlement, so that legacy from the presettlement period still exists on the landscapes (e.g., patches of forests generated after wildfire; Teensma et al. 1991). In other parts of the world, settlement history is much longer, or distinct transitions in resource use did not happen. Fire modeling is well studied, and fire is more feasible than other agents such as insect and disease outbreaks or wind throw to statistical modeling at a large scale. Simulation modeling is a useful tool for the Oregon Coast Range to extrapolate limited information on disturbance regimes to a larger spatial and temporal extent. It is important to quantify uncertainties associated with estimations so that knowledge gaps can be identified and studied by future research (Wimberly 2002).

Multivariate analysis (PCA) was useful to condense the landscape metrics that can be otherwise difficult to comprehend as a whole to determine how different a landscape is from the historical conditions. It was also useful to visualize the relative locations of the landscapes and change over time in relation to the HRV. Because PCA relies on correlation structure among variables to condense the information contained in the variables, when a managed landscape has a different correlation from a historical landscape, the managed landscape can be a multivariate outlier. The synthetic axes obtained from the ordination of historical landscapes might not be as effective to describe the difference between the HRV and a managed landscape. For example, class area and area occupied by the largest patch of the class tend to have a strong positive relationship, but the largest patch size of patchy open in the current landscape was smaller than the historical mean although the current total area was about 7 times larger than the historical. The limitations of multivariate analysis make it important to interpret multivariate results with reference to the original variables.

HRV is also affected by scale (Morgan et al. 1994, Aplet and Keeton 1999). Wimberly et al. (2000) demonstrated that the variability of the amounts of late-successional and old-growth forests in the Oregon Coast Range before settlement was too large at the national forest scale (302,500 ha) or late-successional reserve (40,000 ha) scale to bound the system behavior in a reasonable range. Although studies have not investigated the effects of grain size on HRV estimation, it can potentially be influential especially on characterization of landscape structure by landscape metrics. Grain size is known to influence edge length and patch shape considerably (McGarigal et al. 2002). At the 9-ha scale, the managed landscapes had less intermixing of patches and more simple edges, but the degree or abundance of the characteristics at finer scales cannot be inferred from the data. For example, straight edges around mature forests at year 100 are probably more irregular at a finer scale than what the map at the 9-ha scale suggests. This problem also applies to temporal scales, and Keane et al. (2002) found that long simulations were needed relative to fire frequency to capture full variation and that summary intervals of output data influenced apparent variability. The HRV estimated in this study applies specifically to the particular spatial and temporal scale investigated.

The HRV of landscape conditions depends on classification schemes and metrics used (Li and Reynolds 1994, 1995). If I lumped all old-growth forests (> 200 year) in one class, the landscape would not take 800 years to return to HRV. Likewise, some of the metrics differed in amounts of time required to reach HRV. Baker (1992a, 1995) also found that metrics differed in time to depart from HRV. He attributed to the differences to different sensitivities of the metrics to changes in fire frequency and size although he stated that full explanations need further study (Baker 1995). I found that, in general, metrics that take into account patch type arrangement surrounding a focal patch respond more slowly to change in disturbance regime than metrics that measure only a single type (e.g., IJI and TECI). Inherent sensitivity of metrics to changes in landscape structure are variable among metrics, so that the observed ranges of metric values (standardized range in Table 2.4) does not necessary indicate actual variation in the measured landscape characteristics. Studies that relate metric behavior and change in landscape structure are needed to better understand the potential effects of metrics on the quantification of HRV. Classifications and landscape metrics for an HRV approach should be based on management objectives and ecological and social significance of landscape characteristics at the scale which the heterogeneity is most adequately represented (Kolasa and Rollo 1991, Landres et al. 1999). For example, I focused on diversity of forest ages from very young to very old. These age classes are closely associated with stand structure and habitat for sensitive plants and wildlife found in the region (Johnson and O'Neil 2000).

Management implications

Evaluations and limitations of HRV approaches for management have been discussed elsewhere (e.g., Swanson et al. 1994, Morgan et al. 1994, Landres et al. 1999, Aplet and Keeton 1999, Davis et al. 2001). I highlight some of the points in the context of this study. The coarse-filter strategy for biodiversity conservation assumes that maintaining a representative array of ecosystems in a landscape will provide habitats to the majority of the native species, including the ones poorly known (Hunter et al. 1988, Hunter 1990). HRV in landscape structure guides conservation planning to estimate what are the representative ecosystems and their spatial configuration. Moreover, HRV indicates the importance of ecological processes that drive landscape dynamics and hence variability in habitat conditions. Therefore, HRV has a high potential to provide information to be considered in conservation planning based on the coarse-filter strategy.

HRV in landscape structure provides a management tool not only to objectively assess landscape conditions but also to identify what attributes have departed from the HRV and the magnitude of departure. The HRV approach can distinguish the directions and magnitude of possible consequences from management in relation to the HRV (Morgan et al. 1994). The approach can let managers estimate potential impacts on native biodiversity in the current or possible future landscapes under alternative scenarios.

The results indicate that the landscape of the Oregon Coast Range is outside the HRV and would not return to the HRV for a long period of time (e.g., 800 years) for policy and management. The past 150-year land use history has almost eliminated mature and old growth forests so that the landscape would not respond to changes in harvesting patterns and rates and may exhibit inertia or lag time (Wallin et al. 1994). Does this historical legacy suggest that the HRV approach is not a useful concept for evaluating highly altered landscapes? The use of HRV does not imply that historical conditions are a management goal, but it is rather a guide in developing and evaluating management plans that attempt to maintain ecosystem functions and biodiversity (Bergeron et al. 1999, Cissel et al. 1998, 1999, Swetnam et al. 1999). Analysis of HRV identifies the landscape characteristics that may be lacking at the regional scale, but these deficiencies may not be a concern to all landowners. In addition, climate may change before the landscape returns to a desirable condition, and thus conditions estimated from periods of different climate may become less relevant as a reference (Miller and Woolfenden 1999). Large wildfire may burn the forests in the future and could take the landscape away from the HRV. Departure from the reference condition can serve as one indicator of trends in landscapes and suggest potential landscape attributes that could be changed to meet general biodiversity goals. This information helps provide a context for conservation planning (e.g., Noss 1993) and identify consequences to landscape pattern that may not be desirable.

CONCLUSIONS

This study demonstrated that the HRV approach can provide a reference condition for quantitative evaluations of landscape conditions and alternative management scenarios. The current landscape is outside the HRV, and the management regimes under the current forest policies did not move the landscape within the HRV in 100 years in the simulation. Under the wildfire scenario, the landscape took centuries to return to the HRV. Extensive forest management in the last few decades in the Oregon Coast Range has left legacies, which may persist many decades or centuries on the landscape after change in the management regime. A disturbance-based approach to forest management (e.g., Hunter 1993) suggests basing management regimes such as patch size, configurations, and rotation length on natural disturbance regimes. The approach is theoretically sound and promising, but this study suggests that the positive effects of the approach may not be attained rapidly in highly managed landscapes. In a multi-ownership landscape, HRV is not necessarily the management goal. Therefore, the HRV approaches have more value to biodiversity as an indicator of conditions than a management objective. The HRV approaches require information to reconstruct disturbance regimes, which may not be available in all landscapes. It is also important to note that results depend on scale and characterization of landscape heterogeneity.

HRV is not an inherent property of a landscape, but it is derived from our perceptions and values of forest ecosystems (Lackey 2001, Foster et al. 2003). The HRV approach quantitatively provides reference conditions that facilitate understanding of human effects on the landscape. This understanding may direct management to focus on restoration of ecosystem components that are lacking or are not likely to be maintained without special efforts. There are, however, many other ways to evaluate biodiversity on a landscape and measure the characteristics and heterogeneity (Kolasa and Rollo 1991). Historical studies provide information, and society can use it to help decide what it needs and values from forest landscapes.

CHAPTER 3: HISTORICAL RANGE OF VARIABILITY (HRV) IN LIVE AND DEAD WOOD BIOMASS: A SIMULATION STUDY IN THE COAST RANGE OF OREGON, USA

ABSTRACT

The historical range of variability (HRV) in landscape structure created by natural disturbance has been proposed as a guide for evaluating managed landscapes. Previous studies, however, focused on variability based only on landscape patterns characterized by live trees and did not address variations in stand structure including dead wood. The objective of this study was to investigate the HRV in live and dead wood biomass in the Oregon Coast Range and to examine variability in disturbance history and forest stand structure. I used a stochastic fire simulation model to simulate landscapes for 1000 yrs prior to Euro-American settlement (circa 1850) and calculated biomass as a function of disturbance history. I contrasted spatial pattern of live and dead wood using texture analysis. The HRV was quantified as area of different levels of live and dead wood, and the current condition was compared with the HRV. I compiled 1000-yr fire history information and examined variations in disturbance histories.

The HRV of live and dead wood biomass distributions revealed that the majority of the landscape historically contained 500-700 Mg/ha of live wood and 50-200 Mg/ha of dead wood. The current dead wood condition is outside HRV. Stands with very low (<50 Mg/ha) dead wood are currently the dominant type, covering close to 60% of the region, while it historically occurred only in 2.5% of the region, according to the model results. The model suggested that there was a wide variation in dead wood biomass because of variations in disturbance history. Only a small fraction of the samples experienced stand dynamics described by idealized linear sequence of stand development, and the majority experienced histories characterized by multiple disturbance types during the 1000-yr simulation period. For the 1000-yr period, no stand experienced periodic fires at regular intervals and uniform intensity.

This study suggests that natural disturbance regimes and stand development are characterized by much larger variation than is typically portrayed or appreciated. Spatial and temporal variability in disturbance and forest development was high under the historical fire regime, and incorporating the variability in management is challenging. The HRV approaches to evaluating landscape conditions need to include both landscape and stand characteristics to better represent ecological differences between managed and unmanaged landscapes.

INTRODUCTION

The historical range of variability (HRV) in landscape and forest structure created by natural disturbances has been proposed as a guide for biodiversity conservation in the past decade (e.g., Swanson et al. 1994, Reeves et al. 1995, Aplet and Keeton 1999, Cissel et al. 1999). HRV is defined as the bounded variability of a system within constraints imposed by larger-scale phenomena (e.g., climate, topography) and without significant modern human influence (Swanson et al. 1994, Morgan et al. 1994, Aplet and Keeton 1999, Landres et al. 1999). The HRV in landscape patterns provides reference ranges of conditions to evaluate landscapes for habitat diversity and arrangement (Aplet and Keeton 1999, Landres et al. 1999). It is a "coarse-filter" approach to biodiversity conservation based on the premise that landscapes within HRV can maintain native biodiversity and ecological functions (Noss 1987, Hunter et al. 1988, Hunter 1990, Noss and Cooperrider 1994).

Previous studies have quantitatively examined HRV in landscape patterns that are represented by age classes or general forest types of developmental stages (e.g., stem exclusion vs. understory reinitiation, open vs. closed canopy) or dominant species under historical disturbance regime using fire simulation models (Wallin et al. 1996, Wimberly et al. 2000, in press, Roworth 2001, Wimberly 2002, Keane et al. 2002). The classifications used in the previous studies were based on age and simple overstory structure, and they have failed to include dead wood components. Fires typically leave live and dead wood structures that play important roles in biodiversity and stand development in post-fire stands, and amounts of dead wood are influenced by disturbance history (Spies et al. 1988, Spies and Franklin 1991, Franklin et al. 2002). Snags and logs, two major forms of dead wood, provide nest sites, cover, and foraging substrates to wildlife and growing sites to some plants, and large logs in streams create critical habitat structure for many aquatic species (Thomas 1979, Harmon et al. 1986, Bull et al. 1997, Rose et al. 2001). Forest management has been incorporating dead wood retention to maintain the ecologically important structure (USDA Forest Service and USDI Bureau of Land Management 1994, Rose et al. 2001, Lindenmayer and Franklin 2002). Therefore, there is a need to include HRV of dead wood in analyses to fully understand the HRV and roles of history in variation of forest stand structure in landscapes.

Classifications assume homogeneous stand structure within class, and a limited number of classes often underestimate variability in stand structure in a landscape. High-severity disturbance resets a sequence of stand development in age- or type-based classifications, and therefore, carryover of species and structures from pre-fire stands cannot be fully accounted for. Stand age (i.e. time since stand establishment) does not adequately indicate the structure of live trees for uneven-aged stands and is a poor predictor of dead wood biomass in relatively young stands (e.g., < 150 yrs; Spies et al. 1988). Stand types, such as developmental stages, can incorporate different fire-severity levels to some degree (e.g., Hemstrom et al. 2001, Keane et al. 2002), but it is still difficult to include variation in dead wood associated with each type. Explicitly quantifying live and dead wood components can reflect variability in disturbance history better than categorical classes.

Live and dead wood biomass is a simple measure of forest structure and reflects the cumulative effects of disturbance history on forest structure because change in biomass can continue across multiple disturbances. Forest structure, including live and dead wood, is one of the key attributes of forest ecosystems (Hansen et al. 1991, Spies 1998, Hunter 1999). Forest structure is sensitive to disturbance history, and many different disturbance histories can produce a variety of stand structure (Spies 1998). Many forest structural characteristics follow the general U- or S-shaped curves during stand development and succession (Bormann and Likens 1979, Peet 1981, Spies and Franklin 1988). Live biomass and diversity of tree sizes are examples of an S-shaped curve, which gradually increases over time and culminates in the late stage. Dead wood biomass can follow a U-shaped curve, characterized by high levels in the early and late stages of stand development (Spies and Franklin 1988). Using chronosequence data from unmanaged stands in western Oregon and Washington, Spies et al. (1988) presented general stand-level models of coarse woody debris (CWD) dynamics under three disturbance scenarios. These models predict that the amount of CWD is high in young forests because of mortality from the last disturbance event and is relatively low in intermediate stages after the majority of dead wood from fire mortality has decayed. In older forests, the amount increases due to accumulation of mortality from stands established after the last disturbance. The models representing 3 different fire history scenarios by Spies et al. (1988) show that amounts of CWD can take different pathways, depending on disturbance history, and the variation in disturbance history produces variation in dead wood amounts among the stands of the same age classes.

Disturbance can be characterized as a discrete event that alters physical and biological conditions of a stand (Pickett and White 1985), but disturbance may be better described as a sequence of events because stand structural changes are influenced not only by the most recent event but also by one or more previous events. Moreover, time lags in response of dead wood biomass to disturbance events originate from the slow decay process and delayed input mortality from live biomass. These time lags may create complex relationships between forest structure and disturbance history, which have not been recognized in the previous studies.

The biomass of dead wood in forests is influenced by site productivity, disturbance history, topography, and environmental conditions for wood decay and fragmentation (Harmon et al. 1986, Spies et al. 1988). The effects of interactions of these factors are potentially intricate, and this study focuses on that of disturbance history, more specifically wildfires. Although many factors influence the distribution of biomass on a landscape, a simple model of biomass dynamics as a function of large-scale fires can provide insights into the diversity of disturbance histories and the effects on stand development.

The main goal of this study was to investigate the HRV in biomass dynamics in the Oregon Coast Range and to examine the variability in disturbance history and forest stand structure. I used a simulation modeling approach to scale up stand level biomass models to the regional scale. The model integrates live and dead wood biomass dynamics models in the literature (Bormann and Likens 1979, Peet 1981, Harmon et al. 1986, Spies et al. 1988, Spies and Franklin 1988). The study objectives were to 1) characterize the spatial distribution of live and dead wood biomass under the historical fire regime at the regional scale, 2) characterize the HRV in live and dead wood biomass, 3) compare the current amount of dead wood relative to the HRV of dead wood, and 4) characterize variation in fire disturbance history and stand structure. In this paper, I emphasized dead wood biomass because it illustrates cumulative effects of disturbance better than live wood.

METHODS

Study area

The Oregon Coast Range is a 2 million-ha physiographic province located along the western edge of Oregon, United States (inset in Fig. 3.1). The climate is characterized by mild wet winter and dry cool summer (Franklin and Dyrness 1988). As a result of the geographic setting, the western half of the region has a moister climate than the other half (Fig. 3.1). The topography is characterized by highly dissected mountains, steep slopes, and a high density of streams. The soils are deep to moderately deep and fine to medium texture, derived from sand stone, shale, or basalt (Franklin and Dyrness 1988). Douglas-fir (*Pseudotsuga menziesii*) is a long-lived dominant and is exceptionally important for the forest structure, both living and dead,

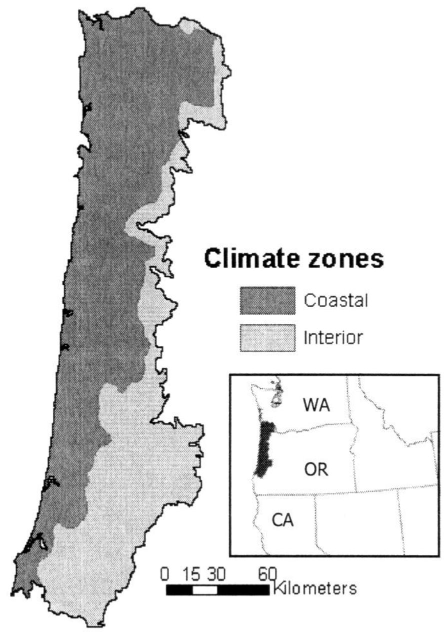

Figure 3.1: The two climatic zones used in the LADS model. The coastal zone is more moist and characterized with infrequent, high-severity fires than the interior zone. The natural fire rotation (NFR) was set at 200 yrs for the coastal zone and 100 yrs for the interior zone.

of the region throughout stand development (Spies et al. 1988, Franklin et al. 2002). This species is fire-resistant and has thick bark that enables individuals to survive low- to moderate-severity fires (Agee 1993). Shade-tolerant, western hemlock (*Tsuga heterophylla*) is another important tree species and gradually fills in the canopy in the late-successional stage (> 200 yrs; Spies and Franklin 1996 and Franklin et al. 2002).

Disturbance regimes

Large-scale wildfire was the most important disturbance that shaped forests of the Oregon Coast Range (Agee 1993, Long et al. 1998, Long and Whitlock 2002). The current fire regime has been present in the last 1000 yrs (Long et al. 1998). In presettlement time, the estimated mean fire-return interval ranged from 150 to 350 yrs for high-severity fires in this region (Fahnestock and Agee 1983, Agee 1993, Teensma et al. 1991, Ripple 1994, Long et al. 1998, Long and Whitlock 2002). Moderate-severity fires occurred often in mixture with high-severity fires (Impara 1997). High-severity fires often led to stand replacement, while moderate-severity fires left unburned forest patches and single trees (Agee 1993, Impara 1997), which can influence subsequent stand development (Goslin 1997, Weisberg in press). Fires tend to be especially severe in the first 30 years of stand development because these stands tend to have high amounts of flammable fuel left after the previous fire (Agee and Huff 1987).

Fires were set by Native Americans in the coastal valleys and adjacent Willamette Valley for agriculture and hunting (Boyd 1996), some of which would have occasionally burned into the foothills of coastal mountains, but the evidence of this is not strong (Agee 1993, Whitlock and Knox 2002). The region experienced more extensive fire occurrences following Euro-American settlement in mid 1800s (Impara 1998, Weisberg and Swanson 2003), and high-severity fires were prevalent in mid 1800 to mid 1900 (Morris 1934, Arnst 1983). Effective fire suppression efforts began in the 1940s in western Oregon (Weisberg and Swanson 2003).

Fires affect dead wood biomass via tree mortality from fire and fire consumption of wood (Agee 1993). No previous studies empirically quantified proportion of tree biomass killed in fire of different severity levels in stands in the Pacific Northwest. A high proportion of fire-killed biomass enters dead wood pool because fire consumption of green trees is relatively small (0-10%; Agee 1993). Fire consumption of dead wood biomass is estimated to range from 20 to 30% (Agee 1993).

Conceptual model of biomass dynamics

The dead wood dynamics model of Harmon et al. (1986) and Spies et al. (1988), which are the basis of the model I used, characterizes the amount of dead wood as a sum of the three major components: carry-over from the pre-fire stand, dead wood created by fire, and mortality from the post-fire stand. Live wood biomass is implicit in their model and an essential companion for the dead wood models to calculate new inputs from mortality in the post-fire stand.

The live and dead biomass of stands is a function of both fire severity and interval between fires in this study (Fig. 3.2). High-severity fires kill all live trees, while moderate-severity fires kill a portion of the trees in a stand. Immediately after high-severity fire, dead wood or legacies from the pre-fire stand are abundant. Reburns, fires recurring in stands within a few decades, may occur and reduce dead and live wood to very low levels. Although I illustrated a few pathways in Figure 3.2, almost any pathway within the shaded area is theoretically possible. The shaded area can be considered as the HRV of biomass dynamics. The actual shape of the possible range depends on disturbance regimes and the rate of tree establishment and growth.

To characterize variation in disturbance history, I classified disturbances into 8 disturbance history types (Fig. 3.3). The types distinguish two severity levels of fires, high (C = catastrophic) and moderate (P = partial), and live wood biomass levels at the time of fire. High-severity fires are stand-replacing events, converting all live wood biomass into dead wood. Moderate-severity fires are assumed to be a mixture of

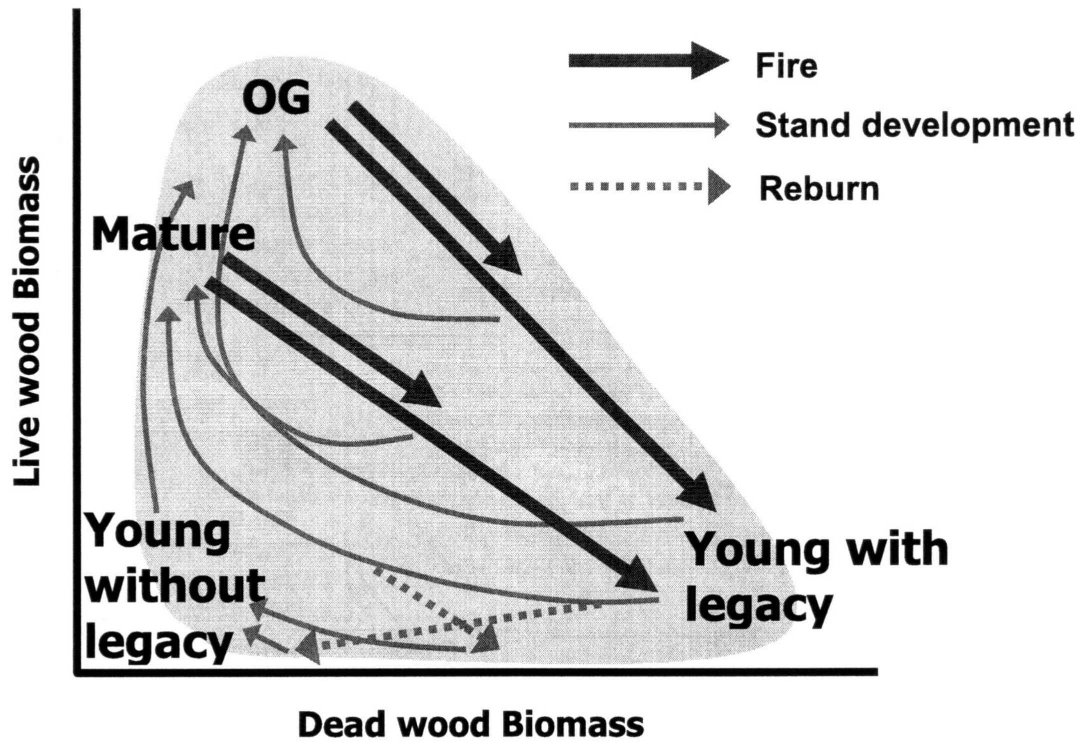

Figure 3.2: Dynamics of live and dead wood biomass in response to different fire severities and frequencies. The thick arrows are fire events, and the short ones are moderate-severity fires, which do not convert all live wood biomass into deadwood. The dotted arrows indicate repeated burns, which returned to the stand when live biomass has not been well developed. The thin arrows indicate stand development over time. "Young with legacy" refers to young stands (< 80 yrs) with high amounts of deadwood, and "young without legacy" refers to young stands with relatively small amounts of deadwood because of reburns. The shaded area conceptually indicates all possible range of pathways under the fire regime and forest growth. Under the historical fire regime, the shaded area can be considered as the HRV of biomass dynamics. Mature = mature forests (80-200 yrs). OG = old-growth forests (> 200 yrs).

