

**Second Edition**

# **WILDLIFE HABITAT MANAGEMENT**

Concepts and Applications in Forestry



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Concepts and Applications in Forestry

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# Contents

Preface.....	xiii
Author .....	xv
 <b>Chapter 1</b> Introduction .....	 1
What Is Habitat? .....	1
Habitat Function .....	2
Habitat for Humans .....	5
Forests as Habitat .....	5
Historical Approaches to Managing Forests as Habitat .....	6
Why Manage Habitat? .....	7
Case Study: The Forests of British Columbia .....	9
Summary .....	11
References .....	11
 <b>Chapter 2</b> Humans as a Forest-Dependent Species.....	 13
Ecological Restoration and Ecosystem Services.....	13
Social Values Associated with Forests and Wildlife.....	15
Environmental Ethics .....	15
Ecological Psychology .....	17
Public Resources on Private Lands .....	17
Case Study: Environmental Activism and Effects on Habitat .....	18
Summary .....	18
References .....	19
 <b>Chapter 3</b> Vertebrate Habitat Selection.....	 21
Hierarchical Selection .....	21
Where in the World Should I Live? .....	22
Looking for a Home in a Nice Community.....	25
Buying a House and Buying Groceries .....	26
What Would You Like for Dinner? .....	26
Density-Dependent Habitat Selection .....	26
Relationship between Habitat Quality and Demographics .....	28
Population Fitness.....	29
Measuring Habitat Selection .....	30
Proximate and Ultimate Cues to Habitat Quality .....	31
Social Cues in Habitat Selection .....	32
Case Study: American Marten Habitat Selection .....	32
Summary .....	33
References .....	34
 <b>Chapter 4</b> Forest Structure and Composition.....	 37
Food and Cover in a Cellulose-Managed System .....	37
Vertical Complexity.....	37

Horizontal Patchiness .....	40
Forage Availability and Quality .....	40
Fruit Production.....	43
Dead and Damaged Trees.....	44
Tree Species and Invertebrate Associations .....	45
Tree Size and Density .....	46
Forest Floor Litter and Soil .....	47
Proximity to Water .....	47
Case Study on Plant Response to Herbivores, or, It Is a (Chemical) War Out There!.....	48
Summary .....	48
References .....	49
<b>Chapter 5</b> Physical Influences on Habitat Patterns .....	53
The Physical Environment .....	53
Geology .....	53
Topography: Slope, Aspect, and Elevation.....	55
Soils .....	56
Climate .....	56
Hydrology .....	60
Vegetation Patterns.....	60
Summary .....	62
References .....	63
<b>Chapter 6</b> Cultural Effects on Habitat Patterns .....	67
Land Use .....	67
Urbanization .....	67
Forest Clearing for Agriculture .....	68
Energy Production and Biofuels.....	68
Climate Change.....	69
Invasive Species .....	71
Synergistic Effects.....	75
Case Study: Passenger Pigeons, Humans, and Forests .....	75
Summary .....	77
References .....	77
<b>Chapter 7</b> Disturbance Ecology and Habitat Dynamics .....	79
Disturbance Size and Pattern .....	79
Disturbance Severity .....	81
Disturbance Frequency.....	82
Disturbance Frequency, Size, Severity Relationships .....	83
Stand Dynamics .....	83
Stand Initiation .....	83
Stem Exclusion .....	85
Understory Reinitiation .....	86
Old-Growth .....	86
Succession as a Continuum of Habitat Elements .....	87
Successional Pathways.....	88

Management Implications from Disturbances .....	89
Summary .....	90
References .....	91
<b>Chapter 8</b> Silviculture and Habitat Management: Even-Aged Systems.....	93
Silviculture as a Forest Disturbance.....	93
Characteristics of Even-Aged Stands .....	94
Considering the Capabilities of the Site .....	95
Choosing a Regeneration Method .....	96
Identifying Legacy Elements to Retain .....	98
Site Preparation Effects on Habitat Elements .....	98
Natural Regeneration and Planting Options .....	99
Vegetation Management Effects on Habitat Elements .....	100
Herbicides and Pesticides .....	101
Direct Effects.....	102
Indirect Effects .....	102
Precommercial Thinning.....	103
Commercial Thinning .....	103
Fertilization .....	107
Rotation Length: Ecological and Economic Trade-Offs.....	107
Case Study: Douglas-Fir Plantation .....	108
Summary .....	111
References .....	112
<b>Chapter 9</b> Silviculture and Habitat Management: Uneven-Aged Systems.....	117
Characteristics of Uneven-Aged Stands.....	117
Considering the Site Potential .....	120
Uneven-Aged Regeneration Methods.....	120
Natural Regeneration and Planting Options.....	120
Uneven-Aged Stand Development.....	121
Habitat Elements in Uneven-Aged Stands .....	122
Vertical Structure .....	122
Horizontal Diversity .....	123
Forage and Browse .....	123
Dead and Dying Trees.....	124
Mast .....	124
Challenges to Using Uneven-Aged Methods.....	124
Nontraditional Management Approaches.....	125
Case Study: Managing a Small Privately Owned Forest .....	126
Summary .....	127
References .....	128
<b>Chapter 10</b> Desired Future Conditions .....	131
Developing the Stand Prescription.....	131
Species Background .....	132
Current Stand Condition.....	132
Desired Future Condition .....	132
Management Actions to Achieve the Desired Future Condition.....	132

Monitoring Plans .....	132
Budget.....	132
Schedule .....	132
References.....	132
Case Study: Growing Red-Cockaded Woodpecker Habitat.....	133
Species Background and Management Options .....	133
Current Stand Condition.....	134
Desired Future Condition .....	135
Management Actions to Achieve the DFC .....	136
Monitoring Plans .....	140
Budget.....	141
Summary .....	143
References .....	143
<b>Chapter 11 Riparian Area Management .....</b>	<b>145</b>
Animal Associations with Riparian Areas .....	146
Gradients within Riparian Zones .....	147
Riparian Functions .....	150
Riparian Buffers.....	152
Managing within Streamside Management Areas .....	154
Beavers: The Stream Managers? .....	155
Case Study: Riparian Area Management in a Patchwork Ownership .....	156
Summary .....	157
References .....	158
<b>Chapter 12 Dead Wood Management .....</b>	<b>161</b>
Primary Cavity Excavators .....	161
Secondary Cavity Users .....	163
Log Users .....	164
Patterns of Dead Wood Following Disturbance.....	165
Changes in Dead Wood Over Time.....	166
Dead Wood during Stand Development .....	167
Management of Tree Cavities and Dead Wood.....	168
Live Cavity-Tree Management in Managed Stands .....	171
Dead Wood Retention and Harvest System Considerations.....	172
Creating Snags and Logs for Wildlife .....	173
Monitoring Cavity Trees, Snags, and Logs .....	174
Case Study: Managing Dead Wood in Oregon Forests.....	174
Summary .....	176
References .....	176
<b>Chapter 13 Managing Fire in Forests .....</b>	<b>181</b>
Effects of Fire on Habitat Elements and Succession.....	181
Species, Fuels, Fire Frequency, and Severity .....	182
Creation and Loss of Dead Wood.....	182
Effects on Species Composition .....	183
Changing Fire Risk through Management.....	183
Salvage Logging .....	183

Effects of Fuels Treatments on Habitat Elements .....	184
Use of Prescribed Fire in Managed Forests .....	185
Case Study: To Salvage Log or Not? A Scientific Debate with a Social Solution.....	185
Summary .....	187
References .....	187
<b>Chapter 14</b> Urban Forests and Habitat Elements .....	189
Defining Urban Forests along an Urban–Rural Continuum .....	189
How Some Species Interpret the “Built Environment?” .....	190
Finding Spaces for Habitat Management .....	191
Habitat Elements Limiting Species in Urban and Suburban Settings.....	191
Urban Streams and Wetlands .....	192
Urban Expansion, Wetlands, and Mitigation .....	193
Managing Trees, Parks, and Forests in Urban Settings .....	194
Summary .....	195
References .....	195
<b>Chapter 15</b> Landscape Structure and Composition .....	197
Landscapes .....	197
Defining the Landscape.....	198
Habitat Quality at the Landscape Scale .....	200
Living on the Edge .....	200
Edge Geometry.....	203
Habitat Fragmentation.....	205
Habitat Area: Species–Area Relationships.....	206
Case Study: Habitat Area or Pattern?.....	209
Summary .....	211
References .....	211
<b>Chapter 16</b> Landscape Connections.....	213
Dispersal.....	213
Understanding the Probability of Successful Dispersal.....	214
Connectivity and Gap-Crossing Ability.....	218
Understanding Physical and Genetic Connectivity .....	220
Management Approaches to Connectivity .....	222
Case Study: Matrix Management for a Wide-Ranging Species.....	222
Summary .....	224
References .....	225
<b>Chapter 17</b> Managing Woodlands in Agricultural Environments .....	227
Value of Woodlots to Landowners .....	228
Edge Effects in Agricultural Settings .....	228
Habitat Elements in Woodlots .....	228
Fencerows and Shelterbelts.....	229
Loss of Forests in Industrial Agricultural Settings .....	229
Field and Farm Management That Influences Woodlot Function.....	230
Annual Crop Management .....	230

Grazing Management .....	231
Land Sparing .....	231
Specialty Crops and Agroforestry .....	232
Organic versus Traditional Agriculture .....	232
Multifunctional Landscapes .....	233
Certification of Agricultural Lands for Wildlife: The Role of Trees .....	234
Summary .....	234
References .....	235
<b>Chapter 18</b> Approaches to Biodiversity Conservation .....	237
What Is Biodiversity? .....	237
Setting Biodiversity Goals .....	239
How Do We Conceptualize “Biodiversity” to Be Able to Conserve It? .....	239
Coarse-Filter Approaches .....	240
Meso-Filter Approaches .....	243
Fine-Filter Approaches .....	243
Challenges to Managing Biodiversity .....	245
Spatial Scale .....	245
Time .....	247
Uncertainty .....	248
Summary .....	249
References .....	250
<b>Chapter 19</b> Landscape Management Plans .....	253
Establishing Goals .....	253
Current Conditions .....	255
Desired Future Conditions .....	255
Pathways to Desired Future Conditions .....	256
Developing the Landscape Management Plan .....	256
Policy Guidelines for Habitat Conservation Plans .....	258
General Structure of the Landscape Management Plan .....	259
Considering Alternative Plans .....	260
Finding Solutions to Land Management Planning Problems .....	261
Plan Effectiveness .....	263
Summary .....	264
References .....	264
<b>Chapter 20</b> Ecoregional Assessments and Conservation Priorities .....	267
Ecoregional Assessments .....	267
Examples of Ecoregional Assessments .....	271
Conducting an Ecoregional Analysis .....	272
Assessing Patterns of Habitat Availability and Quality .....	273
Prioritizing Management and Assessing Policies .....	275
Coarse-Filter Approach .....	275
Integrated Coarse-, Meso-, and Fine-Filter Approaches .....	276
Fine-Filter Approaches .....	277
Utility and Effectiveness of Ecoregional Assessments .....	279
Summary .....	280
References .....	280

<b>Chapter 21</b>	<b>Viable Populations in Dynamic Forests .....</b>	<b>283</b>
	Extinction Risks .....	283
	Goals of PVAs .....	285
	PVA Models .....	285
	Conducting a PVA for a Forest-Associated Species .....	286
	Examples of PVA Analyses.....	289
	Grizzly Bear .....	289
	Marbled Murrelet .....	290
	Neotropical Migrant Birds .....	291
	Model Errors and Uncertainties .....	291
	Poor Data .....	292
	Difficulties in Parameter Estimation .....	293
	Weak Ability to Validate or Verify Models .....	293
	Effects of Alternative Model Structures.....	294
	Interpreting Results from PVA Projections.....	294
	Summary .....	295
	References .....	295
 <b>Chapter 22</b>	 <b>Monitoring Habitat Elements and Populations .....</b>	 <b>297</b>
	Adaptive Management.....	297
	Designing Monitoring Plans .....	299
	Selection of Indicators .....	301
	Describe the Scope of Inference.....	301
	Describe the Experimental Design.....	302
	Sampling Intensity, Frequency, and Duration .....	302
	Monitoring Habitat Elements .....	303
	Monitoring for Species Occurrence .....	304
	Monitoring Trends.....	305
	Cause and Effect Monitoring Designs.....	307
	Are Data Already Available and Sufficient? .....	308
	Making Decisions with Data.....	309
	Examples of Approaches to Monitoring .....	310
	Monitoring Clonal Plants .....	310
	Monitoring the Occurrence of a Small Mammal Species.....	310
	Monitoring Trends in a Salamander Subpopulation.....	311
	Monitoring Response of Neotropical Migrant Birds to Forest Management .....	311
	Monitoring Habitat Elements .....	313
	Summary .....	313
	References .....	313
 <b>Chapter 23</b>	 <b>Forest Sustainability and Habitat Management .....</b>	 <b>317</b>
	Defining the Resources to Be Sustained .....	318
	Scales of Sustainability .....	318
	Humans Are Part of the System.....	319
	Are We Making Progress?.....	319
	Forest Certification.....	320
	Effectiveness of Certification .....	323
	Sustaining and Marketing Ecosystem Services: Habitat Banking.....	324

Summary .....	325
References .....	325
<b>Chapter 24 Regulatory and Legal Considerations .....</b>	<b>327</b>
International Laws and Agreements .....	327
National Laws .....	328
State Laws .....	332
Municipal Policies .....	334
Policy Analysis .....	335
How Decisions in the United States Influence Habitat in the World .....	336
Case Study: The Endangered Species Act and Climate Change .....	337
Summary .....	337
References .....	338
<b>Chapter 25 Should I Manage the Forest? .....</b>	<b>341</b>
What Does Restoration Mean? .....	341
Human Requirements as Constraints on Goals .....	342
Developing a Personal Management Philosophy .....	343
Our Place on Earth .....	344
Living Simply and Sustainably .....	344
Leaving the World a Better Place .....	345
Summary .....	346
References .....	346
<b>Appendix 1: Common and Scientific Names of Species Mentioned in the Text .....</b>	<b>347</b>
<b>Appendix 2: Glossary .....</b>	<b>353</b>
<b>Appendix 3: Measuring and Interpreting Habitat Elements .....</b>	<b>359</b>
<b>Appendix 4: Wildlife–Habitat Relationships Models .....</b>	<b>365</b>
<b>Appendix 5: Projecting Habitat Elements through Time .....</b>	<b>369</b>
<b>Index .....</b>	<b>375</b>



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# Preface

This book is the result of over 35 years of working in academic and research organizations in which foresters, biologists, and individuals from other disciplines collaborated, fought, argued, and occasionally agreed about how forests should be managed. That 35-year period spanned a time that began as a shift from a game-management focus to nongame species management to biodiversity conservation to provision of ecological services. On public lands especially, forestry evolved from commodity-focused even-aged management to green tree retention to ecological forestry. On private forest lands, green certification emerged as a nongovernmental licensing program that recognized commodity production and biodiversity conservation. Some scientists and managers involved in disciplines of wildlife management, forestry, sociology, biology, entomology, zoology, and botany as separate disciplines merged into a common framework for conservation of biodiversity that became known as conservation biology. In the 1980s and 1990s, ecological preservation often became pitted against economic growth. Passion for sustaining traditional ways of life faced passion for sustaining biodiversity. Arguments were intense, but people changed. Eventually, neither philosophy emerged dominant, but rather realization grew that stable economies, viewing humans as part of ecosystems, and sustainability of ecological services were becoming more representative of societal values. Resource professionals, at times reluctantly, also began approaching natural resource management problems in this interdisciplinary light. My own philosophy changed during that time as well. I always felt that the common ground between wildlife biologists and foresters was greater than the chasm of differences, and I tried to represent one of many bridges between the disciplines in my classes, my research, and my interactions with other professionals.

Writing this book stemmed from both a practical need and an emotional desire. Practically, I wanted to be able to use a book in my forestry and wildlife habitat management classes, and hopefully others would use it in theirs. There are excellent texts available, but I had long felt that I could make a contribution by integrating silvicultural and forest-planning principles with principles of habitat ecology and conservation biology. In addition, I wanted a forum where representatives of both disciplines could read my views and use them as a jumping off point for continued discussions, debates, arguments, and perhaps even agreement. The emotional motivations for writing this book came from my two children. Although neither shows an obvious inclination for pursuing a career in natural resources, both exhibit a strong appreciation for nature. I would like to feel that I have done everything I can do to leave them a living inheritance as rich in the diversity of life as the world that I lived in, and ensure that the foundation is laid for their generation to do the same for those who follow.

Many people contributed to the materials in this book. People whom I considered mentors and who shaped my thinking tremendously at various points in my career are Drs. Malcolm Coulter, Mitch Ferrell, Bob McDowell, Bob Noble, Bart Thielges, Logan Norris, John Gordon, Susan Stafford, Anita Azarenko, and Sabah Randhawa. In particular, I would like to thank reviewers of various chapters. Several teaching assistants for my courses read and commented on early drafts of all chapters and to them I am deeply in debt: Misty Cannon, Jesse Caputo, Stephanie Hart, Lori Keyes, and Holly Ober. Reviewers of individual chapters made excellent suggestions and include Matt Betts, Sal Chinnici, Sally Duncan, Cheryl Freisen, Joan Hagar, Sue Koger, Bob Lackey, Josh Lawler, Karl Martin, Bob Mitchell, Randy Rosenberger, Tom Spies, Nobuya Suzuki, John Tappeiner, Denis White, and Ben Zuckerberg. Hundreds of students in my WFCO 564 and FS/FW 453 classes used the first edition of this book and many provided comments and feedback that I used as a basis for improvements. Students in FS/FW 553 reviewed this second edition of the book. JoAnn Smith,

silviculturist on the Kisatchie National Forest in Louisiana, generously provided stand exam data, which were used as the basis for an illustration of achieving a desired future condition.

Photos were generously provided by Susan Campbell, Joan Hagar, Mike Jones, Karl Martin, Bruce McCune, Kevin McGarigal, Jim Petranka, Dave Vesely, Dianna Fisher, Jim Rivers, Katie Williams, and Michele Woodford as well as federal agencies, including U.S. Geological Survey, U.S. Forest Service, National Park Service, and U.S. Fish and Wildlife Service. Numerous publishers and journals generously allowed me to use previously published figures and text and they are cited herein.

I thank the provost of Oregon State University (OSU), Dr. Sabah Randhawa for providing me with the freedom to tackle a second edition of this book while working as the dean of the Graduate School. The University of Massachusetts Amherst provided me with a sabbatical leave to focus on finishing the first edition of this book. Randy Brehm from Taylor & Francis publishers provided outstanding support throughout the process of writing this second edition.

Finally, I thank my family and the friends, especially my partner Dianna Fisher, who have supported me throughout this project, all my many other projects, and the various trials and tribulations of life that have led me to this place and time in my life.

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# Author

**Brenda McComb** is dean of the Graduate School and a professor in the Department of Forest Ecosystems and Society at Oregon State University (OSU). She is an author of over 130 technical papers dealing with forest and wildlife ecology, habitat relationships, and habitat management. She was born and raised in Connecticut at a time and place when the rural setting provided opportunities to roam forests and fields. She earned a BS in natural resources conservation from the University of Connecticut, an MS in wildlife management from the University of Connecticut, and a PhD in forestry from Louisiana State University. She has served on the faculty at the University of Kentucky, University of Massachusetts Amherst, and OSU. She was head of the Department of Natural Resources Conservation at the University of Massachusetts Amherst for 7 years and associate dean for Research and Outreach for 1 year. Most recently, she served 2 years as a head of the Department of Forest Ecosystems and Society at OSU and now serves as dean of the Graduate School at OSU. She also previously served as a chief of the Watershed Ecology Branch in Corvallis for U.S. EPA for 1 year. Her current work addresses interdisciplinary approaches to management of multi-ownership landscapes in Pacific Northwest forests and agricultural areas.



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# 1 Introduction

Aldo Leopold is generally accepted as providing the philosophical basis for wildlife management in the United States. Leopold was trained and employed initially as a forester, and the academic and disciplinary home of wildlife management in the early years was aligned with forestry. But over time, due to the need to be recognized as a discipline in its own right, wildlife biology and management diverged from the forestry profession and in recent years the views of some wildlife professionals have been at odds with views expressed among some forestry professionals. This book is an attempt to bridge the disciplines of wildlife habitat management and forest management. It provides the conceptual bases for stand and landscape management so as to achieve habitat objectives for various species and communities and also provides case studies from across the United States to illustrate how these concepts can be applied. By providing the foresters with an explanation of concepts of habitat selection, habitat relationships, habitat elements, element dynamics in stands and landscapes, habitat permeability, connectivity, and exogenous pressures (climate change, invasive species, development), they can understand how these factors would influence the decisions made during stand and forest management. Further, biologists are provided with explanations of stand and forest landscape dynamics, silvicultural approaches to providing habitat elements, and harvest planning. Case studies in each section of the book provide examples of how these concepts can be applied to achieve habitat goals at stand, landscape, and regional spatial scales. Finally, the information culminates in stand prescription development and forest planning—key prerequisites to sustainable management practices. In addition, this planning process must include the concerns and objectives of various stakeholders. Foresters and wildlife biologists **MUST** work together, cooperatively, with these concerned publics to ensure that management approaches are adaptable to the inevitable social changes and to the competing demands for ecosystem services and aesthetic qualities of forests while also ensuring that current decisions are not likely to forgo future options.

## WHAT IS HABITAT?

Despite the need to work together to achieve mutual goals across forested landscapes, the language of disciplines can simply interfere with success. Throughout this chapter I will try to define terms that could be confusing or misinterpreted between the disciplines. For instance, a brief search of the web using the search terms “forest wildlife habitat” produced 356,000 hits, including the following quotes from resource professionals: “In recent years, an increasing number of landowners have realized the economic importance of timber management as a way to enhance wildlife habitat.” Moreover, another quote states that “Several practices have damaged the wildlife habitat, including habitat fragmentation; past roads, excessive logging and development all worked to fragment large areas of intact habitat.” So who is correct? All 356,000? Well, yes and no. We cannot understand how to manage the forests to provide habitat for wildlife species, or more generally for biodiversity, unless we understand what habitat is and is not. But first, what is wildlife? It is important to first recognize that it is not a singular noun, but rather a plural one. Wildlife encompasses many species. To think of wildlife as one thing is making the mistake of considering us (humans) and them (all other species) as two separate groups—we are all in this together, we people and other species. Biodiversity goes beyond the collection of animals that we often perceive as wildlife to include all forms of life—plants, animals, microbes, and all the bits of the Earth that support them. Leopold (1949, p. 147) suggested that “...to keep every cog and wheel is the first precaution of intelligent

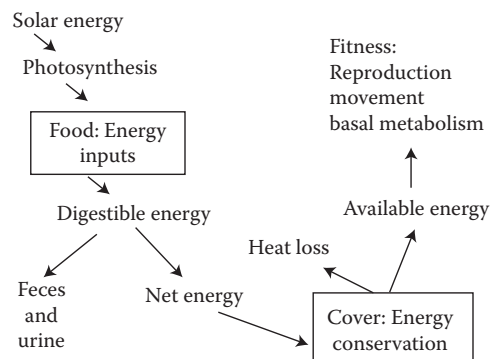
tinkering.” Save the pieces. In our efforts to manage forests to meet wildlife and biodiversity goals, the pieces are the species and the resources are what those species need to survive and reproduce. So let us think about what a species needs as habitat.

Habitat is the place where a species lives. It includes the physical and biological resources necessary to support a self-sustaining population. Each species and each population has its own habitat requirements (Krausman 1999). References to “wildlife habitat” are meaningless, therefore, unless a particular wildlife species is identified because everything is habitat for something. Krausman (1999), Hall et al. (1997), and Garshelis (2000) have made compelling arguments for clarifying the confusion that results from using the term “habitat” to mean the vegetation types or other classes of the environment that are not directly related to a particular species.

Although habitat has been defined in many ways, I define habitat as the set of resources necessary to support a population over space and through time. Hence every species has its own habitat needs, and the term “wildlife habitat” has little real meaning. Further, this definition focuses on populations and not simply individuals. *Populations* are self-sustaining assemblages of individuals of a species over space and through time. Communities, by contrast, are assemblages of populations over space and through time. This definition of habitat is consistent with the approach taken by Hall et al. (1997), but Garshelis (2000) makes the point that quite often foresters and wildlife biologists both will refer to vegetation types or other discrete classes of the environment as habitats. More accurately these are *habitat types* or *cover types* in that some species can be associated with some vegetation types and not with others. But these associations occur only because some or all of the resources needed by the species occur in those types. Consequently, it is important to think about how habitat functions to provide those resources to each species.

## HABITAT FUNCTION

It is useful to think of habitat meeting not only an individual’s needs but also a population’s needs. It is, after all, the population that can be sustained. Individuals, though clearly essential to population maintenance, just come and go in the process. Just as you and I come and go in the process of maintaining a human population. A number of things drive the success of a population, especially energy. Although nutrients, water, and other factors clearly have a role in maintaining the fitness of individuals and populations (e.g., Jones 1992), the lion’s share of the system is driven by energy. A population gains energy from food resources and conserves energy by exploiting cover resources (Figure 1.1).



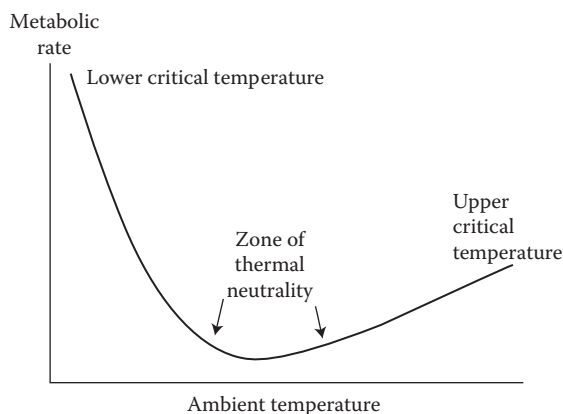
**FIGURE 1.1** The concept of energy flow through individuals to influence individual and population fitness. (Based on Mautz, W.W. 1978. *Big Game of North America, Ecology and Management*. Stackpole Books, Harrisburg, PA, pp. 321–348; adapted from McComb, W.C. 2001. *Wildlife Habitat Relationships in Oregon and Washington*. OSU Press, Corvallis, OR.)

Energy is the currency for population sustainability. Give it more and the population will grow; give it less and the population will decline. When an individual goes energetically bankrupt, he dies. When a population goes energetically bankrupt, it goes extinct.

Food provides the source of energy (and nutrients) for individuals and populations. Food quality matters. Tree species vary in their ability to provide protein and carbohydrates to herbivores. Some parts of plants are more digestible than others and some plant species are more digestible than others (Mautz et al. 1976). It is this digestible energy and the net energy remaining after digestion and metabolism that influences the fitness of individuals and populations. But fitness, the ability to survive and reproduce, is also influenced by cover quality.

It is advantageous for an animal to conserve any energy that it acquires. Mammals and birds maintain a constant body temperature and expend a large amount of energy to maintain that temperature. Cover provides a mechanism for conserving energy. The *thermal neutral zone* is the range of ambient temperatures where an animal has to expend the least amount of energy maintaining a constant body temperature. Thermal cover places the animal closer to the thermal neutral zone. Energy expenditures are minimized in an animal's thermal neutral zone to maintain body temperature (Figure 1.2, e.g., Mautz et al. 1992).

Any departure from the thermal neutral zone results in increased expenditure of energy; so animals often select habitat that reduces climatic extremes. There are upper and lower critical temperatures beyond which exposure for a prolonged period would be lethal. Cover from overheating is especially important to large animals with a low surface-area-to-body-mass ratio because they may find it particularly difficult to release excess heat unless water is available to aid in evaporative cooling. Cover from severe cold is especially important to a species with a high surface-area-to-body-mass ratio (e.g., small birds and mammals). Cover that allows an animal to stay within an acceptable range of temperatures (particularly those that approach the thermal neutral zone) is important to maintaining a positive balance of net energy and hence influences animal fitness. For instance, imagine yourself standing in a field wearing summer clothes in mid-January in Minnesota. Without any measurements, you know that you are expending a significant amount of energy to stay warm. Now imagine you are in a field in Arizona in August at noon. You must expend energy to stay cool and not let your body temperature rise too high (e.g., heat stroke). In either case, moving into a building where the temperature is 18°C (65°F) allows you to spend less energy keeping your body at the appropriate temperature. Refer to Figure 1.2 and plot the metabolic rate for a small mammal or bird at a low temperature and then the metabolic rate for a temperature near the thermal neutral



**FIGURE 1.2** Relationship between metabolic rate and ambient temperature in a hypothetical mammal. (Based on Gordon, M.S. 1972. *Animal Physiology: Principles and Adaptations*, 2nd ed. The MacMillan Co., New York, 591 pp.; adapted from McComb, W.C. 2001. *Wildlife Habitat Relationships in Oregon and Washington*. OSU Press, Corvallis, OR.)

zone. The difference in metabolic rates along the y-axis is an index to the amount of energy that the individual can conserve by staying closer to the thermal neutral zone. For a small animal with a high metabolic rate and high surface area to body mass that conserved energy can mean the difference between life and death on a cold winter night. But there are both behavioral and physiological adaptations that some species have to further conserve energy. Southern flying squirrels (see Appendix 1 for a list of scientific names of all plants and animals used in this book) and some species of cave-dwelling bats often will use communal roosts in winter to collectively maintain a lower surface-area-to-body ratio. Flying squirrels pack many small bodies together to make one bigger, more energetically efficient big body by huddling (Merritt et al. 2001). Other species such as eastern chipmunks hibernate or, as in the case of striped skunks, enter a state of torpor where metabolic rates are reduced and energy is conserved. Black-capped chickadees, a small, 10-g bird that spends winters in very cold climates, will cache food, roost in cavities, and alter their metabolic rates seasonally to cope with temperature extremes (Cooper and Swanson 1994). So the effects of conserving energy through use of thermal cover can be improved even more by these physiological and behavioral mechanisms.

But the relationship portrayed in Figure 1.2 is different for species that do not maintain a constant body temperature. Most reptiles, amphibians, and some nestling birds (birds that have not yet fledged) do not use large amounts of energy to maintain a constant body temperature. They are *ectotherms*—they receive most of their body heat from the surrounding environment, unlike *endotherms* that generate their own body heat. For ectotherms, metabolic rates and food requirements vary as ambient temperature varies. The evolutionary advantage of such an approach is that these ectotherms require less food to survive, but they can be restricted from extreme environments that otherwise would be inhabitable by endotherms (some bird and mammal species). Hence, reptiles and amphibians often use cover to adjust the ambient temperature to allow them to survive, reproduce, and move in places and times when they otherwise would be unable to (Forsman 2000). Consequently, cover is an important component of habitat for these species, to both conserve energy and place them at a temperature where they can be active.

Cover can also refer to the portion of habitat that an animal uses for nesting and escaping from predators. The most significant loss of energy by an animal is conversion of its energy into the energy of its predator! Hiding cover protects an animal from predation. Cottontail rabbits often spend resting hours in dense shrubby cover adjacent to grassy fields and meadows (Bond et al. 2001). The dense shrub cover protects them from predation by red-tailed hawks whose body size and wing spread do not allow them to penetrate dense vegetation. Simple modifications to habitat such as allowing shrubs to proliferate along field edges can lead to increased survival and increased population growth for cottontails in this example.

Nesting cover provides the conditions necessary for raising young—appropriate temperature and protection from predators and competitors. The effectiveness of nest box programs for wood ducks, eastern bluebirds, and other cavity-using species demonstrates that manipulation of the quantity, quality, and availability of nesting cover resources can be an effective management technique (McComb and Lindenmayer 1999). Forest managers can influence habitat for a species by altering food quality, quantity, and/or availability while also altering the quality, quantity, and/or availability of cover. This strategy can lead to drastic changes in habitat quality for the species.

Water is differentially important to animal species. Some species require free water or high humidity (mountain beaver, e.g., have a primitive uretic system) (Schmidt-Neilsen and Pfeiffer 1970). Others species obtain most of their water from their food (e.g., pocket gophers). Some species use water as a form of cover to enhance evaporative cooling (e.g., elk) or to escape predators (e.g., white-tailed deer). Still others such as amphibians require free water or moist environments for reproduction.

The size of habitat is also an important determinant of its suitability for a species. A patch of habitat must be sufficiently large to provide energy inputs and energy conservation features to sustain a population. Habitat may occur in one large unit, but more commonly it is distributed in



patches embedded in other less suitable patches. If these habitat patches are too widely distributed, then the animal expends more energy moving among patches than it receives from those patches. The amount of habitat and its quality and distribution are therefore interrelated. Increasing any one or all of these attributes of habitat increases the net energy available to animals that use this energy to maintain body temperature, move to food and cover, and reproduce.

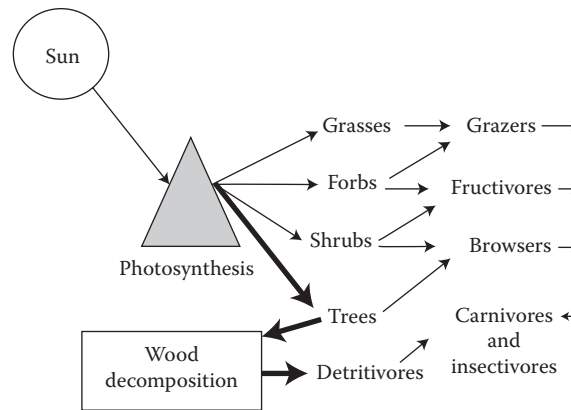
## HABITAT FOR HUMANS

When trying to understand the concept of habitat, it may be helpful to think of your habitat. You are an individual of a species, a mammal that maintains a constant body temperature, and you have your own set of resources that you need to survive and reproduce. Think about your food requirements. You eat a certain number of calories per day and this energy is converted to adenosine-tri-phosphate (ATP) or stored for future use. You use this energy to maintain your body temperature and support your physical being, to move throughout the day from place to place, to raise your children, and to buy or harvest food. Generally human food is rather digestible and high in energy, though in some societies digestible energy can limit not only human health but also survival.