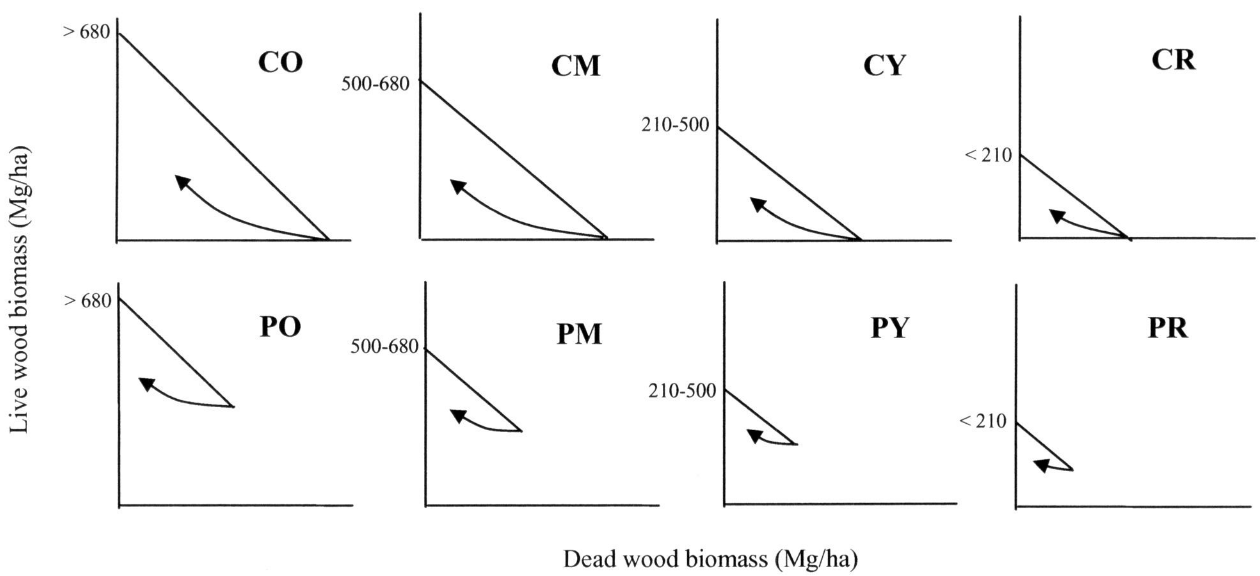

Figure 3.3: Disturbance history types based on initial live biomass and fire severity. There are two classes of fires; C = high-severity, catastrophic fire; P = moderate-severity, partial fire. The cutoff values for live wood biomass were selected to reflect stand structural change with stand age using the chronosequence data; O = old-growth (> 680 Mg/ha), M = mature (500-680 Mg/ha), Y = young (210-500 Mg/ha), and R = reburn (< 210 Mg/ha). Reburn was defined as any fire returning before live wood biomass attains 210 Mg/ha (~30 yrs).

crown and surface fires that convert half of live wood biomass to dead wood (Spies et al. 1988). Live wood biomass increases with time as stand develops, and dead wood biomass is the balance between input from new stands and loss from decay. Live wood biomass roughly follows the length of inter-fire period, and I chose 4 levels to represent different stand biomass types; old-growth (O), mature (M), young (Y), and very young (R = reburn). For example, CO is a high-severity fire in old-growth forests. The age of the stand at the time of fire is a critical variable to explain temporal patterns of large fuels (Lotan et al. 1985, cited in Agee and Huff 1987). The cutoff biomass values were selected based on the growth sub-model derived from field data (Spies et al. in press). The lowest level of live wood represents the effects of reburns.

The simulation model of fire and biomass dynamics

Historical landscapes were simulated using the Landscape Age-Class Dynamics Simulator (LADS), Version 3.1 (Wimberly 2002). LADS is a spatially-explicit, stochastic computer simulation model designed to simulate landscape dynamics under fire regimes specified by the user. The Oregon Coast Range was represented as a grid of 9-ha cells. LADS was parameterized to the historical fire regimes 1000 yrs prior to Euro-American settlement using reconstructed fire boundary maps, dendrochronological and paleoecological studies in the region (for details, see Wimberly 2002). The fire regime has been relatively stable in the last 1000 yrs (Long et al. 1998). The region was subdivided into two distinct climate zones, coastal and interior (Fig. 3.1). The climate of the coastal zone is moist and characterized with a longer natural fire rotation, while that of the interior zone is dryer and historically more frequently burned. Fires were more likely to be severe and larger in the coastal zone than in the interior.

The LADS model was modified to include live and dead wood biomass. The dead wood and live wood biomass dynamics were simulated as a function of time since last disturbance. The components in the model were dead wood and live wood pools, fire mortality, chronic mortality, decay, and fire consumption of wood (Fig. 3.4). Net live wood biomass (LBB_t) in a stand was modeled as a Chapman-Richards function (Richards 1959);

$$LBB_t = a * (1 - e^{(-b*(BIOAGE_{t-1} + STEP))})^c \qquad \text{Equation 1}$$

LBB_t is the live wood biomass at time t. The parameter a is the asymptote of the curve and indicates the maximum possible biomass in Mg/ha which a stand could attain. The parameter b controls rates and c controls time lag for a stand to reach the maximum biomass. $BIOAGE_t$ is the post-fire biological age at time t, and $BIOAGE_{t-1}$ is the biological stand age at the previous time step. $BIOAGE_t$ is calculated from the amount of living biomass left in the stands after a fire event at time t;

$$BIOAGE_t = \log(1 - (LBB_{t(f)}) / a)^{1/c})/ - b \qquad \text{Equation 2}$$

where $LBB_{t(f)}$ is the post-fire live biomass at time t after fire mortality is subtracted from the stand living biomass if there was a fire event at time t.

$$LBB_{t(f)} = (1 - FMORT) * LBB_t \qquad \text{Equation 3}$$

where FMORT is the proportion of live biomass that is killed by fire at time t. Because BIOAGE is a function of live biomass and not an actual stand age, live wood biomass calculated by this equation is independent of time since stand-replacement (i.e. stand age). This scheme allows live wood biomass to increase proportionally to standing biomass (Grier and Logan 1977). STEP is the simulation step length in years.

The live biomass curve was calibrated using existing chronosequence field data from 42 sites in the southern Coast Range ranging in age from 40 to about 525 yrs (Spies et al. in press). The parameters a, b, and c were simultaneously determined by visually fitting a model to the chronosequence data. The parameter values were selected so that the model fits especially well with the data from stands less than 130 yrs old. The stands < 130 yrs old were not likely to have been disturbed since stand establishment and probably represented a reasonable approximation of net growth after stand establishment, compared to that in older stands in this chronosequence. The selected values were, 700, 0.02, and 1.5 for the parameters a, b, and c,

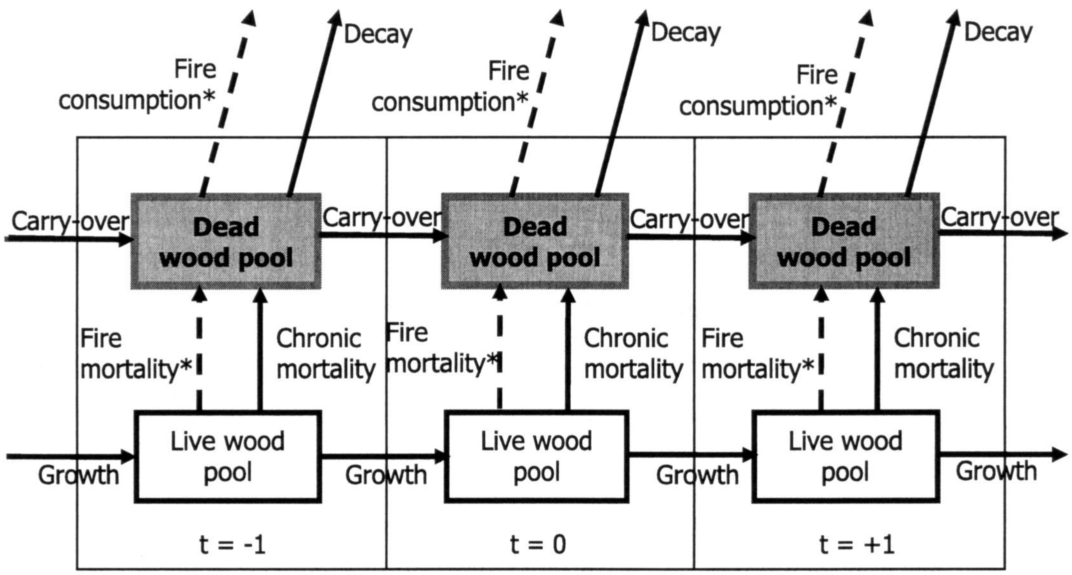

Figure 3.4: The conceptual diagram of the deadwood model in LADS (* if there is a fire event). Dead wood pool carries over across time steps, and certain percentages of the pool decays away at each time step. Live wood pool grows at each time step if it is below the theoretical maximum defined by the growth equation in the model. At each time step, certain percentages of the live wood pool go into the dead wood pool as chronic mortality. When a fire occurs, it consumes 25% of the dead wood pool and put 50% (moderate-severity fire) or 100% (high-severity fire) of the live wood pool into the dead wood pool in the same time period.

respectively. This model was fairly close to the least square fit of the exponential model on the same data by Spies et al. (in press).

The dead wood biomass was computed at each time step as;

$$DWM_t \quad (DWM_{t-1} * e^{(DECAY*STEP)} \quad CMORT * LBB_t) * (1 - CONS) \quad LBB_t * (FMORT)$$

Equation 4

where DECAY is the decay and fragmentation rates per year and STEP is the time interval in the simulation. CMORT is the chronic mortality rates per time step in % biomass. DECAY and CMORT can vary in the model with time since the last fire (Table 3.1). In this study, CMORT was fixed at 0.5% per year for 30 years after high-severity fires because of insufficient information about chronic mortality of trees at different age classes in % biomass in the region. The data from an individual-based model (Zelig; Urban et al. 1999) and literature suggest that average mortality rates in biomass do not vary considerably with age (but fluctuates more in older stages) and rates between 0.5 to 1% are reasonable (Grier and Logan 1977, Sollins 1982, Harcombe et al. 1990, Greene et al. 1992, Wright 1999, Acker et al. 2000, 2002). CMORT was zero for stands < 30 years old because dying trees are the smallest suppressed classes (Cline et al. 1980, Franklin and DeBell 1988, Peet and Christensen 1987) and assumed too small to be recruited as coarse woody debris. CMORT was increased for the first 50 yrs after a moderate-severity fire to reflect elevated mortality due to delayed mortality from fire injury (Spies et al. 1988).

DECAY was set based on literature and expert opinions (Stone et al. 1998, McArdle and Mayer 1961, Harmon et al. 1986, Harmon *pers. comm.*) and attempted to reflect distributions in size, form (snag or log), and species composition across age classes. Spies et al. (1988) documented that, in Douglas-fir forests of western Oregon and Washington, young stands were characterized by the higher densities of snags and logs of all sizes than mature and old-growth stands although largest pieces were most numerous in old-growth stands. Decay rates are lower for larger pieces and remarkably lower for early-successional Douglas-fir than late-successional western hemlock (Graham 1982, Harmon et al. 1986, unpublished data, Stone et al. 1998). Fragmentation rates are considerably higher for snags (Graham 1982, Sollins 1982),

Table 3.1: The model parameters used in this study showing the baseline values and values used for the sensitivity runs.

Parameter	Description	Baseline run	Sensitivity runs –	Sensitivity runs +
Fire regime				
NFR_c	Natural fire rotation for coastal zone (years)	200	160	240
NFR_v	Natural fire rotation for valley margin zone (years)	100	80	120
MFS_c	Mean fire size for coastal zone (km^2)	73	58.4	87.6
MFS_v	Mean fire size for valley margin zone (km^2)	22.2	17.8	26.6
$SDFS_c$	SD of fire size for coastal zone (km^2)	320.5	218.1	384.6
$SDFS_v$	SD of fire size for valley margin zone (km^2)	51	34.9	61.2
SEV(1)	Minimum severity of fires <100 km^2	0	0.0	0.05
SEV(2)	Maximum severity of fires <100 km^2	0.5	0.4	0.55
SEV(3)	Minimum severity of fires 100-500 km^2	0.1	0.01	0.18
SEV(4)	Maximum severity of fires 100-500 km^2	0.8	0.71	0.9
SEV(5)	Minimum severity of fires < 500 km^2	0.7	0.535	0.98
SEV(6)	Maximum severity of fires > 500 km^2	0.95	0.785	1.0
Biomass				
P_a	Chapman-Richards parameter a	700	560	840
P_b	Chapman-Richards parameter b	0.02	0.016	0.024
P_c	Chapman-Richards parameter c	1.5	1.2	1.8
$FMORT_h$	High-severity fire mortality[1]	1	0.9	0.95
$FMORT_m$	Moderate-severity fire mortality[2]	0.5	0.45	0.55
$CONS_h$	High-severity fire wood consumption	0.25	0.2	0.3
$CONS_m$	Moderate-severity fire wood consumption	0.25	0.2	0.3
DECAY	Decay rates for deadwood	varies by age		
$CMORT_h$	Chronic mortality after high-severity fire	varies by age		
$CMORT_m$	Chronic mortality after moderate-severity fire	varies by age		

[1] -10% and -5%.
[2] ±10%.

Table 3.1: continued.

	Baseline run		Sensitivity runs			
			lower		higher	
age class	coast	valley	coast	valley	coast	valley
Decay rates						
0-80	0.05	0.0625	0.04	0.05	0.06	0.075
81-200	0.04	0.05	0.032	0.04	0.048	0.06
201-250	0.035	0.04375	0.028	0.035	0.042	0.0525
251-300	0.03	0.0375	0.024	0.03	0.036	0.045
301-350	0.025	0.03125	0.02	0.025	0.03	0.0375
351-450	0.02	0.025	0.016	0.02	0.024	0.03
451-500	0.022	0.0275	0.0176	0.022	0.0264	0.033
501-550	0.024	0.03	0.0192	0.024	0.0288	0.036
551-600	0.026	0.0325	0.0208	0.026	0.0312	0.039
601-650	0.028	0.035	0.0224	0.028	0.0336	0.042
651-700	0.03	0.0375	0.024	0.03	0.036	0.045
701-750	0.032	0.04	0.0256	0.032	0.0384	0.048
> 750	0.035	0.04375	0.028	0.035	0.042	0.0525
Chronic mortality rates (per decade)						
High severity						
0-30[1]	0	0	0	0	0.1	0.1
> 31	0.05	0.05	0.04	0.04	0.06	0.06
Moderate severity						
0-20	0.15	0.15	0.12	0.12	0.18	0.18
21-50	0.075	0.075	0.06	0.06	0.09	0.09
>51	0.05	0.05	0.04	0.04	0.06	0.06

[1]Trees are assumed too small to be recruited as CWD. The lower run used the same value as the baseline.

and DECAY in the early stage of forest development was elevated to reflect that most dead trees are standing following fires (Agee and Huff 1987, Spies and Cline 1988, Spies et al. 1988). The accumulation of dead wood from new stands peaks in the old-growth stage followed by gradual decline to a lower equilibrium as a result of shifts in species dominance from Douglas-fir to western hemlock, which has smaller diameters and faster decay rates (Grier and Logan 1977, Spies et al. 1988). DECAY was increased for the interior zone by 25% to reflect that environment is more favorable for decay in the interior zone than the wet, coastal zone (Harmon et al. 1986, Spies et al. 1988).

Fire consumption of carryover deadwood at the time of fire events (CONS) is assumed to be 25 % for either high- or moderate-severity fire (Fahnestock and Agee 1983, Agee 1993, Spies et al. 1988). Mortality in living biomass from fire is set at 100% for high- and 50% for moderate-severity fires (Agee 1993). The amounts of green trees left after high-severity fires vary and depend on scale (Isaac and Meagher 1938, Eberhart and Woodard 1987, Delong and Tanner 1996), and I decided to use 100% mortality for simplicity. High-severity fires often occur under extremely dry and windy conditions (Agee 1993, Turner and Romme 1994) and therefore, often lead to high levels of overstory mortality. Issac and Meagher (1933) reported almost complete overstory mortality after the Tillamook fire, a high-severity fire in the region in 1933. The model leaves unburned islands within burned areas of individual or groups of pixels and hence approximates patchy fires at the 9-ha scale. I assumed that fire does not consume newly killed biomass because fire consumption of green wood is relatively small in this region (0-10%; Fahnestock and Agee 1983, Harmon et al. 1986, Agee 1993).

Data creation and analysis

I used 100 simulation runs for 1000 yrs at 10-yr intervals for analysis. Numerous simulation runs were necessary to obtain outputs that represent the full range of possible fire patterns (Wimberly 2002). The 1000-yr simulation length was an appropriate time scale to capture variations in Coast Range forests because 1000-yr is about the longevity of dominant Douglas-fir trees in the region and several times of the mean fire return interval. Also a paleoecological study in the Coast Range suggests that climate has been relatively stable in the last 1000 yrs (Long et al. 1998).

I created two subsets of the data from the outputs. The first set was to quantify spatial pattern and characterize HRV of live and dead wood biomass in terms of area (Objectives 1 and 2) and to compare the current amount of dead wood with the HRV (Objective 3). For this set, I randomly selected one time step from each simulation (a total of 100 maps). The second set of simulation was to characterize variation in fire disturbance history and stand development over 1000 yr periods (Objective 4). For this set, I randomly selected 100 pixels proportionally distributed between the two climate zones (62 pixels in the coastal zone, and 38 in the interior) from each run and compiled 1000-yr disturbance history data for each pixel (a total of 10,000 pixels, 6200 in the coastal zone and 3800 in the interior, from 100 runs, each pixel having 1000-yr fire history). A larger sample size was more desirable to capture the variability, but compilation of 1000-yr history was computationally extensive. Also, temporal and spatial autocorrelation was quite high in the data. Therefore, the results probably underestimated the variability. I used ArcGIS Arc 8.3 (ESRI 2002) to process the outputs and compile the data.

<u>The first data set (100 maps)</u>

For the spatial pattern analysis of biomass, I calculated variance at each pixel location using convolution windows of 4 different sizes (3x3, 11x11, 31x31, 51x51). I used the texture analysis function using variance as the algorithm in ERDAS Imagine 8.6 (Leica Geosystems 2002) to create "texture" maps for the 100 live and dead wood maps. Then, I calculated mean variance for each map as an index of texture or local homogeneity in pattern. Texture indicates smoothness or coarseness based on the spatial repetition period of the local structure (Pratt 1991, Lillesand and Kiefer 2000). Small value of this index implies smooth texture, and high values indicate coarse, rough texture. I used 4 different window sizes because texture measures highly depend on the size of the observation neighborhood (Pratt 1991). I used only the inner part of the region to avoid edge effects.

I classified the amounts of dead and live wood biomass into 6 levels and calculated the % of area in the region occupied by the 36 (6 x 6) biomass combinations. I then examined the % of area occupied by the 6 dead wood classes by age class. I examined dead wood biomass by major stand age classes recognized in the region (Spies and Franklin 1991, 1996, Franklin et al. 2002): 1) very open (0-10 yrs), 2) patchy open (11-20 yrs), 3) young (21-80 yrs), 4) mature (81-200 yrs), 5) old growth (> 201 yrs). I considered the 5^{th}-95^{th} ranges as the HRV of % area occupied by each biomass class.

I compared the current levels of live and dead wood biomass in the study area (Ohmann *unpublished data*) with the HRV. The current vegetation map for the Oregon Coast Range was derived from a statistical model based on satellite imagery, inventory plots, and GIS layers (for details, see Ohmann and Gregory 2002). The database includes live and dead wood biomass variables. The biomass variables were derived from dimensions of wood in the plot data weighted by species specific gravity and, for dead wood, a decay class reduction factor. The correspondence of the map with field plot data with respect to class area of the 6 live, 6 dead, and 36 live and dead wood combined classes was all excellent ($r^2 > 0.99$). Because only one modeled map was available for current conditions, I was not able to quantify the uncertainty around the estimates.

<u>The second data set (10000 pixels with 1000-yr trajectories)</u>

I used this data set to examine variation in disturbance history in the 1000-yr period and characterized the relative frequencies of disturbance histories in two ways: 1) each "disturbance history type" (Fig. 3.3) and 2) "1000-yr disturbance history." I assigned one of the 8 disturbance history types to each time step when fire occurred (Fig. 3.3). I pooled all the simulation runs and quantified the relative frequency of each disturbance history type by climate zones. I then characterized 1000-yr disturbance histories by combinations of 8 disturbance history types (max possible combinations $2^8 = 256$). I quantified the relative frequency of unique 1000-yr disturbance history by climate zone.

Sensitivity analysis

I examined sensitivity of model outputs to changes in the parameter values in terms of % of area in the region occupied by the biomass classes. Most of the parameter values were varied by $\pm$ 20% except the ones with values that cannot be increased or decreased (Table 3.1). For those parameters, values within a reasonable range were selected to examine the effects. Sets of parameter values were altered one at a time, and changes were expressed in % of the baseline mean. SDFS and MFS were varied simultaneously because fire size and its standard deviation are often highly correlated on real landscapes (Wimberly 2002). The SEV parameters were also varied simultaneously so that the proportion of high-severity fires in each size class was varied by 20% (Wimberly 2002). The DECAY and CMORT parameters were also varied simultaneously by each across the age classes. I conducted 10 independent 1000-yr simulation runs at 10-yr intervals for each parameter set. I examined only 10 most abundant classes for sensitivity because % changes relative to the baseline mean were excessively inflated if the baseline mean was small despite small actual changes in mean % area.

RESULTS

HRV in live and dead wood biomass in the Oregon Coast Range

Amounts of live and dead wood were patchily distributed over the landscape in model simulations (Fig. 3.5). The distributions of live and dead wood biomass were largely corresponding to each other at the regional scale but not in detail. The mean variance for the spatial distribution was lower for dead biomass than live biomass at

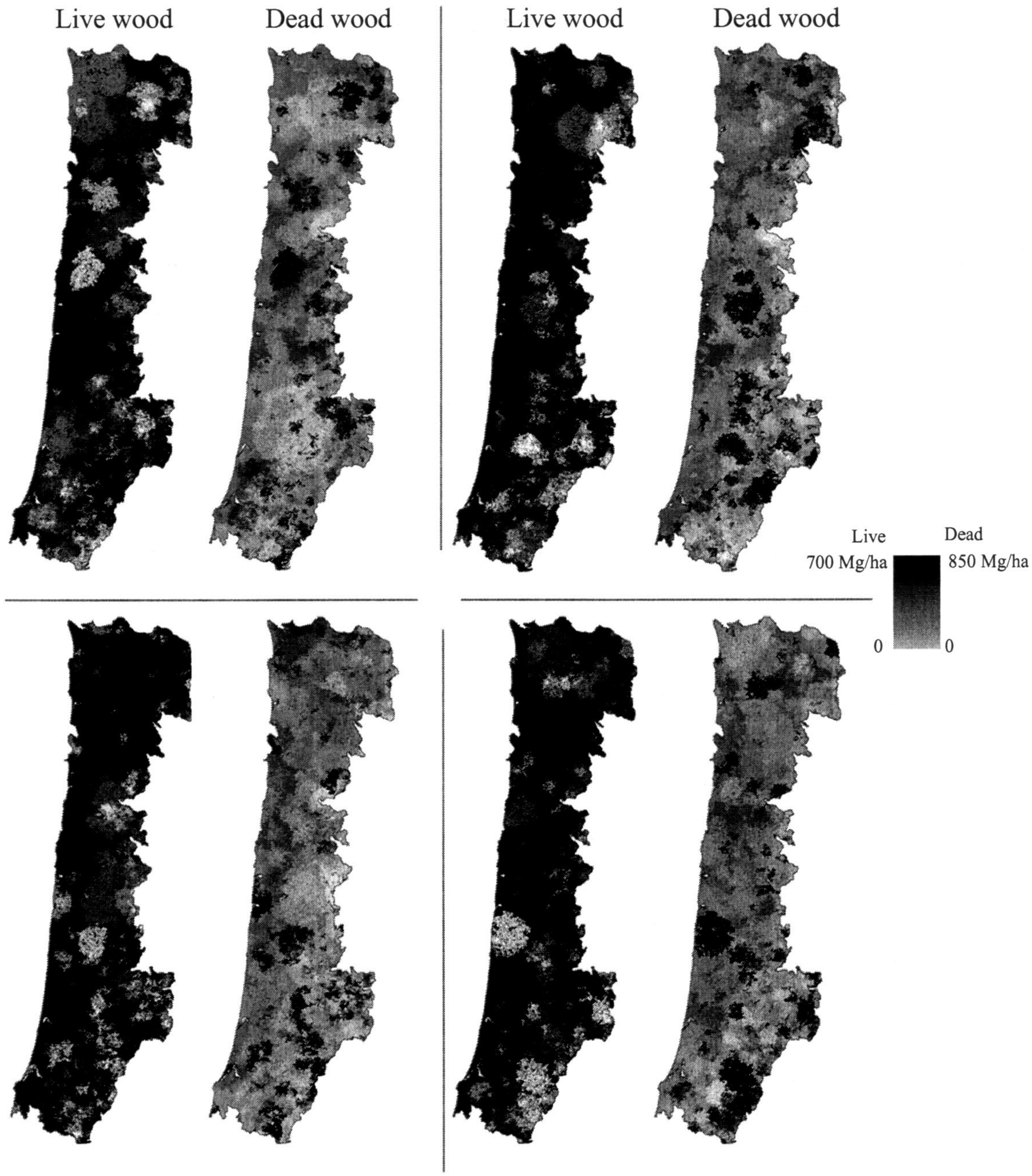

Figure 3.5: Examples of live and dead wood biomass patterns from simulation runs of LADS for the Oregon Coast Range, USA. The display color was stretched to the min and max values.

all 4 scales, which indicates that dead wood biomass is more evenly distributed spatially than is live wood (Table 3.2). Landscape pattern of live and dead wood differed considerably among the maps, indicating high temporal and spatial variability in distribution of live and dead wood biomass in the historical landscapes.