Humans, like most other mammals, also use cover. We have homes where we attempt to keep the ambient temperature as close to our thermal neutral zone as possible. We raise our children there and we use these homes at times as a place of refuge during inclement weather or catastrophic disturbances. We need clean water in adequate quantities so as not to become dehydrated. All of these things must be in close proximity so that we do not spend more energy-acquiring resources than we receive. Substitute nearly any other animal species and we can similarly define the food, cover, and water requirements for that species as we can for our own species. But for each species those requirements differ. No two species are likely to coexist and have the same habitat requirements for very long if resources are limited and species are competing for them. There are instances however when predation pressure on two prey species can allow them to coexist using the same resources, but if predation pressure changes or resource availability changes, then one species will likely outcompete the other (McPeck 1998). Because we humans are so adaptable and because we usurp energy and other resources that could be used by other species, we have a profound effect on the number of species and individuals with which we share this planet. "Saving all the pieces" comes at a price. And it is not a price that society is willing to pay in all instances. There are 7 billion people, which represents 455 million tons of human biomass that must be supported daily on this planet. And two more people are added every second. Saving all the pieces may be a noble goal, but human self-preservation and preservation of life styles can trump that goal quickly, unless we give more thought to our own habitat needs within the context of the needs of the other species with which we share this planet.

## FORESTS AS HABITAT

How we manage forests to partition energy among various forms of life is the essence of the challenge facing foresters and wildlife biologists. It is a challenge because the rate of primary production is fixed over large areas and times, because there is a solar constant, and because climate changes are relatively slow (the current climate change crisis notwithstanding). Further, although herbivores in forests exist in a sea of plant energy, little of it is available for those herbivores to use. Food quantity is often not as important as food quality in a forest (Mautz 1978). Most of the energy in a forest is in cellulose, the wood that society demands, and for many species this wood is not very digestible. Animals can only use the *digestible energy* in food (Figure 1.1); so indigestible portions of food (e.g., cellulose, lignin, chitin, or bones) or compounds in the plants that inhibit digestion (tannins and other phenols) reduce food quality (Robbins et al. 1991). These indigestible portions of a forest can become available as energy to many species if they are made available through decomposing organisms (Figure 1.3). Without the decomposers being available as digestible food for other



**FIGURE 1.3** Energy pathways through a forest. Arrow sizes reflect relative differences in energy availability. (Adapted from Harris, L.D., 1984. *The Fragmented Forest: Island Biogeography Theory and the Preservation of Biotic Diversity*. University of Chicago Press, Chicago, IL, 211 pp.)

species in forests, most of the energy would go unused. Indeed, without decomposers, animal diversity would be reduced to a relatively few species specialized to eating twigs, leaves, fruits, or the nonwoody portions of the forest. The *decomposition pathway* is quite important in maintaining a diverse animal community in many forests. Further, for those species that rely on plant fruits for food, providing plants with adequate sunlight and water to grow and produce flowers and fruits is a key to meeting these food needs.

## HISTORICAL APPROACHES TO MANAGING FORESTS AS HABITAT

Management of forests as habitat for wildlife has been conducted for centuries in many cultures. Native Americans used fire to at least move animals during hunts if not to provide better forage for them (Boag 1992). But it was not until Leopold completed his *Game and Fish Handbook* for the Forest Service in 1915 and the subsequent publication of *Game Management* (Leopold 1933) that active management efforts began on many public and some private forest lands to promote selected species. There have been two common approaches to management of forests as habitat for species: management of individual species and management for biodiversity. Only recently has the focus of land managers shifted from utilitarian goals to protection of rare species to conservation of entire ecosystems. Indeed, the evolution of wildlife management as a discipline was driven largely by the philosophies underpinning forest management and both disciplines have evolved in parallel with regard to the focus of their management efforts.

Both foresters and wildlife biologists viewed that there is a way to manage species to provide the desired plants or animals in a non-declining, sustained yield manner. Harvest scheduling works for black oaks and black bears—we just need to provide the correct habitat, manage the density of the organisms to provide sufficient resources for the remaining individuals, and the system can go on and on and on. Right? Well, maybe, but by taking this approach there is the risk of losing other species that might be of value to society. Managers began to focus management on indicator species (species that are surrogates for other species) or guilds of species (groups of species with similar habitat needs) in order to meet the needs for many other species. These approaches also produced problems for exactly the reasons described in the previous section: each species has its own habitat requirements and each will respond differently to management activities (Mannan et al. 1984).

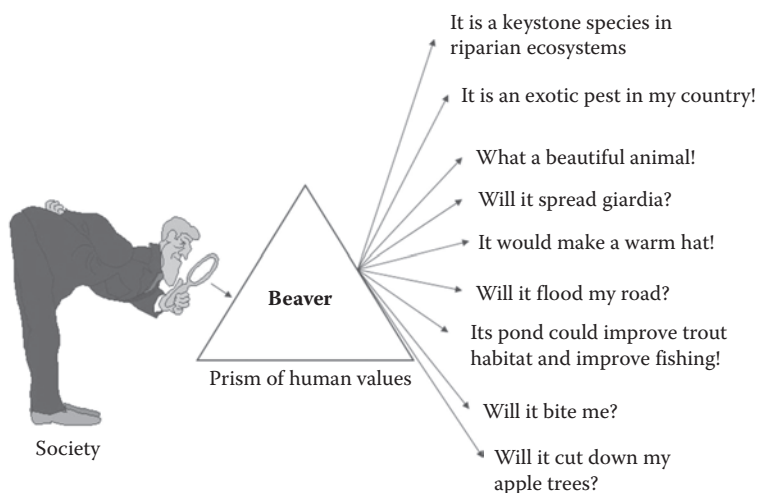
But some approach is needed to produce the desired species of plants and animals while minimizing the risk of losing some pieces that we may need later. The most recent approach to managing to meet societal goals of aesthetics, game, biodiversity, recreation, and timber fall within the realm

of *ecosystem management*—an approach designed to minimize risk to species and maximize the likelihood that the approach will be sustainable (Meffe et al. 2002). One basis for this approach recognizes that forest disturbances change the abundance of individuals in many populations, and those changes also influence the composition of plant and animal communities. Before technologically advanced humans began managing forests, natural disturbances caused the localized extinction of some species and opportunities for recolonization by others. Communities changed as forests regrew following these disturbances. Species tended to be adapted to the range of conditions that occurred under these natural disturbance conditions. Understanding how species respond to these conditions and how management might replicate or depart from those conditions can be useful in understanding the effects of management on a suite of other resources (Landres et al. 1999). In addition, consider that the management of an individual species has consequences for other species in its community. Forest disturbances that benefit black-tailed deer, for example, probably would benefit creeping voles and orange-crowned warblers, two species found in early successional forests, but not Douglas squirrels or pileated woodpeckers, two late-successional forest species.

Natural disturbances such as fires, insect defoliation, and hurricanes notwithstanding, vegetation management by forest-land managers is probably the greatest factor influencing the abundance and distribution of animals in our forests today. By understanding the concepts of habitat function, population change, and habitat patterning, managers can make decisions that can find the appropriate societal balance among commodities, species, and ecosystems.

### WHY MANAGE HABITAT?

We manage habitat for various reasons such as personal goals, corporate objectives, and legal requirements. Policies in the United States, such as the Endangered Species Act and National Forest Management Act, require people in various agencies to manage habitat. Why do we have these policies? Why should we spend time and money managing habitat for species that occur in our forests? Quite simply we do or do not manage habitat because society either cares about these resources or they do not, respectively. Wild animals are public resources that occur on both public and private lands. If society placed no value on a species or group of species, then we would not manage their habitat. Values that society places on animals evolve over time and from culture to culture. Take the beaver for example (Figure 1.4). Clearly, there are many reasons to manage habitat for beaver,



**FIGURE 1.4** Society views natural resources through a prism of values. (Based on discussions with R.M. Muth, University of Massachusetts, Amherst, MA.)

though some segments of society would like to ensure that there are fewer animals and some would like more of them. To complicate matters further, oftentimes people with differing values are neighbors and the beavers do not care where the property line falls!

These values placed on a resource usually change slowly as other aspects of our society change. In some cultures, the species may be viewed as an important economic or otherwise subsistence resource that would be harvested and used for survival. As society becomes less reliant on or less engaged with native species, people may begin to place greater intrinsic value on them or fear them because they are unknown. Finally, the relative importance of a species may change markedly and rapidly as unexpected events occur, leading to rapid changes in societal values that have unanticipated impacts on our ability to manage natural resources. On September 11, 2001, on a west-bound United flight from Hartford to Denver, I was somewhere over Lake Erie at 9:15 a.m. eastern time. I was fortunate to have spent a week in Chicago rather than other alternatives that morning. Those events changed our society's priorities suddenly. They certainly changed mine. Although it did not necessarily diminish the importance that people placed on environmental values, it raised human safety and welfare to a much higher priority than previous to that event. Human and financial resources once used to provide natural resource values for our society were diverted to these higher priorities. We saw a similar response following hurricanes Rita and Katrina in 2005. One can argue that the political decisions were made at the time to achieve ideological as well as humanitarian goals, but changes did occur that impacted many aspects of our ability to meet natural resource and environmental quality goals for society. We are not alone in these struggles.

Overwhelming economic pressures face many parts of the world. Huge loans have been provided by the International Monetary Fund (IMF) to countries such as Argentina. These debts to the IMF, combined with the overwhelming pressure to ensure that people survive on a limited and often declining natural resource base, significantly limit options to maintain environmental values. Forest reserves, popular approaches to biodiversity conservation in wealthy countries and recognized as important by developing countries, are usually an untenable option in much of the world unless significant foreign monetary support is provided. Even so, reserves become only one approach to protecting biodiversity. Indeed, the majority of the land and water resources that could support some components of natural systems are not within reserves and never will be. Social pressures force managers to consider options that are both economically feasible and ecologically sound. If large tracts of forest are managed in a manner that considers the structure and function of the habitat for valued species, while still allowing some economic value to the landowner, then there is a greater likelihood that it will remain as a forest or field. Once the value of a forest falls below that of other land uses, then there is a risk of conversion to a new use (e.g., industrial agriculture, grazing, or housing).

If the forest is managed to consider structure and function to valued organisms, then it may support these species, which otherwise would be found primarily in reserves, thereby complementing effectiveness of the reserve system. For instance, actions that maintain a forest rather than a pasture that likely would be overgrazed will decrease the probability that the site would be lost to desertification in the dry tropics. Active forest management to achieve multiple objectives such as grazing lands (Figure 1.5a), wood products (Figure 1.5b), and habitat for valued wildlife (Figure 1.5c) may be one step toward maintaining economic and ecological values.

We do not know with certainty how to manage all or even most forests to achieve multiple values. But we do understand vegetation dynamics, disturbance ecology, habitat selection, and population dynamics as well as the influence of local, regional, and global economies, cultural mores, and social value systems. If we use this information in a thoughtful manner, then we should be able to develop reasonable management plans to achieve multiple objectives. However, we will need to monitor the effectiveness of the plans to ensure that we are meeting our goals. This *adaptive management* process (listen, synthesize, plan, implement, monitor, learn, listen, etc.) is an integral part of habitat management (Baskerville 1985). My objective is to provide the concepts, processes, and tools that you can use to develop resource management plans that will help you achieve landowner goals now and into the future.



**FIGURE 1.5** In northern Argentina an effort is being made at Salta Forestal to manage native dry tropical forests for grazing (a), wood products (b), and habitat for various species of wildlife (in this case, one of several species of armadillos found in the region) (c) in a manner that helps to support local economies while conserving biodiversity.

## CASE STUDY: THE FORESTS OF BRITISH COLUMBIA

Society directs the way that forests are managed. Over the past 100 years, we have seen marked changes in the principal values associated with forests in nearly every technologically advanced culture. Changes in values and beliefs associated with forest management have followed somewhat parallel courses in the United States, Canada, New Zealand, and Australia, though at different times. The following example is extracted from a paper by Kremaster and Bunnell (1998) and reflects those changing values associated with forest management in British Columbia (BC). Forests in BC extend from the subalpine region of the Canadian Rockies to the boreal forest in the north and south through temperate rainforests along the Pacific Coast to pine forests of the Interior. BC's forests cover an area twice the size of all of the New England states and New York State combined. The forest products industry has been and continues to be very important to the provincial and national economy.

The remaining text in this case study is paraphrased from Kremaster and Bunnell (1998) to illustrate these social changes. "Until the 1940s, forests in British Columbia were seen as inexhaustible suppliers of timber. It was not until after World War II that attention is focused on sustainable forestry, and the public began to expect foresters to grow trees and continuously provide timber over the long term. Major concerns at that time were fire protection and the decline in timber volume as old growth was converted to managed stands. Foresters were expected to manage economically valuable tree species to ages well short of the potential life span of the species.

Wildlife concerns in the first half of the 1900s focused primarily on game species and fish. During the late 1960s and in the 1970s, foresters began to embrace the paradigm of 'multiple use' or managing for many values on each piece of land. In BC, legislation that was passed recognized that many resources were provided by forests and all were important to its citizens. This idea of multiple use remained in place through the mid-1980s. Wildlife concerns expanded to include nongame species, but attention was still focused mainly on game and fish species. Forest guidelines protected unstable soil and some streams, but growing trees and protecting live trees from insect and fire were the main concerns reflected in management guidelines.

During the late 1960s and in the 1970s, silvicultural systems focused on producing and recovering the maximum amount of economic fiber from the forest. Along the Pacific coast, clearcutting was the dominant practice, even as interest in forests expanded beyond trees alone (Figure 1.6).





**FIGURE 1.6** Example of timber harvest in a watershed commonly seen in BC and Alaska in the 1970s.

Typical rotations (the interval between harvests) increased the area of early stages of natural succession and truncated succession well short of tree ages in historical forests. Research began to document changes to wildlife species assemblages associated with various stages of forest development. It was recognized that managed and unmanaged forests change over time in response to natural and human-induced disturbances, and that different vertebrate species were more abundant in different stages of forest development.

Products and values desired from forests have continued to change. Although many people still embrace the notion of multiple use, there is general realization that a given piece of land, unless enormously large, cannot provide all desired, and sometimes competing resources. The current scientific perception is that all parts of the ecosystem are linked and activities that affect one aspect of the system will likely affect others. The current management focus is on managing ecosystems, sustaining biodiversity, and maintaining forests more like historical ones. These approaches are in response to current public concerns, which include loss of species, productivity, future options, and economic opportunities. Sustaining biodiversity has become a fundamental goal. Forest practices and policies have continued to change. Scientists and managers have translated public concerns and their own improved understanding of forest systems into new approaches. Social concerns as well as long time periods and large areas were incorporated into the concept of ‘ecosystem management.’ Concerns about losing species and productivity impelled policy makers to create legislation (e.g., BC Forest Practices Code [FPC]), integrate recent scientific knowledge (e.g., Scientific Panel for Sustainable Forest Practices in Clayoquot Sound [CSP]), and initiate new approaches to planning (e.g., Innovative Forestry Practices Agreements). Legislation and planning try to include recent knowledge but policy continues to precede reliable knowledge.

Legislation and planning processes have been enacted to translate public and scientific concerns into different forest practices. With the advent of the FPC in BC, foresters have a legislated responsibility for sustaining biological diversity when compared with other natural resource managers. Regulations governing agricultural and urban development do not reflect the same concern for maintaining ecosystems, even though these activities have had a greater impact on biological diversity.

The FPC has encouraged less clear-cutting and promoted a range of retention during even and uneven-aged management. Retention of older stages of forest development, maintaining connectivity, and protecting buffers around several stream classes are now legislated. The levers used in FPC regulations reflect forest features that we believe are related to biodiversity and ecosystem productivity. Managers are limited to practical approaches—for example, remove snags or let them stand, and leave live trees to grow old or harvest them at an economic rotation. As a result, these levers include stand structures such as snags, downed wood, species mixtures, and large old trees, and forest-level measures such as seral stage distribution, amount of edge, forest interior, patch size, and corridors. Fortunately, these attributes link to public concerns and to species richness. To evaluate

the effectiveness of these approaches, forest managers need to know ‘How much is enough of specific forest elements?’, ‘What spatial patterns are important?’, and ‘Which species are likely to need more individualized approaches?’”

This example from BC has been repeated in many parts of the United States, Canada, New Zealand, Australia, and elsewhere. And now in BC, as well as in many other parts of the world, climates are changing as are increases in insect defoliators and risk of fire. Saving all the pieces may take considerable effort and active management on at least part of the landscape if species are to persist in landscapes structured by recent timber harvest and facing new environmental conditions. Clearly, social values have shifted to concerns regarding biodiversity conservation, and the management concepts and approaches used by biologists and foresters should reflect those concerns.

## SUMMARY

Habitat is the set of resources necessary to support a population over space and through time. It is a species-specific concept and is different from a habitat type or cover type, which is often a classification of the environment that may or may not be related to the resources necessary to maintain fitness or an individual, a population, or a species. Food represents the inputs of energy and nutrients. Digestible energy is not as available to many vertebrates in forests unless indigestible cellulose is broken down through decomposition. Energy is conserved by a species use of cover. Energy expenditure to maintain a constant body temperature is minimized in the thermal neutral zone; thermal cover allows an animal to provide an ambient temperature closer to the thermal neutral zone. Escape and nesting cover protect an animal from risks of predation or competition.

Habitat was historically managed to increase populations of game species. The focus for habitat management has changed from primarily utilitarian in the mid-1940s in the United States to considering a broad suite of organisms, including endangered species and species enjoyed for their aesthetics. Biodiversity conservation has become the most recent social goal in forest management. Indeed, we manage forests to achieve human needs and desires. Wood products are clearly one reason for managing forests, but providing habitat for desired species, clean water, recreation, and rangeland resources is also paramount. The challenge to private forest landowners is that they are charged with meeting societal goals for a public resource (wildlife) on their private lands. To be effective at meeting both individual and public goals for forests, foresters must work collaboratively with wildlife biologists, conservation biologists, and other resource professionals to develop innovative approaches to forest management. That is the focus of this book.

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## 2 Humans as a Forest-Dependent Species

Products and services from ecosystems are inherent to sustaining life on this planet. Forest ecosystems, in particular, provide a suite of products and services that sustain life for humans and other species. In addition to the obvious products such as wood for building houses, and firewood for heating homes, nontimber forest products are also an economic commodity that benefits many communities. Mushrooms, greens, biochemicals, charcoal, fruits, roots, and tree bark (e.g., cork) all contribute to local economies. In addition, most metropolitan areas of the world receive their drinking water from watersheds that have forested headwaters. Carbon sequestration, shade, and game species harvested for subsistence food are also derived from forests by communities around the world. And the intrinsic beauty of forests attracts hikers, artists, musicians, philosophers, and many others to seek inspiration and solace in forests. And, of course, habitat for many species of biodiversity, including humans, can be found in forested ecosystems.

All of these ecosystem services are valued and are reasons why many people in our societies wish to ensure that forests are protected or managed sustainably. But sustainability is a tricky thing to define. It reflects a suite of societal values such as those listed earlier, and it necessarily implies a time frame. For how long will we be able to sustain these values? Species go extinct. Climate changes. Forests burn and regrow. And throughout these disturbances and other changes that occur in forests, habitat for species is destroyed and regrows, or it may be eliminated or appear in a place where it never occurred before. The one thing that is constant about our environment is change, and the way that species, including humans, have persisted in the face of those changes is through maintaining the genetic diversity and behavioral flexibility to allow some members of each species to persist.

### ECOLOGICAL RESTORATION AND ECOSYSTEM SERVICES

Because of the concern on the part of some people in our societies that humans have degraded some ecosystems in their effort to secure resources, there is a growing interest in ecological restoration. Ecological restoration is the process of managing a system to allow it to provide a certain suite of ecosystem services and products that may have been lost as a result of human use or activities. Degradation is a process that leads to a condition that is less productive relative to a reference or desirable condition. Oftentimes restoration can mean addressing reestablishment of desired plant or animal species and the processes that support them. Restoration can be a difficult if not impossible task in some systems if the inherent potential of the site has been altered markedly, such as the desertification of forest lands in sub-Saharan Africa. Restoration of other sites such as reclamation of surface mines or restoration of urban brownfields can take considerable time, effort, and money and still not fully reflect the ecosystem that was present prior to the impacts of mountaintop removal or urban development. Benayas et al. (2009) analyzed 89 restoration sites from around the world and reported that ecological restoration efforts had increased provision of biodiversity and ecosystem services by 44% and 25%, respectively. Nonetheless, Benayas et al. (2009) found that ecosystem services and biodiversity indicators were both lower in restored ecosystems than in intact reference conditions, and that indicators of biodiversity protection and ecosystem services were positively associated with one another. On the basis of their work, efforts at restoring ecosystems will not

only benefit conservation of biodiversity but also improve the suite of ecosystem services available to humans. The Millennium Ecosystem Assessment Report (2005) provides compelling evidence for the linkage between functional ecosystems and human well-being. Indeed, Butler and Oluoch-Kosura (2006) indicated that functioning societies have the capability to protect or enhance ecosystem services, but societies with impaired well-being are often related to a decline in ecosystem services. They hasten to point out that the socio-political structure of the human population inhabiting an ecosystem is a key to realizing both a healthy ecosystem and a healthy human population.

Enhancing the suite of ecosystems services through acquisition of reserves or through active ecological restoration may be initiated by some societies simply as a way of improving human well-being, but in so doing they also may benefit conservation of biodiversity. Focusing on restoration of ecosystem services may be a more financially viable approach to conserving biodiversity than seeking funding for biodiversity conservation alone. Goldman et al. (2008) reported that ecosystem services projects attract, on average, more than four times as much funding as biodiversity conservation projects. Projects to enhance or maintain ecosystems services also are more likely to include actively managed forests and farms and the people who live there. They demonstrated that projects to maintain or enhance ecosystem services also increase opportunities for conservation of biodiversity while meeting the needs of a diverse set of funders. The success of such projects is often poorly known however because results are rarely monitored. Consequently, at least some biodiversity conservation goals may be met by focusing on ecosystems services. Examples of ecosystem services include but are not limited to (Binning et al. 2001):

- Pollination
- Fulfillment of people's cultural, spiritual, and intellectual needs
- Regulation of climate
- Insect pest control
- Carbon sequestration
- Maintenance and provision of genetic resources
- Maintenance and regeneration of habitat
- Provision of shade and shelter
- Prevention of soil erosion
- Maintenance of soil fertility
- Maintenance of soil health
- Maintenance of healthy waterways
- Water filtration
- Regulation of river flows and groundwater levels
- Waste absorption and breakdown

Because people value services such as these, Costanza et al. (1997) estimated the current economic value of 17 ecosystem services for 16 biomes to be between \$16 and \$54 trillion/year (the average was \$33 trillion/year). The global GNP is approximately \$18 trillion/year. Clearly, ecosystem services are significant contributors to the global economy though they are undervalued on the global market. De Groot et al. (2002) provided a framework for valuing ecosystem services, and recently markets have developed for these services. The most common market available for ecosystem services to date are mitigation approaches whereby impacts (usually development) are mitigated by purchasing and protecting the ecosystems services that would have been present on the impacted site prior to development. Wetlands are a common focus of such efforts. Yet there have been attempts to develop markets for other services. Kroeger and Casey (2007) provided an analysis of ecosystem services markets as they pertain largely to agricultural lands, but the concepts are also applicable to many forest lands. Indeed carbon markets have developed in response to one type of ecosystem service that has been identified as an approach to slowing the rate of climate change (Sedjo and Marland 2003). New markets for these and other ecosystem services are being developed

and, if successful, could provide an economic incentive for private landowners to continue to provide, or indeed enhance, the ecosystem services that many in society have long accepted for free.

## **SOCIAL VALUES ASSOCIATED WITH FORESTS AND WILDLIFE**

Shindler and Cramer (1999) described the changes in social values associated with forests over the past century or so, limiting their discussion largely to values associated with cultures derived from European societies. The evolution of values from utilitarian and more often rural to protectionist and more often urban is one that we have seen creep across the continents of North and South America and Australia following colonization by European cultures. Changes such as these can often produce conflict that can result in a stalemate in the decision-making process, or worse. But the changes that we have seen in European-dominated cultures are only a portion of the full spectrum of values people associate with our environment. Native Americans, First Nations Peoples, and Aborigines possess traditional ecological knowledge (TEK), previously dismissed or ignored by most western societies, that has only recently been accepted and embraced by western-dominated cultures. By including TEK in our ecological value system the spectrum of values and philosophies is expanded and as a result so are our approaches to management, protection, and restoration of natural resources. And as nations change in their cultures, with growing ethnic and cultural diversity (e.g., Hispanic, African American, Asian and other populations in the United States), new values are incorporated into our current cultures and the spectrum of values increase further.

But incorporation of new values into our societies may oftentimes result in conflict that, without careful consultation and introspection, can leave some groups of individuals feeling threatened and marginalized. Trust can be eroded and resolution to conflict can seem difficult if not impossible. Such circumstances are the “wicked problems” described by Shindler and Cramer (1999). Wicked problems are those where conflicting values result in a lack of trust between or among competing values. There are many examples: Timber vs. Spotted owls, to drill for oil or not in the Arctic National Wildlife Refuge, clearing Amazonian rain forests for agriculture, to name a few. Resolution of these conflicts can take years and require rebuilding of trust—trust is something easily lost in a conflict scenario and something that is difficult to regain. Rebuilding trust will require those involved in disagreement to listen to each other’s view points, respect that there are differences and work together to find an acceptable resolution for all involved, if indeed that is even possible. An impartial mediator is often required to initiate the process of resolving these wicked problems.

## **ENVIRONMENTAL ETHICS**

Ethics reflect our values and they guide our behavior. Ethics guide interactions among us and reflect our mutual respect for one another. Ethics can also guide our use of ecosystems and reflect our respect for nature (Taylor 1981). Brennan (2002) provides an excellent overview of the growth of environmental ethics as a discipline that addresses a number of facets of human philosophies toward nature and the natural world. People in different cultures treat the resources on which they rely in a variety of ways. Historically in the United States and much of Europe, we adopted a utilitarian approach to natural systems in which exploitation of resources was common and there was an inherent belief that humans are superior to other organisms. In other cultures and in more recently developed philosophies such as Deep Ecology, humans are viewed as coequal with other organisms and are part of a system in which the various parts and processes have intrinsic value in their own right. Hence, environmental ethics as a discipline has grown to address some of the ethical dilemmas that managers, consumers, and those who appreciate nature find themselves facing as individuals and cultures use resources. A person can be remarkably adept at rationalizing individual decisions because he or she is only one individual in a very large population (e.g., what difference does it make if I turn off my lights if no one else does?) or because the scale of the problem is so huge as to seem incomprehensible to any one individual (e.g., climate change).

Indeed in issues of climate change, biodiversity conservation, spread of infectious diseases, and pollution all can seem unrelated to the day-to-day lives of any one individual. The issue does not seem immediate and it does not seem personal. Compare the U.S. societal response to Hurricane Katrina, the tornado in Joplin, Missouri, the earthquake in Haiti, the tsunami in Japan, or the 9/11 attacks in New York and Washington DC, where there was an immediate societal response to offer aid and assistance. Although you may not have been affected directly, the circumstance was immediate and people could easily empathize with those affected. The increase of  $1^{\circ}\text{C}$  over a decade is not immediate and barely noticeable to most people, neither is the loss of another species, nor the additional kilogram of nitrate fertilizer entering a river, nor the slow but steady spread of a disease such as HIV-AIDS in another country. Ethically all of these issues are ones that do affect human societies, but our reaction to these crises is slow and careful, if there is any at all. Why? Why should we not be as mobilized to save the next species slipping to the brink of extinction as we are to save the next person affected by a natural or human-caused disaster? Maybe you are motivated and do want to help immediately, but if you are and you are in a western civilization, then you are in a minority of society and can feel powerless to enact change. We can easily find ourselves in an ethical dilemma of wanting to take action but unwilling to challenge the social norms to realize the change that you would like to see.

For many people in our western societies, people who take immediate action to address these pressing environmental issues are seen as reactionaries, extremists, or activists. But it is exactly these people, taking more extreme actions on both sides of a polarized issue that pull or push the social agenda in one direction or another. While science and education have contributed significantly to providing the basis for protecting large areas of old-growth forest, tree sitters and peaceful demonstrators have done their part as well (Figure 2.1). Their actions are immediate, they are personal, and they cause a reaction in others. They often evoke a more significant reaction than would a list of the 1500 species of organisms known to be highly associated with old-growth forests. Others write books, essays, poems, and songs to raise awareness and make it personal: Rachel Carson's book *Silent Spring*, Kathleen Dean Moore's essays in *Moral Ground*, Mary Oliver's poem *Sleeping in the Forest*, Natalie Merchant's song *Where I Go*. Each of these works and others like them make the elements of the natural world personal and immediate. These writers, singers, and tree sitters are the "actors" in a social debate about how we treat our planet and the resources that it provides, and they do have an effect on public opinion, policies, and management actions. There are also "do-ers" in the debate. These are individuals who make decisions about how to manage their own lands, public lands, or vote on policies that affect all of us. They set examples for others to follow, make decisions that they feel is a correct action, and freely share their information and



**FIGURE 2.1** A forest activist protects a tree from being cut. (Photo by Reed Wilson. With permission.)

approaches with others. The “actors” and the “do-ers” collectively push and pull social opinion in new directions that at times result in new policies, collective decisions, and social awareness that affects each of us.

## ECOLOGICAL PSYCHOLOGY

Psychology, the science of how organisms behave, is often cast in the realm of human behavior in terms of interpersonal interactions or a person’s role within a group or relative to a social norm. But people have behaviors that reflect their interactions with the natural world as well. Koger and Winter (2010) addressed these issues in depth in their book, *The Psychology of Environmental Problems*. Because of the long evolutionary association that humans have had with other species, and because we indeed see aspects of ourselves in other organisms, issues that address the welfare of wildlife can elicit strong emotional responses among many people. Indeed, Koger and Winter (2010, p. 314) stated that the, “...connection with wildlife draws energy from the deepest core of human feeling.” The degree to which people are passionate about an issue dealing with wildlife conservation or protection is often influenced by an individual’s direct connection to nature. Louv (2008) coined the phrase “Nature Deficit Disorder” to reflect the potential implications of disconnecting individuals, especially people in their formative years, from the natural world. People who have intense experiences, or peak experiences, that ingrain an appreciation for the beauty of nature are those most likely to act on their feelings regarding use of or protection of nature. Whereas information is an important ingredient to individual and collective decision making, it may be necessary but not sufficient. Emotions, spirituality, and deeply held convictions are often more likely to cause an individual or group to act in a certain way than information alone. Indeed as Koger and Winter (2010) point out, the common ground shared by people with disparate views on an environmental issue is centered on each individual’s connection with a place and how it is used. Having a group of people in conflict agree that each cares for a place can be the most important first step in building trust and truly listening to one another as they move on a path toward reconciliation.

## PUBLIC RESOURCES ON PRIVATE LANDS

In the United States, as well as in other countries colonized by England, wildlife species are public resources that often occur on private land. When the public resource is adversely affected or when the public resource adversely affects the private landowner, then a conflict often ensues. These types of issues have been repeated for years. An endangered species is found on a private timber company’s land preventing a planned timber harvest. A black bear population develops a taste for the inner bark of rapidly growing trees and kills trees before they can be harvested. Who is responsible for addressing these problems? Should society expect the landowner to assume the financial burden associated with these issues? In many cases, the expectation is that the private landowner bears the responsibility for providing habitat for animals and use of those animals is controlled largely by the landowner allowing access to her or his land (through trespass laws). But if a landowner is prohibited from managing her land because of the presence of an endangered species and is required to leave economically valuable structures for habitat, structures that increase management costs, or controlling damage to property, in most cases there is not an avenue for compensation for that landowner. Some landowners adopt a stewardship philosophy or a land ethic so that in those cases, conflicts may be minimized. But for many private landowners, private property rights take precedence over serving a public expectation. And in some states in the United States, landowners are compensated if a wolf kills a sheep or cow. But because of the strong emotional feelings of many in society toward wildlife, the strong sense of private property rights, and the inconsistent policies for financial compensation of bearing the financial burden of providing habitat, continued conflicts are inevitable.



## CASE STUDY: ENVIRONMENTAL ACTIVISM AND EFFECTS ON HABITAT

There are countless examples of environmentalists taking a stand against what they perceive as a threat to forests and the ecosystem services that they provide: Tree sitters in the redwoods of California, lawsuits filed against the U.S. Forest Service over intensive forest management practices in Montana, and demonstrations against timber cutting in the Karri forests of southwestern Australia. All had an influence on habitat. In many cases, where environmental activists have been successful, habitat was retained for species associated with late-successional forests, at times at the expense of providing early successional conditions suitable for other species. One example in Thailand dealt with the opposition to loss of forest in several large reserve areas in the face of flooding caused by a proposal to build a large dam that would provide electricity, protect downstream areas from flooding, and provide water for irrigation (Rigg 1991). The Nam Choan Dam was proposed for the Kwae Yai River in western Thailand. Although the project had considerable political and industry support, local public pressure by grass roots activists caused the project to be postponed indefinitely. The opposition to the construction of the Nam Choan Dam was focused on the flooding of large areas of forest in the Thung Yai and Huai Kha Khaeng Wildlife Sanctuaries (Rigg 1991). The resulting reservoir would have formed a barrier separating parts of the two parks and concerns were also raised with regard to increased access to the area by poachers and developers. In this case, and others in Southeast Asia, the success of the environmentalists in opposing the construction of the dam was not based as much on empirical data as the ability of the environmentalists to influence public opinion (Rigg 1991). Initial environmental movements were largely driven by concerns raised from educated urbanites who saw environmental deterioration in the areas outside the cities. Recently, the base for environmental opposition to several economic development proposals has included individuals from a wide range of economic classes and localities in the country. In 1989, environmental pressures on politicians led to a prohibition of logging in the country following flooding, which had been connected to deforestation (Rigg 1991).