Expressed in terms of area, two live-dead wood biomass classes were far more abundant than the other classes (Table 3.3). The most abundant class was very high live wood-moderate dead wood, which occupied 28.6% of the landscape. The second most abundant class was high live wood-low dead wood, which covered 27.4%. Twelve classes, mostly the combinations of high live and dead wood biomass, did not occur on the historical landscapes. Forests with very high dead wood only occurred as a combination with very low live wood class. On average, > 80% of the historical landscapes was in low to moderate dead wood classes (Fig. 3.6). Separate analysis for the two climate zones showed that amounts of dead wood was less in the interior zone than the coastal zone (results not shown). The very low dead wood class was more common in the interior (mean = 6.2%) than in the coastal (mean = 0.2%). About 53% of the coastal zone was in the moderate dead wood class, while 57% of the interior was in low dead wood class.

The current landscape contains lower levels of both live and dead wood biomass than the historical landscapes. The two classes that were historically most abundant have reduced to nil (Table 3.4). Especially, forests with very low live and dead wood occupy more than 25% of the current landscape although it did not occur historically. The areas of all the dead wood classes were outside the HRV on the current landscape (Fig. 3.6). Forests with very low dead wood currently occupy 57.4% of the landscape, compared to 2.5% under the historical regime. The areas occupied by higher dead wood classes were lower than the historical levels. The low and moderate dead wood classes had been particularly reduced from their historical levels.

The historical distributions of dead wood classes differ among age classes (Fig. 3.7). Within each age class, there were variations both in dead wood amounts and in area occupied by the dead wood classes. Higher amounts of dead wood were typical

Table 3.2: The mean variance of live and dead wood biomass from the texture analysis by analysis window size (in number of pixels). The spatial distributions of live and dead wood biomass are significantly different (two-sample t-test $p < 0.01$ at all scales).

Window size	Live	Dead
3x3	121.0 (116.6-125.3)	43.8 (40.7-47.0)
11x11	162.3 (156.7-167.8)	59.5 (55.6-63.4)
31x31	219.3 (212.1-226.4)	80.4 (75.3-85.5)
51x51	259.8 (251.3-268.2)	94.5 (88.5-100.5)

Table 3.3: The mean percent coverage of live and dead wood biomass classes in the Oregon Coast Range. The values in the parentheses are 5^{th} and 95^{th} percentiles. The shaded classes were the most abundant and examined in sensitivity analysis.

Live wood biomass (Mg/ha)		0-50 Very low	51-100 Low	101-200 Moderate	201-400 Moderately high	401-600 High	> 601 Very high
Very high	>680		3.68 (2.69-4.76)	28.63 (19.73-36.43)			
High	501-680		27.43 (20.14-36.73)	4.54 (3.1-6.15)			
Moderately high	351-500	1.94 (1.13-3.27)	7.55 (3.87-14.56)	3.16 (2.27-4.26)	3.71 (2.4-5.15)	0.49 (0.16-1)	
Moderate	201-350	0.51 (0.18-1.05)	2.25 (1.08-4.21)	3.17 (1.59-6.62)	2.27 (1.6-3.01)	0.71 (0.29-1.34)	
Low	101-200	0.02 (0-0.05)	0.33 (0.12-0.7)	1.42 (0.72-2.7)	1.76 (0.62-4.37)	0.06 (0.02-0.11)	
Very low	0-100		0.03 (0-0.1)	0.56 (0.18-1.45)	2.21 (0.91-4.82)	1.89 (0.58-4.42)	1.70 (0.55-4.23)

Dead wood biomass (Mg/ha)

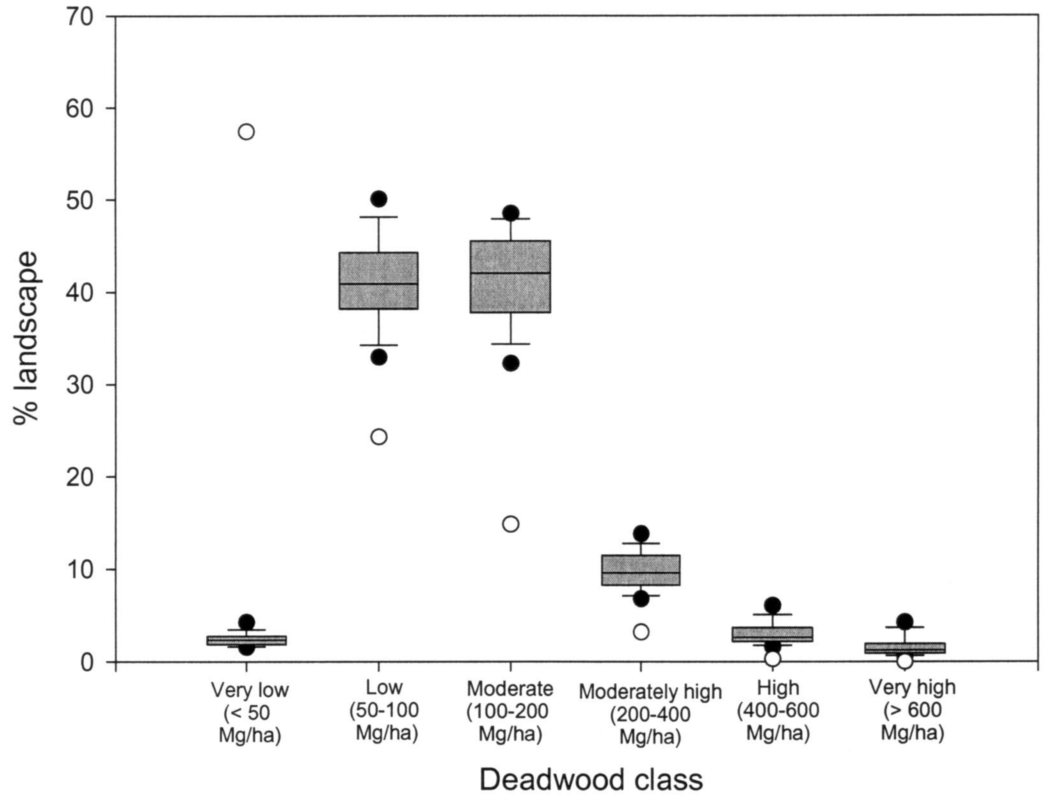

Figure 3.6: The historical ranges of variability in percent of landscape covered by different deadwood biomass classes based on 100 simulated landscapes. The open circles are the current levels of deadwood distribution (Ohmann unpublished data). Boxplots display the median (center line), 25 and 75% quantiles (box), 10 and 90% quantiles (whiskers), and 5 and 95% quantiles (solid circles).

Table 3.4: The mean percent coverage of live and dead biomass classes for the current landscape of the Oregon Coast Range. The numbers in italics are the difference between the current and historical levels (current – historical mean). If the values are positive, there is more area covered by the class on the current landscape than the historical landscapes.

Live wood biomass (Mg/ha)		0-50 Very low	51-100 Low	101-200 Moderate	201-400 Moderately high	401-600 High	> 601 Very high
Very high	>680	0.09 *0.09*	0.64 *-3.03*	0.41 ***-28.21***	0.06 *0.06*	0.27 *0.27*	
High	501-680	1.37 *1.37*	0.71 ***-26.72***	1.00 *-3.54*	0.06 *0.06*	0.04 *0.04*	
Moderately high	351-500	2.74 *0.80*	2.08 ***-5.47***	2.35 *-0.82*	0.81 *-2.90*	*-0.49*	
Moderate	201-350	9.06 ***8.55***	4.20 *1.95*	3.53 *0.35*	0.87 *-1.40*	*-0.71*	
Low	101-200	18.85 ***18.83***	6.33 ***6.01***	3.66 *2.24*	1.14 *-0.62*	*-0.06*	
Very low	0-100	25.31 ***25.31***	10.30 ***10.27***	3.89 *3.34*	*-2.21*	*-1.89*	*-1.70*

Dead wood biomass (Mg/ha)

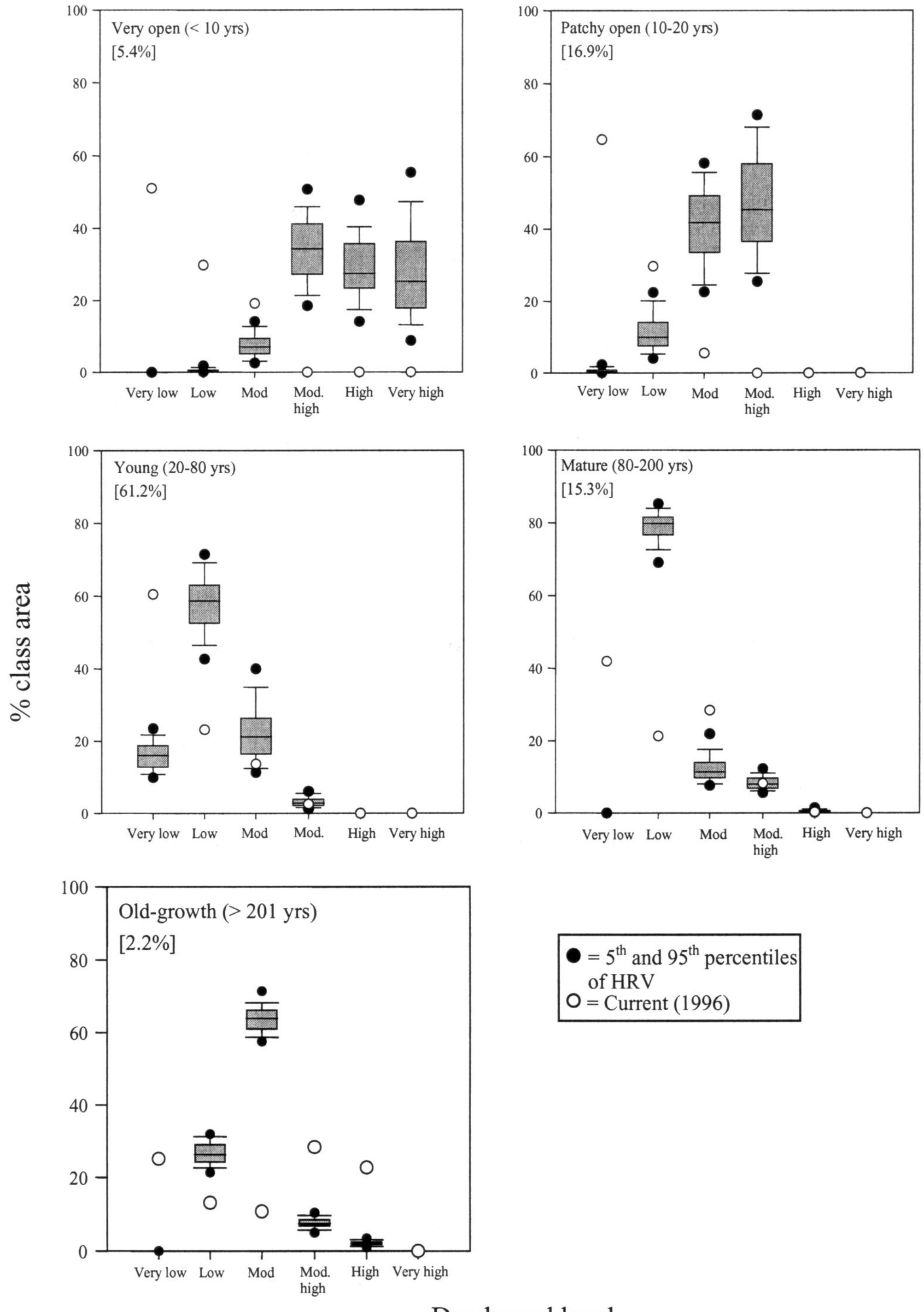

Figure 3.7: The historical ranges of variability by age class in percent of landscape covered by different deadwood biomass classes. The numbers in brackets are percent of current landscape in the age class. The interpretations of boxplots are the same as in Figure 3.6.

of the very open and patchy open classes. Lower dead wood classes became more common in young and mature classes, as about 80% of mature class contained low amounts of dead wood. The moderate dead wood class increased in old-growth forests. The very low dead wood class did not historically occur in very open, mature, and old-growth classes. The very high dead wood class occurred only in very open. The patchy open and young classes contained very low dead wood class, but such condition was rare on the landscape (2.5% of the region; Table 3.2).

The general trends of the current dead wood biomass for all the age classes were similar to the overall except old-growth class (Fig. 3.7). Forests with very low dead wood were consistently greater in the current landscape than the HRV across all the age classes. The areas of higher dead wood classes of the very open and patchy open classes were lower than the historical levels, while the area of greater than moderately high dead wood of the young and mature classes were in or close to the HRV of the landscape. The estimate of the old-growth class should be viewed with caution because only a few inventory plots fell in this currently rare type.

Characterizing variation in disturbance history and stand development

No single disturbance history type dominated fire histories, and the relative frequencies were different between the coastal and interior zones (Table 3.5). Fires in the very young stage or reburns (CR and PR) were less frequent in both zones. Mature and young stages burned more often than old-growth in the interior zone.

At the scale of 1000 yrs, stand histories were diverse. Variations in 1000-yr disturbance histories are numerous and not dominated by any particular disturbance history (Table 3.6). I identified 230, out of 256, unique 1000-yr disturbance histories (e.g., CO-PO, CO-CM-PO-PM, etc.). The frequencies were widely distributed among 1000-yr disturbance histories that occurred, and even the most frequent ones happened only 5 to 6% of fire occurrences in either zone. The overwhelming majority of the 1000-yr disturbance histories were characterized by two or more disturbance history types. The stands that experienced only high-severity fire in the old-growth stage

Table 3.5: The frequency (%) of 8 disturbance history types by the climate zones. For the disturbance history types, see Fig. 3.3.

Coastal

Pathway types	count	%
CO	2169	15.1
CM	2368	16.5
CY	2015	14.0
CR	1281	8.9
PO	1943	13.5
PM	2138	14.9
PY	1712	11.9
PR	737	5.1
Total	14363	100.0

Interior

Pathway types	count	%
CO	1977	6.0
CM	4271	12.9
CY	5106	15.4
CR	2913	8.8
PO	3325	10.0
PM	6447	19.4
PY	6258	18.9
PR	2855	8.6
Total	33152	100.0

Table 3.6: The summary statistics for 1000-yr disturbance histories.

	Coastal	Interior	Coast Range
Total number of samples	6200	3800	10000
No. of unique 1000-yr disturbance histories	213	204	230
Range of occurrence	1-389	1-212	1-399
Range of occurrence (%)	0.02-6.27	0.03-5.58	0.01-3.99
Frequencies of selected 1000-yr disturbance histories (%)			
Single fire type	7.71	0.47	4.96
Only high-severity fire in old-growth stage (Only CO)	4.05	0.08	2.54
Fire only in old-growth stage (Only CO or PO)	13.77	0.71	8.81
Only high-severity fires (Only CO, CM, CY, or CR)	10.68	0.50	6.81
Only moderate-severity fires (Only PO, PM, PY, or PR)	6.37	2.47	4.89
Both high- and moderate-severity fires	82.95	97.03	88.29
No fire	1.11	0.03	0.70
At least one fire in old-growth stage (At least one CO or PO)	97.45	91.47	95.18
At least one reburn (At least one CR)	23.21	47.87	32.58

were merely 4% in the coastal zone and 0.1% in the interior (Only CO in Table 3.6). The stands that experienced fire only in old growth were also uncommon (Only CO and PO), and the majority experienced fire in younger stages as well as in old-growth. However, old-growth forests burned at least once in a 1000-yr period in 95% of the region. The majority of the stands experienced both high- and moderate-severity fires in the 1000 yrs, and stands which experienced no fire were rare (0.7%).

Sensitivity analysis

The relative change in mean % of area in the region for the most abundant 10 biomass classes varied considerably among parameters (Table 3.7). The largest change was 346% (chronic mortality) for very high live-low dead wood biomass, but most of the changes were less than 50%. Although some changes seemed large numerically, the overall trend in the distribution of biomass classes did not change considerably from the baseline run, and the changes were due to shifts to or from neighboring classes. The influential parameters in the tested ranges were Chapman-Richards parameter a (potential maximum live wood biomass), decay rates, chronic mortality rates, and Chapman-Richards parameter b (growth rate). All these parameters had relatively large influence on high live-moderate dead wood class and very high live-low dead wood class (Table 3.7). These influential parameters controlled the live wood biomass growth (Chapman-Richards parameters a and b) and input and output rates of dead wood pool (chronic mortality and decay rates) in the model. The initial time lag in the growth model controlled by Chapman-Richards parameter c within the tested range was not as important. Likewise, the parameters for fire consumption of wood, fire mortality, and fire regime (MFS, NFR, and SEV) were not as important.

Table 3.7: Results from sensitivity analysis of LADS model simulations of historical Coast Range landscapes. The values are percent changes in mean values of biomass classes relative to baseline run.

Biomass			P_a		P_b		P_c		CMORT		CONS		DECAY		$FMORT_h$		$FMORT_m$		MFS		NFR		SEV	
Live wood	Dead wood	Baseline	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low
Moderate	Low	2.25	-21	97	-18	123	34	-8	-22	-5	-9	-17	35	-51	-6	-25	0	-6	-14	-7	-13	9	16	-19
Moderate	Moderate	3.17	-10	-14	9	-33	-53	14	4	10	-6	0	-28	19	0	-4	0	0	-14	8	-14	22	10	-14
Moderate	Moderately high	2.27	-36	93	-17	5	-6	5	19	8	8	-2	-7	43	-3	44	-8	-25	-11	6	-18	28	-17	17
Moderately high	Low	7.55	-60	119	-46	-9	-42	-29	2	-19	-9	-4	-21	-18	-14	-30	6	-3	3	-17	-12	1	21	-18
Moderately high	Moderate	3.16	-49	29	-42	26	4	-7	18	15	4	-4	18	43	5	29	-1	-25	-4	14	-12	20	-8	16
Moderately high	Moderately high	3.71	-23	-80	9	-22	-5	0	19	-21	2	9	-30	28	0	11	-14	25	0	7	-2	0	-19	23
High	Low	27.43	-63	16	-7	4	1	3	-43	7	0	1	12	-37	9	9	1	-1	2	-1	-2	8	5	0
High	Moderate	4.54	19	230	48	23	1	27	268	-58	-4	-1	-65	-91	15	2	-16	34	4	8	-3	0	-15	17
Very high	Low	3.68	168	-100	60	-61	-9	12	13	346	2	1	260	-100	5	3	-7	7	1	2	21	-28	-5	8
Very high	Moderate	28.63	42	-100	15	-18	-1	3	-5	-48	2	-2	-35	-10	0	5	2	3	4	2	18	-20	-7	2

DISCUSSION

Model limitations and assumptions

Disturbance is one of many important factors that determine the distributions of biomass on a landscape (Bormann and Likens 1979, Peet 1981, Shugart 1984, Harmon et al. 1986), and this study focused solely on the effects of fire history. The relative importance of disturbance and other factors, however, depend on scale and geographic locations. Fire history has an important influence on dead wood amounts in the Oregon Coast Range because infrequent stand-replacing fires create a large pulse of dead wood that persists for a long period of time due to large biomass and decay resistance of Douglas-fir trees (Spies et al. 1988). Disturbance history is also important in sub-boreal forests of British Columbia (Clarke et al. 1998) and in southeastern Canadian boreal forests (Hély et al. 2000), where stand-replacing fires are the major disturbance regime. In other locations, where gap-phase dynamics of forests is the major disturbance regime, studies identified autogenic mortality (Rouvinen and Kuuluvainen 2001, Karjalainen and Kuuluvainen 2002, Muller 2003), productivity (Spetich et al. 1999) and topographic gradients (Rubino and McCarthy 2003) as important factor at landscape to regional scales. Important factors other than fire may be incorporated in the future studies, but because these factors are scale-dependent, incorporating many factors in one model would require complex model structure and many parameters to estimate. LADS simulates the processes at the grain size of 9 ha to the extent of the Oregon Coast Range, and the scale limits factors that can be considered.

In the model, growth, mortality, and decay rates were deterministically assigned to pixels, based on the state of pixels. Therefore, it did not include multiple pathways of live and dead wood biomass dynamics in terms of these three aspects. The model represented the live wood dynamics by one equation although there are many factors that affect biomass growth in a stand. Productivity, availability of seed source, and species present after fire are likely to affect the rate at which trees fill in stands and grow (Shugart 1984, West et al. 1981). In the western Pacific Northwest, tree regeneration can sometimes require many decades to reoccupy a site (Franklin and Hemstrom 1981, Huff 1995). Studies found that old growth forests in the region may have started with a low or high density (Tappeiner et al. 1997, Poage and Tappeiner 2002, Winter et al. 2002). Likewise, the model used only one set of decay and mortality rates that was varied with time since the most recent fire. Decay and mortality rates actually change over time and space, reflecting site conditions (e.g., moisture and temperature) and differential mortality rates among locations and species (Muller 2003). Hence, this study probably underestimated variations in biomass dynamics.

Sensitivity analysis indicated that chronic mortality and decay rates were influential factors in the model, and therefore, accurate estimates of the rates are desirable. Size and species of dead wood affects decay rates (Graham 1982, Harmon et al. 1986, unpublished data, Stone et al. 1998), and fragmentation is faster for snags than logs. This model assumed relatively large decay/fragmentation rates early in stand development to reflect abundant snags after fire and small dying trees from suppression, but large pieces of legacy wood could substantially decrease overall decay rates. Stand density affects density-dependent mortality, especially in young stands, but the variation in stand density was not considered in this model. Weighting decay/fragmentation rates by the proportions of snags and species and by size distribution of dead wood will improve the estimation. Stand density may be affected by many factors such as seed availability and edaphic conditions, which may require more process-based modeling schemes (e.g., LANDIS). The structure of LADS could not easily incorporate these factors.

Spatial patterns of live and dead wood biomass in the Oregon Coast Range

Although the same fire histories influence live and dead wood biomass in a stand, the two components of forest structure produced different spatial distribution patterns at the regional scale. The texture analysis indicated that dead wood was more evenly distributed at moderate amounts over the landscape than live wood at all the four spatial scales examined. The distribution of live wood was more contrasting at all scales, suggesting higher spatial variability in live tree structure than dead structure. It was probably because dead wood biomass integrated longer fire history than live wood, making the dead wood biomass more "buffered" against recent disturbances. While growth largely determined live wood biomass, carryover legacies, standing stock, and mortality and decay rates interacted to affect the dead wood pool in the model. This complexity and associated time lag among the factors act to decouple live and dead wood biomass distribution at multiple scales.

At the regional scale, dead wood was historically abundant. The spatial distribution of dead wood biomass suggests that the stands in the Coast Range forests historically contained 50 to 200 Mg/ha of dead wood. There were also large patches of high dead wood biomass. There are many wildlife species that use dead wood in western Oregon (e.g., Maser et al. 1988, Ohmann et al. 1994, Butts and McComb 2000, Johnson and O'Neil 2000), and down logs function as conifer seedling establishment sites as known as "nurse logs" (Harmon and Franklin 1989). Dead wood is an important component of nutrient and hydrologic cycles and geomorphic processes (Harmon et al. 1986). With respect to dead wood amounts, the landscape was well connected for ecological processes at the regional scale.

At the regional scale, the two components of forest structure showed significantly different spatial patterns. However, because spatial resolution of this study is 9 ha, spatial pattern at finer scales cannot be inferred from the results. Amounts of live and dead wood are highly variable within a stand because of variability in environmental and disturbance factors (Bormann and Likens 1979, Harmon et al. 1986, Ohmann and Waddell 2002). The results from this study are specific to the regional scale.

The HRV and current distribution of live and dead wood biomass

The simulations indicated that historical landscapes of the Oregon Coast Range contained a wide variety of live and dead wood biomass. The most abundant biomass classes corresponded to mature and old-growth forests with their characteristic amounts of dead wood. The overall pattern of the distribution was somewhat expected based on the previous studies, which suggested that 50 to 70% of the region was covered with mature or old-growth forests (Wimberly et al. 2000, in press, Wimberly 2002). I also expected low coverage of high to very high dead wood because this type of structure is only associated with a condition shortly after a high-severity fire in mature or old-growth forests. In contrast, although reburns were expected, it was not clear how much they influenced the overall biomass distribution. Likewise, uncertain were the relative influence of moderate-severity fires on dead wood amounts in young and mature forests. Moderate-severity fires were important for the abundance of the classes with moderate to high live and dead wood biomass (the middle part of Table 3.3).