So, were the decisions to prohibit logging and to stop the development of the Nam Choan Dam correct and in the best interest of the people of Thailand? Did the will of the environmental community lead to the correct outcome for the country as a whole? Answers to those questions, and similar questions regarding protection of old-growth Douglas-fir and redwood forests in the Pacific west, or karri trees in Australia are not at all clear. But they do illustrate the effect that environmental activists have had in influencing the forests, their structure, composition, and dynamics in many areas of the world. Cultural values and associated ethics, individual and collective behaviors, and the actions of individuals in environmental movements have resulted in changes in forest policy and management. It seems clear that continued demands for resources by individuals with utilitarian values will remain pitted against those with more biocentric values so long as the demand for resources continues to increase. The skills and involvement of individuals adept at conflict resolution and mediation will be increasingly necessary to ensure that acceptable solutions are reached by both groups working together to care for the planet.

## SUMMARY

Forest ecosystems provide products and services that sustain life for humans and other species. Some ecosystems have become degraded and no longer provide the suite of services and goods that societies require. Ecological restoration is the process of managing a system to allow it to provide a certain suite of ecosystem services and products that may have been lost due to human use or activities. Ecological restoration efforts can also provide benefits for conservation of biodiversity as well as ecosystem services. Societies and their needs for ecosystem services change over time. Incorporation of new values into a society may oftentimes result in conflict. Individuals that feel passionate about protecting or restoring ecosystems and the products and services that they provide often are the outspoken “actors” in a social debate about how we treat natural resources. Others

make decisions about how to manage their lands and are the quiet “do-ers” who set examples for others.

When wildlife, a public resource, is adversely affected or when the public resource adversely affects the private landowner, then a conflict often ensues. Resolution to conflicts may require objective information, but when it comes to making decisions, objective information may be necessary but it is not sufficient. Emotions, spirituality, and deeply held convictions are often more likely to cause an individual or group to act in a certain way than information alone.

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# 3 Vertebrate Habitat Selection

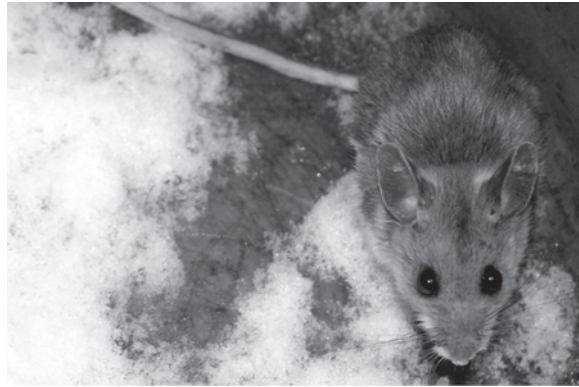
Managing forests to produce a desirable mix of forest resources, including timber products and wildlife species, requires an understanding of how animals respond to habitat in forests. Habitat provided within and among *stands* (units of homogeneous forest vegetation used as the basis for management) over a *landscape* (a complex mosaic of interacting patches including forest stands) can have significant effects on the abundance and distribution of animal species. Management strategies aimed at long-term population change are most likely to succeed if they alter habitat quantity, quality, and/or distribution. Knowing how species select habitat can provide clues as to what habitat elements to provide. *Habitat elements* are those bits and pieces of a forest important to many species, such as vertical structure, dead wood, tree size, plant species, and forage. We will cover these in more detail in Chapter 4.

*Habitat selection* is a set of complex behaviors that a species has developed among individuals in a population to ensure fitness. These behaviors are often innate and have allowed populations to persist under the variable conditions that occur over time in forests (Wecker 1963). These behaviors have also allowed each species to select habitat in a manner that allows it to reduce competition for resources with other species. So the evolutionary selection pressures on each species, both abiotic and biotic, have led species to develop different strategies for survival that link habitat selection and population dynamics. Some species are *habitat generalists*, and can use a broad suite of food and cover resources. These species tend to be highly adaptable and occur in a wide variety of environmental conditions. The deer mouse is a species that exemplifies this strategy in that it can be found in all stages of forest development and in many *forest types* across the United States. Deer mice have high reproductive rates and can demographically take advantage of abrupt increases in food and cover resources (Figure 3.1). This species is also a primary food resource for many forest predators. Hence, providing habitat for deer mice in a forest is quite easy, although they do tend to be more abundant in early successional forests than in late successional forests.

Other species are *habitat specialists*. These species are adapted to survive in forests by capitalizing on the use of a narrow set of resources and the resources that they are better adapted to use than most other species. Consider where you might find spring salamanders in the eastern United States or torrent salamanders in the western United States. Both species occur in clear, cold headwater streams and they tend to be most abundant where fish are excluded from the streams because fish are their predators. Both species are of interest to wildlife biologists due to the concern that forest management activities that reduce canopy cover and raise stream temperatures could threaten populations of these species (Lowe and Bolger 2002, Vesely and McComb 2002). Clearly though, habitat generalists and specialists are simply two ends of a spectrum of species' strategies for survival in forests faced with variable climates, soils, disturbances, competitors, and predators.

## HIERARCHICAL SELECTION

Many studies have been conducted to assess habitat selection by forest wildlife species. The assumption made by biologists is that if we can understand what characteristics of the environment are selected by a species, then we can infer what characteristics we may wish to provide during forest management to accommodate them in our stand or forest. There are some concerns surrounding this assumption that we shall explore later, but the vast majority of information that we have available to manage habitat for species comes from these habitat selection studies. Consequently, we need to understand which level of habitat selection we can influence for a species through our forest

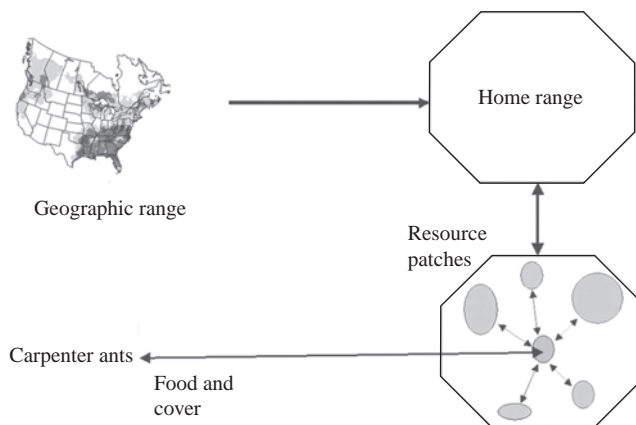


**FIGURE 3.1** Deer mice are habitat generalists that use a wide range of forest conditions. (Photo by Mike Jones. With permission.)

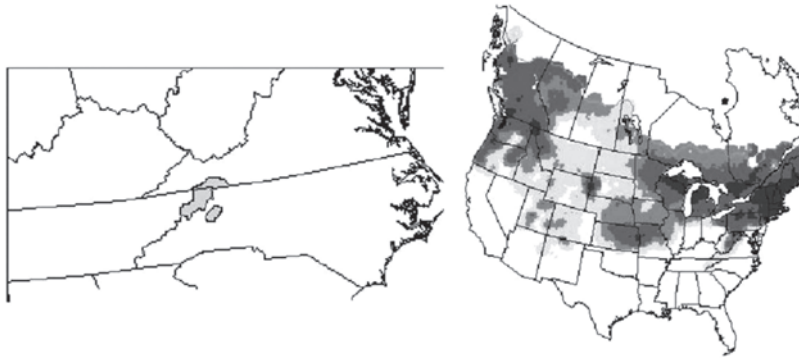
management activities. Johnson (1980) suggested that many species select habitat at four levels and called these levels first-, second-, third-, and fourth-order selection (Figure 3.2).

### WHERE IN THE WORLD SHOULD I LIVE?

*First-order selection* is selection of the geographic range. The geographic range defines, quite literally, where in the world this species can be found. In our example from Figure 3.2, pileated woodpeckers are found in forests throughout the eastern and western North America. Now consider two extremes. In Figure 3.3, I have provided geographic range maps for two species: Weller's salamander, found in spruce forests above 1500 m (5000 feet) in the southern Appalachians, and black-capped chickadees, found throughout the northern U.S. and southern Canada. The geographic range for humans is global, with infrequent excursions to other planetary bodies! Of course, the geographic range for a few other species is also global: cockroaches, Norway rats, and other human *commensals* (species that are typically associated with humans). So why is it that some species occur around the world and others are restricted to a few mountains in the Appalachians? Climate and history have had some effect on the distribution of some species. There is a complex set of geographic distributions for slimy salamanders in the southeastern U.S. that likely result from past



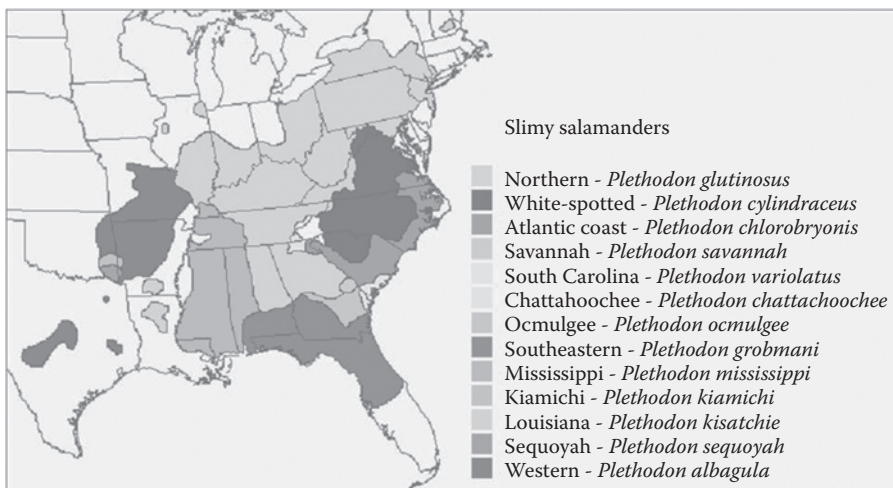
**FIGURE 3.2** Hierarchical habitat selection as described by Johnson (1980). This generalized concept is illustrated using pileated woodpeckers as an example. (Range map from USGS Biological Resources Division.)



**FIGURE 3.3** Range maps for a geographically restricted species, Weller's salamander (left) and a cosmopolitan species, black-capped chickadee (right). (Maps from USGS Biological Resources Division.)

glaciation that has “packed” species into the southern Appalachians, in combination with species that have limited capabilities to disperse from one valley to another (Figure 3.4). Similarly, other species such as the Siskyou mountain salamander may have been more widely distributed during periods when the northwest was cooler and wetter. This species became more and more restricted as the climate has changed and now only occurs in a small region of southern Oregon and northern California.

In another example of the role of barriers as a mechanism for limiting the geographic distribution of organisms, consider that the Columbia River is the dividing line separating the geographic distribution of the western red-backed vole in Oregon and of Gapper's red-backed vole in Washington. One can only guess how this might all change when one or more of them hitches a ride in an RV across the Dalles bridge! Humans, of course, have been important mechanisms for dispersing species into places that are climatically acceptable for a species but barriers had kept species separated until humans moved them. The list of examples is growing rapidly, but includes those well-known such as European starling, tree-of-Heaven, and gypsy moth. Humans are breaking down barriers and allowing opportunities for exotic species to become invasive. Implications for native flora and



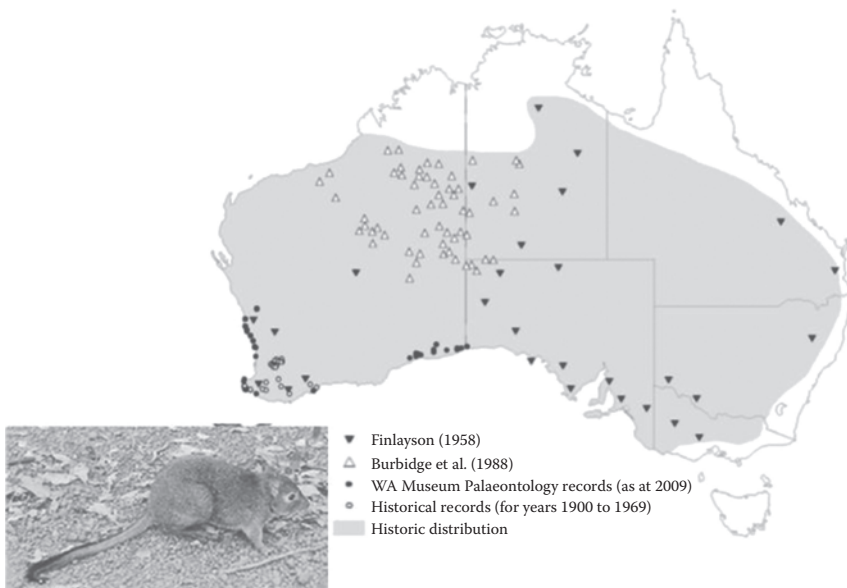
**FIGURE 3.4** Distribution of a complex of slimy salamander species in the southern U.S. (Maps from USGS Biological Resources Division.)

fauna can be huge, and the geographic ranges of some native species can be significantly altered as these invasive species proliferate. The influence of hemlock wooly adelgid on eastern hemlock mortality has led forest managers to extensively salvage dead hemlocks (Howard et al. 2000). This mortality and forest management has led biologists to worry about declines in the distribution and abundance of black-throated green warblers and other hemlock-associated species (Yamasaki et al. 2000). Invasive species can also influence forest wildlife populations by predation. In Western Australia, a species of marsupial known as the woylie was once widespread over Western Australia, but by 1980 had been reduced to three small populations due to the expansion of introduced red foxes (Figure 3.5). With recent widespread control of foxes using warfarin poisons (woylies are not affected by this poison because it occurs naturally in shrubs in their environment), populations have once again begun to expand.

Species geographic ranges have also been influenced by invasive competitors. Recently, barred owls have been found within the geographic range of northern spotted owls in the Pacific northwest of the United States. There is increasing evidence that the northern spotted owls are declining in abundance in the presence of barred owls (Peterson and Robbins 2003, Livezey 2010), and there is clear evidence that the two species are hybridized (Hamer et al. 1994).

Just as humans have been the cause of changes in geographic ranges through species introductions, they have also been responsible for recovering species from areas where they were extirpated. Translocation efforts and re-establishment efforts have been successful in species recovery (Haight et al. 2000). For example, the red-cockaded woodpecker is a threatened species that occurs in forests of the southeastern United States. This species requires old, living pines with heart rot in which to nest. As trees grow and forests age, areas of suitable habitat can be recruited. Rudolph et al. (1992) demonstrated that re-establishment of this species in this newly recruited habitat is possible. There are numerous similar examples of successful reintroductions for game species such as wild turkeys.

Consider the importance of populations of a species at the center vs. the periphery of its geographic range. Populations at the periphery may be in lower quality habitat if either biotic or abiotic factors are limiting its distribution. But recall that environments are not static. They are constantly



**FIGURE 3.5** Past and current distribution of the woylie, a woodland marsupial. The introduction of red foxes reduced the geographic range of the mid-sized marsupial. (Map from the Department of Parks and Wildlife, Western Australia. With permission.)

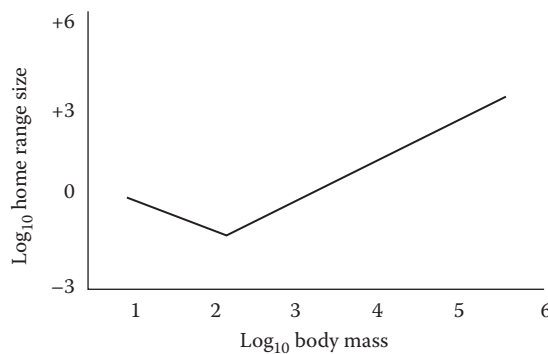
changing. Climate changes, earthquakes change the topography, some species arrive while others leave. It is those populations at the periphery of their geographic range that are on the front line of these changes. Although it may be tempting to think of these peripheral populations as somewhat expendable, they may be critical to population maintenance as large-scale changes in habitat availability occur. Given the rate at which climate is changing, these peripheral populations may be even more important over the next few hundred years.

Although Johnson (1980) does not describe *metapopulation* distribution as a selection level, it is important to realize that within the geographic range, populations oftentimes are distributed among smaller, interacting populations that contribute to overall population persistence, or a metapopulation structure. Hence, these subpopulations may grow, go extinct, and be recolonized as habitat quality changes following forest disturbance and regrowth. The distribution of the subpopulations is important to consider during forest planning because if dispersal among subpopulations is restricted by forest management actions, then the subpopulations that might ordinarily be recolonized may be restricted from doing so.

### LOOKING FOR A HOME IN A NICE COMMUNITY

Johnson (1980) described *second-order selection* as the establishment of a *home range*, an area that an individual or pair of individuals uses to acquire the resources that it needs to survive and reproduce. Not all species have established home ranges, but most do. Species that have nests, roosts, hibernacula, or other places central to its daily activities move in an area around that central place to acquire food, use cover, drink water, and raise young. Home ranges are not the same as territories. A *territory* is the space, usually around a nest, that an individual or pair defends from other individuals of the same species and occasionally other individuals of other species. Territories may be congruent with a home range, smaller (if just a nest site is defended), or may not be present at all. Many bird species, such as eastern bluebirds, defend a territory around a nest that includes the nest site and an area within which the pair finds food to feed their young. Other species such as fox squirrels defend a nest or den site when raising young, but have a home range that overlaps with other individuals. Species such as flying squirrels seem not to establish territories and coexist with other individuals within their home range.

Home ranges vary in size with the body mass of the species (Figure 3.6). Species with larger body mass need more energy to support that mass. Herbivores tend to have smaller home ranges than carnivores of the same size, because energy available to herbivores is more abundant but also



**FIGURE 3.6** Relationship between body mass and home range size. Home range sizes are smallest for those species with body mass of ~100 g, and greater for sizes that depart from this size. (Based on data by Kelt, D.A., and D. VanBuren. 1999. *Ecology* 80:337–340; adapted from Harestad, A.S., and F.L. Bunnell. 1979. *Ecology* 60:389–402.)

because with each increase in trophic level there is a decrease in energy availability. A *trophic level* is the feeding position in a food web: Primary producers are typically plants, primary consumers are herbivores, secondary consumers are carnivores, and tertiary consumers are carnivores that eat carnivores. Hence, there is an energy or biomass pyramid, with more biomass in producers than in herbivores and more biomass in herbivores than in carnivores.

Home range sizes also vary among individuals within a species. As food resources are less abundant or more widely distributed home range sizes increase. But within a species, the home range size has an upper limit that is governed by balancing energy input from food with energy loss by movement among food patches. For instance, Thompson and Colgan (1987) reported larger home ranges for American marten during years of low prey availability than in years of high prey availability.

### **BUYING A HOUSE AND BUYING GROCERIES**

*Third-order selection* is the use of patches within a home range where resources are available to meet an individual's needs. Biologists often can delineate a home range based upon observed daily or seasonal movements of individuals going about their business of feeding, resting, and raising young. But this area is not used in its entirety. Rather there are some places within the home range that are used intensively and other parts of the home range that are rarely used (Samuel et al. 1985). Selection of these patches is assumed to represent the ability of the individual to effectively find and use resources that will allow it to survive and reproduce. But as Garshelis (2000) makes clear, simply the amount of time or number of radio telemetry locations in a particular patch type does not necessarily reflect the importance or lack thereof to an individual. An individual may spend a small amount of time and be represented by relatively few locations in a particular patch type but receive important benefits from that patch type. For instance, you may spend 10% of your time in your kitchen and 30% of time in your bedroom, but the resources that you receive from your kitchen are as important, or more important, than the rest that you receive in the bedroom. It is exactly those resources located in the patch types that are most important to maintaining an animal's fitness.

### **WHAT WOULD YOU LIKE FOR DINNER?**

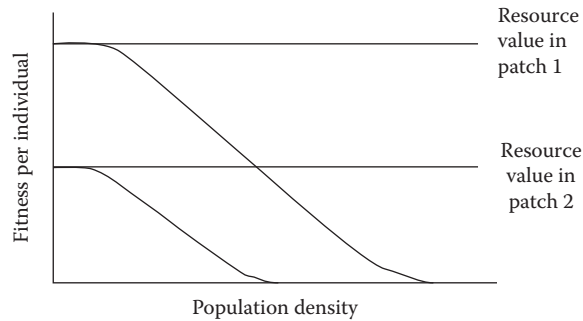
*Fourth-order selection* is the selection of specific food and cover resources acquired from patches used by the individual within its home range. Given the choice among available foods, a species should most often select those foods that will confer the greatest energy or nutrients to the individual. Which food or nest site to select is often a trade-off among availability, digestibility, and risk of predation (Holmes and Schultz 1988). Factors that influence the selection of specific food and cover resources most often tend to be related to energetic gains and costs, but there are exceptions. The need for certain nutrients at certain times of the year can have little to do with energetics and much to do with survival and fitness. For instance, band-tailed pigeons seek a sodium source at mineral springs to supplement their diet during the nesting season (Sanders and Jarvis 2000).

Collectively, these levels of habitat selection influence the fitness of individuals, populations, and species. Habitat quality is dependent not only on the food and cover resources in the stand or forest but also the number of individuals in that stand. Many individuals in one stand means that there are fewer resources per individual. Habitat quality and habitat selection is density dependent. Indeed, even if a patch has excellent, but a fixed quantity, of food and cover quality, too many individuals in the patch can cause some to leave to find other habitat patches of lower quality, but which have fewer individuals.

### **DENSITY-DEPENDENT HABITAT SELECTION**

Fretwell and Lucas (1969) provided the conceptual basis for understanding density-dependent habitat selection. Consider a fixed level of resource availability in two patches, with resource





**FIGURE 3.7** In the above diagram, the top horizontal line represents a fixed amount of resources in patch 1; the lower horizontal line represents the fixed amount of resources in patch 2. The resulting selection of habitat is dependent on the density of individuals in each patch. (Adapted from Fretwell, S.D., and H.L. Lucas Jr. 1969. *Acta Biotheoretica* 19:16–36.)

availability in one patch higher than that in another (Figure 3.7). As the population density in patch 1 increases, the resources available per individual and hence fitness per individual declines. Eventually at a high enough population density, the fitness per individual declines to a point where each individual is afforded a level of resources that would be less than what they received if they moved to patch 2, a patch with lower total resource availability than patch 1. Consequently, the selection of patch 1 and ultimately patch 2 by some individuals is influenced by the populations in each patch.

Under this approach, each individual is free to choose the patch that will provide the greatest energy or other required resources. This concept is called the *ideal free distribution*. But in many populations, especially those that have a dominance hierarchy or are territorial, some individuals are less likely to move to patch 2 and some are more likely to move. Consider the case where the species occupying patch 1 initially is territorial and as more individuals are added to the population each defends a specific territory. Territories help to ensure that individual fitness will not decline, and that each individual or pair in the patch remains fit. Eventually the patch will be saturated with territories and you cannot pack another territory into patch 1. If the territorial individuals are successful in patch 1, and they are successful at defending their territory from interlopers, then they help to ensure their fitness. Of course, the cost of territoriality is the energy expended defending it. Subordinate individuals in the population, those unable to displace an individual already on a territory, are relegated to patch 2. This situation represents an *ideal despotic distribution* where individual fitness is maintained in the highest quality patches at lower than expected densities through territoriality.

If the individuals in a patch are fit enough to support a stable or growing population, that is they are able to sustain that population through reproduction and survival, then they occupy a *source habitat*—one with high individual fitness. If the individuals in a patch have low reproduction and survival rates, often insufficient to maintain the population, then they occupy a *sink habitat*. It is important to keep in mind that in the Fretwell–Lucas example, both patch 1 and patch 2 could be source patches, or sink patches, depending on how the populations are maintained in each patch.

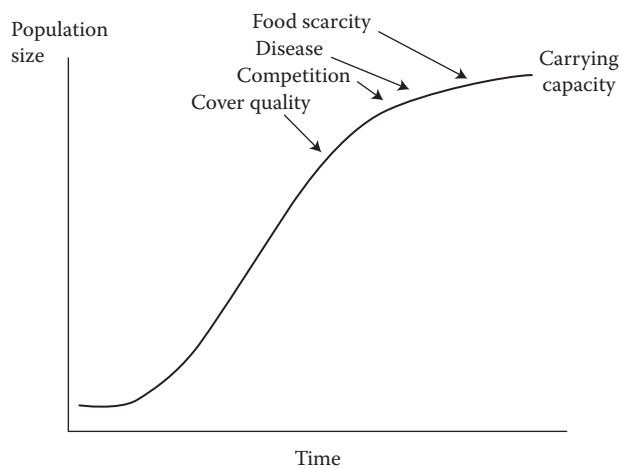
Since resources are already restricted in a sink habitat, increasing the density of territorial species may cause individuals to abandon territorial behaviors, not reproduce successfully, or face a higher risk of mortality. Indeed in some sink habitats, densities of individuals can be much higher than in source patches because in source patches individuals remain territorial, but in sinks they may not be territorial. Consequently, using density as an index to habitat quality may be inaccurate. Animal fitness is a better indication of habitat quality than animal numbers. Reproductive success, survival, and body mass are all indicators of animal fitness (Van Horne 1983).

## RELATIONSHIP BETWEEN HABITAT QUALITY AND DEMOGRAPHICS

Foresters manipulate stand density to ensure that the trees that will eventually be harvested have sufficient resources to grow rapidly, produce seeds, and survive to maturity. Biologists do much the same thing when managing habitat for animals. Populations can be manipulated by modifying habitat and thereby influencing possibilities for survival and reproduction, the two primary indices to fitness. The linkages between animal demography and habitat are complex, but some understanding of these relations is necessary for successful habitat management. Each species has its own potential for population increase, and this potential is described as the *intrinsic rate of natural increase*. There is a solar constant so energy available to plants and animals is limited. Given adequate food, cover, and water populations will grow. But consider what happens as the density of individuals increases. Food becomes scarcer or of poorer quality as the population grows. Cover is occupied by more individuals so the risk of disease and parasitism increases. Intraspecific (among individuals of the same species) competition for resources causes some subordinate individuals to use suboptimal patches. As food, cover, or other resources become limited, the population growth rate decreases, due to either mortality increases or reproduction decreases, or both. This process is termed *logistic growth* (Figure 3.8). If we assume that resources are constant, then the population reaches a point where births equal deaths and growth becomes 0. This point is termed the *carrying capacity* of the habitat for the population.

But resources are not constant; they change daily, seasonally, and annually. Birth rates, death rates, and movement rates are variable over both space and time as habitat changes through forest disturbances and succession. Carrying capacity, consequently, is always changing. The concept of a *dynamic carrying capacity* is useful to land managers because it provides the link between the dynamics of forests, habitat quality, and population growth. Manipulating habitat to change carrying capacity is a particularly effective approach to long-term manipulation of wildlife populations.

But populations do not always reach carrying capacity in relation to habitat quality. Some species, such as voles, snowshoe hares, and ruffed grouse, follow a “boom and bust” population pattern. Populations grow for about 3–6 years and then rapidly decline for another 3–6 years. High-quality habitat usually increases the highs and decreases the lows of a population cycle, but habitat probably



**FIGURE 3.8** Population growth over time assuming fixed resource availability results in a carrying capacity where births balance with deaths and the population remains somewhat stable. In actuality carrying capacity is quite dynamic as resources change over time and space. (Adapted from McComb, W.C. 2001. *Wildlife Habitat Relationships in Oregon and Washington*. OSU Press, Corvallis, OR.)



does not directly mediate these cycles because they occur throughout much of the geographic range of the species (Keith and Windberg 1978).

## POPULATION FITNESS

Individuals are fit when they have a high probability of surviving and reproducing successfully. Population fitness is high when the population is increasing or at least not declining. Individuals with high fitness can occur in populations with low fitness and vice versa. Since it is populations, and not individuals, that are sustainable over the long term, we need reliable indicators of habitat quality using population fitness. Habitat quality refers to the ability of a locality to provide for the long-term persistence of a population over time. Biologists tend to measure the habitat quality based on vital rates of the population. If a population is reproducing at an optimum rate and survival of young and adults is high, the habitat is considered as high quality. *Vital rates* are the demographic parameters that drive population change, primarily birth rates and death rates.

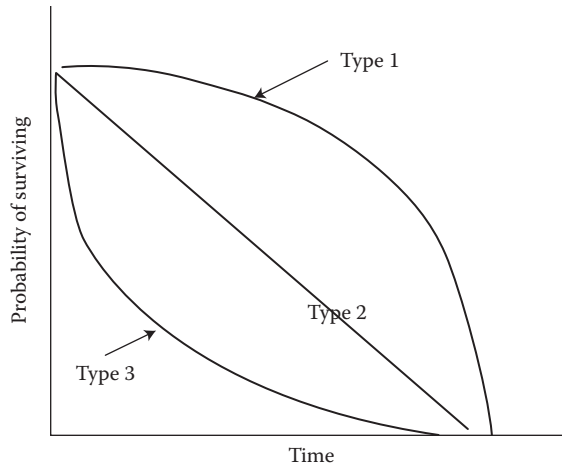
The rate at which animals reproduce is a basic component of population dynamics. Two measures of reproductive fitness are natality and fecundity. *Natality* refers to number of young individuals born or hatched per unit of time. *Fecundity* is the number of young produced per female over a given time period and relates population fitness to the average fitness per female. Usually, 1 year is the time period considered, but for smaller animals, especially those that may breed several times a year, a shorter time period may be selected. Thus, if a population of 1000 female bears produced 200 young in a year, the birth rate, or fecundity, would be  $200/1000 = 0.2$ .

A number of factors affect a population's birth rate. Animals that are young or in poor nutritional condition usually have fewer young and/or breed less often. Age at first reproduction is also an important factor in determining birth rate. Large, long-lived animals typically do not become sexually mature until they are several years of age. A vole might become sexually mature and breed for the first time at 18 days of age. An Asian elephant on the other hand will typically be 9–12 years old when it first breeds. The birth interval is also important in determining birth rates. A vole might produce a litter of young every 30 days during the breeding season, but a grizzly bear may only reproduce every 3 or 4 years. The average number of young produced is of obvious importance in a population's birth rate. Some animals, such as fish or amphibians, produce hundreds or thousands of eggs (not all of them hatch and few survive), while many species only have one or two young at a time (e.g., barred owls). Potential population growth rates are related to fecundity rates. A doubling in the fecundity rate will more than double the population growth rate.

*Mortality rate* is another indicator of population fitness. Mortality rate is measured as the number of animals that die per unit of time (usually 1 year) divided by the number of animals alive at the beginning of the time period. Thus, if 1000 fawns are born in June and 400 are alive the next June, then the mortality rate is 600 (the number that died)/1000 = 0.6 or 60%. Survival and longevity are two other population parameters related to mortality. *Survival* is the number of animals that live through a time period and is the converse of mortality. Thus, if the mortality rate is 0.8 or 80% per year, then survival would be 0.2 or 20% per year. *Longevity* is the age at death of the average animal in a population.

Mortality rates are usually age—and often sex—specific, which means that animals of different ages or sexes die at different rates. In many species, the young and old animals die at faster rates than the mid-aged animals. Often, males have higher mortality rates than females because of activities associated with territorial or mating behavior.

Different species have different *survivorship functions* related to their life-history traits. A type I *survivorship curve* would be typical of animals that have relatively high survivorship until later in life when they become subject to age-related mortality (Figure 3.9). Typically, these are animals with a high degree of parental care. Many larger mammals, such as whales, bears, and elephants, might have type I survivorship curves. Some animals have fairly constant survivorship (type II). Some birds and most reptiles and amphibians probably fit this pattern, although our knowledge



**FIGURE 3.9** Survivorship curves for three example species. Type 1 species have high juvenile survival rates while type 3 species have low juvenile survival rates.

of survivorship in birds is not very complete because they are difficult to study. A type III survivorship curve would be typical of animals with little or no parental care and/or vulnerable young; mortality is high in the young age classes and low in older animals. Insects and fish often have type III survivorship curves.

## MEASURING HABITAT SELECTION

Clearly, we should use estimators of fitness as a measure of habitat quality and selection. Rarely have biologists taken that approach largely because measures of fitness are expensive and difficult to assess for many species. Usually occurrence or abundance is used although some indirect indicator of fitness such as body mass and evidence of breeding may be used instead. Despite not measuring fitness attributes directly for many species, years of habitat selection research have produced repeatable patterns of use and selection for many species.

Assessing habitat selection is scale dependent. Most information available on habitat relationships of species in forests comes from one of two approaches: stand-based assessments or species-based assessments. In stand-based assessments, comparisons of abundance or occurrence are made between stands of different structure or following different treatments. This information can be very useful if the animal response is matched to the scale of the stands. For instance, we would not use number of northern goshawks detected in thinned and unthinned stands that are 10–15 ha in size (second-order selection) as a response variable because the home range of one goshawk could encompass many stands. Rather we might ask if stands were selected by goshawks for foraging (third-order selection) within their home range or if particular nest structures were used in these stands (fourth-order selection). Alternatively, we might place radio transmitters on goshawks and analyze the stand types used in comparison to their availability in each home range. Both approaches provide useful information, and both have potential weaknesses.

When considering how animals use habitat in forests, it is important to differentiate between use and selection. Animals can be found in various types of forest conditions. We occasionally find a species that is typically found in early successional stages occurring in old-growth forests and vice versa. We may find dispersing spotted owls in urban settings. It is important to know why they were found in these settings. Were they forced there by more dominant (and fit) individuals? Did random dispersal bring them there by chance? Are they surviving there? Reproducing? Observations of use can be important information, but they must be placed into the context of why the animal is found in these places.

Selection is the choice of one or more patch types among those that are available. For example, say you were able to make 100 unbiased observations of American marten—80 in old forest and 20 in young forest. Within your study area old forest comprises 40% of the area and young forest comprises 60%; therefore if marten were using the area randomly (no selection), you would expect 60 observations in young forest and 40 in old forest. In this simple example, marten were using old forest out of proportion to its availability and could be said to “select” old forest. Does that mean that young forest was avoided? Or is unimportant? Not necessarily. If marten were eating raspberries in the young forest during the summer then they may not spend much time there, but that food resource was sought (not avoided) and could contribute to marten fitness. Again, it is important to know why marten were found in these forest types. Further, selection can only be assessed among the choices available to an organism. The organism may actually prefer some other conditions that are not available. Given a choice of a beech–maple forest and a pine forest, a gray squirrel may demonstrate a selection for the beech–maple forest, but it would prefer an oak forest if it was available.

Use–availability studies are further complicated in that too often we classify forests by dominant tree species and/or age class and then see which classes are selected by a suite of species. The classes that were created often are done so based on human perceptions of differences (clearcuts, old-growth, hardwood, conifer) and may only be marginally related to providing the habitat elements needed by the species being assessed. Take, for example, Swainson’s thrush. Swainson’s thrushes are associated with shrub cover where they nest and are found in woodlands where shrub cover is dense. That there are hardwoods or conifers or pole-sized trees or old-growth trees in the overstory is somewhat irrelevant. But how often do we humans classify vegetation based on shrub cover beneath the overstory? And of course, habitat is more than just vegetation. Soils, slope, aspect, and so on could all be mapped and classified, but they would need to be mapped and classified differently for each species. And that too rarely happens. The results of selection studies based on *a priori* classifications of forest condition unrelated to habitat elements important to the species of interest should be viewed with caution.

Ideally, experiments that manipulate resources and measure population vital rate responses are most reliable. For example, if an experiment was designed to test the effects of thinning on ruffed grouse, we would randomly identify the located study sites and sample an aspect of fitness (e.g., survival rates and natality) for at least one full population cycle (~10 years) prior to thinning. Study sites would have to be large to ensure that we could sample multiple individuals in each stand. Since the home range for a ruffed grouse is approximately 4 ha (10 acres), then stands might need to be 120 ha (300 acres) in size, or more. Once the pretreatment data were available, then we would thin a randomly selected group of stands and monitor the same vital rates on the thinned stands as well as on untreated controls for another 10 years. Such an approach may be ideal but in most circumstances it is impractical due to expense and logistics. And for some species, such as those protected under the Endangered Species Act, it may be illegal.

*Critical habitat* is defined as specific areas that are essential to the conservation of a federally listed species under the U.S. Endangered Species Act, and which may require special management considerations or protection. If the best available information would suggest that thinning would be detrimental to the species, then the experiment above may simply not be allowed in the United States.

In addition, experiments as described earlier present difficulties when assessing species that show high affinity for an area, also called *site fidelity*. Pairs may return year after year to the same location despite drastic changes in the habitat around them. Effects of the treatment may only be apparent once these pairs are gone, because new breeders may not be recruited to this site because it no longer has the cues they look for in a breeding area.

## PROXIMATE AND ULTIMATE CUES TO HABITAT QUALITY

Use–availability studies often result in evidence for selection of certain habitat types, tree species, or vegetation structures. These structures are often related to the availability of resources that an

animal needs for survival, but not always. The *ultimate* food and cover resources that each species needs are often found by the species using *proximate cues*. Migratory birds are a good example. As they move from breeding areas to wintering areas they must make choices about where to rest or settle such that food and cover will likely be available for them. In these situations, vegetation structure seems to be a key proximate cue to these choices (Cody 1985).

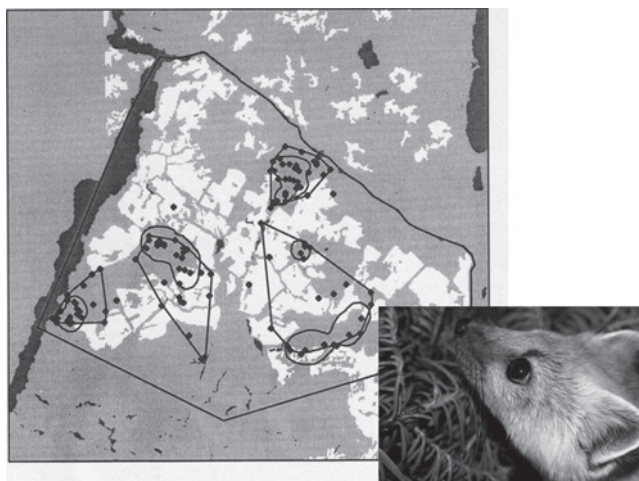
The *structure* of a forest provides a cue to an animal that certain insect or plant food resources might be available, or that nest sites might be available. These cues may cause animals to establish a territory before (e.g., early spring) the ultimate resources (e.g., foliage-dwelling insects) are even available. Animals use visual, aural, and/or olfactory cues when establishing a territory or home range. Managers can identify the habitat elements that may be proximate cues to habitat selection and ensure that these habitat elements are present for those species that are desired in a stand or forest. Managers manipulate aspects of the stand such as stocking levels, tree density, and tree size but need to consider other specific habitat elements that may or may not be related to traditional stand management for timber production. It may be important to grow a large tree, but if the ultimate resources associated with the large tree (proximate cue) are bark-dwelling insects in deeply dissected bark, then simply having large trees may not yield higher quality habitat. Consider a human example. Humans use proximate cues every day. When we are hungry and need food fast (quality may be a separate issue), we do not go into every building and hunt around for a hamburger but instead look for a proximate cue, for example, golden arches. If the place with the golden arches is out of burgers, then a typically reliable cue did not yield the desired resources, and you spent time and energy for nothing. Providing only proximate cues without considering ultimate resources is no different.

## SOCIAL CUES IN HABITAT SELECTION

Say you drive by two restaurants on a Friday evening and the parking lot for one (the Greasy Spoon) is full and the other (The Pie Palace) only has a few cars. Which one would you likely try first? And then you ask your coworkers the next day if they had ever eaten at the Greasy Spoon and they indicate that the food is very good, then you may be even more likely to try eating there. You are using social cues, evidence from your conspecifics that one foraging patch is better than another. Some other animal species are no different. Danchin et al. (2004) described the importance of social cues influencing selection of habitat in a number of species. Subsequent work by Templeton et al. (2009) and Betts et al. (2008) supported the importance of bird song influencing habitat selection. Betts et al. (2008) conducted an experiment testing the effects of vegetation structure (proximate cues) and social information (bird song) in selection of patches by black-throated blue warblers and found that songs emanating from stands with structure completely different from where the bird fledged induced individuals to not only enter the poor quality site to feed but also return the next year and try to breed. It is very clear the social information is a key component of habitat selection for not only humans and warblers but for many other species as well.

## CASE STUDY: AMERICAN MARTEN HABITAT SELECTION

American marten are mustelids, members of the weasel family. Their geographic range extends across North America in boreal forests. Females have a home range of approximately 2.3 km<sup>2</sup> and males, and being larger, have larger home ranges. There has been growing concern that this small carnivore may be adversely affected by intensive forest management in coniferous forests. This case study is based on a study by Potvin et al. (2000), who examined marten habitat selection at stand and landscape scales in intensively managed spruce forests in Quebec. About 10,000 km<sup>2</sup> of Canada's forests are clearcut each year. A consistent finding among many studies conducted throughout the geographic range of marten is that having more than 20%–30% of an area recently cut leads to declines in marten abundance. That does not necessarily mean that clearcuts are not used by marten. Indeed, some types of food, especially berries and other sugar-rich fruits, may be



**FIGURE 3.10** Radio locations, core areas and 95% polygon home ranges of American marten in Quebec boreal forests. White areas have been recently cut and shaded areas are older forest. (Map from Potvin, F., L. Belanger, and K. Lowell: Marten habitat selection in a clearcut boreal landscape. *Conservation Biology*. 2000. 14:844–857. Copyright Wiley-VCH Verlag GmbH & Co. KGaA. Reproduced with permission; Marten photo by Mike Jones. With permission.)

more available in openings, but if openings cover too large an area, then prey (primarily Gapper's red-backed voles) are not sufficiently abundant at other times of the year.

Potvin et al. (2000) attached radio transmitters to 33 marten and they estimated the winter home ranges for each marten. An example is shown in Figure 3.10. Several things are clear from this figure. First, marten did use regenerating stands but much less than would be expected by chance alone. Second, most locations were in the surrounding older forest. Stands of deciduous and mixed deciduous–conifer >30 years of age were selected out of proportion to availability. Conifer forests >30 years of age were used in proportion to availability. Recent cuts <20 years old where young trees had grown to a point where the crowns closed to form a continuous canopy were also used in proportion to availability, but recent cuts that still had an open canopy were used less than expected by chance.

The landscape level analyses showed similar patterns. Home ranges contained more area in forests >30 years of age and less area than expected of young, open forest. Indeed, marten with smaller home ranges had less area of young, open forest than marten with large home ranges suggesting that marten can find more of their required resources in a smaller area when there is less open, young forest. On the basis of these results and results of studies from Maine and Utah, it seems that marten cannot tolerate >30% of a home range in recent clearcuts, but that once young stands form a continuous canopy they will use the area. Consequently, forest planners developing a harvest schedule can use this information to guide where clearcutting could occur to minimize effects adverse on marten.

## SUMMARY

Habitat is selected by many vertebrates at four levels: geographic range, home range, patches within the home range, and the ultimate resources needed for survival. Such selection is assumed to represent a complex set of behaviors that species have evolved to yield high population fitness despite environmental variability. Conspecifics influence selection of habitat in a myriad of ways, including the exchange of social knowledge. Selection of habitat also can be influenced by other species such as competitors and predators. Habitat selection is also density dependent, with the choice of habitat patches influenced by the effects of the population on individual fitness. In territorial species,



subordinate individuals may be forced into sink habitat where survival and reproduction rates may be lower than in source habitat occupied by dominant individuals. Although we often gather information on habitat selection employing use–availability studies, interpretation of these results must be done with caution unless we understand why species are using certain conditions. Lack of selection does not necessarily imply avoidance. Although experimental approaches that document effects of forest management on animal fitness are ideal, they often are impractical. Hence, forest wildlife biologists are usually faced with using information from associational studies to identify the proximate cues to habitat selection are provided during forest management.

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# 4 Forest Structure and Composition

When a forester sees a forest she often will look at the tree species mix, the tree size, tree density, and other clues about how the stand might be managed to achieve wood products or other goals. When a wildlife biologist sees a forest, she sees evidence of deer browse, pellet groups on the forest floor, tracks in the mud, or nests in trees. To effectively manage habitat in a forest, the forester and the biologist must assess the sizes, numbers, and arrangement of a set of habitat elements, the building blocks for habitat within a stand or forest. *Habitat elements* are those pieces of the forest that in certain numbers, sizes, and arrangements meet the food or cover resources for a species. If these are highly variable within and among stands, then the needs for many species can be met. If they are very uniform, then the needs of only a few species can be met. The challenge to the forester and biologist is to walk into a stand and see the same habitat elements. In so doing, the biologist can explain why more or fewer of any set of them are needed to meet a species goal. Similarly, the forester can explain how silviculture might be used to achieve that goal.

## FOOD AND COVER IN A CELLULOSE-MANAGED SYSTEM

Timber management has, for many years, been focused on producing wood products (cellulose) from managed forests. To maximize cellulose production, foresters want to be sure that the growing space for trees in a stand is fully utilized. Cellulose is quite indigestible for most species so unless the trees are allowed to decompose into forms of carbon that are digestible, maximizing timber production can lead to low levels of food for many species. To compensate for this lack of digestible energy, growing space could be allocated to other plant species or habitat elements, or the cellulose must be made available to more species through wood decomposition. Both of these choices result in a decrease in the production of wood for humans. Consequently, the decision to manage habitat elements must come with the understanding that providing some habitat elements in some stands or in some parts of some stands may come at a financial cost to the landowner. The manager must decide which habitat elements can be provided in a way that is compatible with the goals for managing the stand for cellulose, and which will come at a cost, and how much cost the landowner is willing to bear.

Generally, we think of providing these elements by altering the structure and composition of a stand. *Structure* refers to the physical features of the environment such as vegetation, soils, and topography. The complexity of the structure serves as both proximate cues in habitat selection as well as ultimate resources for cover (e.g., nesting sites, resting sites). *Composition* refers to the species of plants, types of soils, and other features that contribute complexity, for instance through plant species richness. It is the combination of vegetation structure and composition that managers can change through their management actions.

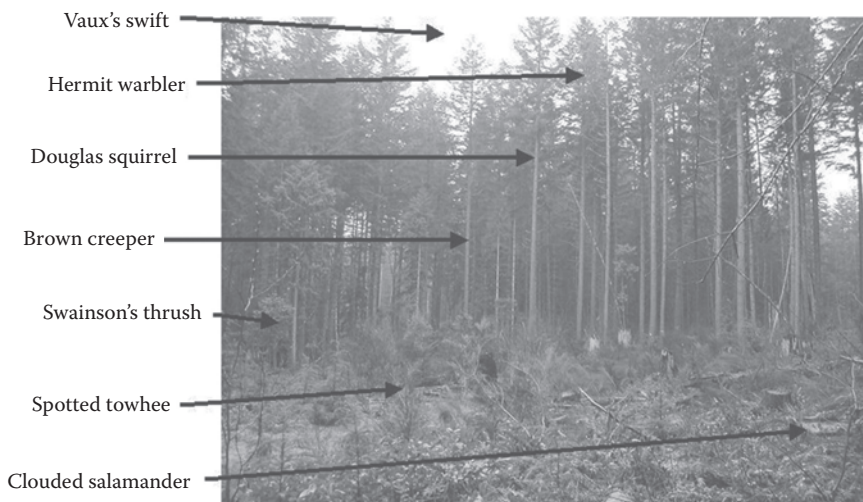
## VERTICAL COMPLEXITY

Read any forest plan or silvicultural prescription and early on there will be reference to forest area: 10, 100, or 1000 ha. But to most animals, forests are not areas. Forests are volumes—they have three dimensions: length, width, and height. One characteristic of forest development that influences the diversity of animals within a stand is the distribution of the foliage vertically within the stand.

Some species use the foliage as a source of food either directly through herbivory (e.g., red tree voles in the Pacific Northwest) or indirectly by feeding on foliage-dwelling invertebrates (e.g., red-eyed vireos in hardwood forests of the southern United States).

Indeed, many bird species distribute themselves vertically within a forest to take advantage of food and cover resources while reducing competition among species for these resources (Figure 4.1). In some stands, the foliage is distributed only in one layer, such as a dense plantation where all of the live foliage is in one canopy layer with little foliage beneath it. These stands typically provide habitat for a narrow range of species, but can still be important for a few species (Titchenell et al. 2011). Mixed species stands have several tree and shrub species and often with foliage distributed in several layers (understory, midstory, overstory). These multilayer stands tend to support more species of vertebrates than a one-layer stand. Wet tropical forests and mixed mesophytic forests develop vertical complexity naturally because of the mixture of tree species that occur within them. The distribution of foliage can be influenced by the shade tolerance of the tree species in the stand. *Shade-tolerant* species are those that can survive under low light conditions. Shade-tolerant species use a strategy of survival and slow growth under low light until an opening or gap is provided in the tree canopy. When that gap occurs from tree fall or tree death, then the shade-tolerant species in that gap grow more rapidly to occupy the opening. These species can occupy lower levels in the stand for many years and often have deep crowns. Sugar maples, hemlocks, and American beech are examples of species that use this strategy. *Shade-intolerant* species do not survive under low light conditions and grow well only under full sunlight. Shade-intolerant species tend to grow rapidly once the seed has germinated. Because leaves are intolerant of shade, they tend to have smaller crowns in dense stands. Aspens, gray birch, and willows are examples of shade-intolerant species. A plantation of shade-tolerant species will have somewhat greater vertical complexity than a plantation of shade-intolerant species at a similar density simply because of the depth of the crowns of the trees.

Silviculture can be used to modify the vertical complexity in a stand in several important ways. Foresters usually will approach management of a stand using either even-aged or uneven-aged strategies. An even-aged stand is one in which most of the dominant trees—those trees comprising the uppermost canopy—are of a narrow range of ages. That is, they all began life at about the same time following a stand-replacement disturbance such as a fire, hurricane, or clearcut. If the trees are



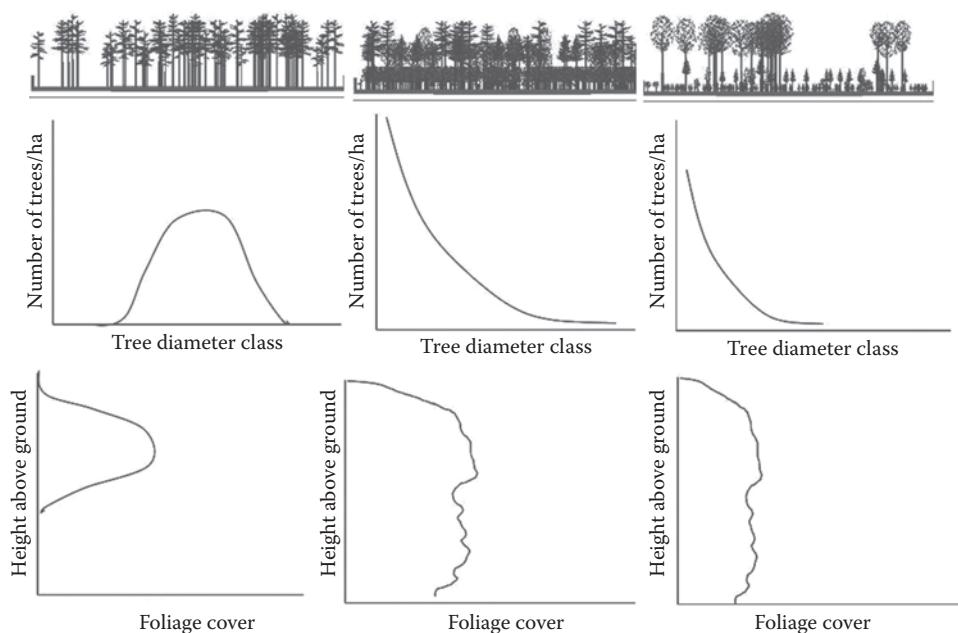
**FIGURE 4.1** Vertical structure of a forest provides niches for many species. This is an example from the Willamette National Forest in Oregon.

all the same age and all the same species, then the tree heights will be very similar and many of the tree diameters will be similar (Figure 4.2). In plantations of one species, the distribution of tree diameters is often a bell-shaped curve. Foliage cover is also represented in one dominant layer and is a bell-shaped curve turned on its side (Figure 4.2).

Even-aged stands that contain a variety of species including those that are shade intolerant and others that are shade tolerant often have more complex diameter distributions and a vertical foliage structure (Figure 4.2). Slower-growing shade-tolerant species in the understory are ‘waiting’ for the intolerant species in the overstory to die. Consequently, the range of tree diameters, heights, and distribution of foliage is more complex than in a single species plantation.

Any disturbances, natural or silvicultural, that cause the diameter distribution of a stand to change from uniformity to uneven sized will lead to more vertical complexity within the stand. In particular, silvicultural practices that create regeneration sites, where seeds can germinate and grow, can result in an uneven-aged stand—one with many tree ages and sizes (Figure 4.2). If these trees of many ages also represent a variety of tree species, then trees with a variety of diameters and heights will be represented in the stand.

Vertical complexity is typically measured using an index to foliage height diversity (FHD). FHD is calculated in a manner similar to species diversity indices, by considering both the number of layers in a stand (comparable to the number of species) and the percent cover by foliage in each layer (comparable to the number of individuals of each species). Taller forests may have many layers and if they have foliage cover in each layer, then FHD will be high (e.g., an old tropical forest). Stands with short stature and all of the foliage in one or a few layers will have a low FHD (e.g., a young single-species plantation). MacArthur and MacArthur (1961) reported that bird species diversity (BSD) was associated with FHD in the temperate forests of the eastern United States. There is a



**FIGURE 4.2** Even-aged, single species stands have a bell-shaped distribution of tree diameters, and one dominant vegetation layer (left). Even-aged mixed species stands have a skewed diameter distribution and a more complex vertical structure (middle). Uneven-aged stands have an inverse-J-shaped diameter distribution and a more complex vertical structure (right). Figures developed using Landscape Management Systems software. (McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

logical conclusion that increasing FHD could lead to an increase in BSD within the stand, but the BSD–FHD relationship has not been found consistently in other forests, such as tropical forests where competition among bird species may be important in structuring the community (Pearson 1975). Further, it is important to remember that not all species of animals benefit from vertically complex stands. Species such as northern goshawks and some bat species are very well adapted to forage beneath the canopy of large even-aged stands that have sufficient flight space beneath the canopy. In general, increasing the vertical complexity of a stand increases the potential number of niches for more species of birds (and possibly bats), but this generalization may not hold in some forest types and no single stand condition is best for all species. Nonetheless, Flather et al. (1992) found that a combination of indices of vertical and horizontal complexity were reliable indicators of bird community integrity in the eastern United States.

### HORIZONTAL PATCHINESS

The variability in tree size, species composition, dead wood, and other habitat elements is often related to the horizontal variability within a stand. Homogenous stands with evenly spaced trees and uniform canopies offer fewer niches so animal diversity is often lower in these uniform stands. The size of openings and arrangement within a stand may positively affect some species and negatively affect others. Chapin et al. (1997) found that American marten use of habitat was more influenced by the combination of vertical and horizontal complexity than by the species composition of the forest. Overwintering birds also seem to be associated with horizontal complexity in the southern United States (Zeller and Collazo 1995). Small openings or variability in tree spacing will likely have few adverse effects on most animal species and may benefit the maintenance and development of important ecological processes (Carey 2003). If openings or other discontinuities are larger than a species' home range, then we may begin to see species occupy the stand that otherwise would not occur there (DeGraaf and Yamasaki 2003), but there may be adverse effects of the openings on forest interior-associated species (Germaine et al. 1997). See Chapter 12 for more information on edge effects in forests.

### FORAGE AVAILABILITY AND QUALITY

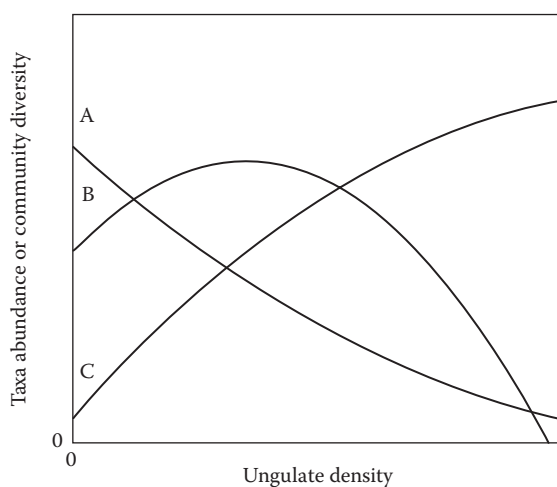
Forage for herbivores is influenced by many aspects of forest composition and structure. Conversely, the forest structure and composition can be significantly altered by herbivory. Vertebrate herbivores such as deer, elk, and moose typically *graze* more heavily on grasses (monocots) and forbs (herbaceous dicot plants) than on woody plants during the growing season. Herbaceous plants are more easily digested than woody plants and can represent 50%–80% of herbivore diets during the growing season. When cold weather or drought kills the upper portion of grasses and forbs, then vertebrate herbivores are forced to *browse* more heavily on woody plants. Browsing is concentrated on the new growth of the woody plants because that part is most easily digested. During the winter, browse can constitute over 90% of the diets of vertebrate herbivores, and for forest managers trying to regenerate forests, this level of browsing intensity can be a significant economic burden and have significant ecological effects. Krueger et al. (2009) indicated that not only can herbivory by white-tailed deer have a significant influence on future forest composition but herbivory interacts with disturbance (e.g., canopy gaps) to influence the spatial distribution of future forest composition.

That herbivores impact the trajectory of stand development is not surprising given the potential for intense browsing during winters. Animals that try to survive a long winter on woody browse face significant stress. Nearly all browsing vertebrates lose body mass during the winter (Mautz et al. 1976). Indeed, browse quality is important in slowing the rate of starvation but not preventing it. Providing high-quality grasses, forbs, and browse is important in the overwinter survival of many herbivores. Controlling the intensity of browse on tree seedlings is important in effectively regenerating forests.

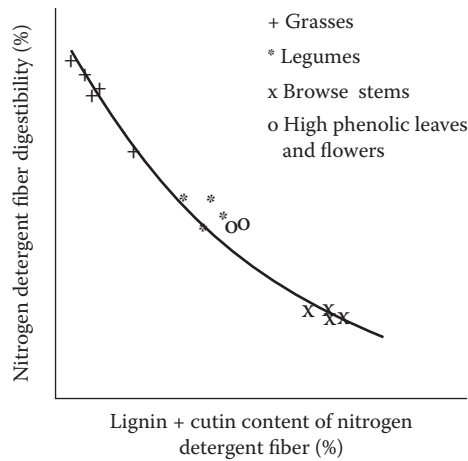
During the growing season, stands that allow more sunlight to support grasses and forbs provide higher quality forage for many species. These grasses and forbs are typically found in greatest

abundance beneath or between overstory trees in open stands (e.g., savannahs) or following a large disturbance such as a fire or clearcut. The species composition of a stand can have profound effects on animal diversity and use by various species. The digestibility of twigs and leaves can be affected by tree species composition (Mautz et al. 1976).

Each species of herbivore is selective of the species of grasses, forbs, and woody browse that they chose to eat. In so doing, herbivores such as deer and elk can have profound effects on forest structure and composition (Rooney and Waller 2003, Figure 4.3). The potential effects of selective browsing include shifts in tree species composition of the forest understory (Strole and Anderson 1992). For instance, white-tailed deer in Illinois preferred to browse on white oak and shagbark hickory; sugar maple was browsed less than would be predicted from its abundance (Strole and Anderson 1992). Horsley et al. (2003) found that bramble abundance in several silvicultural treatments; the density of striped maple in clearcuts; and birch, American beech, and red maple in thinned stands declined in abundance with increasing deer density. In Yellowstone National Park, elk preferentially browse on willows and aspens (Ripple et al. 2001) and moose in Newfoundland selected balsam fir, pin cherry, high-bush cranberry, and white birch over other plant species. Such selection may be influenced by the digestibility of the various plants, but not always. Mautz et al. (1976) compared digestibility among seven plant species eaten by white-tailed deer in the northeastern United States. They found higher levels of digestible energy in hobblebush, eastern hemlock, and balsam fir than in red maples, striped maple, mountain maple, or hazelnut. Despite these differences, white-tailed deer often feed heavily on maples in the winter in New England (Mautz et al. 1976) so factors other than simply digestible energy may be coming into play. Some plant species contain high levels of phenols that reduce their digestibility for many herbivores (Sinclair and Smith 1984, Friesen 1991). Plants that produce high levels of phenols gain some protection against herbivory, but the coevolution of plants and herbivores has resulted in plant defense mechanisms that are less effective for herbivores such as deer. In the battle of coevolution, mule deer have evolved to produce saliva that contains a substance (prolene) that binds with the phenols and reduces their effectiveness (Robbins et al. 1987, Austin et al. 1989). Other chemicals such as lignin and cutin can



**FIGURE 4.3** The change in taxa abundance or community diversity along an ungulate density gradient. Curve A is an idealized representation of taxa or communities that are adversely affected by browsing, and curve C represents taxa or communities that benefit. Curve B represents taxa or communities that benefit from intermediate ungulate densities. (Reprinted from *Forest Ecology and Management*, 181, Rooney T.P., and D.M. Waller, Direct and indirect effects of white-tailed deer in forest ecosystems, 165–176, Copyright 2003, with permission from Elsevier.)



**FIGURE 4.4** Plants with lower lignin and cutin have higher levels of digestibility than those with low levels for elk. (Redrafted from Hanley, T.A., C.T. Robbins, and D.E. Spalinger. 1989. Forest habitats and the nutritional ecology of Sitka black-tailed deer: A research synthesis with implications for forest management. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-230.)

influence digestibility as well. Plants with high content of lignin and cutin have lower digestibility than plants low in these compounds (Figure 4.4).

While some plants have chemical and physical defenses against herbivory (Farentinos et al. 1981), plants also respond to herbivory by altering growth rates. Among many grasses, forbs, and some shrubs, moderate levels of herbivory can actually stimulate growth above the levels of either undisturbed or heavily grazed or browsed plants (Belsky 1986, du Toit et al. 1990). It is widely assumed that browsed plants exhibit compensatory growth at the expense of reproduction and that herbivory, therefore, results in decreased seed production or smaller seed sizes (Belsky 1986).

Herbivores alter forest systems in ways other than consumption. They aid in the dissemination of seeds, and they may help to maintain site quality. Some plants are well adapted to dispersal on animals (e.g., bedstraws). Other plant species (e.g., dogwoods and cherries) are well adapted to scarification that results from passing through animal digestive systems and “direct-seeding” in a packet of fertilizer. Consequently, many fencerows are dominated by cherries, hawthorns, and dogwoods because birds often perch on fences after eating the fruits of these plant species.

Specialized herbivores that feed on fungi, called mycophagists, also play a role in ecological processes in forests. Through symbiotic relationships, mycorrhizal fungi aid vascular plants in the uptake of water and nutrients, and they can be particularly important to early plant growth and survival on harsh sites (Perry et al. 1989). Unlike most other fungi, these fungi produce fruits underground, and they do not rely on aerial spore dispersal as do other fungi. They seem, instead, to be well adapted to animal dispersal. Some fungi known as truffles are important components of the diets of some small mammals, particularly red-backed voles in the United States and woylies in Australia (Maser et al. 1978, Taylor 1992, Figure 2.5). These animals eat fruits and ingest spores, which then pass through the digestive system in a few days and are deposited at a new site. A new fungal mat may then grow from this site and ensure the presence and widespread distribution of mycorrhizae in the soil (Cork and Kenagy 1989). Mixing organic matter in the soil by these burrowing animals also is likely to influence decomposition rates and influences soil processes (Maser et al. 1978).

The activities of some herbivores can have tremendous impact on habitat for other species (Naiman 1988). The activities of American beaver, for example, create early successional riparian forest patches and pools in the stream that can be important to other species. For example, Suzuki and McComb (2004) found very different amphibian and mammal communities associated with areas in the Oregon Coast Range that were impacted by beavers compared with similar areas where

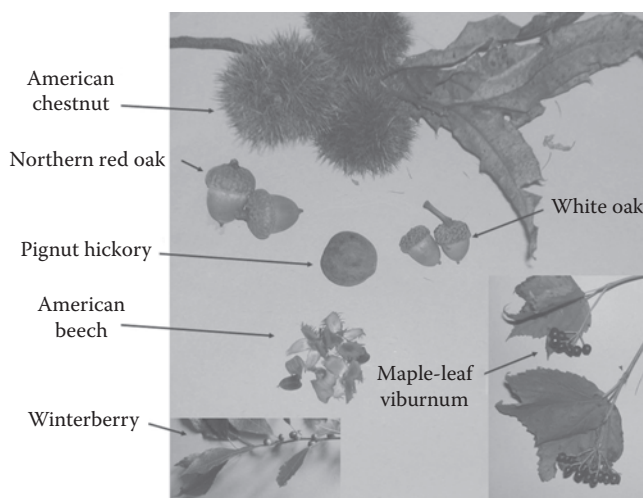


beavers did not build dams. Other examples include black bears that kill patches of trees in plantations, pocket gophers that eliminate regeneration in patches, or elk herds that browse heavily next to riparian zones. All these activities create patchiness or heterogeneity in affected sites, and such patches can be the important resource areas for other species.

## FRUIT PRODUCTION

Fruits that are produced in forests by woody and nonwoody plants provide a key food resource for many species of animals. *Hard mast* are those hard fruits that are produced annually but tend to be highly variable in their production (Healy et al. 1999). Seed production usually is greatest in large, open-grown trees. Generally, those plants in full sunlight with large crowns more regularly produce larger mast crops, but year-to-year variability is high (Healy et al. 1999). Providing a variety of hard mast-producing species in the stand may help to compensate for the variability in fruit production within any one species. For instance, the oaks in the United States are grouped into two subgenera: the white oaks (*Leucobalanus*) and red oaks (*Erythrobalanus*) (Figure 4.5). White oaks flower and are fertilized in the spring, the acorn matures in one growing season, and falls to the forest floor and germinates at the end of the growing season. Red oaks flower and are fertilized in the spring, but the acorn takes two growing seasons to mature before it falls to the forest floor. It then passes through a winter stratification period before germinating the following spring. Red oak acorn production is delayed 1 year after fertilization compared with white oak acorn production. If both red oaks and white oaks occur in a stand, then a late frost that kills flowers in one spring may affect white oak acorn production that fall, but red oaks may still produce abundant acorns from flowers fertilized during the previous year. Similarly, providing a variety of other hard mast-producing species such as hickories, beech, walnuts, and hazelnuts further reduces the risk of a complete mast failure in any one year. Unfortunately, one of the most reliable mast producers once dominant in eastern U.S. forests, American chestnut is now only a stump sprout in our forests due to the chestnut blight fungus. There is some hope that genomic approaches may lead to a recovery of American chestnuts once again (Wheeler and Sederoff 2009).

*Soft mast* are soft fruits such as berries and drupes. These food sources are high in energy and used by many animal species. In a study in South Carolina, McCarty et al. (2002) found that 50% of fruits on 17 species of plants were eaten by vertebrates. Since different trees and shrubs flower and produce fruit at different times of the year, again a variety of species is important. For instance,



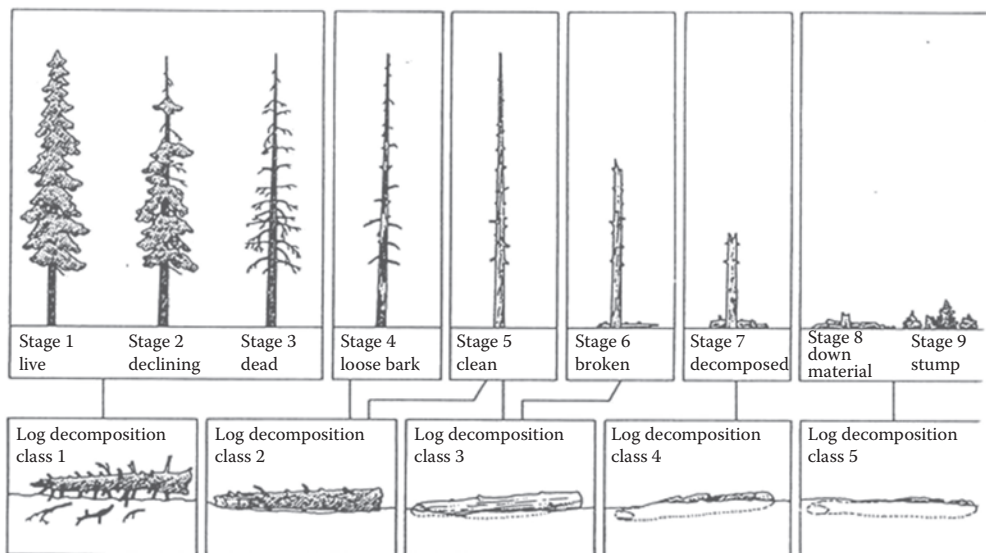
**FIGURE 4.5** Examples of hard and soft mast foods for vertebrates.

serviceberries produce fruits early in the growing season, viburnums in late summer, and hollies retain fruit into the winter (Figure 4.5). These food sources may be particularly important in the winter when other digestible foods are in short supply. Soft mast production is greatest in most species where the fruit-producing plants are receiving full or nearly full sunlight (Wender et al. 2004). Plants in partial shade often allocate most energy to growth and not to fruit production. Consequently, providing patches of forest where sunlight can reach these plants may increase food availability and quality (Perry et al. 1999).