Very low dead wood conditions occurred on the historical landscape, and the amount can be as low as what is observed in young plantation forests. Spies and Cline (1988) reported that intensive plantations may contain only 20 to 40 Mg/ha of dead wood. As the model suggests, reburns created very low dead wood conditions on the historical landscapes, and such condition was associated with low to moderately high levels of live wood, which approximates very young to young forests (10-80 yrs). However, at the regional scale, I found that stands with very low dead wood were not historically common, covering only 2.5% of the region. Therefore, on the current landscape, the dead wood levels found in young plantations can be within the HRV at the stand scale but is outside the HRV at the regional scale.

Dead wood biomass in stands across age classes varied widely around the general U-shape. The variation arose from variation in fire histories (Spies et al. 1988) and partially from climatic difference between the two zones. The variation in fire histories determines the variation in amounts of legacy wood as well as the variation of the magnitude of the peaks in dead wood input in fire events. The history effects are larger in younger stands because, according to the model, most of legacy wood becomes insignificant in the dead wood pool by about 100 yrs, and mortality from the developing stands becomes more important in later seral stages. The climatic difference was also important since 97% of the area with the very low level of dead wood occurred in the interior zone where wood decays faster and reburns were more common.

The large area of very low levels of live and dead wood on the current landscape are likely a result of widespread wildfire in mid-1800s and timber harvesting after WWII, especially clear-cutting, which has prevailed in this region. In addition, short rotations (< 40 yrs) prevent stands from accumulating large wood biomass. Dead wood biomass variation under intensive forest management probably falls mostly outside the HRV (Fig. 3.8). This study supports recent changes in forest management that are designed to retain or increase dead wood in managed forests (the Northwest Forest Plan, Oregon Forest Practices Act, State Management Plans).

Variability in disturbance history

The results indicated that no single disturbance history type dominated the Oregon Coast Range under the historical fire regime. It was somewhat surprising to see that the frequencies were widely distributed among the 8 disturbance history types. I expected that fires in old growth and mature classes would have been more common, given the natural fire rotation (NFR). For example, in the coastal zone the NFR was 200 yrs, but the frequencies were nearly evenly distributed among age classes. Environmental heterogeneity and stochasticity in fire occurrence and spread resulted in fires occurring throughout the developmental stages in the region.

There were also numerous 1000-yr disturbance histories. Although the moderate sampling (10,000 pixels) was likely to result in low variation, the result represented most of the all possible patterns of 1000-yr disturbance history (230 out of 256). The results suggest that the three fire histories presented in Spies et al. (1988),

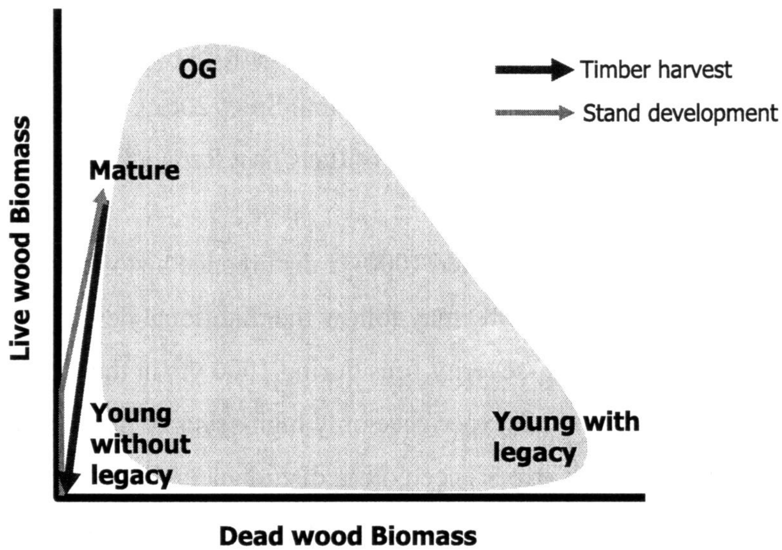

Figure 3.8: A schematic diagram showing comparison between the HRV of biomass under the historical fire regime and typical biomass found in plantations in the Oregon Coast Range.

which represented CO, PO, and CR in this study, were not the most common disturbance histories in the region. Since a variety of stand structure and species composition can arise from various disturbance histories (Noble and Slatyer 1977, Cattelino et al. 1979, Spies 1998, Wimberly and Spies 2002), highly diverse forests structure was probably the characteristic of the Coast Range forests under the historical fire regime.

The examinations of selected 1000-yr disturbance histories showed that only a small part of the landscape (~7%) may follow the traditional developmental pathways characterized solely by high-severity fires during 1000 yrs in the Coast Range, and only a fraction of them (~3%) experience only high-severity fire in old growth stages. The majority of the stands experienced other disturbance history types including moderate-severity fires and reburns in addition to fires in old growth. Frelich and Lorimer (1991) in their simulation study also found that stand development without subsequent disturbance was much less common in the northern hardwood region of Michigan, where about 87% of the landscape experienced partial disturbance after stand establishment. Although the traditional models of forest stand dynamics (e.g., Oliver 1981, Bormann and Likens 1979, Spies and Franklin 1996, Franklin et al. 2002) helped ecologists simplify and understand the patterns and processes in forest stand development, disturbance histories are quite diverse. This variability and complexity is difficult to describe in a linear sequence of stand developmental stages, and "a web of development" may be a better description for multiple pathways in stand development (Frelich 2002).

Management implications

The variation over time and space in dead wood amounts points out the importance of having a regional perspective on achieving forest structure goals. The current landscape is composed of multiple ownership types with different management objectives, and retention of substantial amounts of dead wood is not feasible on intensively managed lands. The variation in dead wood biomass, however, allows flexibility in dead wood restoration at the regional scale. Although the full historical level of dead wood retention would not be an attainable goal on lands managed for timber production, retaining higher amounts of live and dead wood biomass after harvest where management goals are amenable may help maintain wildlife habitat and ecological processes at the stand scale as well as the regional scale.

The temporal variability in disturbance under the historical regime observed in this study suggests some options for incorporating temporal variability in forest management. This study indicates that most sites experienced fires of variable severity at different developmental stages. Variations in rotation intervals and cutting methods (clear-cut vs. partial cut) over time would more closely follow natural disturbance regimes than management regime in which rotation lengths are fixed over time. However, incorporating this high level of spatial and temporal variability into management would be challenging and require thinking beyond a single rotation for a site. Maser (1995) proposed forest management to incorporate variable rotation ages over space and time at a large spatial and temporal scale in his concept of "restoration forestry." In restoration forestry, "sustainability of the forest is maximized and product extraction is optimized" by following "Nature's blueprint." Given our current state of knowledge on ecological functioning and long-term consequences of management practices, HRV can be used as a guide to introduce the idea of temporal and spatial variability in forest management.

The results indicated that the dead wood amounts on the current landscape are well outside the HRV. As long as forests are managed for timber extraction, dead wood inputs are reduced from the level of inputs from fire. The HRV of dead wood is an unattainable objective for timber extraction goals, but the use of HRV does not imply that historical conditions are the target for management. Departure from the reference level can serve as one indicator of trends in dead wood conditions. For example, the condition that is very low in both live and dead wood did not frequently occur historically, and such a condition is probably not a critical habitat type for persistence of native species.

CONCLUSIONS

Quantitative HRV studies to evaluating landscape conditions have focused on landscape structure. Those studies characterized patch types only based on overstory characteristics and neglected dead wood. The results of this study suggest that live and dead wood biomass have different spatial distributions because of lag time in dead wood biomass response to disturbance. This study also suggests that temporal variability in fire return intervals and intensity was historically high in the region, creating variety of forest stand structure and spatial patterns. The use of HRV approaches as a guide for forest management planning has been proposed and helped deciding harvesting rotation and patch size (e.g., Cissel et al. 1999). However, heterogeneity in dead wood distribution at regional scales is often not explicitly considered. This study points out that dead wood patches were also dynamic and had characteristic variation in area with different amounts. The HRV of overstory characteristics alone is not enough to evaluate ecological consequences of management because substantial difference exists in effects of natural disturbance and timber harvesting on stand structure that is not reflected in overstory characteristics such as stand age. The HRV approaches to evaluating landscape conditions need to include both landscape and stand characteristics to better represent ecological differences between managed and unmanaged landscapes.

The historical distribution of live and dead wood biomass in the landscape of the Oregon Coast Range was heterogeneous and dynamic over time. Live and dead wood biomass differently reflected temporal and spatial patterns of wildfire. Dead wood was ubiquitous in the forests at various amounts ranging from very low to very high, depending on fire history. The majority of the landscape was historically covered with low to moderate amounts of dead wood at the regional scale. On the current landscape, the majority contains very low amounts of dead wood probably because of prevalent clear-cutting and short rotations in the last few decades. The HRV approach provides a quantitative reference condition to evaluate management options at regional scales.

CHAPTER 4: VEGETATION CHANGE IN BOTTOMLANDS OF MAJOR COASTAL RIVER VALLEYS OF THE OREGON COAST RANGE FROM CA. 1850 TO PRESENT

ABSTRACT

Valley floodplains have been subject to human use in the Oregon Coast Range before and after Euro-American settlement started in the mid 1800's, but current landscapes of valleys in the region indicate long-term land conversion and development. The vegetation of these valleys before settlement is not well known. Characterizing valley bottomlands in the region is important for both assessing loss in habitat types and prioritizing areas for conservation and restoration because bottomlands provide unique habitats to various native species. The main objectives of this study were to 1) assess the distribution of valley bottoms in the Oregon Coast Range, 2) describe the historical and current vegetation pattern of two major river valleys (Coquille and Tillamook), and 3) compare the historical and current vegetation patterns of these two valleys. Valley bottoms in the region were identified in geographic information systems (GIS) using topographic criteria. I collected historical vegetation information from existing maps and the General Land Office (GLO) survey records. I characterized vegetation types for the two major valleys using an unsupervised classification of satellite images. The result indicated that valley bottoms identified in this study occupied 2.8% of the Coast Range. The two case studies suggested that the Coquille was dominated by hardwood trees and the Tillamook was by conifers. Valley bottoms in both areas differed in vegetation from nearby uplands. Treed areas have declined from 90% to 8% in Coquille and from 69% to 5% in Tillamook as a result of agriculture and development. The historical data offered reference conditions for assessment of changes in biodiversity that have occurred in these unique habitats.

INTRODUCTION

Valley bottom floodplains are one of the most human-altered landscapes in North America, Europe, and elsewhere (Sparks 1995, Klimo 1998, Mitsch and Gosselink 2000, Pedroli et al. 2002). Forests in valley bottomlands have been considerably reduced (Peterken and Hughes 1995, Wigley and Roberts 1997, Knutson and Klaas 1998), and remaining parts of such forests are under threat from continued land conversion, river channelization, and associated changes in disturbance regimes, mainly flood events. For example, more than 70% of an original 10 million hectare of bottomland hardwood forest has disappeared in the Lower Mississippi River Alluvial Valley of the United States (King and Keeland 1999). Mitsch and Gosselink (2000) estimated 53% of the wetlands in the lower 48 states were lost between 1780s and 1980s. In Europe, only a small fraction of original floodplain along the Rhine River remains as flooding surface (Schnitzler 1994).

Large valley bottomlands contain diverse ecosystems and habitats, and the diversity is intricately tied to hydrologic disturbance (Malanson 1993, Naiman et al. 1993, Naiman and Décamp 1997, Pollock et al. 1998). Valley bottoms are usually on alluvial floodplains and can contain various types of wetlands embedded in to create complex hydrologic systems and diverse habitat conditions (Malanson 1993, Naiman and Décamp 1997, Pollock et al. 1998, Mitsch and Gosselink 2000). Valley bottom forests and riparian areas also form unique ecotones between terrestrial and aquatic environment in a landscape (Malanson 1993, Calhoun 1999, Brinson and Verhoeven 1999), where many important ecological processes are maintained (Naiman and Décamp 1997). Wetlands are relatively rare ecosystems in the western United States compared to the eastern part of the country (Mitsch and Gosselink 2000), and thus the rarity of the ecosystem types could exacerbate the consequences of loss and alteration.

In the Coast Range of Oregon, as in other parts of the world, valley bottom floodplains have been centers of human settlement. Presettlement vegetation conditions in these valleys are difficult to infer from the current conditions. Recent studies found that hardwood tree species are more abundant than conifers in unmanaged riparian forests adjacent to streams in the Coast Range (Pabst and Spies 1999, Nierenberg and Hibbs 2000, Hibbs and Bower 2000). Although these studies used "unmanaged" sites, they may not represent historical riparian conditions due to indirect human influences (e.g., fires spreading from settlement, landslides induced by nearby roads). These studies also examined small upper streams, and it is uncertain if the findings generally apply to large valley bottomlands. Large valley bottoms also have wet soils but have geomorphology and fluvial dynamics that differ from relatively narrow flooding zones along upper streams (Malanson 1993, Naiman et al. 2000). There is no study that has examined historical vegetation quantitatively in either small riparian or large valley bottoms using historical records in the Oregon Coast Range. It is essential to quantify vegetation changes occurred in valley bottomlands since settlement for understanding potential diversity of the region.

The objectives of this study were to 1) assess the distribution of valley bottoms in the Oregon Coast Range, 2) describe the historical and current vegetation types of the Coquille and Tillamook Valleys, and 3) compare the historical and current vegetation pattern of the two valley bottomlands. The study period extends from Euro-American settlement, about 150 years ago, to the present. I selected the Coquille and Tillamook Valleys for case studies because they are the major large valleys in the region, and because maps of their historical vegetation cover have already been developed by Benner (1991) and Coulton et al. (1996). Historical vegetation was reconstructed for the two valleys using the General Land Office (GLO) Public Land Survey (PLS) (Stewart 1935) and the Donation Land Claim (DLC) records from the 1850s to 1870s.

METHODS

Study areas

The Oregon Coast Range

The Oregon Coast Range is a 2-million-ha physiographic province in Oregon, USA (Fig. 4.1) and is an ecologically complex and socially diverse region. It is characterized by highly dissected mountains, steep slopes, and a high density of streams. Elevations range from the sea-level to about 1250 m. The climate of the region is characterized by mild wet winter and dry cool summer. Two major vegetation types, the *Picea sitchensis* (sitka spruce) Zone and *Tsuga heterophylla* (western hemlock) Zone, cover the region (Franklin and Dyrness 1988). The *Picea sitchensis* Zone is found at lower elevations along the coast, extending about 5 km inland from the Pacific Ocean and can extend further along valleys. The major tree species found in this zone are sitka spruce, western hemlock, and western redcedar (*Thuja plicata*). The *Tsuga heterophylla* Zone lies further from the ocean, and the moisture and temperature regime of this zone is more extreme relative to the previous zone. The major tree species are western hemlock and Douglas-fir (*Pseudotsuga menziesii*). Hardwoods are not common but found in riparian and disturbed areas and at lower elevations. The major species are red alder (*Alnus rubra*) and bigleaf maple (*Acer macrophylla*) (Franklin and Dyrness 1988). Historically, hardwoods were mostly restricted to riparian areas but have increased its abundance in uplands due to forestry practices in the early to late 20th century that provided regeneration sites (Spies et al. 2002a).

Before the advent of effective fire suppression around 1940s in western Oregon (Weisberg and Swanson 2003), infrequent, high-severity fires burned the forests in the *T. heterophylla* Zone (Agee 1993, Fahnstock and Agee 1983, Teensma et al. 1991, Ripple 1994). In the *P. sitchensis* Zone, forests burn rarely due to its moist microclimate, and wind is the major local disturbance (Agee 1993).

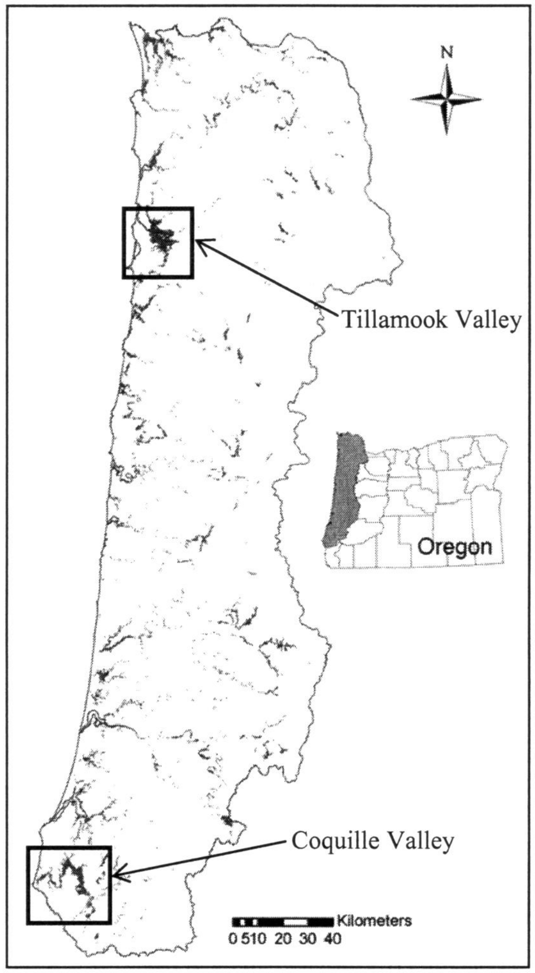

Figure 4.1: The map of the Oregon Coast Range and the valleys identified in this study. The Coquille and Tillamook Valleys are the case study areas.

Native American peoples are known to have resided in the Coast Range before Euro-American settlers arrived in the region and often built their campsites and villages near major rivers and estuaries (Boyd 1999, Zybach *personal communication*). The GLO surveyors noted villages of Native Americans along the coast in Tillamook (Coulton et al. 1996). Use of fire by native peoples is not as well documented as other parts of North America probably because of high precipitation and ample marine resources, but some evidence suggests that they used fire to enhance deer hunting and berry production in major valleys in the region (Boyd 1999). Large to small prairie existed in the two valleys when the survey was conducted (Benner 1991, Coulton et al. 1996).

Coquille Valley

The Coquille River in southwestern Oregon forms one of the major valleys in the Oregon Coast Range (Fig. 4.1). It flows through Coos County, Oregon, and drains into the Pacific Ocean. The valley bottomland delineated in this study (see Methods section) was about 7699 ha and about 50 km long east to west with flat bottomland of variable widths ranging from about 0.2 km to 4 km. The hydrology of the river close to the mouth is largely affected by the ocean, where an extensive estuary has developed (Benner 1991, Hall 1995). The vegetation of the area falls in the *Picea sitchensis* Zone (Franklin and Dyrness 1988). Settlement by Euro-Americans had already begun when the GLO surveyors arrived between 1850 and 1870.

The landform of the valley bottom is mostly floodplains and terraces, and the soils are typical of the two landforms (Haagen 1990). The floodplain soil is poorly drained to very poorly drained with silty and clayey soils. The soils on the terraces are somewhat better drained with similar texture. Plant rooting is likely to be limited by water table for water-intolerant species except on the 100-year floodplain.

Tillamook Valley

Tillamook Valley is located in the northwestern part of the region (Fig. 4.1). It is an extensive valley bottom formed by several rivers—the Kilchis, Wilson, Trask, and Tillamook—all draining into the Tillamook Bay. The valley bottomland delineated in this study (see Methods section) was about 7915 ha, about 20 km long north to south, and about 10 km wide at the widest point. Like the Coquille Valley, the hydrology of the river is affected by the ocean, and a large estuary is located at the mouths of the rivers. The vegetation of the area also falls in the *Picea sitchensis* Zone (Franklin and Dyrness 1988). Settlement started around 1851, and farming was the primary land use (Coulton et al. 1996).

The landform of the western half of the valley bottom is stream bottoms and of the eastern half is old alluvium terraces (Bowlsby and Swanson 1964). Soil drainage varies from poorly drained to well-drained, but the majority of the valley bottom is well drained. The soils are strongly acidic and high in organic matter.

Developing the valley map for the Oregon Coast Range

I developed a GIS coverage of major valley floors in the Oregon Coast Range using several topographic criteria in GIS. The objective of this mapping was to quickly identify large valley floors in the region. The mapping criteria were 1) elevation less than 200 m above the sea level, 2) slope less than 2 degrees, 3) proximity ($\leq$ 200 m) to 5th or higher order river channels, and 4) area $\geq$ 100 ha. I used the first 3 criteria to identify pixels of valley bottomlands associated with large rivers. Then I clustered the valley pixels and used the 4th criterion to eliminate single or clusters of pixels representing < 100 ha. The Willamette Valley and associated small valleys were excluded from the analysis because they reside in different climatic conditions and are probably characterized by different vegetation. The resultant patterns of valley floors (Fig. 4.1) reasonably matched the outlines of valleys on the USGS topographic maps.

Historical vegetation

Benner (1991) and Coulton et al. (1996) developed the historical vegetation cover maps for the Coquille and Tillamook Valleys, respectively. The maps were developed using vegetation descriptions from GLO PLS and DLC records collected between 1857 and 1872 for Coquille and between 1856 and 1857 for Tillamook. The GLO survey records are a valuable resource for studying pre-settlement vegetation (Nelson et al. 1994, 1997, Manies et al. 2001, Schulte and Mladenoff 2002), and historical vegetation maps have been constructed from the GLO survey records for elsewhere in United States. The DLC records are similar to PLS and covered lands that were already claimed by settlers before the implementation of the PLS system, but the coverage is limited in the case study areas (Loy et al. 2001). The GLO surveys were conducted along township and section lines (but property lines for DLC), and the surveyors noted in their field journals vegetation types, soil suitability to cultivation, abrupt changes in vegetation cover, and some landmarks they observed (General Land Office 1851, Stewart 1935). Benner (1991) and Coulton et al. (1996) delineated historical vegetation types according to vegetation descriptions, soil types, and topography. Soil survey and USGS topographic maps were also used to help delineation and to confirm the validity of descriptions (e.g., ascending/descending slopes, soggy soils) in the survey records (Benner *personal communication*). The periods covered by the source data corresponded to early Euro-American settlement in the two valleys. The original map of the Coquille Valley had 12 vegetation classes, and that of the Tillamook valley had 10 classes. I manually digitized the original maps in Arc/Info (ESRI 1995).

The historical vegetation maps from the previous studies described the vegetation types qualitatively but did not quantify species composition and forest structure. I used the witness tree information from the GLO survey records[1] to estimate tree density, basal area, and relative frequency by species. I summarized tree

[1] Tree data were transcribed from the original surveyors' journals by P. Benner in the process of historical vegetation map development.

species composition, size distribution, and the estimated variables by vegetation class in the historical maps (Table 4.1 and 4.2). I used the Point-Centered Quarter (PCQ) method (Cottam and Curtis 1956, Mueller-Dombois and Ellenberg 1974) to estimate tree density based on distance from corners to trees. This method is often used for this purpose (e.g., Fralish et al. 1991, Schafale and Harcombe 1983, Manies and Mlandenoff 2000, Manies et al. 2001). I did not use corners that were located on or very close to the borders of vegetation types because the accuracy of the map was not high enough to determine the exact locations of the borderlines of the vegetation types. I also did not include trees recorded for meander corners and line trees in calculating tree density because meander corners were purposefully located in riparian areas and the PCQ method cannot apply to line trees. This procedure left 454 trees for analysis for Coquille and 400 for Tillamook. For each species, I calculated importance values (IV), which were calculated as the average of relative basal area and relative frequency (Curtis 1959). I also examined tree data from uplands surrounding the valleys for comparison with bottomland vegetation. The numbers of trees in uplands included were 799 for Coquille and 511 for Tillamook. I included a subset of the vegetation types on the historical maps—5 Vegetation Types (VT) for Coquille and 7 for Tillamook— in the analysis because some types occurred on uplands or coastal dunes (Table 4.1).

Current vegetation

I developed the current vegetation maps for the two valleys using remotely sensed imagery from Landsat Thematic Mapper (TM) taken in mid 1990s with digitized color aerial photographs and Digital Ortho Quadrangles (DOQs) from the same period as reference. I classified the TM images by using an unsupervised classification in ERDAS IMAGINE (Leica Geosystems 2002). The cover classes were 1) water, 2) developed/bare ground, 3) agriculture/sparse low vegetation cover, 3) broadleaf tree dominated, 4) conifer tree dominated, and 5) mixed tree cover. Developed/bare ground included the towns of Coquille and Tillamook, residential areas, and bare ground for uses other than agriculture. Agriculture/sparse low vegetation cover included agricultural fields and open areas with low vegetation. Some of the non-agricultural bare ground was difficult to distinguish spectrally from bare agricultural fields, and I used subjective criteria during photo interpretation to differentiate the two classes. Likewise, areas covered with low, sparse vegetation were difficult to distinguish from agricultural fields with hay. I was not confident with distinguishing the two types and thus lumped agriculture and sparse vegetation into a single class. Although potential ecological roles played by the two types would be different, I considered it acceptable for vegetation cover comparison. When broadleaf and coniferous trees were well mixed, I assigned mixed tree cover. Accuracy of the classifications was 88% for Coquille and 76% for Tillamook as determined by assessing the accuracy of class assignment at 100 randomly selected points with the aerial photographs and DOQs.

Quantifying vegetation cover change in the valleys

I further combined vegetation classes into either treed or not-treed types for comparison because the classification schemes were different between the historical and current maps. I used the "treed" category to generally represent the areas that contained tree canopy cover at various densities (Table 4.1). For current vegetation, the treed category included any of broadleaf, conifer, and mixed forests, while not-treed category included any of agriculture, open/bare ground, developed and low vegetation for both valleys. I lumped the historical vegetation types based on the vegetation descriptions and tree density. For the Coquille Valley, I combined VT 2 (timbered swamp with brush), 3 (timbered swamp with grass), 4 (timbered swamp with water), and 5 (wooded bottomland) into the treed category (Table 4.1). Although tree density was low for VT 5 in Coquille, I included it in the treed category following the description by Benner (1991). Only VT1 (marsh prairie) was in the non-treed category in Coquille. For the Tillamook Valley, I combined VT 2 (tidally-influenced forest), 3 (flood plain bottomland), 4 (timbered floodplain), and 5 (timbered valley

Table 4.1: Descriptions of the vegetation types on the historical maps and calculated vegetation characteristics using the witness tree data for the Coquille and Tillamook Valleys.

Vegetation type	Abbreviated name referred in the text	treed or not-treed	Brief description[a]	% valley[b]	Range of tree density[c] (trees/ha)	Quadratic mean diameter (cm)	Basal area (m^2/ha)	Elevation range (m)	Average slope (degrees)
Coquille Valley									
1 Marsh prairie	Marsh prairie	not-treed	Trees and woody vegetation was scarce. The area was subject to overflow. Generally, it was swampy and wet.	9.7	0.3-0.5	25.3	0.015-0.025	0.4-6.9	0.64
2 Timbered swamp with brushy understory	Timbered swamp with brush	treed	The areas were timbered with trees with understory predominantly composed of various brush and briers.	30.3	2.0-46.2	41.6	0.27-6.3	1.3-14.6	0.45
3 Timbered swamp with coarse grass understory	Timbered swamp with grass	treed	The soil was very mucky. The surveyors did not record standing water on surface except beaver ditches.	18.4	126.4-172.9	26.4	6.9-9.5	1.5-39.4	0.36
4 Timbered swamp with coarse grass understory	Timbered swamp with	treed	The soil was very mucky. The surveyors recorded standing water on surface with many pond lilies.	15.3	63.6-106.0	24.3	3.0-4.9	1.5-15.6	0.45
5 Wooded bottomland, mostly floodplain	Wooded bottomland	treed	The areas were generally not wet, and no standing water on surface was recorded. The areas were mostly within 100-year floodplain.	26.3	0.3-7.6	48.4	0.055-1.4	0.5-26.7	0.73

Table 4.1: continued.

Vegetation type	Abbreviated name referred in the text	treed or not-treed	Brief description[a]	% valley[b]	Range of tree density[c] (trees/ha)	Quadratic mean diameter (cm)	Basal area (m^2/ha)	Elevation range (m)	Average slope (degrees)
Tillamook Valley									
1 Grassy tidal marsh	Grassy marsh	not-treed	This land was regularly inundated by Bay waters.	5.2	0.2-2.3	61.6	0.06-0.68	0-19	0.35
2 Tidally-influenced forest	Tidally-influenced forest	treed	This land was forested with a variety of tree species. The surveyor described the ground surface as broken with tide fissures. The land was subject to inundation in heavy storms.	10.2	30.3-49.3	66.4	10.5-17.1	0-20	0.19
3 Main valley floodplain bottom land	Main bottomland	treed	The land was described as river bottom lands. The soils were described as good quality ("1st rate"). The land was forested with a variety of tree species. The land could be subdivided into areas within the upper reaches of the tidal river or beyond the area of typical tidal influence.	32.9	33.8-35.3	116.8	36.2-37.8	0-39	0.34
4 Upper valley timbered floodplain	Upper floodplain	treed	The land was outside of the river terraces and was forested with a variety of tree species except cottonwood, which indicates less occuurrence of flood events.	3.6	93.3-126.6	63.1	29.1-39.5	5-58	0.83
5 Timbered valley lands	Timbered valley	treed	The land was predominantly covered with timber.	22.5	11.8-41.0	130.3	15.7-54.7	0-65	0.84
6 Prairie lands	Prairie lands	not-treed	There was a very strong correlation between soil types and the prairie locations. More than 50 % of the corners (26 out of 44) marked with a stake or a mound.	20.0	0.3-2.7	86.2	0.18-1.6	0-52	0.67
7 Marshy and brushy swampland	Swampland	not-treed	Outside of tidal marsh lands and tidal bottomlands that appear to have been predominantly wetland. Only a few trees recorded.	5.6	Too few trees recorded to estimate vegetation characteristics.				

[a] Benner (1991) for Coquille Valley and Coulton et al. (1996) for Tillamook Valley.
[b] Estuary and water were excluded from % area calculation.
[c] The range was obtained by increasing distance by 25%, 50%, and 100% of the maximum distance recorded in the same vegetation type to the "nearest trees" in the calculations for corners without trees recorded .

Table 4.2: The most probable species for the plant names mentioned in the surveyors' field journals.

	The most probable species	
As recorded	Common name	Scientific name[a]
alder	red alder	*Alnus rubra*
ash	Oregon ash	*Fraxinus latifolia*
bearberry	bearberry	*Myrica californica* [b]
cedar	western redcedar	*Thuja plicata*
chittam	cascara buckthorn	*Rhamnus purshiana*
cottonwood or balm	black cottonwood	*Populus trichocarpa*
crabapple	western crab apple	*Pyrus fusca*
dogwood	Pacific dogwood	*Cornus nuttallii*
elder	elderberry	*Sambucus racemosa*
fir	Douglas-fir	*Pseudotsuga menziesii*
hazel	hazelnut	*Corylus cornuta*
hemlock	western hemlock	*Tsuga heterophylla*
maple	bigleaf maple	*Acer macrophyllum*
myrtle	Oregon myrtle	*Umbellularia californica*
sagwood[c]	?	*?*
spruce	sitka spruce	*Picea sitchensis*
tasselwood	silktassel	*Garrya fremontii*
vinemaple	vine maple	*Acer circinatum*
willow	willow	*Salix spp.*
yellow fir	grand fir	*Abies grandis*

[a] Nomenclature is from Hitchcock and Cronquist (1973) and Jensen and Ross (1999).
[b] There are multiple species that could be called "bearberry" in the region. Given that the surveyors were instructed to record trees > 5 in in diameter, this species is more probable than the common shrub species called "bearberry" (*Arctostaphylos uva-ursi*) in this region. *M. californica* can be a shrub or small tree.
[c] I could not find the corresponding species to "sagwood." This name seemed to have not been mentioned commonly before (Personal communication with Drs. Ed Jensen and Mike Newton, College of Forestry, Oregon State University).

Table 4.3: The percent of valley area covered by vegetation types on the current landscapes of the Coquille and Tillamook Valley based on an unsupervised classification of satellite imageries taken in mid 1990's.

vegetation type	% valley area	
	Coquille	Tillamook
Agriculture and sparse/low vegetation	89.5	87.2
Broadleaf	7.0	0.8
Conifer	0	0.7
Broadleaf/conifer mixed	0.6	3.7
Developed and non-agricultural bare ground	2.8	7.7

lands) into the treed category and VT 1 (grassy tidal marsh), 6 (prairie lands), and 7 (swampland) into the not-treed category (Table 4.1).

There was considerable variation within the treed category. I lumped areas with sparse trees into the treed category because 1) tree density in many vegetation classes ranged widely and overlapped with each other, 2) the GLO surveyors recorded some trees within reasonable distances (e.g., < 20 m), and 3) these structure types provided by sparse trees have different functions as wildlife habitat from areas with no trees, and 4) the simple, treed or not-treed classes are adequate to draw ecological conclusions because treed areas have been considerably reduced in the valleys.

RESULTS

Valley bottomlands in the Oregon Coast Range

Using the selection criteria, I identified 87 valleys, which occupied about 2.8% of the region (Fig. 4.1). The current vegetation map of the region from another source shows that non-forest covers 65.5% and forest covers 34.5% of the valleys.[2] Close to 90% of the valleys are owned by non-industrial private landowners (Fig. 4.2). The federal (Forest Service and Bureau of Land Management) and state governments manage only 3.8% of the valleys. This ownership pattern contrasts with that of upland forests, where non-industrial private, industrial ownerships, and collectively the three government agencies share a similar proportion (Fig. 4.2). The 87 valleys varied in size, ranging from 108.5 ha to 7915.3 ha. Tillamook Valley was the largest valley followed by the Coquille Valley.

---

[2] Johnson, K. N., Brooks, J. P., Biesecker, R. and Goodwin, J. In prep. Accuracy assessment of open/semi-closed classification. This map was not used for the comparison between the current and historical conditions of the valleys because the classification was not based only on cover types but also on land use and ownership types.

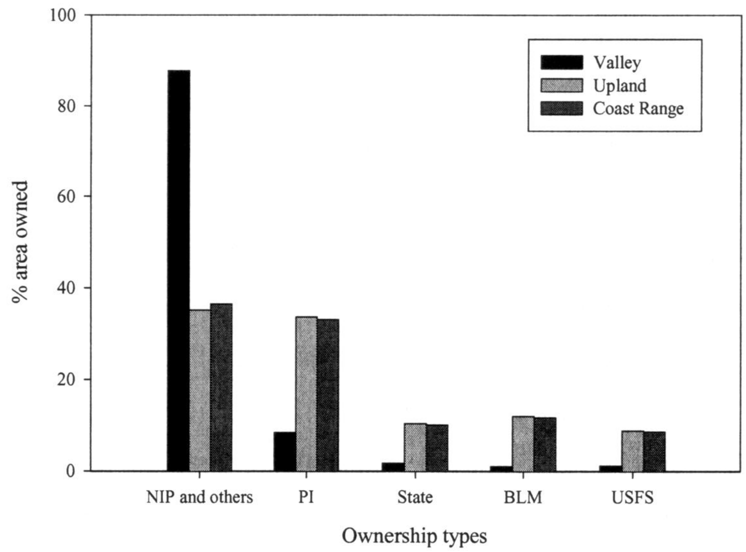

Figure 4.2: The allocation of ownerships in the valleys, uplands, and the Oregon Coast Range. The majority of the valleys are owned by non-industrial private or other ownership types, which is contrasting with upland and the whole region. NIP = non-industrial private, PI = private-industrial, State = State of Oregon, BLM = Bureau of Land Management, USFS = Forest Service.

Historical vegetation cover

Coquille Valley

General descriptions of the vegetation recorded by the surveyors suggest that the bottomland was generally swampy except narrow ridges along the riverbank and was generally timbered at various densities (Benner 1991). Trees historically covered 90% of the bottomland at various densities, suggesting that the canopy cover ranged from open to moderately closed (Table 4.1; Fig. 4.3a).

The common species found were maple, myrtle, willow, ash, alder and spruce (Fig. 4.4; for scientific names see Table 4.2). Maple and myrtle were more abundant in wooded bottomlands (VT 5), whereas ash and alder were common in other wetter types. Willow was abundant in timbered swamps. More than 50% of the trees recorded were less than 30 cm in diameter (Fig. 4.5a). Only wooded bottomlands (VT 5) had a relatively even distribution of diameters up to the 70-cm class. Trees larger than 100 cm (40 inches) in diameter were found only in timbered swamp with brush (VT 2) and wooded bottomlands (VT 5). Most of the trees were found within 20 m (100 links) of corners and > 50% of trees were found within 8 m (40 links) except marsh prairie (VT 1). Density estimates based on distances to trees suggested that marsh prairie (VT 1) was very open while the other vegetation types were covered with open to moderately closed canopy (Table 4.1).

The overall species composition was similar between marsh prairie (VT 1) and timbered swamp with brush (VT 2), and the species with higher IVs were spruce, ash, and alder (Table 4.2). These two vegetation types were located closer to the mouth of the Coquille River (Fig. 4.3a). The timbered swamp with grass (VT 3) and timbered swamp with water classes (VT 4) were characterized by similar species composition, and high IV of willow, a species that is tolerant to saturated soils. The species composition of wooded bottomlands (VT 5) was distinct from those of other vegetation types, and the most important species were maple and myrtle, which are less tolerant of saturated soils, followed by willow. Topographic differences were

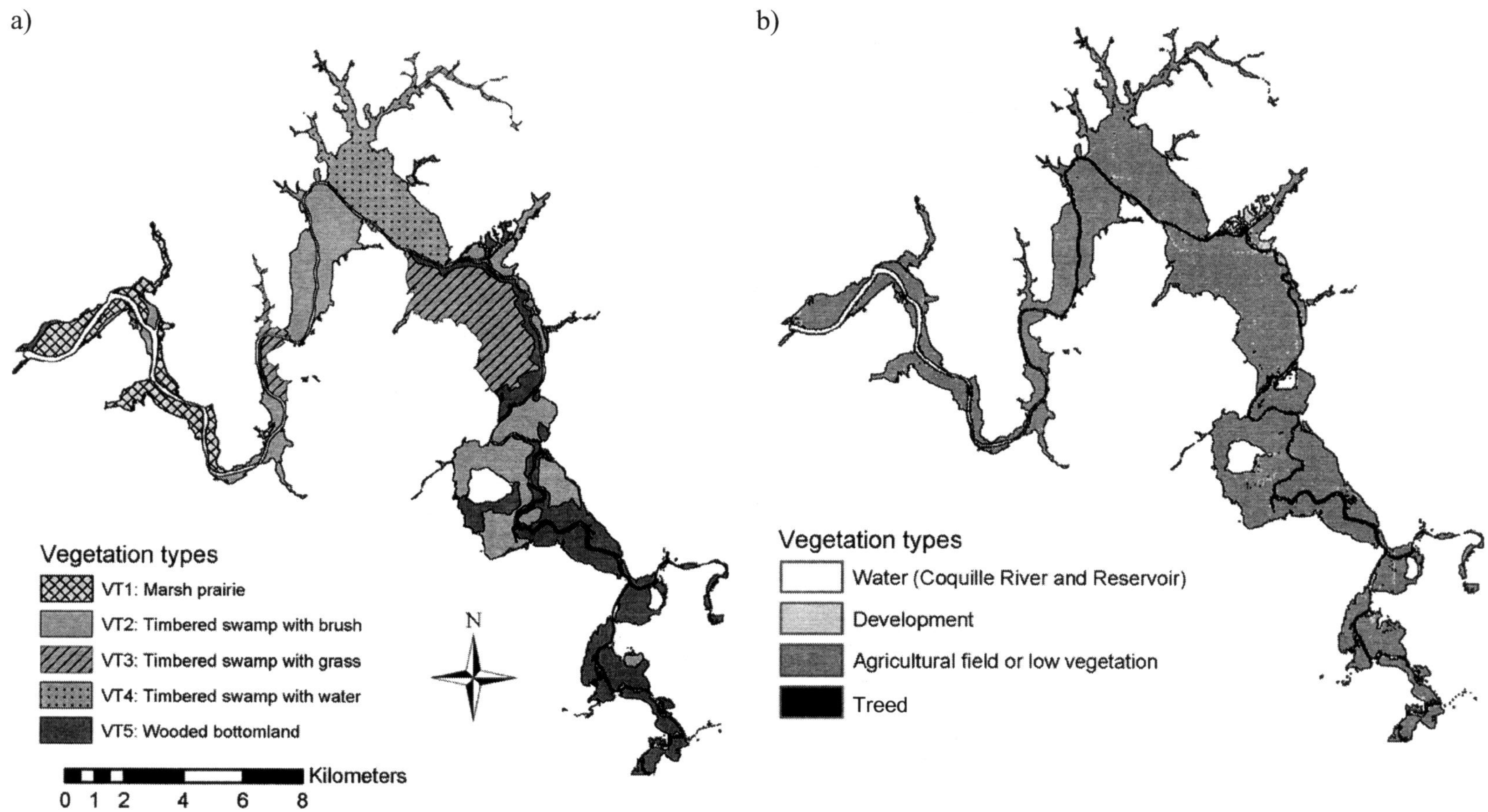

Figure 4.3: The historical (a) and current (b) vegetation maps of the Coquille Valley. The historical map was modified from Benner (1991). The descriptions of the vegetation types are in Table 4.1. VT = Vegetation type.

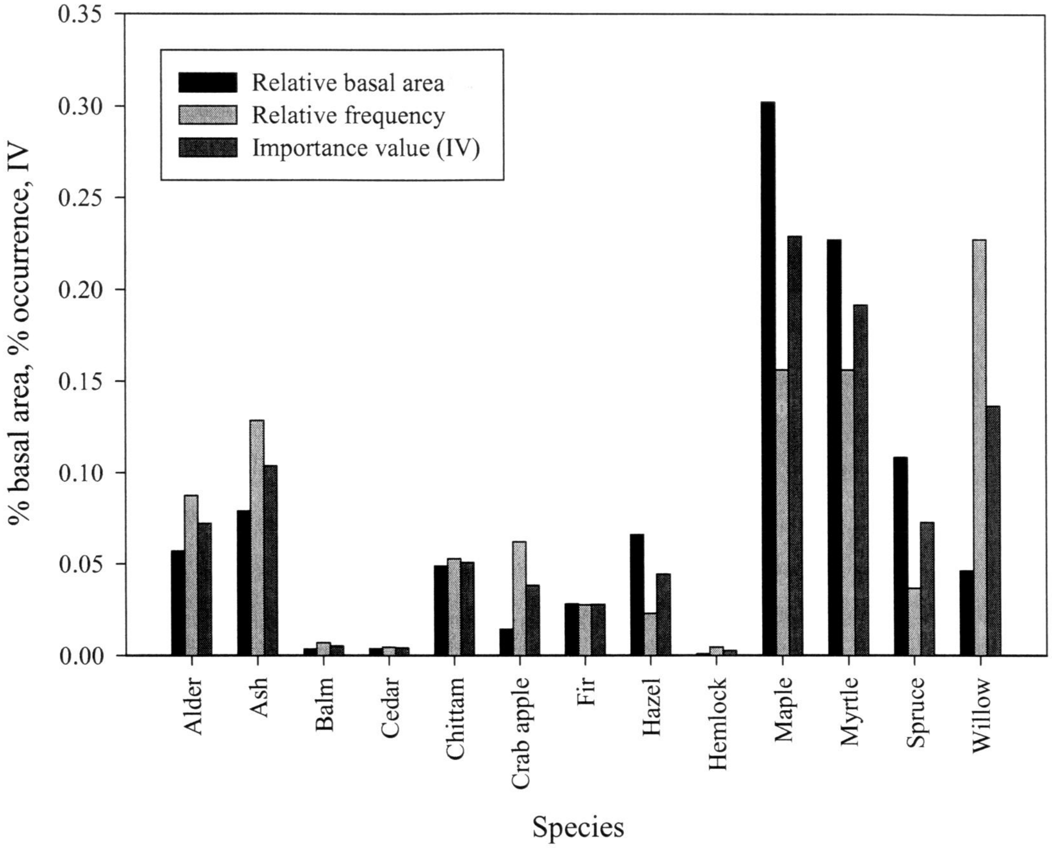

Figure 4.4: Relative frequency, relative basal area (m^2), and importance value (IV: average of relative frequency and relative basal area) of species recorded by the surveyors in the Coquille Valley. Several minor shrub species were not included (bearberry, elder, sagwood, tasslewood, and vine maple). In the Coquille Valley, the surveyors used “balm” for black cottonwood. See Table 4.2 for the scientific names.

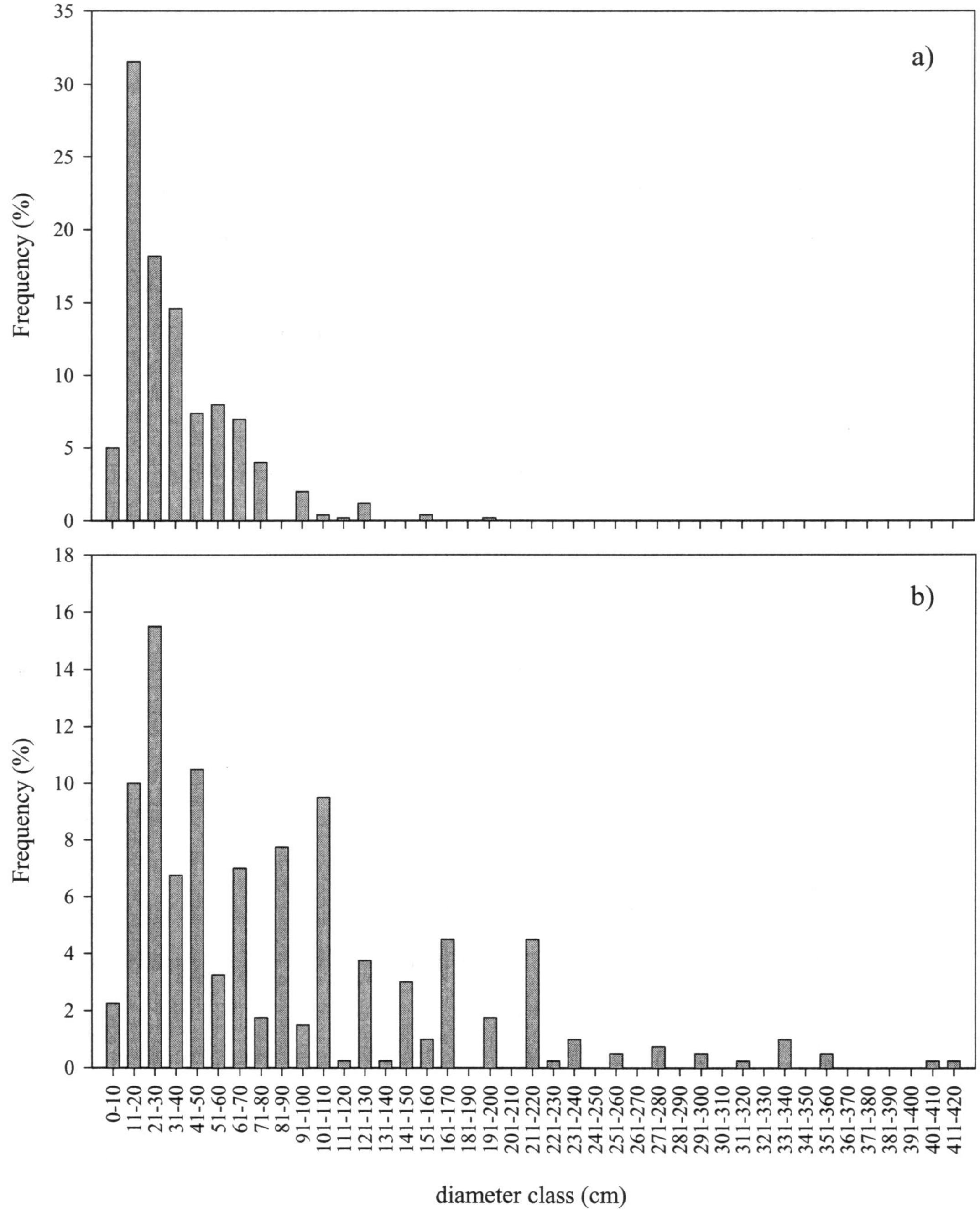

Figure 4.5: Relative frequency of trees by diameter class for a) the Coquille Valley and b) the Tillamook Valley. Note the different values on the y-axes.

minimal except for slightly higher elevations and steeper slopes of wooded bottomlands (VT5) (Table 4.1).

The species of uplands differed from the valley. Uplands were dominated by Douglas-fir and hemlock, followed by alder, cedar, maple, and spruce (Table 4.2). About 50% of the recorded trees were less than 30 cm in diameter, and another 40% were between 30 and 100 cm. The tree density of the uplands was 98.6 trees/ha. The distances to witness trees were more consistent in uplands, which indicates that tree cover was patchier in valley bottoms than in surrounding uplands.

Tillamook Valley

Sitka spruce and western hemlock dominated the Tillamook Valley until the turn of the century (Coulton et al. 1996). The surveyors noted the conditions of soils and timber as good quality ("1st rate") for most of the survey lines. Trees historically covered 69% of the valley bottomland at various densities, indicating that canopy cover ranged from open to moderately closed (Table 4.1; Fig. 4.6a). There were also extensive prairies. About 70% of the recorded trees were conifer species.

Spruce was the most common species (> 50% of the recorded trees; Fig. 4.7), and some of them were very large (> 2.5m in diameter, Fig. 4.5b). Hemlock was relatively abundant along upper streams and areas further from the rivers, and > 30% of hemlock was > 100 cm in diameter. Tree density was higher in upriver floodplain (VT 4) than the other vegetation types and was low in grassy tidal marsh (VT 1) and prairie lands (VT 6) (Table 4.1).