## DEAD AND DAMAGED TREES

Trees provide a basis for food and cover resources for various species while they are alive and growing. The value of trees to some species wildlife extends well beyond this period, however, and for many species, the value of a tree only begins after the tree has died. Dead trees in various stages of decay offer sites for nesting, resting, and foraging for many species of vertebrates and invertebrates (Figure 4.6). Species vary in their use of dead wood size and decay classes (McComb and Lindenmayer 1999). Those species that use standing dead trees, or *snags*, are often separated into two groups. *Primary cavity nesters* are those species that can excavate a cavity in dead wood or trees with heart rot decay (Figure 4.7). Woodpeckers are the best example; indeed, most woodpecker species must excavate a cavity in a tree or snag before they will complete the nesting ritual (for instance, they will usually not nest in nest boxes unless the box is filled with sawdust so that they have something to excavate). *Secondary cavity users* use cavities that were created either by primary cavity nesters or in natural cavities (Figure 4.7). This group of species can be extremely diverse and includes parrots, tree frogs, tree-hole mosquitoes, and black bears. All of these species rely on either primary cavity nesters or trees with natural cavities for survival.

Snags go through a process of decay that allows primary cavity nesters to excavate cavities. Snags that are not well decayed are still hard wood and only a few species of woodpeckers can excavate in these snags. As the snags decay and become soft snags, other species can then excavate



**FIGURE 4.6** Stages of decay of trees and logs. As wood decays, the types of food and cover change for various species. (From Maser, C., R.G. Anderson, and K. Cromack, Jr. 1979. *Wildlife habitats in managed forests: The Blue Mountains of Oregon and Washington*. Pages 78–95. USDA For. Serv. Agric. Handb. No. 553.)





**FIGURE 4.7** Primary cavity nesters have used this snag (left), and these cavities are now available to secondary cavity nesters. Secondary cavity nesters can also use cavities created by fungal decay, such as the one in this live tree used by porcupines (right).

in the snags. If snags of both types are provided in a stand, then there are more potential nest sites for more species. Snags are particularly important in conifer forests. Hardwood forests, especially with large trees, often have large dead limbs that provide many of the same benefits as snags.

Tree species vary in their propensity to decay following the death of a tree or a wound to a live tree. Some species tend to be more prone to forming *natural cavities* that result from tree injuries. Many of the conifers (e.g., pines) are poor producers of natural cavities. Tree cavities are important den sites for secondary cavity-using species (those species that cannot excavate their own cavity as woodpeckers do). Hollow trees are formed through top breakage and subsequent heart rot. Large hollow trees are especially important for species such as fisher, bears, and some species of bats and swifts. Species such as Oregon oak, Pacific madrone, and bigleaf maple in the western United States and red maples and blackgum in the southern United States seem to produce many natural cavities and dead limbs (effectively, elevated snags) that are used by cavity-nesting animals (McComb et al. 1986, Raphael 1987, Gumtow-Farrier 1991).

Fallen logs also provide cover and nesting sites for a wide range of species, including many amphibian and reptile species, small mammals, and a few species of ground nesting birds (Butts and McComb 2000). Large logs provide more cover and nesting opportunities for more species than small logs so the production of large trees that can fall to the forest floor should be given consideration during silvicultural activities (McComb 2003). Hollow logs can only occur if a hollow tree falls to the ground (they do not decay into hollow logs after they have fallen); so retention of some decaying hollow trees in a forest is necessary to provide hollow logs. Trees that fall into streams and lakes also play a role in habitat quality for many aquatic and semiaquatic species (Naiman et al. 2002). Logs in a stream divert water and cause pools either from plunging over the log or from scouring under the log. Dead wood also provides cover for fish and amphibians and is used as a substrate upon which some salamanders lay their eggs. Large logs also are often the basis for a beaver dam in a stream. More information on managing dead wood is provided in Chapter 12.

### TREE SPECIES AND INVERTEBRATE ASSOCIATIONS

MacArthur and MacArthur (1961) also found that BSD is associated with plant species diversity within stands, probably because of the additional niche space provided in stands with more plant species. Vertebrates that feed on invertebrates associated with the leaves and needles of trees and shrubs

select certain species over others for feeding (Holmes and Robinson 1981, Muir et al. 2002). Selection is probably dependent on the food resources available, competition among species, and the foraging adaptations of each species. Insect abundance and species richness tend to be higher in hardwood than conifer stands, but clearly there are species well-adapted to gleaning insects from both types of trees (Muir et al. 2002). Hardwood composition in conifer stands is associated with the abundance and occurrence of several bird species in the northwest United States (Huff and Raley 1991).

Tree species also vary in their ability to support bark-dwelling insects, an important overwintering food supply for some bird species (Mariani 1987). Rough-barked trees provide more cover for these insects and support higher insect biomass than smooth-barked trees (Brunell 1987) (Figure 4.8).

### Tree Size and Density

Trees of different sizes play various roles as vertebrate habitat elements in stands. Seedlings provide browse for deer (much to the chagrin of some foresters!), nest sites for shrub-nesting birds, shade for forest floor amphibians, and hiding cover for many species of birds and mammals. Saplings provide browse for larger herbivores such as moose, and pole-sized trees may provide cover for ungulates. Large trees, especially those that grow beyond marketable size, can significantly influence the quality of a stand as habitat for some species. Trees in some managed stands are designated as legacy trees and left to grow to maturity and die through natural processes (Carey and Curtis 1996). For instance, Douglas-firs >125 cm (50 in.) in dbh (diameter at breast height) are used by marbled murrelets (Singer et al. 1991), red tree voles, and northern spotted owls. Large trees also add to the vertical structure within forests.

Large trees that add large surfaces of deeply fissured or scaly bark are used by bark-foraging birds such as brown creepers (Mariani 1987) and they support lichens, an important food source for species such as northern flying squirrels (Martin 1994). Designating a variety of tree species as legacy trees in forests would provide a range of growth rates and bark surfaces and contribute to complexity in the stand. These legacy trees provide an ecological link to the previous stand structure and composition.

Tree density also influences the production of many habitat elements. Dense stands with many trees may exclude sunlight from the forest floor, producing an open sub-canopy condition. Sparsely stocked savannah stands leave much sunlight and moisture available for grasses, forbs, and browse. Manipulation of stand density is probably the most significant influence that a forester can have on habitat availability for a wide variety of species (Carey 2003).



**FIGURE 4.8** Example of a rough-barked tree (white oak, left) that supports higher densities of overwintering invertebrates than a smooth-barked tree (red maple, right).

## FOREST FLOOR LITTER AND SOIL

Forest floor and below-ground conditions influence habitat quality for ground-foraging and burrowing species. The type and depth of leaf litter have been shown to be associated with the community structure and abundance of invertebrates (Bultman and Uetz 1982). Consequently, the leaf litter characteristics are associated with species that find food or cover on the forest floor, such as ovenbirds in the northeastern United States (Burke and Nol 1998). Insectivorous mammals also are assumed to be associated with litter type and depth. Leaf litter has been adversely affected by several invasive nonnative species, such as earthworms that can cause a reduction in leaf litter on the forest floor, and the loss of leaf litter is associated with declines in species such as amphibians. Maerz et al. (2009) reported that this effect may in part be due to declines in populations of small arthropods, which are an important food source for the salamanders.

Some terrestrial mammals and amphibians remain active below-ground during the summer. For instance, red-backed salamanders use earthworm burrows (Caceres-Charneco and Ransom 2010), and rough-skinned newts use logs and burrow systems of voles and shrews as summer daytime refuges (McComb and Chambers, unpublished data). Burrow systems of mountain beavers, gopher tortoises, and pocket gophers are used by many other species (Maser et al. 1981, Figure 4.9). These below-ground conditions are often not considered during forest management so the next time that you walk through a forest, think of the unseen animal community that lives beneath your feet.

### Proximity to Water

Intermittent and permanent streams, seeps, springs, vernal pools, ponds, swamps, marshes, and lakes all provide water in a setting that can be critical to habitat quality for many species of aquatic and semiaquatic organisms. Although we have little control over how close a stand or forest is to water, we do have control over the function of the water body as habitat for a variety of species. For aquatic and semiaquatic species, the temperature, sediment load, and chemical concentrations in water may be influenced by the surrounding forest. Trees and shrubs over the water influence the temperature of the water by providing shade, by the influx of nutrients through litter fall, and by the degree of erosion through root strength. Some species of amphibians require clear, cold water for survival and have greatest fitness in water bodies where there are no fish (predators) (Lowe and Bolger 2002). These non-fish-bearing streams often are overlooked as a potential habitat for animals because they may be dry at some times of the year and may appear no different than the surrounding uplands. In many settings, especially on federal lands in the United States, buffer strips are provided to retain habitat for species associated with these sites (Vesely and McComb 2002). More information on managing riparian areas is provided in Chapter 11.



**FIGURE 4.9** Gopher tortoises are associated with certain soil conditions and their burrows are used by a wide variety of other species. (Photo by Mike Jones. With permission.)

## **CASE STUDY ON PLANT RESPONSE TO HERBIVORES, OR, IT IS A (CHEMICAL) WAR OUT THERE!**

Some plants produce chemicals in their leaves and twigs that reduce consumption by herbivores (herbivory). Others produce spines, thorns, and physical barriers to herbivory. Consequently, the effect that herbivores have on plant communities can be altered depending on the ability of the plants to cope with or avoid being eaten. A well-known herbivore, beaver, cuts trees of a range of sizes to feed on the bark and to use in building dams. They are selective of certain sizes and species of plants that they cut and so influence the riparian forest composition and structure considerably. A study by Martinsen et al. (1998) examined interactions between beaver, leaf beetles, and cottonwoods. Cottonwoods felled by beavers sprout vigorously, and these sprouts contain higher levels of defensive compounds than the original stem that was cut. This is an important chemical strategy for the cottonwood because it can repel generalist herbivores (those that eat a wide variety of plants), but not a specialist insect herbivore, the leaf beetle, that sequesters these chemicals for its own defense. Martinsen et al. (1998) found 15 times as many adult beetles on resprouts following beaver cutting as on uncut cottonwoods. Resprout cottonwoods have twice the concentration of phenolic chemicals as uncut stems. Several indices of beetle fitness were also higher on resprout growth than on uncut cottonwoods. This is fascinating but what does this have to do with managing habitat in forests? There are several implications from this work. Phenols are highly toxic to some mammals. As little as 1 g of some phenolic compounds can kill a human (Budavari 1989). Phenolic concentrations are often higher in juvenile plants than in mature plants. For instance, snowshoe hares prefer to feed on mature willows and poplars, avoiding juvenile trees. Through plant breeding or nursery practices, if nursery stock can be developed that contain higher levels of phenolic compounds, then seedling damage by herbivores could be reduced. But it may not be that simple. Deer saliva contains a glycoprotein that has large amounts of proline, glycine, and glutamate/glutamine that binds with tannins, and potentially other defensive compounds, to reduce the effectiveness of these compounds. So although increasing defensive compound concentrations in seedlings may be a reasonable strategy to reduce some forms of herbivory, it may not work well for all herbivores.

In addition, this study points to the fact that browse is not browse is not browse. Species vary in their production of these defensive compounds and hence in the quality of browse for herbivores. And individual plants vary in production of these chemicals depending on if they have already been browsed, if they are growing rapidly or slowly, or growing in shade or sun (Martinsen et al. 1998). Consequently, we can create literally tons of browse per hectare following a clearcut, but if the species composition is such that the resulting browse is of low quality then herbivores may be at a disadvantage eating this browse compared with another site with higher quality, but less quantity of browse.

## **SUMMARY**

Managing habitat for vertebrates in forests often entails manipulating a set of habitat elements that are important to many species. The sizes, density, and distribution of plants; vertical structure; horizontal complexity; forage; dead wood, large trees; leaf litter; soil; and water contribute to habitat quality for many species. Habitat is not just vegetation but includes soils, water, and below-ground structure. Managers have control over the structure and composition of vegetation; so by manipulating the density, sizes, and distribution of trees and shrubs in a stand, foresters can have a tremendous influence on the availability of these habitat elements to vertebrates. Further, manipulation of vegetation can also influence the quality of these habitat elements. Browse resources that are high in lignin, cutin, phenols, and tannins reduce digestibility for many herbivores. Managing in a way that provides not only abundant browse resources but high-quality browse resources can have the biggest benefit to ungulates. Similarly, providing large pieces of dead wood or large decaying trees and stands representing a range of vertical and horizontal complexities can also benefit a wide variety of species.



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# 5 Physical Influences on Habitat Patterns

Habitat elements, those pieces of the environment that in certain sizes, numbers, and distribution influence habitat quality for vertebrates, are not uniformly distributed across stands and forests. Some stands have high vertical diversity and others are rather simple in their vertical structure. Some support a deep litter layer and others virtually none. Browse quantity and quality vary tremendously depending on the plant species, growth rates, shade, and past browsing. So what is it that influences these patterns of habitat elements? In this chapter, we will explore the physical factors influencing patterns of habitat elements.

Although habitat is not simply vegetation, vegetation is shaped by natural disturbances and is the part of the environment that we can influence by silviculture and harvest planning. The patterns and dynamics of vegetation have a significant effect on the patterns and dynamics of habitat elements. Most of this chapter will focus on the physical processes influencing the pattern of vegetation and habitat elements associated with vegetation, and then in the Chapters 6 and 7 we will address how cultural features and disturbances change those patterns.

## THE PHYSICAL ENVIRONMENT

Probably the greatest overriding effects of the physical environment on habitat quality for many species are those of geology, climate, soils, and topography on vegetation patterns. Associations between the physical environment and vegetation patterns are complex and they vary regionally. These associations have a significant influence on patterns of habitat elements such as vertical complexity, horizontal complexity, and forage resources. In the following subsections, we will cover both direct and indirect effects of physical factors that influence habitat quality for vertebrates. Although major factors are presented independently, it should be clear that all of these factors interact to a greater or lesser degree to influence habitat quality for species.

## GEOLOGY

Vegetation structure and composition are often highly associated with the underlying geology. The glacial outwash materials of the northeast, the volcanic basalts of the Cascades, and the erosion of sedimentary rocks all interact with climate and animal activities to give rise to conditions for different plant communities. For instance, in the Klamath-Siskiyou Province of northwestern California and southwestern Oregon, diverse geologic conditions and soils produce a rich array of plant communities (Coleman and Krukeberg 1999). Soils developed from serpentinized rock may be relatively enriched in various toxic metals, including nickel, magnesium, barium, and chromium, and lacking in important nutrients such as calcium (Smith and Diggles 1999). Consequently, serpentine geology provides the basis for a unique plant structure and composition that includes Jeffrey pine savannas, xeric shrub types, and serpentine barrens, as well as hygric *Darlingtonia* fens (Coleman and Krukeberg 1999). Vegetation structure and composition, and habitat element characteristics are also related to surficial geology and topography over larger areas. For instance, the complex surficial geology and topography of northeastern Ohio created a variety of conditions favoring pre-settlement forests composed of American beech and maples on some sites and oaks and hickories

on others (Whitney 1982). The structure of these forests and their ability to produce mast, browse, and other resources varies in part because plant species composition differs between sites with different geologic histories, slope, and aspect. The relationship between geology and plant associations is not always direct. Hypotheses were generated years ago that plant species richness (the number of plant species found in a particular area) is low in areas with low net primary productivity (in some cases due to nutrient restrictions) and also at high levels of productivity (where competition allows the most successful species to dominate) and is highest in areas of intermediate net productivity. But this hypothesis is not well supported; Adler et al. (2011) examined patterns among 47 plant communities on five continents and found no consistent relationships between net primary production and plant species richness. Of course, soil fertility and underlying geology are only two aspects of net primary production (the ability of a plant community to accumulate carbon), but the message is clear that there are many interacting processes driving the number of plant species on a site.

In addition to the effects that geology has on topography and soil formation, the interaction of geology with water can lead to direct effects on habitat for some species. Richards et al. (1996) reported that physical characteristics such as catchment area, clay soils, and glacial outwash materials had strong influence on stream habitat structure as some land use activities. Wilkins and Peterson (2000) related amphibian occurrence and abundance in headwater streams draining second-growth Douglas-fir forests to two geologic substrates: sedimentary and basalt formations. Streams traversing basalt had almost twice the abundance of the Pacific giant salamanders as streams flowing over marine sediments. They concluded that habitat quality for headwater amphibians in western Washington was strongly influenced by basin geology. In parts of North America, karst geology often occurs in association with limestone formations, or other carbonate rocks that dissolve more easily than surrounding rock formations (Figure 5.1). These karst geologies are prone to formation of caves, sink holes, and subterranean water flow. In Alaska, some species seem to seek karst features and the stable environment provided within caves (Baichtal 1993). Caves are used by a wide variety of species (Blackwell and Associates, Ltd. 1995). Deer are known to rest in the vicinity of



**FIGURE 5.1** Locations of karst geology in the United States. These areas are prone to caves and subterranean streams and often require special consideration during forest management. (Reprinted from the National Cave and Karst Institute, USDI National Park Service.)

caves during both the summer, when air from the caves is cooler, and in winter, when cave entrance air is generally warmer (Blackwell 1995). Cave systems also are used by many species of bats for roosting and hibernation. Cave environments provide specific air circulation patterns, temperature profiles, humidity, cave structure, and locations relative to feeding sites that some species of bats require (Hill and Smith 1992).

Although manipulation of vegetation does not directly influence geological features such as caves and headwater streams, the microclimatic characteristics of these environments near the surface of the ground can be altered by forest manipulation. For instance, these associations provide managers an opportunity to consider headwater amphibian conservation strategies by prioritizing stream segments based on geology with respect to their likely amphibian fauna and providing more or less shade to these systems depending on the needs of the species being managed.

### TOPOGRAPHY: SLOPE, ASPECT, AND ELEVATION

Vertebrates use behavior to modify the ambient temperature around them and move themselves closer to their thermal neutral zone as they seek appropriate thermal cover. Invertebrates also seek topographic features when selecting habitat (Weiss et al. 1988). Quite often thermal cover may not only entail vegetative cover but also be located on particular aspects and slope positions that provide warmth or cooling. Variation in slope and aspect affect the spatial variability in solar radiation incident on the ground and also ground surface temperature and wind speeds (Porter et al. 2002). These areas of slope and aspect can particularly influence habitat quality for ectotherms (those species that regulate their body temperature by their behavior) (Thomas et al. 1999). Endotherms also can use behavior to modify their ambient environment. For instance, Pearson and Turner (1995) studied ungulates (deer, elk, and bison) in Yellowstone National Park and found that grazing occurred most often in burned areas at low elevation, drier sites, and on steep southerly slopes. In addition to the direct effects of slope and aspect on ambient temperature, forest vegetation can moderate extreme temperatures on those sites receiving direct sunlight, or lack of vegetation can cause cooler sites to be warmer in the day and cooler at night. Slope and aspect in combination with vegetation can influence snow depths and winter habitat quality for some species of ungulates (D'eon 2001), and large mammals will use topography as a form of hiding cover (Ager et al. 2003, Sawyer et al. 2007). Silvicultural activities that consider slopes and aspects as potential thermal environments or hiding cover for some species can significantly influence habitat quality for these species.

Volcanic extrusions in combination with glacial activities have left cliffs (nest sites for common ravens and peregrine falcons), caves (used by some bat species for hibernacula), and a diverse topography producing vegetation gradients over relatively small areas. Distributions of some animal species also follow the complex topographic features that resulted from glacial recession. For instance, the distribution of Bicknell's thrush is highly associated with the higher elevations in the Berkshires, Green, and White Mountains, Adirondacks and Catskills that generally represent rock formations resistant to the forces of the past glaciers.

Some species also benefit from improved nutrition in areas with certain topographic characteristics. The body weight of red deer in Norway was correlated with access to the variability in topography and aspects (Mysterud et al. 2001), likely due to the variability in plant species represented over a relatively small area. Vegetation patterns often are significantly influenced by changes that occur along elevational gradients. Changes in elevation of less than a meter influence vegetation patterns in bottomland hardwood forests (Wall and Darwin 1999). In mountainous terrain of the western United States and Canada, vegetation patterns are significantly structured by patterns of precipitation and temperature that change with elevation. As air is moved over mountains, it increases in elevation, cools, and moisture precipitates from the air as rain or snow. This *orographic precipitation* produces marked patterns in vegetation, which in turn influences the occurrence and abundance of some vertebrates (e.g., Terborgh 1977). For instance, the distribution of several species of reptiles and amphibians, such as Cascades frogs are highly associated with elevation, temperature,

and snowfall in the Cascades Mountains. Some species of *Anolis* lizards also seem to separate along elevational gradients in tropical forests (Buckley and Roughgarden 2005). Elevation also can interact with other physical variables such as ultraviolet radiation due to ozone depletion (UV-B) to influence abundance of some species. Declines in California red-legged frog populations exhibit a strong positive association with elevation, as well as with percentage of upwind agricultural land use (i.e., chemicals), and local urbanization (Davidson et al. 2001). These declines in frog abundance along an elevational gradient are consistent with increased levels of UV-B radiation.

## SOILS

Maintaining long-term soil productivity in managed forests is critical to producing wood products and habitat elements (Fox 2000). Given the short-term positive effects of forest fertilization (Brockway 1983) and negative long-term effects of soil compaction (on certain soils) (Kozlowski 1999) on tree growth, it is easy to see that soil structure and composition are critical to sustaining various ecosystem services (Knoepp et al. 2000). Bedrock and surficial geology interact with climatic variables and hydrology to produce soils varying in their ability to support growth of certain species and clearly alter growth rates of trees. On sites where trees grow faster, large trees and vertical complexity can develop more quickly. Soil structure, nutrient composition, moisture content, and temperature clearly influence the vegetation species composition and quality of habitat elements such as tree species composition and browse quality. For instance, the chemical constituency of flowering dogwood and red maple foliage changes over subtle gradients of soil moisture and nutrient availability such that the production of phenolic compounds was highest on sites of greatest plant stress (Muller et al. 1987). Soils can also have direct effects on vertebrate habitat quality. The minerals available in some soils can meet direct dietary needs. Pabian and Brittingham (2011) found a positive relationship between soil calcium levels and ovenbird territory density, clutch size, and nest density in Pennsylvania forests. Species such as moose, deer, caribou, and band-tailed pigeons seek sodium-rich soils and springs to meet demands for this nutrient (e.g., Kennedy et al. 1995, Sanders and Jarvis 2000). Areas of eucalypt forest in Australia on high nutrient sites have trees with higher foliar nutrients, making them more suitable for some arboreal marsupials (Pausas et al. 1995). This relationship presents a challenge to forest managers wanting to maximize wood production and protect high-quality habitat for these species.

Burrowing animals are clearly affected by soil structure. As an example of the extensive nature of burrow systems, Askram and Sipes (1991) found approximately 16,000 holes and 7.5 km (4.7 miles) of above-ground runways per 0.4 ha (1 acre) created by voles in an orchard in the northwestern United States. Gopher tortoises are adept at making burrows in sandy coastal plain soils (Ultsch and Anderson 1986, Figure 4.9), Gapper's red-backed voles in boreal forests develop elaborate networks of burrows, and mountain beaver develop extensive burrow systems used by many other species. Regosin et al. (2003) suggested that small mammal burrow densities could influence abundances of spotted salamanders in northeastern U.S. forests. Ground-based logging equipment and other forest management activities can disrupt soil structure and reduce burrow availability. Protection of soils from disruption, compaction, and erosion can be a key step in providing habitat for many species that spend their time below ground.

## CLIMATE

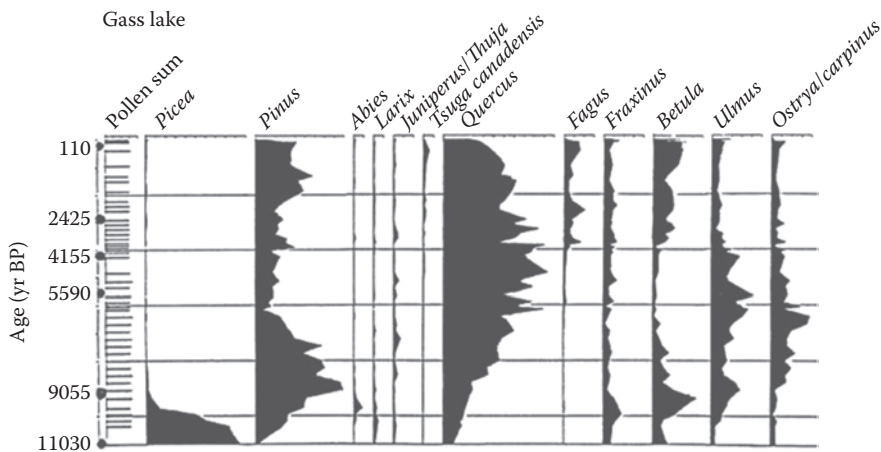
Climatic conditions are a predominant force influencing the distribution of organisms. Temperature and precipitation, separately and in combination, have significant effects on vegetation patterns. The physical environment of North America has been altered by climatic processes extending hundreds of thousands of years into the past. Most notably glaciation and subsequent climate change have influenced surficial geology, soils, and hydrology over large parts of North America. For instance, about 12,000 years ago, there was a 2-km-thick chunk of ice sitting over much of the northern

United States and Canada. At that time, the sea level was considerably lower and forests once grew on parts of what is now the continental shelf. As climates changed, the receding glaciers left deeply scoured valleys, outwash plains, wind-blown deposits of sands, drumlins, and erosion-resistant ridges of granite and basalt. The distribution of tree species found in forests then was much different than it is today and patterns of forest species changed markedly as the glaciers receded (Figure 5.2, Jacobson et al. 1987). The soils that were left behind as glaciers receded represented a complex mosaic of sands, gravels, and clays that structures vegetative communities quite markedly.

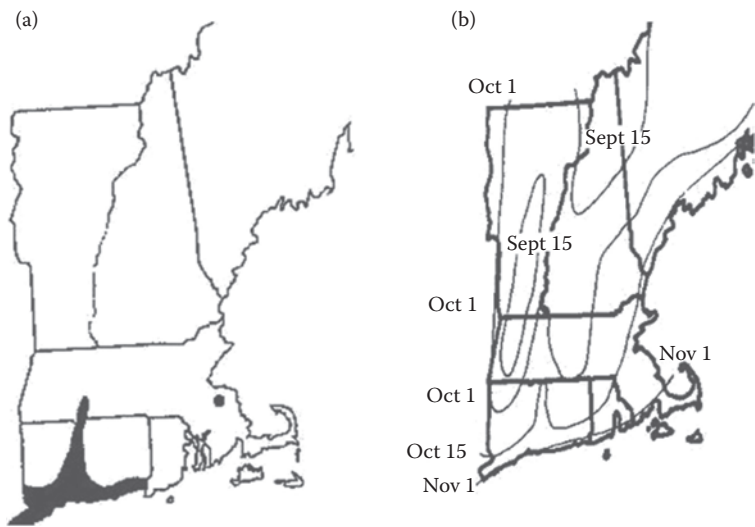
Climatic conditions not only influence the vegetation patterns of a region but also directly influence the ability of species to survive in an area. Consider the geographic distribution of the northern copperhead in New England (Figure 5.3a). The northern limit of the geographic range for this species, an ectotherm sensitive to prolonged cold, closely corresponds with the mean dates of the first and last frost (Figure 5.3b).

Indeed many species of ectotherms are influenced by air, soil, or water temperatures in forests (e.g., Nussbaum et al. 1983, Welsh and Lind 1996). Some endotherms are also highly influenced by changes in ambient temperature. Temperature at roosts and maternity sites of some bat species seems to be particularly important for survival and growth of adults and young (Agosta 2002). In fact, there is even evidence that sex ratios of bats and some reptiles can be influenced by ambient temperatures (Ewert et al. 1994, Ford et al. 2002). Consequently, considering the effects of forest management on temperature regimes in forests can be quite important to providing adequate habitat for species such as these.

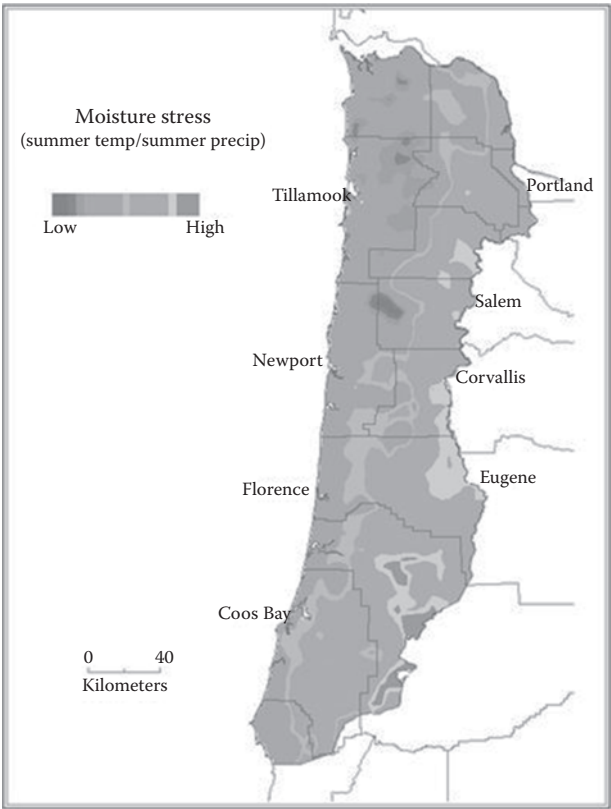
Moisture plays a critical role in structuring the patterns of vegetation and often is more important than temperature in structuring the regional environment (e.g., the Pacific Northwest). Clearly, precipitation and temperature are interrelated, particularly as they influence moisture stress on plants. Consider the map of moisture stress in Figure 5.4 for western Oregon. Moving from west to east, patterns of tree dominance changes from Sitka spruce and western hemlock in areas with low moisture stress, to Douglas-fir and western hemlock, to Douglas-fir and grand fir to Oregon oak and grand fir, as you move east along this gradient. Moisture stress seems to be an important factor associated with vegetation patterns over much of the northwest (Zobel et al. 1976, Ohmann and Spies 1998) and possibly the continent (Goward et al. 1985). Under predicted



**FIGURE 5.2** Patterns of tree pollen from the sediments in a Wisconsin Lake. Note that as spruces declined about 10,000 years ago, pines increased in dominance, until about 6000 years ago, and then oaks dominated. American beech did not begin to have a significant presence in the area until about 4000 years ago. (Reprinted from Webb, S.L. 1987. *Ecology* 68:1993–2005. With permission of the Ecological Society of America.)

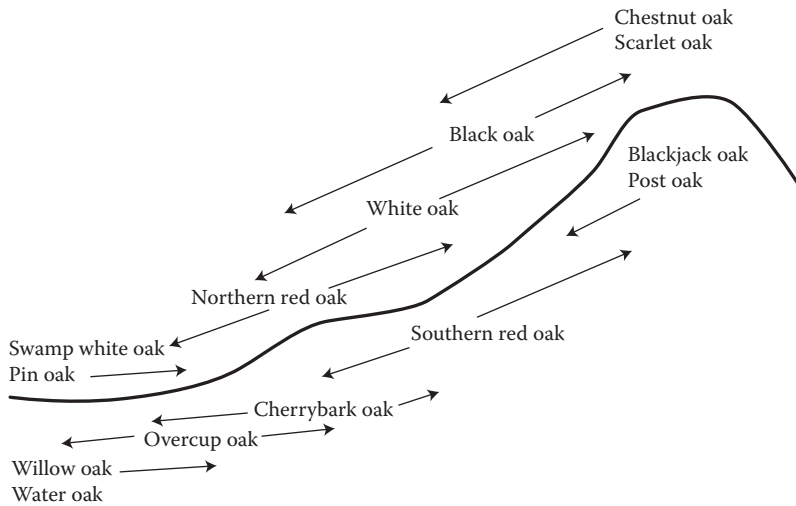


**FIGURE 5.3** The geographic range of northern copperheads (a) and the isotherms for date of first frosts (b). (Reprinted from DeGraaf, R.M., and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution*. University of Press of New England, Hanover, NH. With permission from the University Press of New England.)



**FIGURE 5.4** Map of moisture stress in the Oregon Coast Range. Plant communities seem to be structured in large part by this variable. (Reprinted from a figure developed by the USDA Forest Service Pacific Northwest Research Station Coastal Landscape Analysis and Modeling Study.)





**FIGURE 5.5** Gradient of oak species along a topographic and moisture stress gradient in the eastern United States.

climate change, increased moisture stress can lead to significant shifts in tree species distributions (Coops et al. 2011).

Local patterns of vegetation are also driven by moisture stress gradients (Figure 5.5). In the eastern United States, oak species segregate along topographic features that are related to a moisture stress gradient. Sites with standing water are dominated by pin oak and swamp white oak, and as you move upslope, these are replaced with northern red oak and white oak, then black oak, and finally chestnut oak and scarlet oak at the ridgetops. The ability of a forest to produce mast is highly influenced by site conditions that reflect moisture stress. Further it is important to recognize these patterns when regenerating forests during management. Matching plant species to sites is a key step in successfully regenerating a stand.

Precipitation also influences soil moisture and hydrology for an area (see previous and following sections) and consequently plays a key role in structuring habitat quality for many species. Recently though, there also have been noticeable effects of acid precipitation on habitat quality for species such as amphibians and waterfowl (Stenson and Ericksson 1989). The effects of acid deposition on vertebrates are complex. There are direct effects of reduced pH such as reduced reproduction or survival of fish and probably aquatic amphibians. There also are indirect effects, such as a shift of top predators from fish to invertebrates and a reduced decomposition rate due to decreased abundance of animals that contribute to the decomposition of organic matter by consuming wood, leaves, and other plant materials (detritivores) (Stenson and Ericksson 1989). In addition, productivity and turnover rate of nutrients can be reduced and there often is an increase in water transparency, which influences predation effectiveness. In addition, acid precipitation can have indirect effects on terrestrial systems through changes in vegetation structure and composition resulting from alteration of nutrient exchange capacity in the soils (Schreiber and Newman 1988).

There are also direct effects of rain (Waltman and Beissinger 1992) or snow (Kirchhoff and Schoen 1987) on habitat quality for many species. Many species seek shelter from precipitation due to the evaporative cooling effect on their bodies. Snow, in particular, can influence movements and choice of foraging and resting areas by deer and other ungulates (Nelson 1998).

In the past decade there has been increasing attention paid to the effects of climate change on conservation of biodiversity around the world. Predictions include significant changes in geographic ranges for species with narrow niches (a narrow range of conditions in which they can occur, especially at high latitudes and high elevations). But specific changes that we might see are difficult



to predict because changes are a result of not only changes in the physical environment but also range shifts of competitors, predators, and diseases. To complicate things even more, changes in climate may lead to conditions that we have not seen before in a specific location or in former analog conditions (Fitzpatrick and Hargrove 2009). How will a species that is strongly influenced by climatic conditions, many reptile species for instance, respond to entirely new climatic conditions? Uncertainty in predictions of species ranges increases with the degree to which a species is highly associated with specific climatic conditions and the nonanalog conditions depart from the historic conditions under which a species has evolved.

## HYDROLOGY

Watersheds influence the physical structuring of the environment and consequently habitat elements and habitat quality for a number of species. We will spend much more time addressing issues of riparian vegetation and management on habitat elements in a later chapter. Clearly though, the proximity of habitat elements to water and groundwater conditions can influence vegetation patterns markedly. Some tree species are very well adapted to growing in flood-prone or saturated soils (e.g., baldcypress, eastern cottonwood, and water oak). Considering management effects on the regeneration and growth of these species is key to effective management for many bottomland species. In particular, waterfowl managers in the southern United States have used greentree reservoirs to provide timber and waterfowl habitat (Wigley and Filer 1989). Greentree reservoirs allow water levels to be manipulated in order to flood bottomlands in the winter to attract waterfowl feeding on acorns. Water then is drawn down during the growing season to allow rapid growth of oaks that are then harvested for timber.

The hydrologic features of an area can also have more direct effects on some species. Seeps and steep headwater streams provide large boulder and rock substrates, cool water, and highly aerated water that provide habitat for species such as spring salamanders in the east and torrent salamanders in the west (Sheridan and Olson 2003, Olson et al. 2007). As the watershed area increases, headwater streams that may be intermittent become permanently flowing. If there are no barriers to fish movement, we may find brook trout in the eastern streams and cutthroat trout in western streams, which in turn influence the distribution of salamanders in those streams. As the gradient (slope) of the stream declines and water volume increases, we see sediments deposited and more meandering stream courses. These mid-watershed stream reaches provide the substrate for dens for muskrats and beavers. Farther along the gradient where wide valley floors and annual floods characterize the large river systems, backwater sloughs, swamps, side channels, and flooded wetlands become important nest sites for species such as wood ducks, great-blue herons, and western pond turtles. In fact, the large bottomland hardwood forests of the Mississippi River floodplain and Gulf coast provide habitat for many species, including the ivory-billed woodpecker, which may have been recently rediscovered (although that remains to be confirmed). Finally, as the river empties into an estuary, the interface between fresh and salt water provides a set of conditions that are ideal for many species such as American black ducks, muskrats, and many species of wading birds.

Kettle holes that resulted from the melting of large ice blocks embedded in glacial soils may allow the formation of vernal pools. Vernal pools hold water for a period in the spring, but dry out later in the summer. These sites are key breeding habitat for species such as marbled salamanders and wood frogs. Large mountain lakes that result from scouring from glaciers also provide habitat for species such as loons, Cascade frogs, and common goldeneyes. Since each of these hydrologic features provides habitat for a different suite of species, each must be considered differently when managing adjacent forest lands.

## VEGETATION PATTERNS

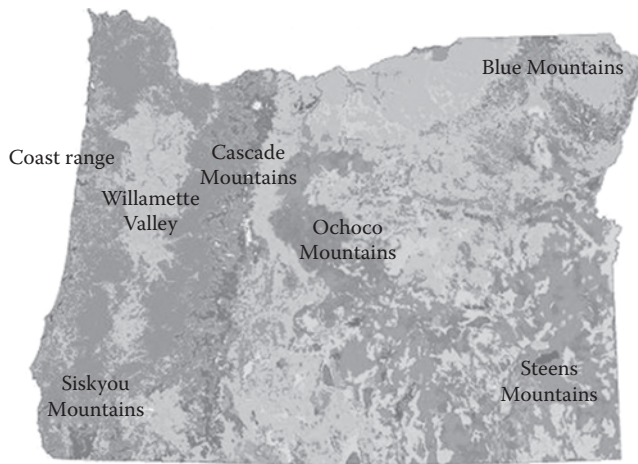
Vegetation patterns are associated with these physical features of the environment. The potential vegetation of a region represents the dominant vegetation that would be present in a region in the



**FIGURE 5.6** Potential vegetation of North America. (From Adams, J. *North America during the Past 150,000 Years*. Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN. <http://www.esd.ornl.gov/projects/qen/nercNORTHAMERICA.html>.)

absence of culturally induced changes (Figure 5.6). Clearly these vegetation patterns influence the patterns of habitat elements over the region. Wildlife habitat types have been delineated in many states and provinces to facilitate the understanding of historic and current patterns of vegetation and topography. For instance, in Oregon, 33 habitat types representing vegetation, water, and geologic features are a way of understanding how species might respond to these features (Figure 5.7). It should be clear by now, though, that these broad categories driven by physical forces may only be crudely related to the distribution of some species because every species has its own habitat requirements. Indeed, Cushman et al. (2008) found that habitat types were a very weak indicator of habitat for any particular species of bird. Indeed it is the interaction of individual plant species with the physical environment as well as the interactions with other species and other individuals of the same species that are better predictors of where a species might occur on a landscape.

Vegetation patterns that we see today were quite different historically. The changes in vegetation patterns resulting from glaciation and climate changes occurred relatively slowly in the past and



**FIGURE 5.7** The mosaic of historic habitat types for the state of Oregon, US. These patterns have been changed through recent cultural activities associated with land use. (Redrafted from a map produced by the Institute for Natural Resources, Oregon State University. With permission.)

encompassed thousands of generations of vertebrates. The slow rate of change usually allowed species to adapt to these changes. Habitat selection certainly also would have been influenced by these changes and led to genetic advantages for those individuals most adaptable to the new conditions. Habitat selection also would have facilitated the changes in vegetation pattern. Movement of heavy-seeded tree species (e.g., oaks) was facilitated by birds and mammals (e.g., jays and squirrels). Similar dispersal mechanisms are seen in high altitude areas where whitebark pine occurs. Dispersal of whitebark pine seeds by Clark's nutcrackers may be critical to the long-term survival of both species as climates change (Schrag et al. 2008). The current distribution of vertebrates may be associated with vegetation patterns in a broad sense (e.g., mink frogs are associated with boreal forest) or temperature (Figure 5.3) or by the distribution of vegetation in conjunction with temperature (e.g., Carolina wren, see DeGraaf and Yamasaki 2001). For many species it is the interplay among these factors that led to regional patterns of distribution in vegetation, habitat elements, and geographic ranges of species.

The potential patterns of plant species are particularly important for many species because they represent the potential of a system to provide energy to the trophic levels within the ecosystem. Systems vary considerably in their production of biomass. Biomass represents stored energy sources for consumers and detritivores. Mature deciduous forests can support over 475 tons/ha (190 tons/acre) of biomass (Whittaker et al. 1974), while deserts support only 5–50 tons/ha (2–20 tons/acre) of biomass (Noy-Meir 1973). Consequently, human activities that influence the direct physical factors of geology, soils, climate, or hydrology, or vegetation patterns through land use, can have huge impacts on the distribution of vertebrates by changes in habitat quality.

## SUMMARY

The physical factors of a region that include geology, soils, topography, climate, and hydrology interact to influence the potential for a region to support a plant community and the elements of habitat used by vertebrates. These physical factors also influence the quality of habitat for some species directly by their influence on providing suitable microclimates, burrowing substrates, cover, and stream features. The greatest effect of these physical factors is in structuring plant communities from continents to stands. It is the forest vegetation and the associated habitat elements that we can influence through forest management decisions. In Chapter 7, we will spend more time discussing the role of disturbance in driving patterns of vegetation across large regions.

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# 6 Cultural Effects on Habitat Patterns

In addition to the physical influences on habitat patterns discussed in Chapter 5, there is a historical context associated with human activities that must be understood to explain changes in the vertebrate communities that we have observed over the past several hundred years and the patterns that we see today. Native Americans likely maintained more open landscape conditions through use of agriculture and fire than what may have been first described by European settlers (Boag 1992). Humans are a part of ecosystems and they have been on some continents for millennia. The influx of European humans into the North American environment led to changes in forest cover and distribution that were quite different from the historic conditions that occurred up to that point. There are several factors that have occurred since the arrival of European humans onto the North American continent that set the context for management of habitat in North American forests.

The distribution of vegetation is faced with three dominant current pressures that might change habitat quality for many species at a much more rapid rate than has occurred historically: (1) land use, (2) global climate change, and (3) invasive species. These forces represent a significant common ground between foresters and wildlife biologists. Discussions about how to manage forests for products and habitat for species fall silent when forests are replaced by other systems.

## LAND USE

### URBANIZATION

The effects of land use on habitat patterns have been apparent for centuries. But development, especially as reflected in urban sprawl, is occurring at a remarkable rate in many of our forests. In Massachusetts, 16 ha (40 acres) per day are converted from forest to housing (Foster et al. 2005). The rate is similarly as great in urbanizing areas across North America. One only need fly over Mexico city, Phoenix, Seattle, or Vancouver to see the effects of development and sprawl on forest, grassland, and desert ecosystems. Urbanization homogenizes what was a more complex mosaic of vegetation and physical features providing resources for multiple species (Pauchard et al. 2006). As human populations increase, the urban–rural interface expands and the effects of urbanization extend beyond that of the individual house footprints. The proliferation of roads, utility infrastructure, and human use of remaining fragments of forest land lead to marked changes in the function of these forests as habitat for many vertebrates (Theobald et al. 1997). Some species increase in abundance and expand their distribution in response to these changes. Two bird species, Carolina wrens and tufted titmice, have increased in abundance by 17% and 7% over the past 40 years in Massachusetts (Sauer et al. 2005). Many more species have declined significantly over that same time period, such as wood thrushes and black-and-white warblers (Sauer et al. 2005). Conversion of forest to subdivisions probably has at least some role in these changes, and consequently some native species face habitat loss from development. Certainly we see loss of potential production of wood products from these lands. The greatest threat to forest sustainability and biodiversity is conversion of forests to other land uses, which often results when markets value forest systems and the ecosystem services they provide less than the economic value of houses, industries, and production agriculture (NCSSF 2005). As urban areas are expanding and the remaining land and water resources are becoming more constrained, more attention is being paid to meeting the needs for

species in urban areas. City planners, urban ecologists, and others are beginning to explore biodiversity conservation as one ecosystem service that some cities can provide with careful attention to urban design, roads and associated under- and overpasses, connected reserves, and riparian buffers. Such efforts mandate both ecological and social solutions to the challenges associated with providing these features in urban landscapes. We explore this topic more in Chapter 14.

## **FOREST CLEARING FOR AGRICULTURE**

Agriculture currently occupies over 40% of Earth's land area and consumes 70% of available freshwater (McLaughlin 2011). Hansen et al. (2008) estimated that 27.2 million ha of humid tropical forest was cleared between 2000 and 2005. Forest clearing is concentrated in only a small part of the tropical forest biome though, so impacts in countries such as Brazil and Indonesia as well as some countries in Africa are particularly significant. In West Africa, approximately 80% of the original extensive tropical forest area is now an agriculture–forest mosaic (Norris et al. 2010). Clearing and conversion to agriculture can be to enable production of food crops, grazing areas, or biofuel production. Net loss of forest land over time is offset partially by land abandonment and return to a secondary forest; however, secondary forests do not typically function as a habitat for some species as well as primary forest, at least during the first few decades of recovery. The rate of recovery of those functions varies from one tropical forest system to the next and the degree to which the remaining forest has been fragmented. Secondary forests can recover some habitat functions for some species in 20–40 years and consequently can play an important role in biodiversity recovery, particularly when located near primary forests (Dent and Wright 2009). But land clearing continues to occur in many areas at a greater rate than agricultural land abandonment. Tilman et al. (2011) predicted a 100%–110% increase in global crop demand from 2005 through 2050 but they point out that this demand can be met in different ways with profound impacts on our ability to conserve primary forests for biodiversity conservation. If we continue current behaviors of greater agricultural intensification in rich nations and greater land clearing in poor nations, then Tilman et al. (2011) predicted that approximately 1 billion ha of land would be cleared globally by 2050. However, if this crop demand was met by intensification on croplands of all nations then less land might be cleared, approximately 0.2 billion ha. Regardless of the scenario, clearing of lands for agriculture will constrain our ability to conserve biodiversity associated with forests in many parts of the world.

## **ENERGY PRODUCTION AND BIOFUELS**

The effects of climate change are evident in many parts of the world, and especially near the poles and at high elevations. The more direct effects of climate change on biodiversity conservation will be discussed in the next section, but these effects have led to a search for energy sources that are an alternative to fossil fuels. Biofuels represent any fuel feedstock that is derived from a biological source. Wood pellets, hog fuel, and firewood are biofuels, as is ethanol produced from corn, algae, switchgrass or other crops, including cellulose from trees and shrubs. Typical biofuel feedstock production is highly agricultural in design so agricultural lands are often used to produce fuel, replacing land that once produced food. As was noted earlier, demand for food will increase over the next 30–40 years, so if we produce more crops for fuel, we will need more land on which to do that. Reclearing lands that are now secondary forest is one alternative, but in many areas that is not sufficient to meet demands so primary forest is also cleared. Palm oil plantations are an example of a biofuel crop that now covers over 13 million acres of former native forest, largely in Southeast Asia (Danielson et al. 2009). Many of the impacts of biofuel production on biodiversity conservation are described in an earlier section on agricultural land clearing. Such impacts simply polarize those parts of our society wishing to find alternatives to fossil fuels from those trying to conserve biodiversity when both are often polarized from those that endorse greater fossil fuel production.

Tilman et al. (2009) summarized mechanisms for producing biofuels in a manner that would minimize these adverse impacts and produce up to 500 million tons of biofuel feedstocks per year in the United States alone:

1. Grow perennial plants on degraded lands abandoned from agricultural use.
2. More fully utilize crop residues.
3. Sustainably harvest wood and forest residues.
4. Double crops and use mixed cropping systems.
5. Utilize municipal and industrial wastes.

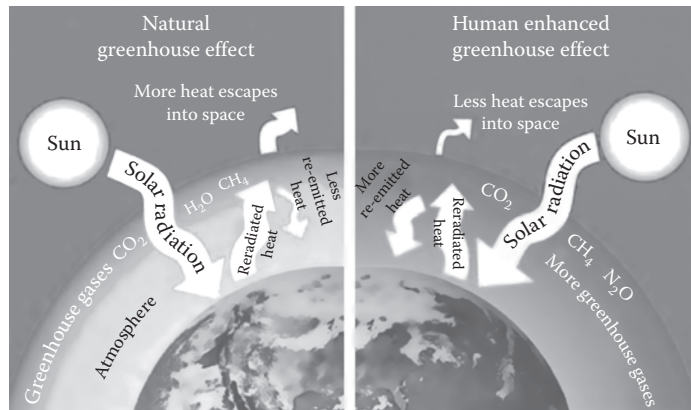
From a forest management perspective, sustainable harvest of wood and forest residues seems attractive, but even that approach must be taken with caution. Continual and intense removal of woody residues following a timber harvest has the potential to alter soil productivity and soil biota as well as species that use the forest floor (Anderson 2006). Consequently, guidelines have recently been developed to ensure that the adverse effects of biofuels production in forests and plantations can be minimized or reduced (Abbas et al. 2011). Abbas et al. (2011) categorized guidelines as those that would pertain to two conditions: Energy plantations (typically willows, cottonwoods, and eucalyptus) and forest residue removal in forest and timber management. Unfortunately, there are very few studies that have been conducted that document the relationship between wood biomass and habitat for one or more species, but that information is needed in order to develop guidelines for managers. The effects of biomass removal on forest floor fauna have been investigated in Scandinavia (Gunnarsson et al. 2004), but there has been a little work done recently to address effects elsewhere in the world.

## CLIMATE CHANGE

In forested environments carbon is present in living and dead biomass, in soil, and in the atmosphere that surrounds the forest. Forests remove carbon from the atmosphere through photosynthesis, but forests also return a significant amount of carbon to the atmosphere through respiration. As trees die and decay or are burned, carbon is released back to the atmosphere in a pulse, and then the forest regrows and fixes carbon again until the next large disturbance. As forests age and as the trees slow in their growth, carbon fixed by photosynthesis is largely offset by losses due to respiration. Nonetheless, these old forests can store a large amount of carbon for long periods of time between disturbances and when a disturbance does occur that kills many of the trees, decomposition of wood is slow and so carbon is released slowly over time, even following large forest fires.

Burning of fossil fuels is adding carbon to the atmosphere that is in addition to what is released through respiration, decomposition, and other mechanisms in carbon cycling that has occurred for thousands of years (Figure 6.1). Land use that converts forests (which store large amounts of carbon) to agriculture or urban areas, contributes to additional atmospheric carbon, but burning of fossil fuels continues to be the largest human-caused addition (IPPC 2001). Why is addition of carbon in the form of carbon dioxide, methane, and other forms important? These gases allow less heat to escape into space and in so doing influence changes in temperature, airflows, and other processes on the surface of the Earth. Temperatures may increase in some areas (the poles seem to be a good example now), or be more variable as changes occur in the jet stream, as offshore ocean currents alter, and other global processes respond to the fact that less heat is escaping the atmospheric envelope around the Earth (Figure 6.2). Consider the change in CO<sub>2</sub> concentration in the atmosphere over the past few thousand years (Figure 6.3), and the contributions of carbon to the atmosphere are continuing.

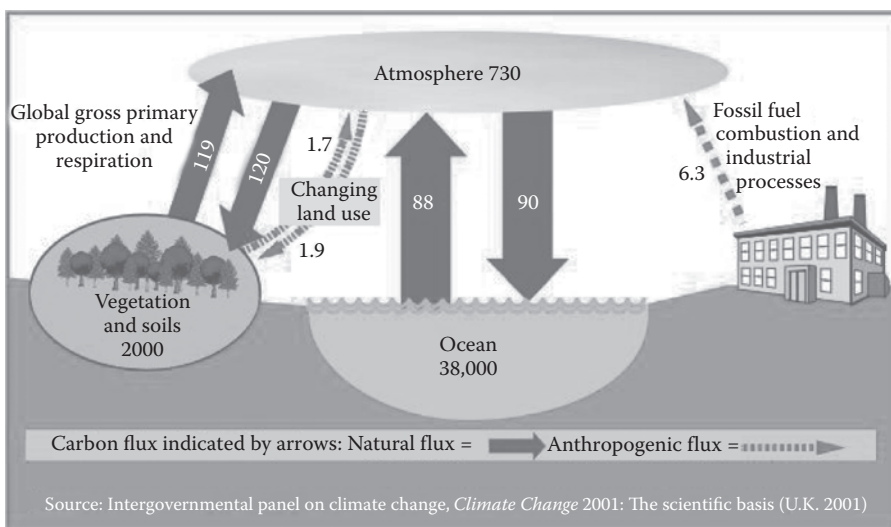
Climate change has the potential to alter the distribution of animals through changes in patterns of vegetation as well as through changes in the physical environment. These changes are quite likely to lead to marked changes in the ability of plants and animals to tolerate conditions as temperature and precipitation patterns change more rapidly than they have historically. The result will likely be a shift north for many southern species or a shift to higher elevations for species currently restricted



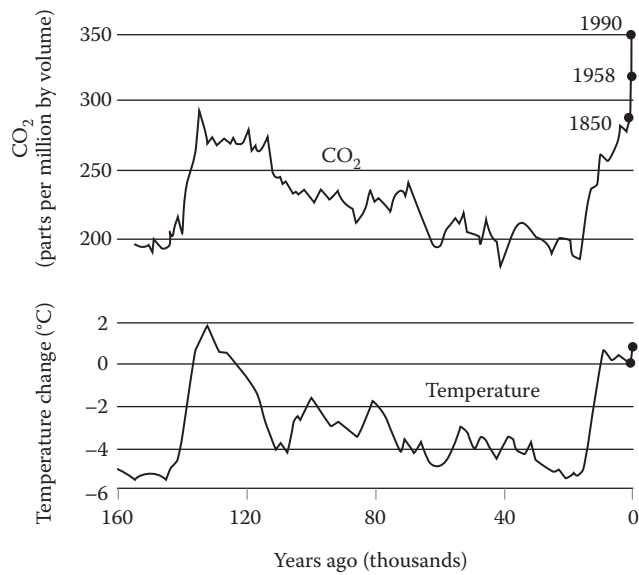
**FIGURE 6.1** Contributions to atmospheric carbon from various sources. (From <http://www.esrl.noaa.gov/research/themes/carbon/>)

to lower elevations. The area that would be available to meet the needs of species at high elevations or high latitudes will be increasingly compressed.

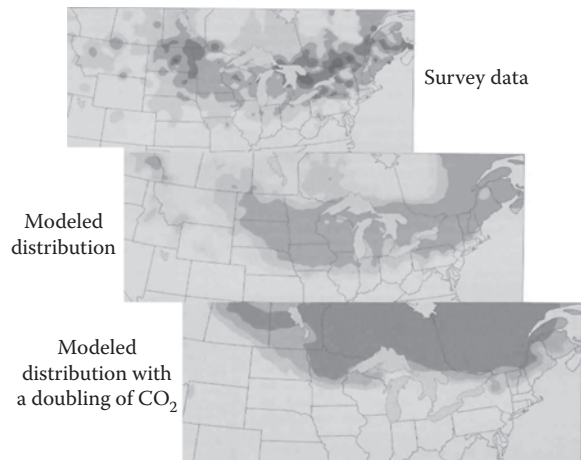
These shifts in distribution are assumed to occur if the organism is mobile enough and adaptable enough to allow movement in response to these climatic changes. For example, changes in vegetation as a result of elevated temperatures have been linked to the current and likely future distribution of animal species (e.g., Figure 6.4). Clearly, plant species face issues of coping with movement rates that will keep up with changing temperatures, but even some vertebrates (e.g., salamanders) likely will not respond quickly enough. Further, these organisms face obstacles as they are forced to change in their geographic ranges (e.g., roads, farms, and cities). Of course as urban and agricultural areas expand, and as fossil fuel demands increase worldwide, the effects of climate change will likely worsen until alternative energy sources begin to dominate. The changes that occur due to increases in carbon dioxide concentration will likely be largely irreversible for 1000 years after emissions stop (Solomon et al. 2009). It is clear that conservation biologists and wildlife managers will need to work



**FIGURE 6.2** Impacts of increase greenhouse gases on the Earth's climate. (From <http://www.nps.gov/goga/naturescience/climate-change-causes.htm>.)



**FIGURE 6.3** Carbon dioxide concentrations over the past 160,000 years. Circles represent changes due to human activities. (Reprinted from Schneider, S.H., and T.L. Root. 1998. *Status and Trends of the Nation's Biological Resources*. Pages 89–116. USDI U.S. Geol. Surv. publication, Reston, VA.)



**FIGURE 6.4** Current range of the northern bobolink (upper frame), predicted range based on current carbon dioxide levels (middle frame), and predicted range under doubled carbon dioxide concentrations (lower frame). (Reprinted from Schneider, S.H., and T.L. Root. 1998. *Status and Trends of the Nation's Biological Resources*. Pages 89–116. USDI U.S. Geol. Surv. publication, Reston, VA.)

with forest ecologists and forest managers to develop the active management approaches to address these profound and long-lasting changes that we will see over many human generations.

### INVASIVE SPECIES

We are homogenizing the planet. As we move plants, animals, microbes, and the materials in which they occur purposefully or inadvertently to different parts of the Earth, we allow some of them to find conditions in that new part of the world to be suitable for them. For many species, that is not the

case. But for some, landing in a new place at a similar elevation or latitude to their native geographic range (e.g., starlings in the United States once formerly restricted to Europe) allows them to proliferate. Once in that new environment they may fill a niche that was not previously occupied or may be a better competitor than a native species for a given niche. These species new to the scene and having a competitive advantage are considered invasive in that they often will displace one or usually many native species. The number of invasive species occurring across ecoregions is increasing. Alien species are the second leading cause of extinction in the United States and cost approximately \$120 billion annually (Crowl et al. 2008). These invasive species are often exotic, those that are brought into an area from other countries or continents. Chestnut blight fungus, gypsy moth, Scotch broom, European starlings, and house sparrows are examples of exotic species that have become invasive in North America.

Invasive species can also be excellent predators, or they can introduce new parasites or diseases into populations that had not previously encountered these new diseases. When red foxes and house cats were introduced into Australia, mid-sized marsupials in the drier parts of that country were vulnerable to the more effective and efficient placental predators than they had been to native marsupial predators (Johnson and Isaac 2009). Cat and fox control measures in conjunction with relocating some mid-sized marsupial species to offshore islands that did not have these nonnative predators allow many, but not all, species of native marsupials to survive, though for many species their current range is only a fraction of their geographic range prior to the arrival of these predators (Kinnear et al. 2002).

Invasive organisms can also come in the form of bacteria, viruses, and fungi that lead to morbidity and mortality in native populations. Frick et al. (2010) describe a recent emerging disease, white-nose syndrome, in North American bats that probably originated in Europe and was spread to the United States. West Nile virus in birds, myxomatosis in rabbits, and chytridiomycosis in amphibians all represent the forms of pathogenic pollution spread from one location to another, most likely by humans as vectors. Emergence of infectious diseases seems to be related to loss of biodiversity. Biodiversity loss may increase disease transmission, but on the other hand, areas with high levels of biodiversity may serve as a source for new pathogens (Keesing et al. 2010).

But species need not come from another continent to be invasive. Species native to a continent can also be invasive when they are placed in a new location. Bullfrogs moved to the west coast from eastern North America are likely responsible for declines in some western native amphibians (Figure 6.5). And not all exotics are invasive. In fact most are not. Ring-necked pheasants are an example of an exotic species that colonized the Midwest but that probably does not displace any



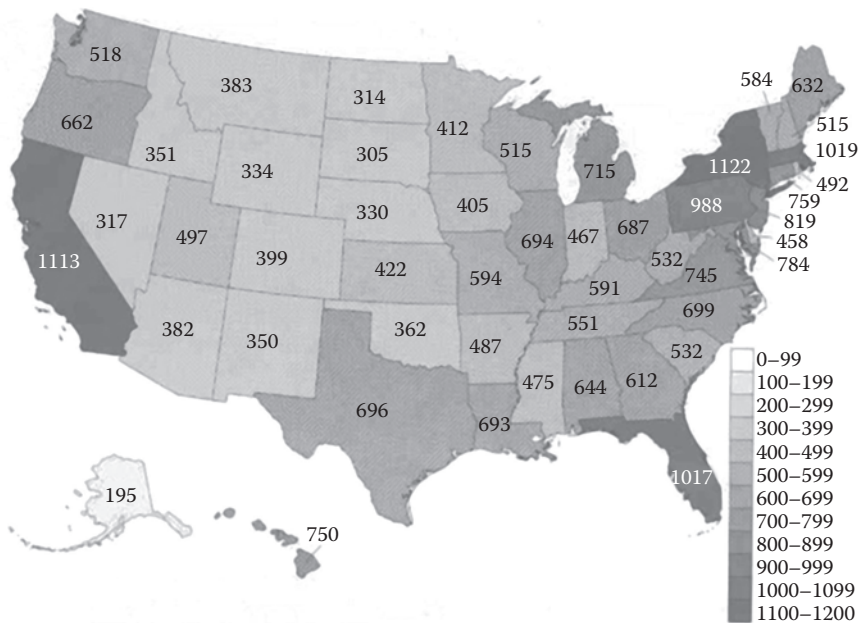
**FIGURE 6.5** Bullfrogs were introduced to the west coast from eastern North America and are likely responsible for declines in some western native amphibians. (Photo by Mike Jones. With permission.)



native species. The landscape planting industry annually uses thousands of species of plants that are not native and only a few escape to become invasive. Increasing expansion of suburbia leads to increasing spread of landscaping plants around houses and can become invasive, such as Norway maple. In addition, as climate changes, some invasive plants and animals that may not be a problem now may become a problem under new climatic conditions, or the current problem may be reduced under new climatic conditions.

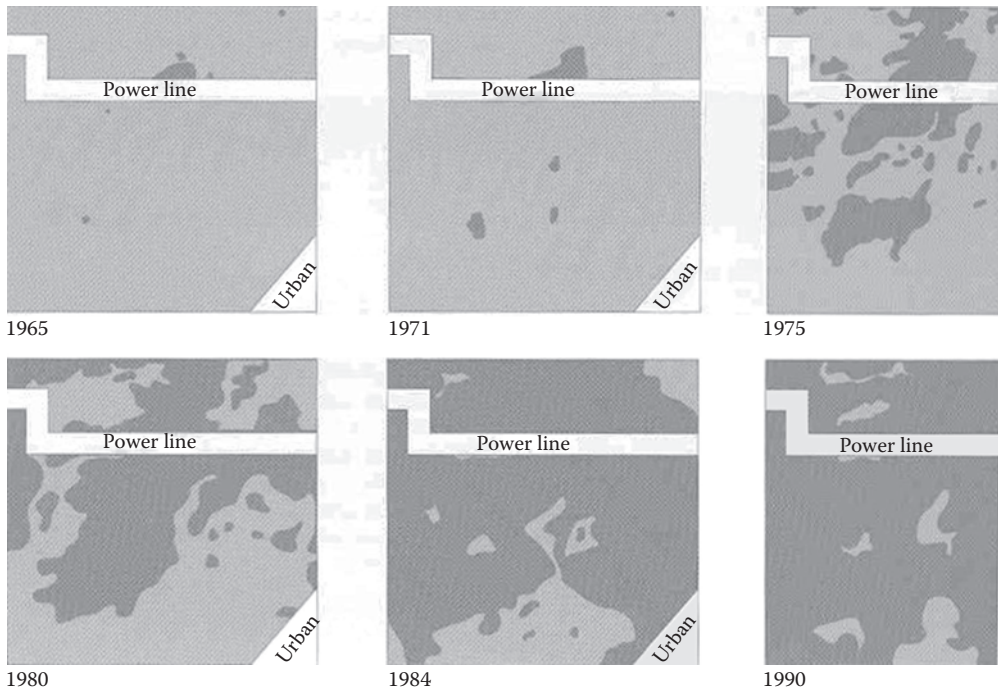
How do these exotic plants and animals find their way into our forests? Most arrive on this continent at port cities (Figure 6.6). States that have ports rank high in the number of these species that have been introduced. Some were introduced for purposes of soil stabilization and wildlife habitat improvement: multiflora rose and autumn olive, for example. Now biologists are trying to eradicate the very species that were imported and planted for habitat purposes! The continuation of introductions of exotic species in the landscaping industry and some forest products industries is raising more and more concerns about homogenization of our globe that could lead to a net loss in biodiversity (Richardson et al. 1997). The direct effects of invasive species on habitat quality can be quite apparent. The competitive advantage that invasives have over other species can lead to homogenization of the site and the displacement of native species into more isolated patches. For instance, pines were once only found in the northern hemisphere, but over 19 species are now established in the southern hemisphere through use of exotic species in plantations and for erosion control (Richardson 1998). The Australian paperbark tree was introduced in Florida during the early 1900s. Prolific seed production, flood tolerance, and rapid regrowth following fire enabled this species to invade wetlands and eliminate native plants and the animal species that rely on them (Figure 6.7; Hofstetter 1991). Once established, invasive species such as false brome can eliminate native vegetation and can be very difficult to control (Figure 6.8).

If society wishes to maintain habitat for various wildlife species, then biologists and foresters must first work together to address issues of development, climate change, and invasive species. Else the discussion of how to manage forests becomes moot.



**FIGURE 6.6** Numbers of nonnative plant species that have been introduced into each state in the United States. (Reprinted from Williams J.D., and G.K. Meffe, 1998. *Status and Trends of the Nation's Biological Resources*. Pages 117–130. USDI, U.S. Geol. Surv. publication, Reston, VA.)





**FIGURE 6.7** Changes in native plant species cover over time following invasion by the Australian paperbark tree (dark gray). (Reprinted from Williams J.D., and G.K. Meffe, 1998. *Status and Trends of the Nation's Biological Resources*. Pages 117–130. USDI, U.S. Geol. Surv. publication, Reston, VA.)



**FIGURE 6.8** Understory of a Douglas-fir stand dominated by the invasive exotic grass, false brome. Note the absence of native understory plants in this stand.