Spruce was the most important species in all the vegetation types. Grassy tidal marsh (VT 1) had fewer species than other types. Hemlock was relatively important for upriver floodplain (VT 4) and timbered valley lands (VT 5). Cedar was a relatively important species only for upriver floodplain (VT 4). Tree density was lower, and mean diameter was larger in main bottomland (VT 3) and timbered valley lands (VT 5). Although timbered valley lands (VT 5) and prairie lands (VT 6) were both on the old alluvial terraces (Bowlsby and Swanson 1964), hemlock was relatively more frequent in timbered valley lands (VT 5), while alder and willow were more

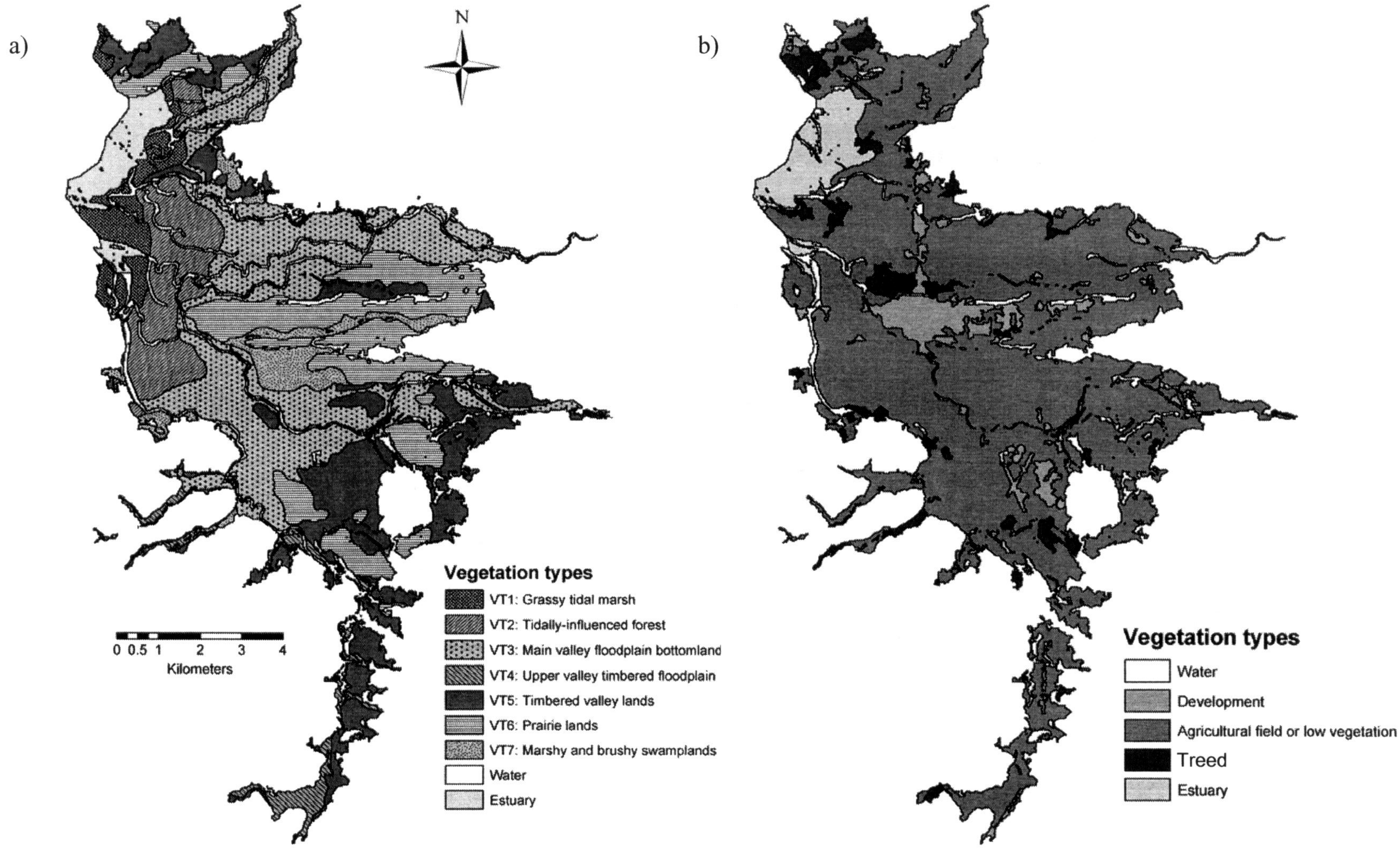

Figure 4.6: The historical (a) and current (b) vegetation maps of the Tillamook Valley. The historical map was modified from Coulton et al. (1996). The descriptions for the vegetation types are in Table 4.1. VT = Vegetation type.

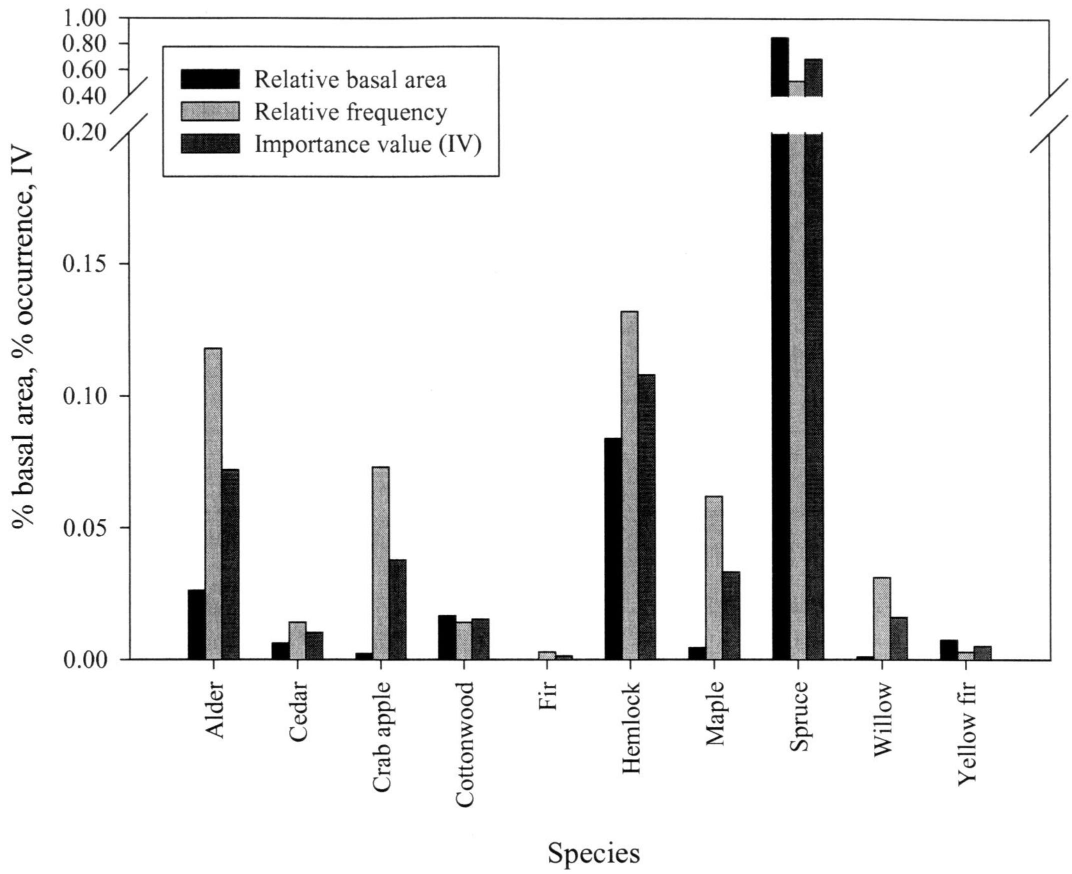

Figure 4.7: Relative frequency, relative basal area (m^2), and importance value (IV: average of relative frequency and relative basal area) of species recorded by the surveyors in the Tillamook Valley. Several minor shrub species were not included (bearberry, dogwood, and vine maple). See Table 4.2 for the scientific names.

frequent than hemlock in prairie lands (VT 6). Topography was similar among the first 3 and among the latter 3 vegetation types (Table 4.1).

The relative importance of spruce and hemlock was reversed in the uplands compared to the valley. Hemlock accounted for 55% of the recorded trees in the uplands and spruce occurred with a frequency of about 20%. About 25% of the recorded trees were less than 30 cm in diameter, and about 15% of the trees were >100 cm in diameter. The tree density of the uplands was 73.8 trees/ha. The wide range of distances to trees in the valley bottomlands indicated that tree distribution was patchier in the valley bottom than in surrounding uplands.

Current vegetation cover and change since settlement

Coquille Valley

The valley bottomland with tree cover was about 8% (Table 4.3). Trees were mostly broadleaf, and conifer trees were very rare. Most of the broadleaf tree vegetation occurred along the Coquille River as narrow riparian strips. About 90% of the valley bottom was agricultural fields or covered with low, sparse vegetation (Fig. 4.3b). About 97% of the valley without trees was in agricultural use. The developed areas included the town of Coquille and commercial sites. Since settlement, treed area has decreased from 90% to 8%, while non-treed has increased from 10% to 92% (Fig. 4.8).

Tillamook Valley

The valley bottomland with tree cover was about 5% (Table 4.3), most of which occurred in small patches. About 87% of the valley bottom was agricultural field or covered with low, sparse vegetation (Fig. 4.6b). About 92% of the valley without trees was in agricultural use. The developed areas included the city of Tillamook, the airport, bare ground, and roads. Since settlement, treed area has decreased from 69% to 5%, while non-treed has increased from 31% to 95% (Fig. 4.8).

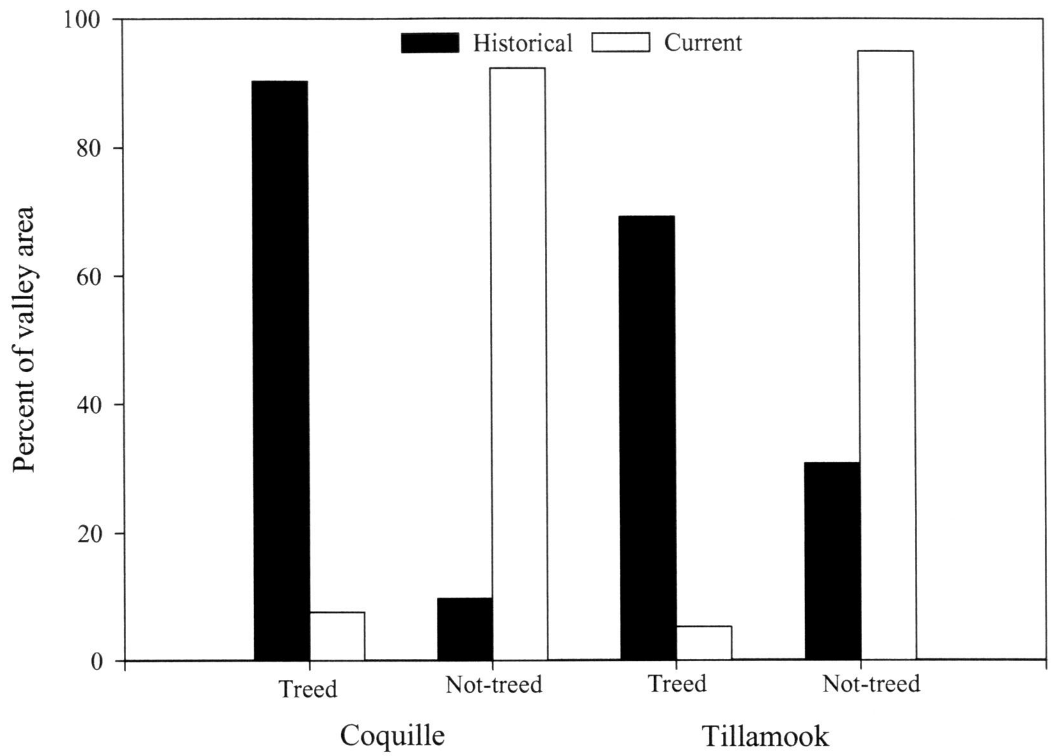

Figure 4.8: Change in percent valley area covered with trees or no trees in the Coquille and Tillamook Valleys. Estuary and water were excluded from the area calculation.

DISCUSSION

Changes in the valleys

The landscapes in the Coquille and Tillamook Valleys have dramatically changed since the early period of Euro-American settlement. The main trend was land conversion to agriculture for both valleys. On the current landscapes, trees are mostly confined to narrow strips along the rivers and streams or small park-like areas in the Coquille Valley. In contrast, small patches of conifer, broadleaf, and mixed forests are present in the Tillamook Valley. The bottomland ecosystems were historically maintained by natural disturbance and hydrologic regimes in addition to the influence of native people, but most of such important processes have been altered, if not lost, from the ecosystems of the two valleys. Hydrology of the valleys has been altered by flood control, drainage for agriculture, and changes in vegetation cover. In Coquille, straight ditches and remnant stream channels are visible on the aerial photos (Benner 1991). Levees and dikes have been constructed to maximize area of land for agriculture and to control flooding, which disconnects the river from the floodplains, disrupting processes occurring across the two systems. Sedell and Froggatt (1984) found similar changes in Willamette Valley, Oregon. Conversion to agriculture and associated alteration in hydrology are the main changes occurred in the valleys in the last 150 yrs.

Historical landscapes in the two valleys

Although the Coquille and Tillamook Valleys are similar in landform, broadleaf trees were historically dominant in the Coquille Valley, while conifers were dominant in the Tillamook Valley. Tree density was estimated to be quite low in most vegetation types in both valleys. The difference in species composition is probably a result of difference in soil drainage. Most of the soils found in Coquille are very poorly drained, and well-drained soils are limited. Species that prefer well-drained soils were found only in wooded bottomlands (VT 5) in the Coquille Valley, and species tolerant of poor drainage (ash, alder, and willow) were more common in wetter vegetation types. On the other hand, the soils in the Tillamook Valley are mostly well-drained, which allowed establishment of many spruce trees and high growth in the fertile soil. Many spruce trees attained large diameter (e.g., 2-4 m) probably because of the species' long longevity.

Another explanation for the difference may be attributed to the difference in fire regime. The Coquille Valley might have burned more frequently than the Tillamook Valley. The Coquille Valley is located in southwestern Coast Range, where fire regime is more frequent than the northwestern parts. Many hardwood species can resprout from stumps after fire. Sitka spruce and western hemlock are not fire tolerant, so that long fire return intervals are needed to attain large size. Time since the most recent fire might have been very long in Tillamook so that short-lived hardwood species might have been declining.

It is unlikely that the difference in historical tree species composition between the two valleys could have been caused by selective logging by early settlers. In Tillamook, the first settler arrived in 1851, and farming was important in early settlement (Coulton et al. 1996). Trees were merely obstacles to the development of farmland in that period. The first sawmill did not start operations until 1863 in Tillamook (Coulton et al. 1996) and 1865 in Coquille (Benner 1991). Clearing for agriculture is a possible reason, but if early settlers indiscriminately felled trees to cultivate lands, then relative importance of conifer and hardwood should be more or less preserved. Thus logging was probably not yet the factor affecting the vegetation because most of the survey was done by that time. Therefore, it is reasonable to assume that anthropogenic effects were minimal to cause the compositional difference between the two valleys.

Implications of landscape change in the valleys for regional biodiversity

Valley bottomlands historically contributed to regional biodiversity in the Oregon Coast Range. Valley bottomlands are ecologically unique and diverse systems that are quite different from adjacent upland forests in terms of vegetation and disturbance regimes. These ecosystems are especially important in the region because valley bottoms are rare in the region due to steep, highly dissected topography. Coastal valleys may form estuaries as in the Coquille and Tillamook Valleys, providing unique habitats to various species of both aquatic and terrestrial (Johnson and O'Neil 2000). Burnett (2001) found that coho salmon (*Oncorhynchus tshawytscha*) and chinook salmon (*O. kisutch*) use rivers and streams in valley bottomlands than constrained upper streams in western Oregon. The river continuum concept predicts that lower floodplain ecosystems contain different community assemblages and habitats from upper locations of a stream network and contribute to landscape heterogeneity within a watershed (Vannote et al. 1980). Floodplains are structurally, compositionally, and functionally different from narrow riparian areas along upper streams (Swanson et al. 1998).

Valley bottomlands are also important for habitat and ecosystem diversity at the local landscape scale. The Coquille and Tillamook Valleys contained multiple and distinct ecological units in terms of hydrology, cover type, tree species composition, and forest structure before settlement. The flood-pulse concept describes the dynamics of floodplain systems and the importance of floods on diversity and productivity (Bayley 1995). Animals, fish, and plants in such systems have adapted to the dynamics, forming floodplain habitat complex (Malanson 1993). The GLO surveyors noted many beaver dams and presence of elk in both valleys (Benner 1991, Kauffman et al. 2000). A large number of reptile, amphibian, bird, and mammal species that occur in the Pacific Northwest depend on riparian and wetland habitats (Kauffman et al. 2000). Riparian and wetland habitats contain a greater number of species than uplands and provide critical breeding and foraging habitats for many bird species in the region (Kauffman et al. 2000). Therefore, implications of the changes that occurred in the valleys to regional biodiversity extend from the local to the regional scales. At the local scale, the dynamic mosaic of different types of wetlands created various microhabitat conditions in bottomlands, while the unique vegetation and hydrology increased ecosystem diversity at the regional scale.

Uncertainty related to the data and density calculation

Estimated tree densities were used only as ancillary information in this study to aid categorizing vegetation into treed or not-treed because tree density calculated with small sample size using the PCQ method was not as reliable. It was also not clear how to handle corners with no trees recorded. The density estimation is also highly sensitive to large distances between points and trees. In addition, it was unclear whether three different types of corners (section corner, quarter-section corner, and meander corner) represented the same information about the characteristics of vegetation in the valleys. The surveyors were required in this part of the country to record 4 trees at section corners, one in each quadrant, and two trees at quarter section corners, one on each side, if trees were available. Meander points were established when surveyors crossed bodies of water, and they recorded two trees near the corners if available. The section and quarter-section corners can be considered as systematic, regular sampling, but meander corners could not. Meander corners were likely to fall in riparian areas and would not be representative of vegetation over the valleys (Nelson et al. 1994). I did not use meander corners because of the bias. I calculated tree densities using different sets of distances from corners that had no trees recorded and presented the ranges in Table 4.1.

Additional uncertainty stemmed from the fact that the surveyors did not always record the nearest trees to the corners because of the instructions and for practical reasons (Bourdo 1956). The minimum tree diameter to be recorded was 5 inches although smaller trees were sometimes recorded. Because the purpose of recording trees was for later identification of particular parcels of land for settlement, the surveyors tended to select trees that would remain alive and standing for a reasonable period and sometimes selected rare species to make the corners easier to find (Schafale and Harcombe 1983). They were also instructed to "blaze" trees and inscribe townships and other information on the trees, so that they might have avoided trees of small diameter or thick bark (Manies et al. 2001, Schulte and Mladenoff 2001). I assumed that they recorded the nearest trees at the corners throughout the calculation. Because the estimates were highly sensitive to large distances, use of the estimates should be made with caution if the sample size is small.

Uncertainty associated with the GLO data is sometimes impossible to quantify. The purpose of the survey was not ecological but legal (Manies et al. 2001). The potential errors and biases in the GLO survey have been well studied (Bourdo 1956, Galatowitsch 1990, Manies and Mladenoff 2000, Manies et al. 2001, Schulte and Mladenoff 2001), and it was suggested that drawing broad inferences in vegetation distribution over a large landscape from the GLO data be reasonable (Schulte and Mladenoff 2001).

Limitations and constraints in this study

Settlement has already begun in the valleys around the time of the survey, and therefore, it was not strictly presettlement. Settlement was, however, in its early period, and the data used in this study is the only well-documented source of historical information about vegetation in the case study areas. This historical information is a reasonable source to establish a reference condition to assess the current conditions of the two valleys.

Benner (1991) and Coulton et al. (1996) used hydrology, degrees of soil saturation, and vegetation cover to delineate and classify vegetation types on the historical vegetation maps. The current vegetation maps were developed strictly based on cover types using remotely sensed imagery. The difference between the historical and current maps can arise from the use of two different techniques. The change in vegetation pattern was large and vegetation classes were general so that the conclusion of loss of tree cover is reasonable.

The valley bottomlands were quite heterogeneous, ranging from prairies to moderately closed forests with patchy tree distribution, and therefore, lumping such cover types into two categories underestimated the diversity. Given the available information and dramatic changes occurred in the two valleys, the comparison using the two cover types was useful to gain quantitative information about the change occurred since settlement.

Direct extrapolation of the findings from the two coastal valleys to other valleys in the Coast Range would be speculative. The other valleys identified in the region are smaller and are more inland. This study suggested that soil drainage can be an indicator of historical abundance of hardwoods in valley bottoms. Conversion to agriculture and development is suggested to be widespread in valleys in the region since agricultural fields and highways are ubiquitous in valleys today.

CONCLUSIONS

Historical vegetation records are a valuable source of information to examine landscape change that has occurred after the arrival of Euro-American settlers (Schulte and Mladenoff 2001, Manies and Mladenoff 2000). The baseline condition provided by historical data can be used to assess current conditions and quantify the change for better understanding of habitat loss. Valley bottomlands are rare ecosystems in the Oregon Coast Range and are predominantly owned by private non-industrial landowners. The landscapes of the Coquille and Tillamook Valleys have changed dramatically in the last 150 years since the early period of Euro-American settlement. The majority of treed areas are currently without trees, mostly agricultural field. Historically, the Coquille valley was dominated by hardwood species, while conifers were dominant in the Tillamook Valley. Difference in soil drainage between the two valleys provides clues to potential vegetation types of other valley bottoms in the region. Species composition and structure were different from those of uplands, and the potential effects of loss of such unique habitats on the regional biodiversity are not known. Valley bottomlands have been intensively utilized for human use, and therefore, the ecosystems are one of the most threatened habitat types in the Oregon Coast Range.

CHAPTER 5: GENERAL CONCLUSIONS

History is an invaluable source of information to understand and evaluate management influences on contemporary ecosystems and landscapes. The current landscape of the Oregon Coast Range has changed since Euro-American settlement about 150 yrs ago. The patch structure of the current landscape is outside the historical range of variability (HRV) and characterized by abundant young forests and lack of old growth forests (Chapter 2). The historical landscape was highly heterogeneous with large patches of complex shapes and various age classes and a wide range of patch sizes. Old growth forests historically occurred more than 50% of the landscape on average, while old growth forests currently occupy only 2%. On the other hand, young forest (21-80 yrs) is > 3 times as more abundant currently as it was historically. Forests of < 20 yrs old were historically rare, but on the current landscape, this age range occupies greater than one-forth of the landscape. Fires left numerous unburned remnant island patches within burned areas, which may have acted as dispersal nuclei of organisms into disturbed areas (e.g. Sillett et al. 2000, Wimberly and Spies 2001).

The dead wood abundance on the current landscape is outside HRV (Chapter 3). The distribution of live and dead wood biomass on the current landscape is highly skewed to lower levels, and close to 60% of the current landscape is characterized by very low amount of dead wood. Historically, very low dead wood occurred only in a fraction of the landscape. High amounts of dead wood was characteristic of very young stages (< 20 yrs), and dead wood amounts were lower in young and mature forests, as the general U-shaped curve describes the dynamics. Decades of intensive forest management for timber production has led to stand conditions with very low dead wood. The simulation suggested that high- and moderate-severity fires were equally important for the biomass dynamics in the Coast Range. During the 1000-yr simulation, most of the stands experienced multiple fires of different severity at different return intervals. This temporal pattern is contrasting to intensive forest management in this region, where forests are harvested at regular rotations.

The current landscapes of the Coquille and Tillamook Valleys are quite different from historical conditions that the historical survey records described (Chapter 4). In these bottomlands, hardwood and conifer forests and woodlands have been converted to agricultural fields and development. Interestingly, the Coquille Valley was dominated by hardwood species, while the Tillamook was by conifer species, especially large spruce trees. Soil drainage is poor in the Coquille, and the difference in soil conditions may be responsible for the vegetation difference. Valley bottom ecosystems are rare in the Oregon Coast Range, occupying only 2.8% of the region. Soil drainage may be the clue to historical vegetation types in other valley bottomlands.

The forest policies currently implemented in the region did not return the landscape within HRV in 100 yrs in simulation (Chapter 2). The vast young forests and lack of old growth forests on the current landscape hindered the landscape to reach HRV for centuries in the simulation after the wildfire regime was reinstated. In addition, boundaries of different ownership types may constrain the patch patterns in the region because of highly contrasting management objectives among ownership types. Past forest management left legacies on the landscape that would take for a long time to be erased by disturbance and forest development.

All three chapters concluded that the current landscape conditions of the Oregon Coast Range are quite different from the historical ones. Multiple ownership groups partition the current landscape with strongly contrasting management objectives. Commodity production is an important objective on the State and the private land, and the Federal ownerships put a priority on protection of late-successional forests and riparian reserves for native species, especially northern spotted owl. Although the Northwest Forest Plan, with which the federal lands comply, had the regional perspective in mind, it is uncertain that what landscape patterns the different management regimes collectively will create. This study indicated that in 100 yrs patch characteristics on the landscape will reflect the boundaries of ownership groups. Mature and old growth forests would mostly exist in the federal and state lands, and younger forests on the private lands would comprise the matrix of the older forest patches. Given the demand for timber, management objectives, and past and future human effects, using HRV as a management goal may not be sustainable socially and economically. For example, achieving HRV in dead wood amounts is not feasible for timber production goals. However, HRV approaches can provide reference conditions, and departure from the reference can serve as an indicator of landscape conditions.