## SYNERGISTIC EFFECTS

The climate will change and some of those changes could last for 1000 years or more. The population of humans on the planet is increasing and each will demand a place to live and food to eat so land use will continue to shift to cities and agriculture. People continue to move around the planet and carry with them diseases, propagules, and animals, homogenizing our flora and fauna. All a bit overwhelming isn't it? Now consider that these three stressors, caused by people, interact to create even further uncertainty in the future of the planet. Consider the likely effects of climate change on invasive species. Hellmann et al. (2008) identified five possible effects of climate change on invasive species: (1) novel mechanisms for species introduction, (2) changes in the conditions that allow establishment of new invasive species, (3) emerging impacts of invasive species on the ecosystems in which they occur, (4) shifts in the distribution of established invasive species, and (5) changes in strategies needed to control invasives. And on top of the changes induced by climate change, consider how changes in our climate will influence other forces such as disturbances (see Chapter 7) and water levels in oceans, streams, and rivers. And the changes are not always in the direction of exacerbating invasive species issues—in some places, changes will help reduce the risk of invasion or spread while in other areas these risks will increase (Bradley et al. 2010). Indeed the common approach to addressing invasives in the face of such uncertainties is to try to assess the relative risks of changes we might see and of control measures we might use and select those that are most likely to be effective with the least risk.

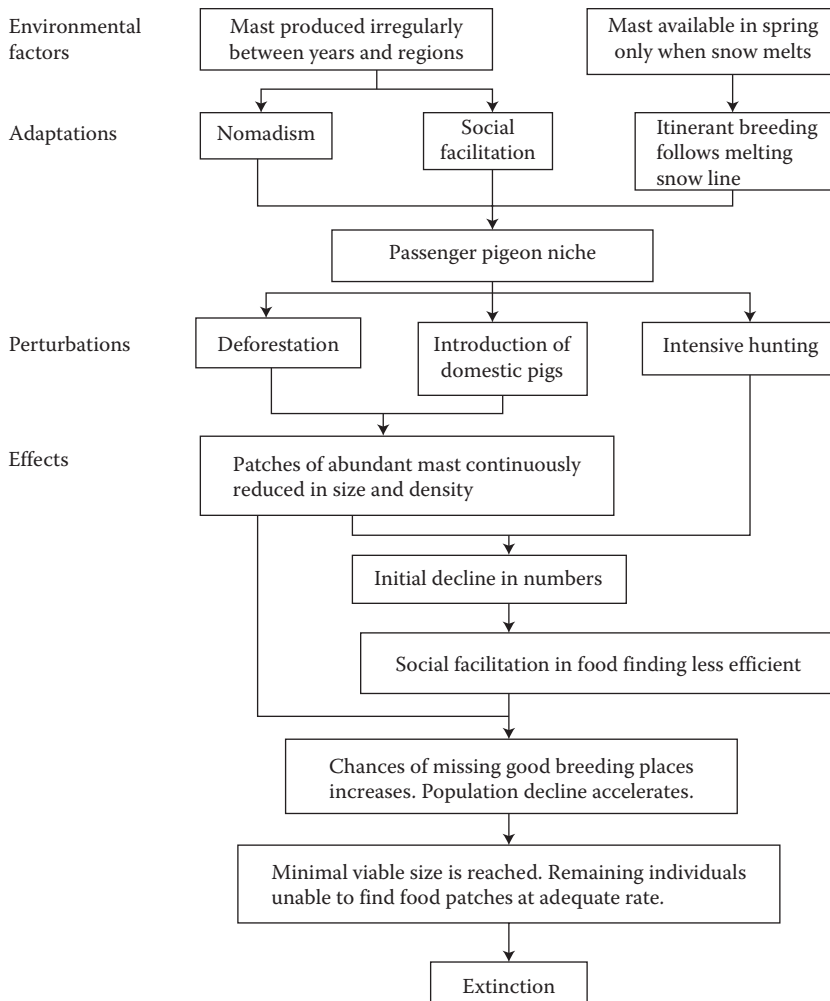
For instance also consider the effects of changing land use on invasive species spread. As we urbanize areas and bring exotic plants into environments for landscaping purposes, some may naturalize and spread. Some that are here now may find a new pathway to spread as climate changes. There are feedback loops as well. Simply the conversion of forests to alternative land uses can add greenhouse gas contributions to the atmosphere and exacerbate the rate of climate change. What can we possibly do to conserve biodiversity or manage habitat for certain species in the face of such an uncertain future? Some scientists view these complex problems as “wicked” having many disciplinary issues, all interacting, with social, biological, and physical components (Mooney et al. 2009). The irony of today's predicament is that we merely need to look at our own histories to see similar synergistic effects of environmental stressors that led to extinction of species.

## CASE STUDY: PASSENGER PIGEONS, HUMANS, AND FORESTS

Contemporary decision makers could learn about the roles of physical and cultural influences on habitat quality for selected species from patterns and changes in habitat elements that have occurred during the recent history of the United States. The passenger pigeon extinction represents a classic example of how a species was not able to persist when faced with a suite of pressures on the populations, especially changes in habitat conditions that were imposed by European humans.

Prior to European settlement, passenger pigeons were nomadic, occurring in flocks of millions of birds moving over vast areas of contiguous eastern deciduous forests (Ellsworth and McComb 2003) (Figure 6.9). Acorn production in the forests varied considerably from year to year and place to place (Healy et al. 1999) so the large flocks of pigeons provided many eyes to search for available food. Once a member of the flock found food, the remainder of the flock would follow, using a process known as social facilitation to locate patches of high acorn production near nesting and roosting areas. In the spring, pigeons followed the receding snowmelt northward to the nesting areas, relying largely on red oak acorns and beechnuts as food during these movements (Bucher 1992). Red oaks, unlike white oaks, undergo a winter stratification period and germinate in the spring. White oaks germinate in the fall so are less available to pigeons following snowmelt. Consequently, pigeons were well adapted to the extent and variability in patterns of acorn production in oak forests across the eastern United States and southern Canada.

European settlers cleared the forests initially for agriculture and eventually for cities and industries. Farmers often allowed domestic pigs to forage for mast (acorns, chestnuts, and beechnuts) in



**FIGURE 6.9** Factors leading to the extinction of the passenger pigeon in North America. Although many processes were at work, habitat loss was a primary driver. (With kind permission from Springer Science+Business Media: *Current Ornithology*, The causes of extinction of the passenger pigeon, 9, 1992, 1–36, Bucher, E.H.)

the forest, and pigs are a bit like 200 lb mammalian vacuum cleaners in the forest, eating mast, and digging roots (Henry and Conley 1972). The combination of clearing of oak forests, foraging by domestic livestock for acorns, and increased levels of harvest of pigeons as food caused pigeons to be less abundant. Patches of food became more widely dispersed and despite the pigeons' social facilitation behavior, food became more difficult to find. Hunting of the pigeons reduced their numbers, thereby making social facilitation as a mechanism for finding food patches even more ineffective because there were fewer eyes to find the more dispersed food. Nest sites also became less available, and because passenger pigeons only laid one egg per year, and both parents helped with incubation and rearing, there was a high energy investment in reproduction but a low reproduction rate. This low reproductive rate exacerbated the issues associated with reduced abilities to find food and nesting sites and so populations began to decline. Declines accelerated as the population entered what population biologists call the *extinction vortex* (Westemeier et al. 1998). Before long the population was simply not able to persist. The last nesting birds were seen in the Great Lakes region in the 1890s. The last individuals were killed in the wild in 1900, but some individuals remained in captivity until 1914. Martha, the last passenger pigeon, died at the Cincinnati Zoo on September 1, 1914. The

ultimate demise of the passenger pigeon was more a result of habitat loss than other factors, although overhunting contributed to the declines. Habitat loss occurred through cultural activities on a template of physical features and vegetation to which the species was well adapted. This classic example of species extinction should be one that we continue to learn from and consider how we should “save all the pieces” if we do not want additional species to have a similar fate.

## SUMMARY

Humans have had a remarkable impact on the patterns of vegetation on the planet. Historically, these have largely been through land use changes that continue to proliferate as more and more humans inhabit Earth. Development pressures, proliferation of invasive species, and climate change all threaten the extent and function of forests in the world, and hence will influence our ability to provide wood products and habitat elements to support conservation of biodiversity. These pressures on our forests provide common ground for foresters and wildlife biologists to work together if society continues to demand both wood products and biodiversity conservation.

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# 7 Disturbance Ecology and Habitat Dynamics

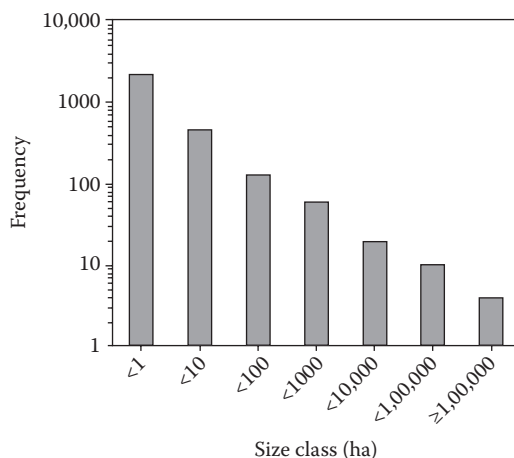
Stuff happens: fires, hurricanes, volcanoes, floods, and earthquakes. On average, approximately 450,000 ha are burned in the United States annually, over 1 million ha are affected by hurricanes and over 20 million ha are affected by insects and pathogens (Dale et al. 2001). The economic cost to society is over 1 billion dollars/year in the United States (Dale et al. 2001). To most people, these events are catastrophes. They can kill people, destroy property, and they can be catastrophic for other organisms too. Wind uproots trees, fires burn dead wood, and floods erode streambanks. They can also be events that renew habitat for other species. Indeed biodiversity conservation depends on disturbance. Wind adds dead wood to a forest, fires open the tree canopy and initiate a new forest, and floods create a new seedbed for willows and cottonwoods. Disturbances to forests have occurred for as long as there have been forests. Animals living in forests have adapted to many of these disturbances and some species rely on disturbances to provide food, cover, and water for survival. Understanding characteristics of disturbances and how disturbances influence habitat elements in stands and over forests can provide information that forest managers can use to provide habitat for selected species or to aid in conserving biodiversity. Knowledge of natural disturbances can help when developing silvicultural systems that might meet the needs of forest-associated wildlife (Franklin et al. 2002).

There are several characteristics of disturbances that can be used to understand potential effects on forest development, forest function and the sizes, numbers and distribution of habitat elements: type of disturbance, size and pattern, frequency, and severity. Disturbance type is also important, with changes in habitat elements being quite different depending on the cause of the disturbance (e.g., fire vs. wind). Estimating these characteristics of natural disturbances can facilitate prediction of forest recovery and the subsequent development of vegetation structure.

## **DISTURBANCE SIZE AND PATTERN**

Disturbances come in many shapes and sizes. The size of a disturbance can influence animal species that either remain in or recolonize after a disturbance (Rosenberg and Raphael 1986). Nearly 3 million ha of forest burned in the United States in 2002, a particularly bad fire-year. Some such as the Biscuit fire in southern Oregon were 200,000 ha in size. Most however were much smaller. In fact, there are usually many more small fires than large fires (Figure 7.1). Similar negative exponential distributions of disturbance size have been reported for wind disturbances (Foster and Boose 1992) and tree-fall gaps (Foster and Reiners 1986). Generally, there are many more small nonhuman disturbances than large disturbances across most forested landscapes.

Species that benefit from a disturbance seem to be associated with disturbances of different sizes. Black-backed woodpeckers, elk, and bison colonize forests following large severe fires as millions of dead trees and millions of kilograms of forage become available (Figure 7.2). On the other extreme, white-footed mice (Figure 4.1) are favorably affected by openings of 0.1 ha in size, and decrease in abundance in larger openings (Buckner and Shure 1985). A similar species, deer mice, was not found in these small openings but increased in abundance in larger openings (Buckner and Shure 1985). Disturbances also can increase the probability that certain invasive species might become established (Hobbs and Huenneke 1992) and how long they may persist (Blair et al. 2010).



**FIGURE 7.1** Histogram of the fire-size distribution for 2898 lightning fires during 1980–1993. Data are shown as counts on a logarithmic scale, in logarithmic bin-widths. (From Cumming, S.G. 2001. *Canadian Journal of Forest Research* 31:1297–1303. With permission.)

For species adversely influenced by a disturbance, the likelihood that they would be displaced by severe disturbances increases as the disturbance approaches and surpasses the size of the species home range (Gosse et al. 2005). But they may not be displaced if the disturbance is sufficiently small relative to the home range size (Hayward et al. 1999, Leonard et al. 2008). Some species that inhabit forests have been able to persist by including fine-scale disturbances within their home ranges, or by recolonizing stands of sufficient size that regrow to the point where necessary habitat elements become available.

Related to the disturbance size is the shape of the area created by a disturbance. We will cover edge effects in Chapter 16, but the function of an opening can be quite influenced by its departure from a circle because circles have the least edge per unit area of any regular shape. Consequently, two disturbances, both of 100 ha in size, may function quite differently for some species if one is circular and the other is shaped like an amoeba. More complex shapes can provide better opportunities



**FIGURE 7.2** Bison (shown here), elk, and other herbivores benefited from the large and severe fires in Yellowstone National Park.



for cover adjacent to food for some herbivores (Clark and Gilbert 1982) and can exacerbate the likelihood of colonization by invasive species (Cumming 2001).

Disturbance pattern is the spatial arrangement of the disturbance patches. Pattern is related to the size of a disturbance but, in addition to the areal extent of a stand or landscape affected by a disturbance, the pattern created by disturbance can influence the distribution of resources within and among potential home ranges for a species. Clumped distributions of fine-scale disturbances (e.g., clusters of root-rot pockets that kill trees) may result in a cumulative decrease in habitat availability within an individual's home range. A more random or uniform distribution of disturbances (e.g., lightning strikes) may allow that individual to tolerate the same disturbance density because only a small portion of any one individual's home range in the stand would be affected.

## DISTURBANCE SEVERITY

Crown fires and volcanoes are severe disturbances. A tree falling in a forest is not a severe disturbance. The severity of a disturbance reflects the impact on the stand or forest. Is the stand completely replaced or are only a few trees removed providing more growing space to the remaining trees? The severity of a disturbance influences the amount of organic material destroyed and redistributed by the disturbance and hence the amount and form (living or dead) of material that remains after the disturbance. Mount St. Helens was a severe disturbance, but even after its eruption, many pieces of the previous forest persisted. Trees were buried in ash, but so were seeds, fungi, and many species of animals that survived below ground. These biological legacies and the chance occurrence of them in places where they could grow and recolonize give rise to a landscape that is now on a trajectory to establish a new forest (Nash 2010). It is these same legacies that remain following a disturbance that may directly or indirectly provide the habitat elements needed for various species. Tree-fall gaps in later stages of forest development produce snags and logs, and sites where regeneration can become established in the openings allowing vertical complexity to increase. Similarly, following a hurricane or a fire, we see abundant dead wood and the establishment of a new cohort of plants and animals. The residual structures following severe disturbances such as these may persist into the next stand and provide the large trees and snags used by some species as the new stand grows (McComb and Lindenmeyer 1999; Figure 7.3). The diverse early successional conditions represented by disturbances leaving a diverse suite of biological legacies are rare in many landscapes as natural disturbances are partially replaced by uniform approaches to forest management (Swanson et al. 2011).

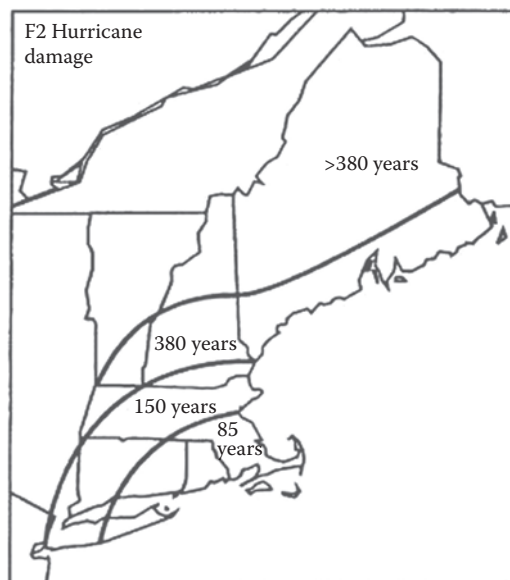


**FIGURE 7.3** Legacy trees, snags, and logs retained following a timber harvest in the Blue River Ranger District, Willamette National Forest, Oregon. (Photo by Bruce McCune. With permission.)

The legacy from the previous stand can be represented in the amount of dead wood, number of live trees remaining, depth of the leaf litter, and the vertical and horizontal complexity in the stand (Spies 1998, Franklin et al. 2002). The residual organic material that remains after disturbance can influence the direction of succession and the rate of subsequent development following the disturbance (Franklin and Halpern 1989). Biological legacies such as these can also provide a seed source for the new stand and ensure sources of mycorrhizal fungi are available to reinnoculate the disturbed site (Dahlberg 2001). Legacies can also be particularly important to species later in the development of forests. Northern spotted owls, for instance, are associated with biological legacies such as remnant large old trees that provide nest sites in many northwestern U.S. forests (North et al. 1996).

## DISTURBANCE FREQUENCY

The frequency with which a disturbance occurs in a forest will influence the tree species composition and the amount of living and dead organic material present on the site over time. Hurricane frequency, for instance, can influence the proportion of large areas in early vs. late stages of forest development (Figure 7.4). Frequent intense disturbances can delay the onset of forest development, or may preclude it. Infrequent low severity disturbances may lead to development of vertical structure in a stand. Frequency can be characterized in several ways: disturbance rate, percent of a stand disturbed on an annual basis, or the return interval (time between disturbances). The return interval for disturbances of certain types and intensities varies considerably among forest types. Fire return intervals may be as frequent as once every 2–5 years in some savannah systems (Harrell et al. 2001), and as long as once every 300–400 years in northwestern coniferous forests (Wimberly et al. 2000). Return intervals between management events in managed forests also vary, and when the return interval for a managed forest departs significantly from the return interval that has occurred naturally for thousands of years, then the risk of losing habitat elements and associated species can increase (Hansen et al. 1991).



**FIGURE 7.4** Zones of hurricane frequency in New England showing mean recurrence intervals between consecutive hurricanes of sufficient force to blow down forests. (From Boose, E.R., K.E. Chamberlin, and D.R. Foster. 2001. *Ecological Monograph* 71:27–48. With permission of the Ecological Society of America.)

## DISTURBANCE FREQUENCY, SIZE, SEVERITY RELATIONSHIPS

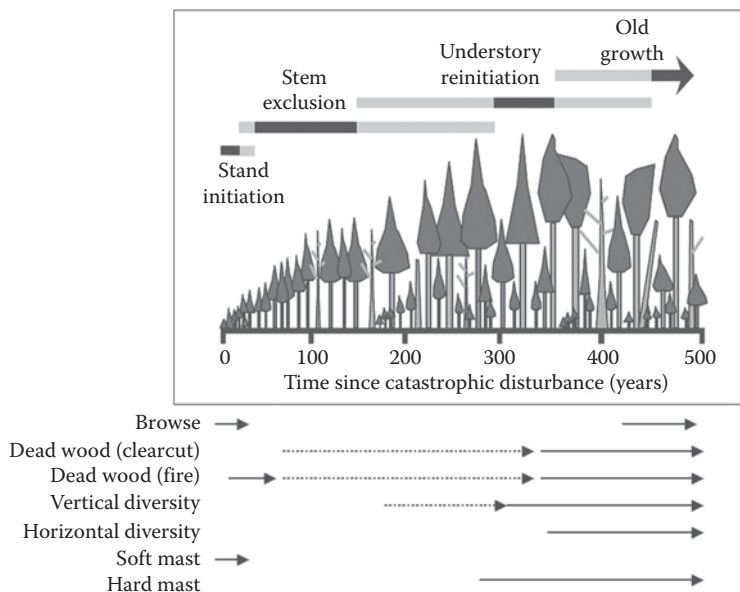
Imagine a forest of 1000 ha in size and uniformly old and unmanaged. Then consider the frequency with which tree-fall gaps occur in that forest. Big trees die and fall, creating an opening in the forest probably many times each year. Disturbance sizes are small, severity is low (little biomass removed) and frequency is high. Now consider the frequency with which a stand-replacement fire or hurricane might occur in that forest. On average perhaps once every 100 years? 200 years? Longer? When it does occur it may affect the entire 1000 ha and be quite intense. Although a generalization, frequent small-scale disturbances are often of low severity. In general, large severe disturbances are infrequent. In situations where we may prevent a disturbance from occurring as frequently as it might ordinarily occur, then the severity can be unusually high when it does occur. For instance, by controlling and preventing fire in many western U.S. forests, trees become more susceptible to insect defoliation and the accumulation of large fuels. So when a fire does occur, then it is unusually large and intense. Indeed, balancing disturbance frequency, severity, and size through natural disturbances and silviculture is a key to providing habitat elements, water, timber, and other ecosystem services from these forests.

## STAND DYNAMICS

A fire burns a forest, a hurricane blows over half of the trees in a stand, or an ice storm causes damage to a northern hardwood stand, reducing timber quality and value. How do stands respond to these disturbances? Early ecological perceptions of vegetation change following a disturbance provided the basis for the concept of ecological succession (Palmer et al. 1997), which presumed that there is a somewhat predictable change in the structure and composition of a stand following a stand-replacement disturbance. Although subsequent ecological research confirms that the recovery process in forests is not so deterministic (Palmer et al. 1997), simple concepts of ecological succession are a place to start to understand how forests change following a disturbance. Forest development is a continuum that we often break into arbitrary classes to help us simplify the complexity of forest change. After disturbances, forests develop through four general physiognomic stages: “stand initiation,” “stem exclusion,” “understory reinitiation,” and “old growth” (Oliver 1981, Oliver and Larson 1990) (Figure 7.5). Disturbance severity and frequency determine which species will dominate the forest during each of these stages. It is important to keep in mind, however, that these stages are merely convenient ways of understanding a *sere*, which is a set of forest communities that develop during stand regrowth following a disturbance.

### STAND INITIATION

A disturbance usually kills or damages trees and consequently creates both growing space for the remaining trees or sites for germination and growth of new trees. Forests often contain large amounts of dead wood and live plants following most natural disturbances, and the disturbance often creates a suitable site for regeneration by seedlings (regeneration from seed) or sprouts (vegetative regeneration). This stage of stand development is called stand initiation. In many forests, regeneration occurs naturally through *advance regeneration* present in the stand prior to the disturbance or natural regeneration that occurs from seedling establishment or sprouting following the disturbance. Forest managers may choose to control the species and number of regenerating trees by artificial regeneration, which involves planting seedlings at a particular spacing. The early growth and survival of regeneration can be controlled, ensuring that the future stand will be composed of the trees species and trees sizes desired by the land manager. Plantations of black spruce in Canada, Douglas-fir in the Pacific Northwest, and loblolly pine in the Gulf Coast are examples of artificial stand initiation. We will discuss this process in more detail in Chapter 8. In many forests, planting trees is unnecessary because it is expensive and there is an abundant natural regeneration. Following an intense



**FIGURE 7.5** A conceptual timeline portraying developmental stages for temperate rain forests of southeast Alaska. Shaded bars represent temporal overlap among developmental stages. (From Norwacki, G.J., and M.G. Kramer. 1998. *The Effects of Wind Disturbance on Temperate Rain Forest Structure and Dynamics of Southeast Alaska*. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-421.)

disturbance in central and northern hardwood forests of the eastern United States, it is common to expect 8000–100,000 woody stems/ha (including trees and shrubs) within a few years following a disturbance (Annand and Thompson 1997, Schuler and Robison 2002). Unoccupied growing space for plants following a disturbance will become occupied if water, light, and nutrients are sufficient. From the standpoint of providing habitat elements, this early stage of stand development tends to be a grazing-based system, with much of the net primary production available to herbivores through forage. Foraging also occurs below ground with species such as pocket gophers feeding on root systems of the newly established plants. However, if the disturbance killed but did not remove trees from the previous stand as might occur in a low severity fire or a wind storm, then much of the energy available in the dead wood is also available to other organisms through wood decomposition.

Trees remaining after a disturbance, or on the edges of disturbances, if not too severely damaged by the disturbance are suddenly free from competition by other trees and can respond by expanding their crowns and/or root systems and growing rapidly (Chen et al. 1992). In low severity disturbances, which tend to be more frequent, this “thinning effect” in forests allows remaining trees to continue to grow, rather than succumb to competition. In so doing, large trees develop in these stands and when they die, larger snags and logs are produced than would likely be produced at the same time in the absence of disturbance.

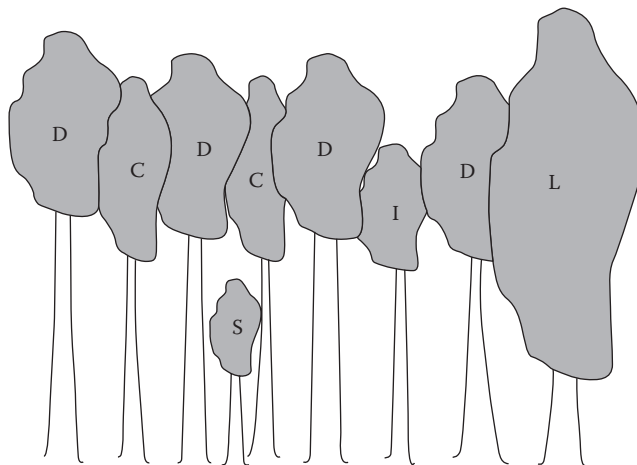
As plants begin to regenerate a site following a disturbance, eventually all the growing space in the site will be occupied. Foresters often use *basal area* as an index to the area of the stand occupied by trees. Basal area is the cross-sectional area of all trees on an acre or hectare. So imagine a hectare of forest (an area of 10,000 m<sup>2</sup> or a square 100 m on each side; 2.47 acres). If you cut all of the woody stems off at 1.4 m (4.5 ft) above ground and measured the area of all the cut surfaces and summed these areas, then you would have an estimate of basal area in hectare. The maximum basal area that a site can support will depend on moisture, growing season length, tree species, and nutrients, among other things. Through stand development, basal area will increase rapidly at first and then slow and finally reach a point where it fluctuates around a certain upper level. Two very

similar sites can support the same basal area but have very different tree densities. One site can have high basal area in a few large trees and the other site can have the same basal area represented in many small trees. Once *stocking*, or the degree to which a site is occupied by trees of various sizes, reaches a certain combination of tree density and basal area, then competition between the trees begins to greatly influence the structure and dynamics of the stand.

### STEM EXCLUSION

Once all of the growing space is occupied and plants begin to compete for light, soil moisture, or nutrients, then competition for those sparse resources begins to occur. Due to species characteristics such as growth rates, shade tolerance, and moisture tolerance, some plants are better able to use the resources of the environment than others. The plants that are not as fit in this competitive environment begin to grow more slowly and eventually die. The trees that remain during this process begin to stratify into crown classes that represent the various abilities of the trees in the stand to cope with competition (Figure 7.6). *Dominant* trees are the most fit in this environment and they form the uppermost canopy. They typically receive sunlight from above and the sides and have deep crowns and hence high leaf area (the area on the ground covered by leaves in the trees overhead) so they can photosynthesize well and grow rapidly. *Codominant* trees receive sunlight from above and so grow well and form most of the primary canopy layer. Codominant trees may have somewhat smaller crowns and may not grow as fast as the dominant trees, but they still contribute significantly to stand structure. *Intermediate* trees have smaller crowns, grow more slowly, and may contribute little to the upper canopy. *Suppressed* trees grow very slowly and have small crowns; they often die due to lack of light or moisture. *Wolf trees* are those legacy trees from the previous stand that may have very deep crowns because they were open-grown during the stand initiation phase.

The stand eventually develops a uniform canopy, further limiting the light available to plants beneath the canopy. During this stage, we see vertical structure become simplified into one dominant canopy layer and forage resources decline markedly. In hardwood forests of the eastern United States, it is common to see 90%–99% of woody stems die during these early stages of stand development. Although there can be instances where insect herbivory in these dense stands can be high (e.g., spruce budworm irruptions), most of the energy available for animals is through decomposition and the forest is now a detrital-based system. However, because most of the trees that die are smaller than the dominant and codominant trees, pieces of dead wood produced during this phase are small and decay rapidly.



**FIGURE 7.6** Crown class differentiation during development of an even-aged stand.



### UNDERSTORY REINITIATION

As the trees in a stand begin to continue to differentiate into crown classes, and some of the larger trees begin to die, the openings created by the dead trees create gaps in the canopy, allowing sunlight to enter to the forest floor. As sunlight enters, shade-tolerant understory plants can begin to establish and we begin to see a somewhat more complex forest structure. In many cases, seedlings will become established on the forest floor without a noticeable gap and survive until a gap is formed nearby, giving them a boost of growth. The gap may close and the shade-tolerant tree may have to wait for one or more additional gap events to reach the canopy. Less commonly, establishment may come from seeding into a tree-fall gap from the surrounding trees or may be the result of seeds stored in the soil in a *seedbank* waiting for the correct light and moisture conditions to allow germination. Forage resources begin to return to the stand, though they are often low in quantity and quality. Any dead wood remaining from the disturbance that created this new stand has largely decayed by this time. Consequently, dead wood abundance is quite low at this stage, though a few large snags and logs are starting to form in the stand both through competition and from insects, disease, wind, and other small-scale disturbances.

### OLD-GROWTH

As regeneration in the tree gaps grows and eventually replaces the dominant and codominant trees that have died in the stand, a new stand structure develops. Old growth or a *shifting gap phase* allows a structurally more complex stand to grow. This structure is created as trees age and die or are killed by diseases, insects, and other factors that lead to individual tree death in the stand, allowing regeneration to form in gaps throughout the stand. The stands begin to accumulate large pieces of dead wood, have trees of a variety of sizes and species, are vertically more complex, and increase in horizontal patchiness. Although the primary energy pathway in these stands is still through decomposition, small patches of forage develop. These forage patches in old forests can be quite variable in quality depending on the species and growth rates of the plants that fill the gaps. Frequent low-intensity fires, for instance, can create “gapiness” in forests and maintain a high level of forage availability (Figure 7.7). In some situations, tree seedlings

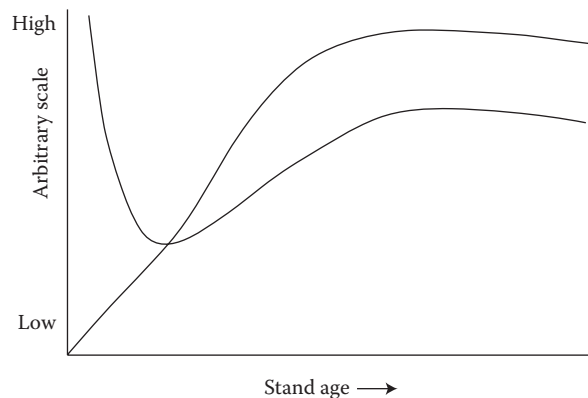


**FIGURE 7.7** Repeated low severity fires in some forests can increase forest gapiness, renewing forage availability for species such as this white-tailed deer.

growing in partial shade have higher levels of defensive chemicals in their foliage than seedlings in direct sunlight (Tucker et al. 1976). Happe et al. (1990) found higher levels of defensive compounds in plants in clearcuts than in old-growth forests however. Clearly, these plant defensive strategies vary among forest types.

## SUCCESSION AS A CONTINUUM OF HABITAT ELEMENTS

Stages are convenient ways of understanding stand dynamics. These stages are idealized and may occur at various spatial scales within and among stands. A continuous process of disturbance and regrowth occurs at various times and over various areas so a continuum is the appropriate context in which to place these stages. Habitat elements and other aspects of forest structure and composition also change over this continuum (Figure 7.5). Following a stand replacement disturbance such as a fire or hurricane, forage and soft mast production are typically highest in the earliest stages of development and decline markedly as crowns close. Once gaps begin to form as large trees die, then a modest recovery of these elements can be expected. Large pieces of dead wood also are most abundant following a disturbance, but they decay and are absent in the middle stages of succession. Abundant dead wood is added once again as large trees begin to die late in stand development. Hard mast production starts to occur once mast-bearing tree species reach maturity and develop large crowns during mid- to late stages of development. Spies (1998) provided two generalized curves to understand some of these changes in forest structure and composition over time following a stand replacement disturbance in Douglas-fir forests (Figure 7.8). Similar trends can be expected in other forest types, although the time scale would likely be different for other forests. Elements of forest structure that follow curve 1 (U-shaped) include the amount of dead wood, horizontal complexity, plant and animal species diversity, and susceptibility to fire. For instance, after a disturbance, we may expect to find large amounts of dead wood, diverse plant communities, and edges between disturbed and undisturbed sites. As tree crowns close and shade out many plant species and as dead wood decomposes, we reach a low point in both factors. Then as trees age and die and gaps form, we see greater diversity and more dead wood in the stand. Factors that follow trend line 2 (sigmoidal) include the diversity of tree sizes, vertical complexity, average tree size, incidence of tree damage and hollow trees, leaf litter depth, surface area of bark per tree, and live plant biomass. All of these factors increase over time as the stand moves from stand reinitiation to old-growth. In some forest types, curve 2 can also represent changes in dominance by shade-tolerant species.



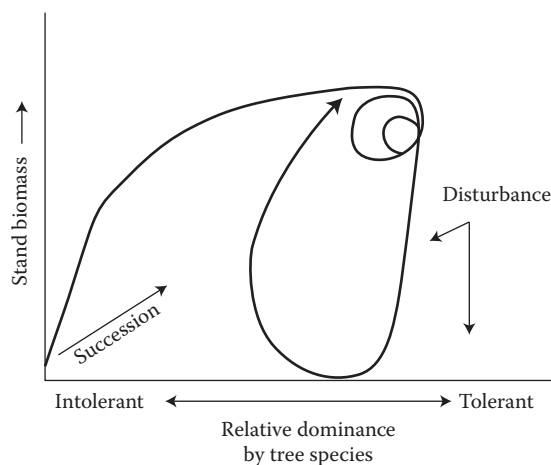
**FIGURE 7.8** Generalized patterns of habitat element change over time in Douglas-fir forests. The time scale is different in other forest types. (From Spies T.A.1998. *Northwest Science* 72:34–39.)