This thesis explored several methods for using historical information to evaluate landscapes. Chapter 2 and 3 used spatially-explicit simulation modeling to establish HRV. The advantage of using simulation models is that models can simulate many possible landscapes under a historical fire regime so that the variability of landscape characteristics under the regime can be quantified. I established 90% confidence ranges of historical variability in landscape structure and biomass distribution that would be possible under the historical fire regime in the region. Remotely sensed images and historical information usually do not have this temporal depth to characterize variability (Swanson et al. 1994, Swetnam et al. 1999). The HRV presented in Chapter 2 reflected dynamic landscapes within which native species have evolved capacities to cope with environmental fluctuations. Another advantage of simulation modeling of disturbances is the versatility in linking to other processes for which disturbance is an important driver. In Chapter 3, spatial and temporal dynamics of disturbance was combined with live and dead wood biomass dynamics by simple relationships. Nutrient release by fire, carbon storage, and wildlife habitat potential etc. can be incorporated in simulation modeling to examine historical dynamics of the patterns over time and space.

Chapter 4 used historical information (the GLO survey) and remotely sensed imageries (aerial photos and satellite images) and utilized a GIS for spatial analysis. This method compared conditions at two points in time. The historical data from the previous study by Benner (1991) and Coulton et al. (1996) and the GLO surveyors' journals provided detailed information about the vegetation of the valley bottoms around the time of settlement in the Coquille and Tillamook Valleys. Potential bias in survey data and lack of long-term record are the main limitations (Bourdo 1956, Swanson et al. 1994, Morgan et al. 1994, Swetnam et al. 1999). In this study, difficulties arose for comparisons between two vegetation maps developed using different data sources (survey data vs. classified satellite imageries) because the spatial scale at which vegetation was described was likely to be different. I was able to make a comparison between the current and historical landscapes, but this method did not allow me to infer how common the condition similar to the current one would be historically.

HRV can be defined for any dynamic characteristics at any scales. However, processes are scale-dependent so that they have to be examined at temporal and spatial scales that are appropriate to the processes that are driving the pattern of interests. This study examined only the effects of wildfire on landscape characteristics, and the results need to be interpreted at the specific scale. In reality, wind throw, landslides, and insect and disease outbreaks are important disturbances in the Oregon Coast forests and create heterogeneity at finer scales.

Scale is an important concept in ecology (Peterson and Parker 1998). The landscape dynamics simulated by the model in this study is specific to the scale—grain and extent—at which the process was modeled. The focal level was the region of the Oregon Coast Range, and birth, death (same as birth in this case), and growth of stands at the pixel scale (i.e. 9 ha) give rise to the dynamic pattern seen at the regional scale. The hierarchy theory suggests that the level below the focal level explains the mechanisms for the phenomena observed at the focal level, and the level above the focal level puts constraints on the behaviors of the system at the focal level (O'Neill et al. 1986, King 1997, Turner et al. 2001). The quantification of landscape structure using the model at the regional scale is meaningful but that at the stand scale is not. In other words, properties at specific locations cannot be used to draw conclusions, and conclusions can be drawn only to the focal scale. Therefore, the HRV estimated in this thesis can serve as a quantitative reference at the regional scale but cannot provide specific recommendations that can be implemented at the stand scale.

The HRV of landscapes created by natural disturbance has been proposed as a guide for evaluating managed landscapes, and I agree that HRV is a useful reference to evaluate landscape conditions. The HRV approach is a more objective way to evaluate landscape conditions than setting arbitrary standards, but we still need to decide how to characterize and quantify landscape patterns and at what scales to do so. These criteria can all affect the conclusions. For example, future landscapes under the current policy scenario might have been within HRV if the patch shape and arrangement metrics had not been included. Scientific knowledge can help to choose key elements and processes and their appropriate scales of examination. Ultimately, interests and concerns in society are important to decide what should be protected and maintained in landscapes.

Key findings:

1. The current landscape of the Oregon Coast Range is outside the HRV in terms of both landscape structure and dead wood abundance.
2. The historical landscape was dynamic, composed of patches of various sizes and age classes ranging from 0 to > 800 yrs, including numerous small unburned island forests. Old growth forests (> 200 yrs) occupied about 50% of the landscape on average, while younger age classes were also important.
3. Neither the current policy scenario nor the wildfire scenario would return the landscape condition within the HRV in 100 yrs because of lack of old growth forests and overabundance of young forests. Under the wildfire scenario, the landscape took 800 yrs to return to the HRV.
4. Historically, the majority of the stands contained > 501 Mg/ha of live wood biomass and 50-200 Mg/ha of dead wood biomass. Stands with < 50 Mg/ha of dead wood biomass, about the amount that can be found in plantations, was very rare historically, but it is almost 60% of the current landscape.
5. There was a wide variation in dead wood biomass within age classes because of variation in disturbance history.
6. Historically, high- and moderate-severity fires were equally frequent, and stands experienced multiple fires of different severity at different developmental stages. A variety of 1000-yr disturbance history was observed in the simulation.

7. Valley bottomlands are relatively rare (i.e. 2.8%) in the Oregon Coast Range and have experienced disproportional impact of human activities since Euro-American settlement relative to uplands. Private landowners predominantly own the valley bottomlands.
8. Coquille and Tillamook Valleys historically contained more trees at various densities, and species composition and forest structure were distinct from surrounding uplands. Most of treed areas have been converted to agriculture. The historical records and aerial photos indicated alteration of hydrology in the valleys.
9. Historical vegetation in the Coquille and Tillamook Valleys was not similar. Hardwoods were common in Coquille and conifers in Tillamook. Soil drainage may be the reason for the difference.

BIBLIOGRAPHY

Acker, S. A., C. B. Halpern, M. E. Harmon, and C. T. Dyrness. 2002. Trends in bole biomass accumulation, net primary production and tree mortality in Pseudotsuga menziesii forests of contrasting age. Tree Physiology 22:213-217.

Acker, S. A., P. A. Harcombe, M. E. Harmon, and S. E. Greene. 2000. Biomass accumulation over the first 150 years in coastal Oregon Picea-Tsuga forest. Journal of Vegetation Science 11:725-738.

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K. and M. H. Huff. 1987. Fuel succession in a western hemlock/Douglas-fir forest. Canadian Journal of Forest Research 17:697-704.

Andison, D. W. and P. L. Marshall. 1999. Simulating the impact of landscape-level biodiversity guidelines: a case study. Forestry Chronicle 75:655-665.

Angelstam, P. 1997. Landscape analysis as a tool for the scientific management of biodiversity. Ecological Bulletins 46:140-170.

Aplet, G. H. and W. S. Keeton. 1999. Application of historical range of variability concepts to biodiversity conservation. Pages 71-86 *in* R. K. BayDack, H. Campa, III., and J. B. Haufler, editors. Practical approaches to the conservation of biological diversity. Island Press, Washington, D.C., USA.

Arnst, A. 1983. Tillamook revisited. American Forests 89:26-29, 44-52.

Attiwill, P. M. 1994. The disturbance of forest ecosystems: the ecological basis for conservative management. Forest Ecology and Management 63:247-300.

Baker, W. L. 1989. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area, Minnesota. Ecology 70:23-35.

Baker, W. L. 1992a. Effects of settlement and fire suppression on landscape structure. Ecology 73:1879-1887.

Baker, W. L. 1992b. The landscape ecology of large disturbances in the design and management of nature reserves. Landscape Ecology 7:181-194.

Baker, W. L. 1993. Spatially heterogeneous multi-scale response of landscapes to fire suppression. OIKOS 66:66-71.

Baker, W. L. 1995. Longterm response of disturbance landscapes to human intervention and global change. Landscape Ecology 10:143-159.

Bayley, P. B. 1995. Understanding large river-floodplain ecosystems. BioScience 45:153-158.

Benner, P. A. 1991. Historical reconstruction of the Coquille River and surrounding landscape. *In* Near Coastal Waters National Pilot Project: The Coquille River, Oregon. Action plan for Oregon Coastal Watersheds, Estuary and Ocean Waters, 1988-91. Oregon Department of Environmental Quality for the US Environmental Protection Agency.

Bergeron, Y., B. Harvey, A. Leduc, and S. Gauthier. 1999. Forest management guidelines based on natural disturbance dynamics: stand- and forest-level considerations. Forestry Chronicle 75:49-54.

Bergeron, Y., A. Leduc, B. D. Harvey, and S. Gauthier. 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forests. Silva Fennica 36:81-95.

Bettinger, P. and M. Lennette. 2002. LAndscape Management Policy Simulator (LAMPS), version 1.1. Oregon State University, Corvallis, Oregon, USA.

Bissionette, J. A. 1997. Wildlife and landscape ecology. Springer-Verlag, New York, New York, USA.

Black, A. E., P.Morgan, and P. F. Hessburg. 2003. Social and biophysical correlates of change in forest landscapes of the interior Columbia Basin, USA. Ecological Applications 13:51-67.

Bormann, F. H. and G. E. Likens. 1979. Pattern and process in a forested ecosystem. Springer-Verlag, New York, New York, USA.

Bourdo, E. A. Jr. 1956. A review of the General Land Office Survey and of its use in quantitative studies of former forests. Ecology 37:754-768.

Bordelon, M. A., D. C. McAllister, and R. Holloway. 2000. Sustainable Foresty Oregon Style. Journal of Forestry 1:26-34.

Bowlsby, C. C. and R. C. Swanson. 1964. Soil survey of Tillamook Area, Oregon. Soil Conservation Service. United States Department of Agriculture, Washington, D. C., USA.

Boyd, R. 1999. Indians, fire, and the land in the Pacific Northwest. Oregon State University Press, Corvallis, Oregon, USA.

Bull, E. L., C. G. Parks, and T. R. Torgersen. 1997. Trees and logs important to wildlife in the interior Columbia River Basin. USDA Forest Service, Pacific Northwest Research Station. General Technical Report. PNW-GTR-391.

Burgess, R. L. and D. M. Sharpe. 1981. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, New York, USA.

Burnett, K. M. 2002. Relationships among juvenile anadromous salmonoids, their freshwater habitat, and landscape characteristics over multiple years and spatial scales in the Elk River, Oregon. PhD Dissertation. Oregon State University, Corvallis, Oregon, USA.

Butts, S. R. and W. C. McComb. 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. Journal of Wildlife Management 64:95-104.

Calhoun, A. 1999. Forested wetlands, Pages 300-331 *in* M. L. Hunter Jr., editor. Maintaining biodiversity in forest ecosystems. Cambridge University Press, Cambridge, UK.

Cattelino, P. J., I. R. Noble, R. O. Slatyer, and S. R. Kessell. 1979. Predicting the multiple pathways of plant succession. Environmental Management 3:41-50.

Cissel, J. H., F. J. Swanson, W. A. McKee, and A. L. Burditt. 1994. Using the past to plan the future in the Pacific Northwest. Journal of Forestry 92:30, 31, 46.

Cissel, J. H., F. J. Swanson, G. E. Grant, D. H. Olson, S. V. Gregory, S. L. Garman, L. R. Ashkenas, M. G. Hunter, J. A. Kertis, J. H. Mayo, M. D. McSwain, S. G. Swetland, K. A. Swindle, D. O. Wallin. 1998. A landscape plan based on historical fire regimes for a managed forest ecosystem: the Augusta Creek Study. USDA Forest Service, Pacific Northwest Research Station. General Technical Report. PNW-GTR-422.

Cissel. J. H., F. J. Swanson, and P. J. Weisberg. 1999. Landscape management using historical fire regimes: Blue River, Oregon. Ecological Applications 9:1217-1231.

Clark, D. F., D. D. Kneeshaw, P. J. Burton, and J. A. Antos. 1998. Coarse woody debris in sub-boreal spruce forests of west-central British Columbia. Canadian Journal of Forest Research 28:284-290.

Cline, S. P., A. B. Berg, and H. M. Wight. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management 44:773-786.

Cohen, W. B., T. A. Spies, R. J. Alig, D. R. Oetter, T. K. Maiersperger, and M. Fiorella. 2002. Characterizing 23 years (1972-95) of stand replacement disturbance in western Oregon Forests with Landsat Imagery. Ecosystems 5:122-137.

Committee of Scientists. 1999. Sustaining the People's Lands: Recommendations for Stewardship of the National Forests and Grasslands into the Next Century. United States Department of Agriculture, Washington, D. C., USA.

Cottam, G. and J. T. Curtis. 1956. The use of distance measures in phytosociological sampling. Ecology 37:451-460.

Coulton, K. G. and P. B. Williams, with P. A, Benner. 1996. An environmental history of the Tillamook Bay estuary and watershed. The Tillamook Bay National Estuary Project, Garibaldi, Oregon, USA.

Crow, T. R., H. E.George, and D. J. Mladenoff. 1999. Ownership and ecosystem as sources of spatial heterogeneity in a forested landscape, Wisconsin, USA. Landscape Ecology 14:449-463.

Curtis, J. T. 1959. The Vegetation of Wisconsin. University of Wisconsin Press, Madison, Wisconsin, USA.

Cushman, S. A. and D. O. Wallin. 2000. Rates and patterns of landscape change in the Central Sikhote-alin Mountains, Russian Far East. Landscape Ecology 15:643-659.

Cushman, S. A. and K. McGarigal. 2003. Landscape-level patterns of avian diversity in the Oregon Coast Range. Ecological Monograph 73:259-281.

Davis, L. S., K. N. Johnson, P. S. Bettinger, and T. E. Howard. 2001. Forest management: to sustain ecological, economic, and social values. Fourth edition. McGraw Hill Companies, New York, New York, USA.

Delcourt, H. R. and P. A. Delcourt. 1988. Quarternary landscape ecology: relevant scales in space and time. Landscape Ecology 2:23-44.

DeLong, S. C. and W. B. Kessler. 2000. Ecological characteristics of mature forest remnants left by wildfire. Forest Ecology and Management 131:93-106.

Delong, S. C. and D. Tanner. 1996. Managing the pattern of forest harvest: lessons from wildfire. Biodiversity and Conservation 5:1191-1205.

Eberhart, K. E. and P. M. Woodard. 1987. Distribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research 17:1207-1212.

ESRI (Environmental Systems Research Institute, Inc.). 1995. ARC/Info. Redlands, California, USA.

Engstrom, R. T., S.Gilbert, M. L. Hunter Jr., D. Merriwether, G. J. Nowacki, and P. Spencer. 1999. Practical applications of disturbance ecology to natural resource management. Pages 313-330 *in* R. C., Szaro, N. C. Johnson, W. T. Sexton, and A. J. Malk, editors. Ecological Stewardship: a common reference for ecosystem management. Elsevier Science, Oxford, UK.

Fahnestock, G. R. and J. K. Agee. 1983. Biomass consumption and smoke production by prehistoric and modern forest fires in western Washington. Journal of Forestry 81:653-657.

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management, an ecological, economic, and social assessment, Report of the Forest Ecosystem Management Team (FEMAT), 1993-793-071. Government Printing Office, Washington, D.C., USA.

Fralish, J. S., F. B. Crooks, and F. M. Harty. 1991. Comparison of presettlement, second-growth and old-growth forest on six site types in the Illinois Shawnee Hills. American Midland Naturalist 125:294-309.

Franklin, J. F. and D. S. DeBell. 1988. Thirty-six years of tree population change in an old-growth *Pseudotsuga-Tsuga* forest. Canadian Journal of Forest Research 18:633-639.

Franklin, J. F. and C. T. Dyrness. 1988. Natural vegetation of Oregon and Washington Oregon State University Press, Corvallis, Oregon, USA.

Franklin, J. F. and M. A. Hemstrom. 1981. Aspects of succession in coniferous forests of Pacific Northwest, Pages 212-229 *in* D. C. West, H. H. Shugart, D. B. Botkin, editors. Forest Succession: Concepts and Application. Springer-Verlag, New York, New York, USA.

Forman, R. T. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge, New York, USA.

Forman, R. T. T. and M. Godron. 1986. Landscape Ecology. Wiley, New York, New York, USA.

Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw , D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. BioScience 53:77-88.

Franklin, J. F. 1993. Preserving biodiversity: species, ecosystems, or landscapes? Ecological Applications 3:202-205.

Franklin, J. F. and R. T. T. Forman. 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecology 1:5-18.

Franklin, J. F., T. A. Spies, R. Van Pelt, A. B. Carey, D. A. Thornburgh, D. R. Berg, D. B. Lindenmayer, M. E. Harmon, W. S. Keeton, D. C. Shaw, K. Bible, J. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir as an example. Forest Ecology and Management 155:399-423.

Frelich, L. E. 2002. Forest dynamics and disturbance regimes: studies from temperate evergreen-deciduous forests. Cambridge University Press, Cambridge, UK.

Frelich, L. E. and C. G. Lorimer. 1991. Natural disturbance regimes in hemlock-hardwood forests of the Upper Great Lakes Region. Ecological Monographs 61:145-164.

Galatowitsch, S. M. 1990. Using the original land survey notes to reconstruct presettlement landscapes in the American west. Great Basin Naturalist 50:181-191.

Garman, S. L., F. J. Swanson, and T. A. Spies. 1999. Past, present, and future landscape patterns in the Douglas-fir region of the Pacific Northwest. Pages 61-86 *in* Rochelle, J.A., L. A. Lehmann, and J. Wisniewski, editors. Forest Fragmentation. Koninklijke Brill NV, Leiden, Netherlands.

Godron, M. and R. T. T. Forman. 1983. Landscape modification and changing ecological characteristics. Pages 12-28 *in* H. A. Mooney and M. Godron, editors. Disturbance and ecosystems: components of response. Springer-Verlag, New York, New York, USA.

Goslin, M. N. 1997. Development of two coniferous stands impacted by multiple, partial fires in the Oregon Cascades: establishment history and the spatial patterns of colonizing tree species relative to old-growth remnant trees. M.S. Thesis. Oregon State University, Corvallis, Oregon, USA.

Graham, R. L. L. 1982. Biomass dynamics of dead Douglas-fir and western hemlock boles in mid-elevation forests of the cascade range. PhD Dissertation. Oregon State University, Corvallis, Oregon, USA.

Greene, S. E., P. A. Harcombe, M. E. Harmon, and G. Spycher. 1992. Patterns of growth, mortality, and biomass change in a coastal *Picea sitchensis - Tsuga heterophylla* forest. Journal of Vegetation Science 3:697-706.

Grier, C. C. and R. S. Logan. 1977. Old-growth *Pseudotsuga menzisii* communities of a western Oregon watershed: biomass distribution and production budgets. Ecological Monographs 47:373-400.

Gustafson, E. J. 1998. Quantifying landscape spatial pattern: what is the state of the art? Ecosystems 1:143-156.

Haagen, J. T. 1990. Soil survey of Coos County, Oregon. Soil Conservation Service in cooperation with Oregon Agricultural Experiment Station. United States Department of Agriculture, Washington, D.C., USA.

Hall, R. L. 1995. People of the Coquille Estuary: native use of resources on the Oregon coast: an investigation of cultural and environmental change in the Bandon area employing archaeology, ethnology, human biology and geology. Words & Pictures Unlimited, Corvallis, Oregon, USA.

Hansen, A. J., T. A. Spies, F. J. Swanson, and J. L. Ohmann. 1991. Conserving biodiversity in managed forests. BioScience 41:382-392.

Harcombe, P. A., M. E. Harmon, and S. E. Greene. 1990. Changes in biomass and production over 53 years in a coastral *Picea sitchensis - Tsuga heterophylla* forest approaching maturity. Canadian Journal of Forest Research 20:1602-1610.

Hargis, C. D., J. A. Bissonette, and J. L. Dacid. 1998. The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landscape Ecology 13:167-186.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, Sedell. J. R., G. W. Lienkaemper, K. Jr. Cromack, and K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133-302.

Harmon, M. E. and J. F. Franklin. 1989. Tree seedling on logs in *Picea-Tsuga* forests of Oregon and Washington. Ecology 70:48-59.

Harvey, B. D., A. Leduc, S. Gauthier, and Y. Bergeron. 2002. Stand-landscape integration in natural disturbance-based management of the southern boreal forest. Forest Ecology and Management 155:369-385.

Hély, C., Y. Bergeron, and M. D. Flannigan. 2000. Coarse woody debris in the southeastern Canadian boreal forest: composition and load variations in relation to stand replacement. Canadian Journal of Forest Research 30:674-687.

Hemstrom, M. A., J. J. Korol, and W. J. Hann. 2001. Trends in terrestrial plant communities and landscape health indicate the effects of alternative management strategies in the Interior Columbia River Basin. Forest Ecology and Management 153:105-126.

Hessburg, P. F., B. G. Smith, and R. B. Salter. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications 9:1232-1252.

Hibbs, D. E. and A. L. Bower. 2001. Riparian forests in the Oregon Coast Range. Forest Ecology and Management 154: 201-213.

Hooge P. N., W. Eichenlaub, and E. Solomon. 1999. The animal movement program. USGS Alaska Biological Science Center, Gustavus, Alaska, USA.

Huff, M. H. 1995. Forest age structure and development following wildfires in the western Olympic Mountains, Washington. Ecological Applications 5:471-483.

Hunter, M. L. Jr., G. L. Jr. Jacobson, and T. I. Webb. 1988. Paleoecology and the coarse-filter approach to maintaining biological diversity. Conservation Biology 2:375-385.

Hunter, M. L. Jr. 1990. Wildlife, forests, and forestry. Prentice Hall, Englewood Cliffs, New Jersey, USA.

Hunter, M. L. Jr. 1993. Natural fire regimes as spatial model for managing boreal forests. Biological Conservation 65:115-120.

Hunter, M. L. Jr. 1999. Maintaining biodiversity in forest ecosystems. Cambridge University Press, Cambridge, UK.

Impara, P. C. 1997. Spatial and temporal patterns of fire in the forests of the central Oregon coast range. PhD Dissertation. Oregon State University, Corvallis, Oregon, USA.

Isaac, L. A. and G. S. Meagher. 1938. Natural reproduction on the Tillamook burn: four years after the fire. USDA Forest Service, Portland, Oregon, USA.

Johnson, D. H. and T. A. O'Neil. 2000. Wildlife-Habitat Relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Karjalainen, L. and T. Kuuluvainen. 2002. Amount and diversity of coarse woody debris within a boreal forest landscape dominated by *Pinus sylvestris* in Vienansalo Wilderness, Eastern Fennoscandia. Silva Fennica 36:147.

Kauffman, J. B., M. Mahrt, L. A. Mahrt, and W. D. Edge. 2000. Wildlife of riparian habitats. Pages 361-388 *in* D. H. Johnson and T. A. O'Neil, managing directors. Wildlife-Habitat Relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Keane, R. E., R. A. Parsons, and P. F. Hessburg. 2002. Estimating historical range and variation of landscape patch dynamics: limitations of the simulation approach. Ecological Modelling 151:29-49.

King, A. W. 1997. Hierarchy theory: a guide to system structure for wildlife biologists. Pages 185-214 *in* Bissonette, J. A., editor. Wildlife and Landscape Ecology. Springer-Verlag, New York, New York, USA.

King, S. L. and B. D. Keeland. 1999. Evaluation of reforestation in the lower Mississippi River Alluvial Valley. Restoration Ecology 7:348-359.

Klimo, E. 1998. History, condition and management of floodplain forest ecosystems in Europe. Pages 173-183 *in* K. Sassa, editor. Environmental forest science: proceedings of the IUFRO Division 8 Conference Environmental Forest Science. Kyoto University, Kyoto, Kyoto, Japan.

Knutson, M. G. and E. E. Klaas. 1998. Floodplain forest loss and changes in forest community composition and structure in the Upper Mississippi River: a wildlife habitat at risk. Natural Areas Journal 18:138-150.

Kolasa, J. and C. D. Rollo. 1991. Introduction: the heterogeneity of heterogeneity: a glossary. Pages 1-23 *in* Kolasa, J. and S. T. A. Pickett, editors. Ecological heterogeneity. Springer-Verlag, New York, New York, USA.

Krummel, J. R., R. H. Gardner, G. Sugihara, and R. V.O'Neill, and P. R. Coleman. 1987. Landscape patterns in a disturbed environment. OIKOS 48:321-324.

Kuuluvainen, T. 2002. Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. Silva Fennica 36:98-125.

Lackey, R. T. 2001. Values, policy, and ecosystem health. BioScience 51:437-443.

Landres, P. B., P. Morgan, and F. J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Leica Geosystems. 2002. ERDAS IMAGINE 8.6. Heerbrugg, Switzerland.

Lertzman, K. and J. Fall. 1998. From forest stands to landscapes: spatial scales and the roles of disturbance. Pages 339-367 *in* Peterson D. L. and V. T. Parker, editors. Ecological Scale: theory and applications. Columbia University Press, New York, New York, USA.

Lertzman, K., J. Fall, and B. Dorner . 1998. Three kinds of heterogeneity in fire regimes: at the crossroads of fire history and landscape ecology. Northwest Science 72:4-23.

Li, H. 1989. Spatio-temporal pattern analysis of managed forest landscapes: a simulation approach. PhD Dissertation. Oregon State University, Corvallis, Oregon, USA.

Li, H. and J. F. Reynolds. 1994. A simulation experiment to quantify spatial heterogeneity in categorical maps. Ecology 75:2446-2455.

Li, H. and J. F. Reynolds. 1995. On definition and quantification of heterogeneity. OIKOS 73:280-284.

Lillesand, T. M. and R. W. Kiefer. 2002. Remote sensing and image interpretation. 4th edition. John Wiley & Sons, New York, New York, USA.

Lindenmayer, D. B. and J. F. Franklin. 2002. Conserving forest biodiversity: a comprehensive multiscaled approach. Island Press, Washington, D.C., USA.

Long, C. J., C. Whitlock, P. J. Bartlein, and S. H. Millspaugh. 1998. A 9000-year fire history from the Oregon Coast Range, based on a high-resolution charcoal study. Canadian Journal of Forest Research 28:774-787.

Long, C. J. and C. Whitlock. 2002. Fire and vegetation history from the coastal rain forest of the western Oregon Coast Range. Quaternary Research 58:215-225.

Lotan, J. E., J. K. Brown, and L. F. Neuenschwander. 1985. Role of fire in lodgepole pine forests. Pages 133-152 *in* Lodgepole pine: the species and its management. Washington State University, Seattle, Washington, USA.

Loy, W. G., S. Allan, A. R. Buckley, and J. E. Meacham. 2001. Atlas of Oregon. University of Oregon Press, Eugene, Oregon, USA.

Lundquist, J. E., L. R. Lindner, and J. Popp. 2001. Using landscape metrics to measure suitability of a forested watershed: a case study for old growth. Canadian Journal of Forest Research 31:1786-1792.

Malanson, G. P. 1993. Riparian Landscapes. Cambridge University Press, Cambridge, UK.

Manies, K. L. and D. J. Mladenoff. 2000. Testing methods to produce landscape-scale presettlement vegetation maps from the U.S. public land survey records. Landscape Ecology 15:741-754.

Manies, K. L., D. J. Mladenoff, and E. V. Nordheim. 2001. Assessing large-scale surveyor variability in the historic forest data of the original U.S. Public Land Survey. Canadian Journal of Forest Research 31:1719-1730.

Manley, P. N., G. E. Brogan, C. Cook, M. E. Flores, D. G. Fullmer, S. Husari, T. M. Jimerson, L. M. Lux, M. E. McCain, J. A. Rose, G. Schmitt, J. C. Schuyler, and M. J. Skinner. 1995. Sustaining ecosystems: a conceptual framework. USDA Forest Service, Pacific Southwest Region. R5-EM-TP-001.

Martin, K. J. and W. C. McComb. 2002. Small mammal habitat associations at patch and landscape scales in Oregon. Forest Science 48:255-264.

Maser, C. 1995. The redesigned forest. R. & E. Miles, San Pedro, California, USA.

Maser C., S. P. Cline, K. Jr. Cromack, J. M. Trappe, and E. Hansen. 1988. What we know about large trees that fall to the forest floor. Pages 25-46 *in* C. Maser, R. F. Tarrant, J. M. Trappe, and J. F. Franklin, technical editors. From the forest to the sea: a story of fallen trees. USDA Forest Service and USDI Bureau of Land Management. PNW-GTR-229.

Matlack, G. R. 1994. Plant Species Migration in a Mixed-History Forest Landscape in Eastern North America. Ecology 75:1491-1502.

McArdle, R. E. and W. H. Meyer. 1961. The yield of Douglas-fir in the Pacific Northwest (Techinical Bulletin No. 201). United States Department of Agriculture, Washington, D. C., USA.

McCune, B. and M. J. Mefford. 1999. PC-ORD Version 4.20: Multivariate analysis for ecological data. MjM software design, Gleneden Beach, Oregon, USA.

McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities. MjM software design, Gleneden Beach, Oregon, USA.

McGarigal, K. and W. C. McComb. 1995. Relationships between landscape structure and breeding birds in the Oregon Coast Range. Ecological Monographs 65:235-260.

McGarigal, K., S. A. Cushman, and S. G. Stafford. 2000. Multivariate Statistics for Wildlife and Ecology Research. Springer-Verlag, New York, New York, USA.

McGarigal, K., S. A. Cushman, M. C. Neel, and E. Ene. 2002. FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps Computer software program produced by the authors at the University of Massachusetts, Amherst. www.umass.edu/landeco/research/fragstats/fragstats.html

Miller, C. I. and W. B. Woolfenden. 1999. The role of climate change in interpreting historical variability. Ecological Applications 9(4):1207-1216.

Milne, B. T. 1992. Indications of landscape condition at many scales. Pages 883-895 *in* D. H. McKenzie, D. E. Hyatt, V. J. McDonald, editors. Ecological indicators Volume 2. Elsevier Applied Science, New York, New York, USA.

Mitsch, W. J. and J. G. Gosselink. 2000. Wetlands. John Wiley & Sons, New York, New York, USA.

Mladenoff, D. J., M. A. White, J. Pastor, and T. R. Crow. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications 3:294-306.

Montreal Process Working Group. 1998. The Montreal Process. http://www.mpci.org/home_e.html

Morgan, P., G. H. Aplet, J. B. Haufler, H. C. Humphries, M. M. Moore, and W. D. Wilson. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. Journal of Sustainable Forestry 2: 87-111.

Morris, W. G. 1934. Forest fires in western Oregon and western Washington. Oregon Historical Quarterly 35:311-339.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and Methods of Vegetation Ecology Wiley and Sons, New York, New York, USA.

Muller, R. N. 2003. Landscape patterns of change in coarse woody debris accumulation in an old-growth deciduous forest on the Cumberland Plateau, southeastern Kentucky. Canadian Journal of Forest Research 33:763-769.

Naiman, R. J., H. Décamps, M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3: 209-212.

Naiman, R. J. and H. Décamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28: 621-658.

Naiman, R. J., R. E. Bilby, and P. A. Bisson. 2000. Riparian ecology and management in the Pacific coastal rain forest. BioScience 50: 996-1011.

Nelson, J. C., A. Redmond, and R. E. Sparks. 1994. Impacts of settlement on floodplain vegetation at the confluence of the Illinois and Mississippi Rivers. Transactions of the Illinois State Academy of Science 87:117-133.

Nelson, J. C. 1997. Presettlement vegetation patterns along the 5th principal meridian, Missouri Territory, 1815. American Midland Naturalist 137:79-94.

Nierenberg, T. R. and D. E. Hibbs. 2000. A characterization of unmanaged riparian areas in the central Coast Range of western Oregon. Forest Ecology and Management 129: 195-206.

Noble, I. R. and R. O. Slayer. 1977. Post-fire succession of plants in Mediterranean ecosystems. Proceedings of the symposium on the environmental consequences of fire and fuel management in Mediterranean climate ecosystems. USDA Forest Service Gen Tech Rep. WO-3:27-36.

Noss, R. F. 1987. From plant communities to landscapes in conservation inventories: a look at The Nature Conservancy (USA). Biological Conservation 41:11-37.

Noss, R. F. 1993. A conservation plan for the Oregon Coast Range: some preliminary suggestions. Natural Areas Journal 13:276-290.

Noss, R. F. and A. Y. Cooperrider. 1994. Saving Nature's Legacy: Protecting and Restoring Biodiversity. Island Press, Washington, D.C., USA.

Ohmann, J. L. and M. J. Gregory. 2002. Predictive mappting of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, USA. Canadian Journal of Forest Research 32:725-741.

Ohmann, J. L. and K. L. Waddell. 2002. Regional patterns of dead wood in forested habitats of Oregon and Washington. Pages 535-560 *in* W. F. Laudenslayer Jr., P. J. Shea, B. E. Valentine, C. P. Weatherspoon, T. E. Lisle, technical coordinators. Proceedings of the symposium on the ecology and management of dead wood in western forests. 1999 November 2-4. Reno, NV. Gen. Tech. Rep. PSW-GTR-181. USDA Forest Service, Pacific Southwest Research Station. Albany, California, USA.

Oliver, C. D. 1981. Forest development in North America following mojor disturbances. Forest Ecology and Management 3:153-168.

Oliver, C. D. and B. C. Larson. 1996. Forest Stand Dynamics. Updated edition. John Wiley & Sons, Inc., New York, New York, USA.

O'Neill, R. V., C. T. Hunsaker, S. P. Timmins, B. L. Jackson, K. B. Jones, K. H. Riitters, and J. D. Wickham. 1996. Scale problems in reporting landscape pattern at the regional scale. Landscape Ecology 11:169-180.

Oregon Department of Forestry. 1996. Forest Practice Administrative Rules Oregon Department of Forestry, Salem, Oregon, USA. http://www.odf.state.or.us/fp/rules.

Pabst, R. J. and T. A. Spies. 1999. Structure and composition of unmanaged riparian forests in the coastal mountains of Oregon, USA. Canadian Journal of Forest Research 29: 1557-1573.

Pedroli, B., G. de Blust, K. van Looy, and S. van Rooij. 2002. Setting targets in strategies for river restoration. Landscape Ecology 17:5-18.

Peet, R. K. 1981. Changes in biomass and production during secondary forest succession. Pages 324-338 *in* D. C. West, H. H. Shugart, D. B. Botkin, editors. Forest Succession: Concepts and Application. Springer-Verlag, New York, New York, USA.

Peet, R. K. and N. L. Christensen. 1987. Competition and tree death. BioScience 37:586-595.

Peterken, G. F. and M. Game. 1984. Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. Journal of Ecology 75:477-512.

Peterken, G. F. and F. M. R. Hughes. 1995. Restoration of floodplain forests in Britain. Forestry 68:187-202.

Peterson D. L. and V. T. Parker. 1998. Ecological Scale: theory and applications. Columbia University Press, New York, New York, USA.

Pickett, S. T. A. and K. H. Rogers. 1997. Patch dynamics: the transformation of landscape structure and function. Pages 101-127 *in* J. A. Bissonette, editor. Wildlife and landscape ecology. Springer-Verlag, New York, New York, USA.

Pickett, S. T. A. and P. S. White. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, San Diego, California, USA.

Poage, N. J. and J. C. I. Tappeiner. 2002. Long-term patterns of diameter and basal area growth of old-growth Douglas-fir trees in western Oregon. Canadian Journal of Forest Research 32:1232-1243.

Pollock, M. M., R. J. Naiman, and T. A. Hanley. 1998. Plant species richness in riparian wetlands—a test of biodiversity theory. Ecology 79:94-105.

Pratt, W. K. 1991. Digital image processing Wiley, New York, New York, USA.

Quigley, T. M., R. W. Haynes, and R. T. Graham. 1996. Integrated scientific assessment for ecosystem management in the Interior Columbia Basin, and portions of the Klamath and Great Basins. USDA Forest Service, Portland, Oregon, USA.

Reeves, G. H., L. E. Benda, K. M. Burnett, P. A. Bisson, and J. R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium 17:334-349.

Richards, F. J. 1959. A flexible growth function for empirical use. Journal of Experimental Botany 10:290-300.

Riitters, K. H., R. V. O'Neill, C. T. Hunsaker, J. D. Wickham, D. H. Yankee, and S. P. Timmins. 1995. A factor analysis of landscape pattern and structure metrics. Landscape Ecology 10:23-39.

Ripple, W. J. 1994. Historic spatial patterns of old forests in western Oregon. Journal of Forestry 92:45-49.

Rose, C. L., Marcot, B. G., T. K. Mellen, J. L. Ohmann, K. L. Waddell, D. L. Lindley, B. Schreiber. 2001. Decaying wood in Pacific Northwest forests: concepts and tools for habitat management. Pages 580-623 *in* D. H. Johnson and T. A. O'Neil, managing directors. Wildlife-Habitat Relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Rouvinen S, K. T. K. L. 2002. Coarse woody debris in old *Pinus sylvestris* dominated forests along a geographic and human impact gradient in boreal Fennoscandia. Canadian Journal of Forest Research 32:2184-2200.

Roworth, E. T. I. 2001. Quantifying the variation in landscape structure under a natural fire regime in the Southern Rocky Mountains. M.S. Thesis. University of Massachusetts, Amherst, Massachusetts, USA.

Rubino, D. L. and B. C. McCarthy. 2003. Evaluation of coarse woody debris and forest vegetation across topographic gradients in a southern Ohio forest. Forest Ecology and Management 183:221-238.

Schafale, M. P. and P. A. Harcombe. 1983. Presettlement vegetation of Hardin County, Texas. American Midland Naturalist 109:355-366.

Schieck, J., K. Stuart-Smith, and M. Norton. 2000. Bird communities are affected by amount and dispersion of vegetation retained in mixedwood boreal forest harvest areas. Forest Ecology and Management 126:239-254.

Schnitzler, A. 1994. Conservation of biodiversity in alluvial hardwood forests of the temperate zone: the example of the Rhine valley. Forest Ecology and Management 68:385-398.

Schulte, L. A. and D. J. Mladenoff. 2001. The original US public land survey records: their use and limitations in reconstructing presettlement vegetation. Journal of Forestry 99:5-10.

Schulte, L. A., D. J. Mladenoff, and E. V. Nordheim. 2002. Quantitative classification of a historic northern Wisconsin (USA) landscape: mapping forests at regional scales. Canadian Journal of Forest Research 32:1616-1638.

Seaman, D. E. and R. A. Powell. 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. Ecology 77:2075-2085.

Seymour, R. S., A. S.White, and P.G. Demaynadier. 2002. Natural disturbance regime in northeastern North America--evaluating silvicultural systems using natural scales and frequencies. Forest Ecology and Management 155:357-367.

Shugart, H. H. 1984. A theory of forest dynamics. Springer-Verlag, New York, New York, USA.

Sillett, S. C., B. McCune, J. E. Peck, T. R. Rambo, and A. Ruchty. 2000. Dispersal limitations of epiphytic lichens result in species dependent on old-growth forests. Ecological Applications 10:789-799.

Simons, N. P., F. E. Schwab, and R. D. Otto. 2002. Songbird abundance in clear-cut and burned stands: a comparison of natural disturbance and forest management. Canadian Journal of Forest Research 32:1343-1350.

Sollins, P. 1982. Input and decay of coarse woody debris in coniferous stands in western Oregon and Washington. Canadian Journal of Forest Research 12:18-28.

Sparks, R. E. 1995. Need for ecosystem management of large rivers and their floodplains. BioScience 45:168-181.

Spetich, M. A., S. R. Shifley, and G. R. Parker. 1999. Regional distribution and dynamics of coarse woody debris in Midwestern old-growth forests. Forest Science 45:302-313.

Spies, T. A., J. F. Franklin, and T. B. Thomas. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology 69:1689-1702.

Spies, T. A. and J. F. Franklin. 1991. The structure of natural young, mature, and old-growth Douglas-fir forests in Washington and Oregon. Pages 91-109 *in* L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir Forests. USDA Forest Service, Pacific Northwest Research Station. General Technical Report. PNW-GTR-285. Portland, Oregon, USA.

Spies, T. A., W. J. Ripple, and G. A. Bradshaw. 1994. Dynamics and pattern of a managed coniferous forest landscape in Oregon. Ecological Applications 4:555-568.

Spies, T. A. and J. F. Franklin. 1996. The diversity and maintenance of old-growth forests, pp. Pages 296-314 *in* R. R. Szaro and D. W. Johnston, editors. Biodiversity in managed landscapes: theory and practice. Oxford University Press, New York, New York, USA.

Spies, T. A. 1998. Forest structure: a key to the ecosystem. Northwest Science 72:34-39.

Spies, T. A. and M. G. Turner. 1999. Dynamic forest mosaics. Pages 95-160 *in* M. L. Hunter Jr., editor. Maintaining biodiversity in forest ecosystems. Cambridge University Press, Cambridge, UK.

Spies, T. A., J. F. Franklin, G. Spycher, and R. J. Pabst. In press. Aboveground biomass and carbon storage in natural Douglas-fir stands in relation to stand and site characteristics in western Oregon and Washington.

Spies, T. A. and S. P. Cline. 1988. Coarse woody debris in forests and plantations of coastal Oregon. Pages 5-24 *in* C. Maser, R. F. Tarrant, J. M. Trappe, and J. F. Franklin, technical editors. From the forest to the sea: a story of fallen trees. USDA Forest Service and USDI Bureau of Land Management. PNW-GTR-229.

Spies, T. A. and J. F. Franklin. 1988. Old growth and forest dynamics in the Douglas-fir region of western Oregon and Washington. Natural Areas Journal 8:190-201.

Spies, T. A., D. E. Hibbs, J. L. Ohmann, G. H. Reeves, R. J. Pabst, F. J. Swanson, C. Whitlock, J. A. Jones, B. C. Wemple, L. A. Parendes, B. A. Schrader. 2002a. The ecological basis of forest ecosystem management in the Oregon Coast Range. Pages 31-67 *in* S. D. Hobbs, J. P. Hayes, R. L. Johnson and others, editors. Forest and stream management in the Oregon Coast Range. Oregon State University Press, Corvallis, Oregon, USA.

Spies, T. A., G. H. Reeves, K. M. Burnett, W. C. McComb, K. N. Johnson, G. Grant, J. L. Ohmann, S. L. Garman, and P. Bettinger. 2002b. Assessing the ecological consequences of forest policies in a multi-ownership province in Oregon. Pages 179-207 *in* J. Liu and W. W. Taylor, editors. Integrating landscape ecology into natural resource management. Cambridge University Press, Cambridge, UK.

Stanfield, B. J., J. C. Bliss, and T. A. Spies. 2002. Landownership and landscape structure: a spatial analysis of sixty-six Oregon Coast Range watersheds. Landscape Ecology 17:685-697.

Stewart, L. O. 1935. Public Land Surveys: history, instructions, methods. Collegiate Press, Inc., Ames, Iowa, USA.

Stone, J. N., A. MacKinnon, J. V. Parminter, and K. P. Lertzman. 1998. Coarse woody debris decomposition documented over 65 years on southern Vancouver Island. Canadian Journal of Forest Research 28:788-793.

Swetnam, T. W., C. D., Allen, and J. L., Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189-1206.

Swanson, F. J., J. F. Franklin, and J. R. Sedell. 1990. Landscape patterns, disturbance, and management in the Pacific Northwest, USA. Pages 191-213 *in* I. S. Zonneveld and R. T. T. Forman, editors. Changing landscapes: an ecological perspective. Springer-Verlag, New York, New York, USA.

Swanson, F. J. and J. F. Franklin. 1992. New forestry principles from ecosystem analysis of Pacific Northwest forests. Ecological Applications 2:262-274.

Swanson, F. J., J. A. Jones, D. O. Wallin, and J. H. Cissel. 1994. Natural variability-implications for ecosystem management. Pages 80-94 *in* M. E. Jensen and P. S. Bourgeron, technical editors. Volume II: Ecosystem Management: Principles and Applications. USDA Forest Sservice Pacific Northwest Research Station. General Technical Report. PNW-GTR-318.

Swanson, F. J., T. K. Kratz, N. Caine, and R. G. Woodmansee. 1988. Landform effects on ecosystem patterns and processes: geomorphic features of the earth's surface regulate the distribution of organisms and processes. BioScience 38:92-98.

Swanson, F. J., S. L. Johnson, S. V. Gregory, and S. A. Acker. 1998. Flood disturbance in a forested mountain landscape. Bioscience 48:681-689.

Swanson, F. J., J. H. Cissel, and A. Reger. 2003. Landscape management: diversity of approaches and points of comparison. Pages 237-266 (Chapter 9) *in* R. A. Monserud, R. W. Haynes, and A. C. Johnson, editors. Compatible Forest Management. Kluwer Academic Publishers. The Netherlands.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189-1206.

Tappeiner, J. C., D. Huffman, D. Marshall, T. A. Spies, and J. D. Bailey. 1997. Density, ages, and growth rates in old-growth and young-growth forests in coastal Oregon. Canadian Journal of Forest Research 27:638-648.

Teensma, P. D. A., J. T. Rienstra, and M. A. Yeiter. 1991. Preliminary reconstruction and analysis of change in forest stand age classes of the Oregon Coast Range from 1850 to 1940. Technical Note. T/N OR-9. USDI Bureau of Land Management. Portland, Oregon, USA.

Tinker, D. B., W. H. Romme, and D. G. Despain. 2003. Historic range of variability in landscape structure in subalpine forests of the Greater Yellowstone Area, USA. Landscape Ecology 18:427-439.

Turner, M. G., W. H. Romme, R. H. Gardner, and R. V. O'Neill. 1993. A revised concept of landscape equilibrium: disturbance and stability on scaled landscapes. Landscape Ecology 8:213-227.

Turner, M. G. and W. H. Romme. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology 9:59-77.

Urban, D. L., M. F. Acevedo, and S. L. Garman. 1999. Scaling fine-scale processes to large-scale patterns using models derived from models: meta-models. Pages 70-98 *In* D. Mlandenoff and W. Baker, editors. Spatial Modeling of Forest Landscape Change: Approaches and Applications. Cambridge University Press, Cambridge, UK.

Urban, D. L., R. V. O'Neill, and H. H. Shugart Jr. 1987. Landscape Ecology: a hierarchical perspective can help scientists understand spatial patterns. BioScience 37:119-127.

USDA Forest Service and USDI Bureau of Land Management. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl; Standards and guidelines for management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. Portland, Oregon, USA.

Vale, T. R. 2002. Fire, native peoples, and the natural landscape. Island Press, Washington, D.C., USA.

Vannote, R. L., G. W. Minshall, K. W. Cimmins, J. R. Sedell, and C. E. Cusing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Wallin, D. O., F. J. Swanson, and B. Marks. 1994. Landscape pattern response to changes in pattern generation rules: land-use legacies in forestry. Ecological Applications 4:569-580.

Wallin, D. O., F. J. Swanson, B. Marks, J. H. Cissel, and J. Kertis. 1996. Comparison of managed and pre-settlement landscape dynamics in forests of the Pacific Northwest, USA. Forest Ecology and Management 85:291-309.

Weisberg, P. J. and F. J. Swanson. 2003. Regional synchroneity in fire regimes of western Oregon and Washington, USA. Forest Ecology and Management 172:17-28.

Weisberg, P. J. In press. Importance of non-stand-replacing fire for development of forest structure in the Pacific Northwest. Forest Science.

West, D. C., H. H. Shugart, and D. B. Botkin. 1981. Forest succession: concepts and application. Springer-Verlag, New York, New York, USA.

Whitlock, C. 1992. Vegetational and climatic history of the Pacific Northwest during the last 20,000 years: implications for understanding present-day biodiversity. The Northwest Environmental Journal 8:5-28.

Whitlock, C. and M. A. Knox. 2003. Prehistoric burning in the Pacific Northwest: human versus climatic influences. Pages 195-231 *in* T. R. Vale, editor. Fire, native peoples, and the natural landscape. Island Press, Washington, D.C., USA.

Wigley, T. B. and T. H. Roberts. 1997. Landscape-level effects of forest management on faunal diversity in bottomland hardwoods. Forest Ecology and Management 90:141-154.

Wimberly, M. C. 2002. Spatial simulation of historical landscape patterns in coastal forests of the Pacific Northwest. Canadian Journal of Forest Research 32:1316-1328.

Wimberly, M. C., T. A. Spies, C. J. Long, and C. Whitlock. 2000. Simulating historical variability in the amount of old forests in the Oregon Coast Range. Conservation Biology 14:167-180.

Wimberly, M. C. and T. A., Spies. 2002. Landscape- vs. Gap-level controls on the abundance of a fire-sensitive, late-successional tree species. Ecosystems 5:232-243.

Wimberly, M. C. 2003. Species response to dynamic landscape patterns: theoretical simulations and management considerations (a conference abstract). Page 141 *in* 4th North American Forest Ecology Workshop held at Corvallis, Oregon.

Wimberly, M. C., T. A. Spies, and E. Nonaka. In press. Using natural fire regime-based criteria to evaluate forest management in the Oregon Coast Range. *In* Perera, A. H., L. J. Buse, and M. G. Weber, editors. Emulating Natural Forest Landscape Disturbances: Concepts and Applications. Columbia University Press, Cambridge, New York, USA

Winter, L. E., L. B. Brubaker, J. F. Franklin, E. A. Miller, and D. Q. DeWitt. 2002. Canopy disturbances over the five-century lifetime of an old-growth Douglas-fir stand in the Pacific Northwest. Canadian Journal of Forest Research 32:1057-1070.

Worona, M. A. and C. Whitlock. 1995. Late quaternary vegetation and climate history near Little Lake, central Coast Range, Oregon. GSA bulletin 107:867-876.

Wright, P. J. 1999. The effect of fire regime on coarse woody debris in the west central Cascades, Oregon. M.S. Thesis. Oregon State University, Corvallis, Oregon, USA.

Zackrisson, O. 1977. Influence of forest fires on the north Swedish boreal forest. OIKOS 29:22-32.

Zonneveld, I. S. and R. T. T. Forman. 1990. Changing Landscapes: an ecological perspective. Springer-Verlag, New York, New York, USA.