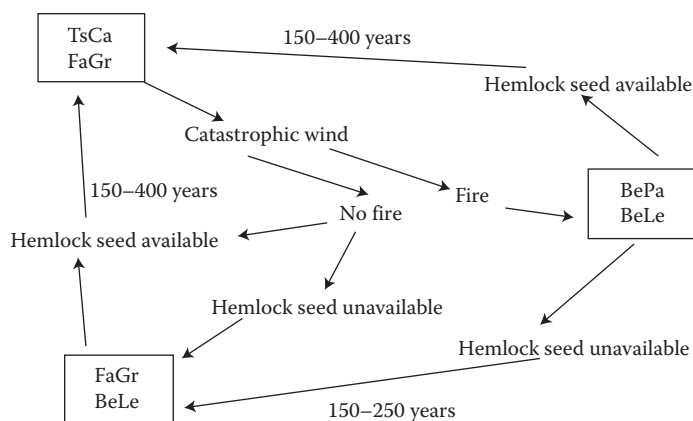
## SUCCESIONAL PATHWAYS

The discussion of stand dynamics so far has assumed that there is a reasonably predictable change in structure and composition over time following a disturbance. This is consistent with the view of succession proposed by Clements (1916). But succession is not quite so deterministic. Consider the theoretical changes in biomass following a disturbance that leaves soil bare (Figure 7.9). As plants occupy the site and grow, the plant community increases in biomass. Simultaneously, the plant community composition changes over time from one dominated by shade-intolerant species to include more shade-tolerant species. An intense disturbance may set succession back to a point where there is no living biomass, but roots and seeds persist in the soil (e.g., an intense fire in a hardwood forest). If there are existing seedlings, a seedbank, or a source for sprouts, then sprouts and seedlings may allow the site to recover with dominance by more shade-tolerant species than would have been there if these features were removed in a disturbance (a volcano or landslide). Disturbances that remove less biomass also recover but tend to be dominated by more shade-tolerant species. The availability of seed species in the soil seedbank, changing climate, soil pH, nutrient availability, and other factors continue to shift the final condition represented following full recovery after a disturbance (Figure 7.9). Variability in vegetation structure and composition following disturbances is to be expected due to the inherent variability in the severity, frequency, and size of disturbances affecting a forest as well as continual changes in climate, soils, and hydrology (Donato et al. 2012). Consequently, we often see a set of successional pathways that vegetation on a site could follow over time as disturbances occur and vegetation responds (Cattellino et al. 1979, Lebrija-Trejos et al. 2010).

In Figure 7.10, a site in which the potential vegetation is eastern hemlock–American beech may see a number of different vegetation states resulting from disturbance and regrowth. Following an intense hurricane and a subsequent fire or other disturbance that leaves bare soil, paper and black birch usually are the first tree species to dominate the site. Over time in the absence of a hemlock seed source, the site may move to a beech–birch stand. Should hemlock seed be available then a beech–hemlock stand may develop. From this diagram it should be clear that the same site can see several relatively long-lasting forest conditions that develop following disturbances and that these conditions each provide a different suite of habitat elements.



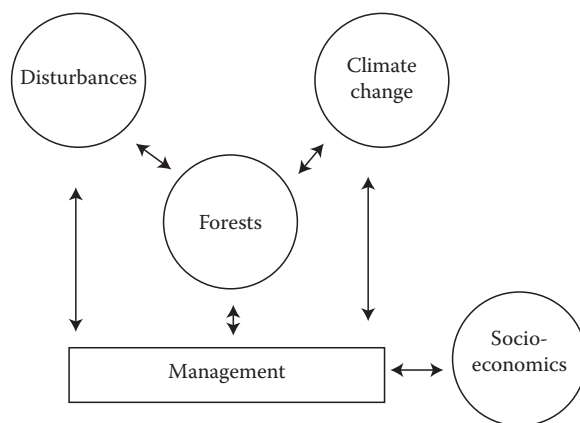
**FIGURE 7.9** Theoretical changes in forest states during succession and disturbances of varying frequencies and intensities. (From Spies, T.A. 1997. *Creating a Forestry for the 21st Century: The Science of Ecosystem Management*. Pages 11–20. Island Press, Washington, DC.)



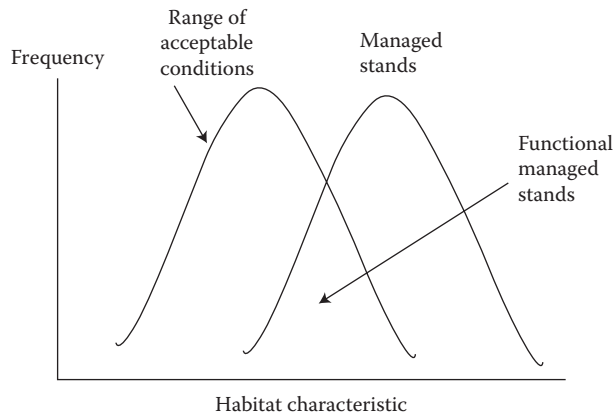
**FIGURE 7.10** Successional pathways for a southern New England Forest. TsCa = eastern hemlock, FaGr = American beech, BeLe = black birch, BePa = paper birch. (Courtesy of Anthony Damato 2006.)

### MANAGEMENT IMPLICATIONS FROM DISTURBANCES

Management activities do not replace the natural disturbances but they can be complementary to them. Knowledge of the frequency, severity, and size of natural disturbances, and the various successional pathways that emerge from them can offer clues to management strategies that might be effective in achieving a variety of goals (Long 2009). And a departure from natural disturbances can move landscapes into new conditions that no longer reflect the conditions under which species have evolved and persisted (Cyr et al. 2009). Disturbances influence forest structure and composition, as do the myriad of physical factors (Figure 7.11). Management actions driven by the goals of society, be they economic, cultural, or spiritual, interface with the dynamic processes of disturbance and climate change (Vierikko et al. 2008). When considering the diverse suite of habitat elements that might be needed to maintain biodiversity within a forested region, it is often useful to use variability in forest structure and composition to our advantage whenever possible. Forest managers producing wood products to meet industry and societal needs want to minimize uncertainty in the production process. But from the standpoint of providing a suite of habitat elements, uncertainty and the variability that it produces is something that should be embraced and worked with and not avoided. No single stand management



**FIGURE 7.11** Interactions among forest disturbance, climate change, and management. (Dale, V.H. et al., Climate change and forest disturbances, *BioScience*, 2001, by permission of Oxford University Press.)



**FIGURE 7.12** Hypothetical range of conditions in a habitat element that might be seen during stand development following a natural disturbance and during stand development using silvicultural practices. (From McComb, W.C. 2001. *Wildlife Habitat Relationships in Oregon and Washington*. OSU Press, Corvallis, OR.)

system will precisely match the variability inherent in natural forests that resulted from a variety of disturbance regimes. The compatibility of management and habitat goals is scale dependent both in space and time, and management often occurs over a much narrower range of space and time than scales associated with natural disturbances. Some of the variation can be incorporated into the managed forest landscapes of the region by using a variety of silvicultural systems and forest management strategies. The choice of these systems will depend on the biological, social, and economic objectives for the stand and the landscape, and they will imitate natural disturbances to varying degrees (Vierikko et al. 2008; Figure 7.12). Indeed, the basis for development of existing silvicultural systems for timber objectives was that these systems reflect the regeneration and growth strategies of the commercially important tree species in a region. Intensive timber management as currently practiced leaves less dead wood and noncommercial plant species than natural disturbances (Hansen et al. 1991) so it may not imitate natural disturbances for other forest resources as well as it does for timber. The management strategies that include goals for habitat elements for certain species or for biodiversity conservation goals require consideration of more factors than are necessary for production of commodities, but it is commodity production that can pay for the management activities needed to achieve certain habitat goals. The two goals should be complementary. In the following chapters, we will explore methods of stand and forest management that can achieve both commodity and habitat goals.

## SUMMARY

On the template of the physical and cultural landscape from which vegetation arises, vegetation is further altered by disturbances. Disturbance severity, size, and frequency interact to influence the sizes, amounts, and distribution of habitat elements such as vertical complexity, forage, dead wood, horizontal complexity, and plant species composition. Vegetation regrowth following disturbance in the very simplest sense follows several stages of stand development following a stand replacement disturbance: stand initiation, stem exclusion, understory reinitiation, and shifting gap phase. These conditions occur in various scales of time and space and represent somewhat arbitrary points on the continuum of successional change. Indeed a variety of successional pathways can be seen following a disturbance depending on the severity and frequency of disturbance and the regrowth potential of the vegetation. Seed and sprout availability, shade or moisture tolerance, and time interact to determine what plant community is likely to develop and be maintained on a certain site following disturbances of varying intensities and frequencies. This knowledge of disturbance and succession can be used to craft management strategies to achieve multiple goals.

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# 8 Silviculture and Habitat Management

## *Even-Aged Systems*

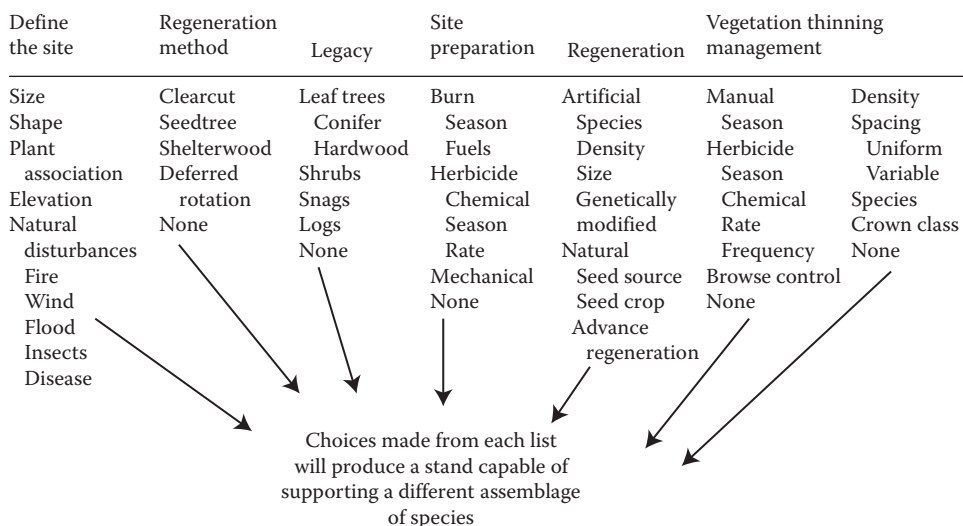
The forest disturbances described in the last chapter continue to influence the structure and composition of forests. Shift happens. But how contemporary human societies chose to use the forest and manage them for various species has an ever-increasing influence on the structure, function, and composition of forests. Although culturally induced pressures of development, invasive species proliferation, and climate change will continue to determine whether we have forests to use for other purposes, two forest management practices currently have a significant impact on habitat for many species: silviculture and fire management (both fire protection and prescribed burning). *Silviculture* is the art and practice of managing forest stands to achieve specific objectives for the landowner or land manager. These objectives could include timber production, recreation, habitat for various wildlife and fish species, biodiversity conservation, aesthetics, and nontimber forest products. Indeed many nonindustrial forest landowners in the United States own and manage land for reasons other than timber production. But it is the economic value of the forest that allows many landowners to manage their forests to achieve other objectives. Consequently, managing forest lands for habitat requires a basic understanding of silvicultural principles to be effective in achieving habitat goals. Similarly, foresters charged with recruiting, maintaining, or removing various habitat elements must understand how their silvicultural prescriptions are likely to influence habitat elements.

### SILVICULTURE AS A FOREST DISTURBANCE

Silvicultural activities are forest disturbances and as such have particular sizes, severities, frequencies, and patterns across a landscape. These human-caused activities interface with the suite of natural disturbances also representing various sizes, severities, and frequencies. Managers can decide the degree to which silvicultural activities emulate natural disturbances (Fenton et al. 2009) or depart from characteristics of regional natural disturbances. In either case, the disturbances caused by people are at least in part usually additive to the disturbances caused by nonhuman forces.

Forest managers can select the types and rates of disturbances that will meet specific resource objectives and, in so doing, influence successional pathways to achieve specific goals. Some habitat management issues that foresters and wildlife biologists face may result from insufficient consideration of the size, frequency, severity, and patterning of silvicultural disturbances on a landscape. A range of management decisions can be made on any given site that will result in stand conditions and plant communities that support only certain species (Figure 8.1).

Consider, for example, an Oregon Coast Range site managed with the following combination of decisions: clearcut; no retention of logs, snags, or green trees; no site preparation; rely only on natural regeneration; no vegetation management; and no precommercial thinning. The result would probably be a red alder or salmonberry plant community with few Douglas-firs or other conifers. A similar approach in the Gulf Coastal Plain would likely result in an area dominated by oaks and sweetgum but with few pines. In northern hardwoods, however, birch–beech–maple would probably dominate the site, and in spruce–fir-dominated boreal forests in which balsam fir advance regeneration occurs, the next stand would be dominated by balsam fir. Now consider the same sites managed



**FIGURE 8.1** Decisions made by a land manager from each list will lead to development of very different forest structure and composition on a site. These choices are reflected in the availability, number, and sizes of habitat elements. (Redrafted from McComb, W.C. 2001. *Wildlife Habitat Relationships in Oregon and Washington*. OSU Press, Corvallis, OR.)

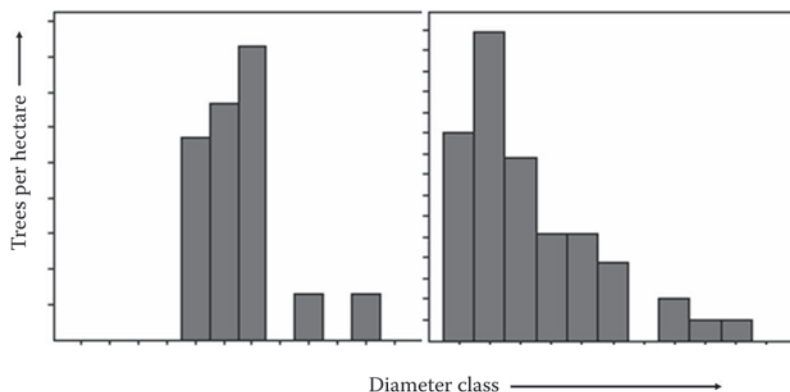
with the following decisions: clearcut, snag legacy, plant the commercially important conifer of the region, seedling release with herbicides, and thin to 750 trees/ha (300 trees/acre). The result would probably be a conifer-dominated stand condition with a grass–forb understory (sparse shrubs) during the early stages of stand development. Each condition is a habitat for a different suite of species on the same site managed in one of two ways. Many possible decisions could be made early in stand development that could produce a wide range of stand conditions (Figure 8.1).

## CHARACTERISTICS OF EVEN-AGED STANDS

Once the site characteristics and stand objectives have been identified, silviculturists usually first decide whether to use an even-aged or uneven-aged silvicultural system to regenerate and maintain the stand. These two systems ensure that the desired species of new young trees will replace the trees that are cut and that they grow to achieve the landowner's objectives. Even-aged stands are those in which all or most of the trees in the stand are approximately the same age. If the trees in an even-aged stand are also all the same species, then at least during the early stages of development, most of the trees will be of similar diameter and height (Figure 8.2). The ideal distribution of tree diameters in an even-aged stand is a bell-shaped curve but often there is some departure from a normal curve. If the new stand is composed of a variety of tree species, all with somewhat different growth rates, then a different pattern emerges. Within the first few decades, the distribution of tree diameters will begin to depart from a bell-shaped curve as some trees grow rapidly in diameter and height and others grown more slowly (Figure 8.2). From the standpoint of considering vertical structure, horizontal complexity, forage resources, and other habitat elements, it is often more useful to think about the stand size-class distribution than the age distribution in mixed species stands.

Uneven-aged stands are those in which there are typically three or more cohorts or age groups represented simultaneously in the same stand. Typically, there are many more small trees than medium-sized trees and many more medium-sized trees than large trees; so the distribution of tree sizes is similar to an even-aged, mixed species stand (Figure 8.2). We will cover uneven-aged management in Chapter 9.





**FIGURE 8.2** Distribution of live tree diameters in an even-aged stand with one species, white pine (left), and with multiple species (right) at 60 years of age as predicted using the Landscape Management System. (Adapted from McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

Even-aged stands typically develop through the stages of stand development described in Chapter 7: stand initiation, stem exclusion, understory reinitiation, and shifting gap phase. Depending on the goals for a stand, such as pulpwood, sawtimber, aesthetics, or habitat for certain species, this sequence of stand developmental stages may or may not be truncated. For instance, a manager growing pulpwood and providing habitat for species typically found in the stem exclusion stage may never allow the stand to enter the understory reinitiation or shifting gap-phase stages.

There are three classic methods to establish even-aged stands that will be described in more detail in the following section: clearcut, seed tree, and shelterwood. These methods can be used to initiate the stand development from stand initiation through shifting gap stages. Depending on the goals for the stand, natural disturbances can be allowed to further modify the stand structure over time if the manager decides not to salvage the dead and damaged trees in the stand.

In this chapter we also will cover two-aged stands as a modification of even-aged systems. Two-aged stands are not explicitly planned for in traditional silviculture methods, but can be quite useful when trying to retain a legacy from the previous stand or to recreate some aspects of natural disturbances such as fire and wind. Because these structures apparently occur quite commonly following natural disturbances and because they have many favorable aspects from the viewpoint of habitat element complexity, they should be standard options to be considered when planning silvicultural treatments.

## CONSIDERING THE CAPABILITIES OF THE SITE

Before deciding which regeneration method to use to achieve silvicultural goals, the site must be evaluated. The size of the stand, its shape, access, slope, aspect, soils, and all of the other physical variables discussed in Chapter 5 must be taken into consideration. These factors will not only influence the type of vegetation the site can support, they also will influence the function of the vegetation in providing habitat elements on that site. For instance, an opening placed on a dry, south-facing slope with rocky ledges may be ideal for increasing habitat quality for some reptile species but may not be nearly so effective on a north-facing slope. Forest management along streams may influence the water temperature and sedimentation rates, but future stand development may be affected by beavers. So before planning any silvicultural activities, the potential for the site to support certain plant and animal species should be assessed. Ask yourself, “Can the goal that I set for the area be achieved given the physical environment in which it occurs?” If your goal could be achieved, then you should ask if it can be achieved at an appropriate cost of time and money. For instance,

converting a forest to a salmonberry field in Washington or a mountain laurel field in New England may be easy and cost-effective (at least in the short run) by harvesting all trees and releasing these shrubs. On the other hand, converting these shrub fields to forests may simply not be possible without a significant initial investment of time and money.

The dominant disturbances in the area must also be considered. If your goals for the site may be compromised by a high probability of incidence of root disease, insect defoliation, browse, fire, ice, or wind, then alternative goals might be more appropriate. An excellent example is the inability to establish adequate regeneration of many commercial tree species in hardwood stands in many parts of Pennsylvania simply because of deer browsing (Marquis 1974). Attempting to grow eastern hemlock where hemlock wooly adelgid (an exotic defoliating insect) is likely to occur is another example of an impending disturbance that would compromise your ability to achieve a habitat goal. Similarly, growing dense conifer stands in wind-prone or ice-prone environments may cause much of the stand to break or be blown over (Bragg et al. 2003). So be sure that your goals for an area are consistent with the ability of the site to allow you to achieve those goals effectively and efficiently.

### CHOOSING A REGENERATION METHOD

Clearcutting is probably the most commonly used silvicultural regeneration method designed to initiate an even-aged stand. This term should not be confused with a harvesting system. Harvesting systems are the means of removing the trees from the site to a landing where they can be taken to a mill, regardless of the regeneration method used. Harvesting systems include horses, skidders, cable systems, and helicopters. *Clearcutting* is a silvicultural decision designed to ensure adequate growing space for regeneration of a new stand. It is usually done to allow establishment of a plantation, though occasionally direct seeding is used to establish a new stand.

The boundary of the area to be clearcut is typically marked and then all commercial (and often noncommercial) trees are cut. If noncommercial trees and shrubs are not cut, then they are usually killed after harvest to allow rapid growth of the desired species. The size of the clearcut is usually determined by a combination of ownership boundaries, economics, and law. From simply an economic standpoint, larger clearcuts are more efficiently harvested and established than smaller ones. There are fixed costs associated with harvesting, planting, vegetation management, and thinning that are all reduced when work is in one large area rather than in many small ones. State policies and federal land management plans often dictate the upper limit of clearcut sizes that are allowed by law. Habitat concerns, especially minimum opening sizes for some species or adequate edge conditions for others will also influence clearcut size and shape.

If natural regeneration can be assured by leaving some trees after the harvest to provide a seed source to the newly created growing space, then the *seed-tree regeneration method* can be used. This approach is particularly effective for species that produce abundant seeds and produce seeds on a regular basis. Loblolly and shortleaf pines may be regenerated using this approach because they have winged seeds that disperse from the seed trees, they produce seeds regularly, and if the site is appropriate, sufficient seeds germinate and grow. Using this approach, dominant trees in the stand that have deep crowns and seem to have regular cone or seed production are identified prior to harvest and marked for retention. The number of seed trees retained depends on the dispersal capabilities of the seeds. For instance, in shortleaf pine, approximately 85% of seeds fall within 50 m (155 ft) of seed-producing trees (Burns and Honkala 1990). Seed tree spacing for this species can probably be as much as 50 m among trees, but closer if there is concern regarding seed production following harvest. Once regeneration is established, then the seed trees can be removed or retained depending on the goals and objectives for the stand. Retaining seed trees can have beneficial effects for species, such as red-cockaded woodpeckers, that use widely scattered mature trees in a savannah-like structure (Conner et al. 1991). If seed trees are retained, then there can be a reduction in growth of new seedlings for some tree species (e.g., longleaf pine; Boyer 1993).

Where newly germinated seedlings need some protection from direct sunlight, dessication, or frost, then a *shelterwood* regeneration method may be needed. A shelterwood system is designed to provide shelter or protection of the regeneration (newly established trees) by leaving a sparse canopy cover—enough to provide protection but not so much canopy cover that seedlings do not receive enough sunlight to survive and grow. This technique is most often used with tree species that are not too shade intolerant. This system is used in central hardwood oak forests (Annand and Thompson 1997), ponderosa pine (Anderson and Crompton 2002), upland mixed pine hardwood stands, mixed species jack pine stands, and northern hardwood stands, among others (Holloway and Malcolm 2006). By retaining a somewhat denser overstory to provide protection, some species that cannot survive in seed-tree stands can tolerate shelterwood stands (Taulman and Smith 2004). Establishing regeneration using the shelterwood system usually takes two to three steps. First a *preparatory harvest* is made to thin the stand and allow sunlight to strike the crowns of the dominant trees. This allows trees to increase in diameter and crown size, which increases the probability of having adequate seed production for many tree species (Dey 1995). Once seed crops seem likely or sufficient, then an *establishment or seed cut* is made to further release the trees with abundant seed production and to provide growing space for the new even-aged stand of seedlings. Finally, once the seedlings are well established and growing and can tolerate full sunlight or frost, then the overwood, or overstory, trees are removed. At least some *overwood removal* is usually necessary to allow the newly established regeneration to grow rapidly, but the level of removal is dependent on the goals for the stand.

Another type of even-aged method that has been proposed as a way of retaining some structure in these even-aged stands is the *deferred rotation* method described by Smith et al. (1986). Silviculturists also will refer to this a *clearcut with reserves*. This system retains some trees through two complete growing cycles, or rotations, in the stand and benefits certain species of animals by allowing some open-grown trees to grow large and old (Thompson and Desseker 1997, Chambers et al. 1999a). Deferment harvesting is now a commonly used technique in the central Appalachians (Thomas-Van Gundy and Schuler 2008). Rapid growth of the residual trees in combination with abundant and diverse regeneration leads to a more complex stand structure than would be seen in a traditional clearcut (Thomas-Van Gundy and Schuler 2008). Whichever system is chosen, the stand usually proceeds through site preparation, stand reestablishment, vegetation management, and stand-density management before it is ready for harvest at the end of the *rotation*, or growing cycle. Decisions at each stage influence stand structure and composition and, in turn, habitat quality for the wildlife species present at various stages of stand development (Figure 8.1).

The selection of a regeneration method will have an effect on the stand structure during the early stages of stand development (Cimon-Morin et al. 2010). Seed-tree and shelterwood systems at least initially leave vertical structure until the seed trees and overwood are removed wholly or in part. Regardless of whether natural or artificial regeneration is used to establish the new stand, the newly cleared area produces a flush of grasses, forbs, shrubs, as well as small tree, but Craig and MacDonald (2009) found that retaining more than 10%–20% of the previous stand in a boreal forest had significant effects on understory vegetation responses. Shade provided by overwood can limit production of herbs and forbs (Graham and Jain 1998). Hence, shelterwood or deferred rotation systems that, with high levels of retention, may leave an understory more similar to the preharvest stand, while retaining fewer trees may lead to an increase in understory development. The duration of this stand initiation phase, in which herbaceous plants occur and can dominate, can also be influenced by the choice of regeneration method. Because of the difficulties associated with vegetation management when residual trees are present, early vegetation control may not be possible in shelterwood stands. Consequently, the duration of the shrub condition in shelterwood stands may be longer than in clearcuts where shrubs are often controlled using herbicides. Further, the degree to which a stand is successfully regenerated can influence the duration of the stand initiation phase. Stands with high densities of tree seedlings that close crowns quickly will shade out other species, whereas low-density regeneration will allow sunlight to penetrate through crowns for a longer period of time before crown closure.

Woody and herbaceous plant species associated with these even-aged regeneration methods are often very to moderately shade intolerant, although this can be adjusted with overwood retention in shelterwood systems and with artificial regeneration. Consequently, both the structure and composition of stands can be quite variable and diverse in early stages of stand development depending on the degree to which trees are retained from the previous stand. Even-aged systems also produce stands that often create abrupt edges depending on the stature of adjacent stands. These high contrast edges are beneficial to some species (McGarigal and McComb 1995), but not to others, especially some species of amphibians (Martin and McComb 2003).

### **IDENTIFYING LEGACY ELEMENTS TO RETAIN**

During a harvest, the land manager may decide to leave certain structural components of the previous stand on the site and into the next rotation (Franklin et al. 2007). This legacy from the previous stand is a means of adding more complexity to the new stand and allowing some of the structure, composition, and processes from the previous stand to be carried forward into the new stand. Snags are the most visible type of legacy left or created on many sites and they can significantly influence the occurrence of a number of cavity-nesting species (Bull and Partridge 1986, Schreiber and DeCalesta 1992, Chambers et al. 1997, McComb and Lindenmayer 1999). Logs, living conifers, and hardwoods (and the lichens, bryophytes, fungi, and other species associated with them) left on the site can provide structural and compositional features that create conditions in the new stand more typical of those found after natural disturbances (Pharo and Lindenmayer 2009). Recall that after most stand replacement natural disturbances, there is a considerable carryover of dead wood and also some live trees and shrubs into the new stand. Animal communities associated with stands that include these features probably would be more complex than communities in stands that lack similar components (Thompson and Desseker 1997, Chambers et al. 1999a). Diverse early successional conditions are indeed a rare feature on many landscapes where forest management has replaced natural disturbances over large areas (Swanson et al. 2010). Legacy features, however, may interfere with site preparation, vegetation management, and growth rates of the trees in the new stand, so they may only be desirable where land-management goals include resources besides timber.

Legacy trees are retained during overstory removal cuts from one rotation through most or all of the next rotation, with the new stand growing up around the reserve trees and retained habitat elements. The retained trees are not the same as seed trees. Seed trees are specifically left to provide a seed source for the next stand and then may or may not be removed. When seed trees are retained, they function in much the same way that a green-tree retention stand might (Sullivan and Sullivan 2001), but retention trees are specifically left as legacy structure in a stand although they may also provide seeds to the site. Green-tree retention and clearcut stands in Oregon seem to provide a similar habitat for many bird species when <25 trees/ha (<10/acre) are retained in retention stands (Chambers et al. 1999b). However, more species of birds typically use retention stands than clearcuts, at least in the breeding season (Vega 1993, Chambers et al. 1999b). More species were detected in uncut stands than in clearcut or retention stands during the winter (Chambers et al. 1999b). Other investigators reported similar findings in that animal responses to the retained structures were variable and dependent on the number of trees retained and their spatial pattern. The degree to which retention trees function in older stands that are nearing rotation age have not been so well studied.

### **SITE PREPARATION EFFECTS ON HABITAT ELEMENTS**

Once the site is harvested and legacy structures have been left, then often the site must be prepared as a suitable seed bed or planting spot for the new regeneration. Site preparation may not be necessary on some sites, especially if advance regeneration is adequate. On the other hand, to ensure successful survival and growth of newly planted seedlings in plantations, site preparation can be

quite intense. When managing southern pines or Douglas-fir, it is common to see a site burned and then mechanically manipulated to prepare the site for renewal. In the southern United States, mechanical site preparation may include bedding (plowing soil into raised beds where the trees are planted), chopping with a large rolling drum behind a bulldozer, or scraping the unwanted vegetation into piles or windrows with a bulldozer. This type of scarification may significantly affect the below ground structure of the stand by temporarily removing burrows and compacting soils (Todd and Andrews 2008). Burrowing species such as gopher tortoises in the southern Gulf Coast (Figure 3.9) do not seem to be directly affected by such activities, but the species that use their burrows may be (Mendonça et al. 2007). The intensity of site preparation may also affect plant communities that develop after the disturbance. Intense scarification, burning, or some herbicides may reduce shrub development in subsequent stages of stand development, at least temporarily (Newmaster et al. 2007). Alternatively, use of light fires without mechanical site preparation may proliferate the sprouting of shrubs and affect the presence or abundance of grasses and forbs in a young stand (Burton et al. 2010). The choice of site preparation influences not only the trajectory of the plant community that develops on the site, but also the level of residual “legacy” that remains after the treatment. Intense burns or mechanical scarification, for example, will usually either reduce levels of dead wood on the site or concentrate it into piles and windrows (Swift and Bell 2011). These intense treatments may be a desirable method for manipulating the habitat of some species that may influence regeneration success (such as mountain beaver), but it also may have adverse consequences for other species. In Oregon, species such as creeping voles and vagrant shrews increase after intense site preparation, but others such as Pacific and Trowbridge’s shrews, ensatina salamanders, and Pacific giant salamanders decrease after site preparation (Cole et al. 1997, 1998).

## NATURAL REGENERATION AND PLANTING OPTIONS

Land managers can determine the composition of developing stands by deciding which plant species will be re-established after site preparation. In many northern hardwood and boreal forests, advance regeneration may already be present. Clearcutting simply releases the existing regeneration (actually an overstory removal) and provides adequate light, water, or nutrients for the small trees to begin growing. In addition, the added light and heat striking the newly exposed forest floor can lead to abundant natural regeneration through germination of seed in the soil (i.e., seed bank) and through sprouting of roots and stumps. The species of plants that regenerate following a clear-cut can be quite diverse due to the availability of light, moisture, and nutrients, and the relative lack of competition. Within a short period of time, however, all growing space is occupied and there are simply too many plants competing for resources for all to grow quickly. In some instances, these plants may not be the species desired to meet habitat or timber objectives. Consequently, artificial regeneration is often used in conjunction with even-aged management approaches.

Plantations help to ensure that the appropriate species of trees are regenerated on a site to meet future objectives. Plantations can be, and often are, of a single species. From the standpoint of growing a certain product for a market, growing one species to a uniform size during one time period is economically efficient. For some animal species, the conditions provided in these uniform stands are desirable. For other species, the simple structure and composition found in monoculture plantations may not provide the habitat elements needed for survival and/or reproduction (Kerr 1999). For a variety of reasons, mixed tree species plantations are becoming more common but still represent a small fraction of the total plantation area in the world (Nichols et al. 2006). Planting one species over large areas can lead to increased risk of crop loss due to insects or disease. Swiss needle cast causes Douglas-fir trees to become defoliated but does not seriously affect western hemlock or Sitka spruce (Hansen et al. 2000). A species of invasive root rot is fatal to Port-Orford cedar but not Douglas-fir (Jules et al. 2002). Southern pine beetle may be a serious threat to loblolly or slash pines but not so seriously affect longleaf pines (Burns and Honkala 1990). So depending on the goal for the stand and the risks associated with losing certain species of trees or shrubs during stand



development, mixed species stands may carry less risk and also provide certain benefits in terms of plant diversity and vertical profile diversity as the stand develops.

Plantation establishment must not only consider the species to be planted, but also the nursery stock to be used and the spacing of seedlings in the plantation. Typically, tree seedlings are transplanted to a forest after 1–2 years in a nursery. Seedlings with larger root collar diameters (stem size at the ground line) survive and grow faster than smaller seedlings (South et al. 1995, Rose and Ketchum 2003). Genetic variability and genetic resistance to insects and disease can also be considered at this period of plantation establishment. By selecting planting stock from local seed sources adapted to the local conditions, trees are more likely to survive and grow well. Genetically modified (GM) tree seedlings may also be used and it is unclear what the long-term implications of using GM plants might be to development of habitat elements in plantations. Current efforts are focused on developing herbicide- and disease-resistant strains of trees so that competing vegetation can be more effectively controlled and trees can survive common diseases. If control of competing vegetation can be made more effective, then availability of grasses, forbs, and shrubs would be reduced significantly during the earliest stages of stand reestablishment. There is also the potential to use the GM trees to influence the concentrations of secondary metabolic compounds such as phenols and tannins in seedlings and make them less palatable to vertebrate and invertebrate herbivores. The risks associated with using GM materials include the potential for gene escapement to native plants that may have unwanted and unexpected effects on their growth, survival, and value as food to herbivores (Johnson and Kirby 2001).

Seedlings grown in a nursery are often rich in nutrients to give them the best opportunity to grow and survive when they are out-planted. In addition, newly planted seedlings may be fertilized or mulched to ensure survival and growth. These treatments tend to increase their value as browse for some species because they represent a nutrient-rich source of food for some herbivores (Crouch and Radwan 1981). In many circumstances, seedlings must be protected from browse and girdling by many species of herbivores, such as deer, elk, moose, voles, gophers, mountain beavers, and American beavers (Keeton 2008). But even intensive protection treatments such as placing vexar tubes around seedlings may not be effective in all circumstances (Brandeis et al. 2002).

One factor that can have a significant effect on the production of habitat elements later in plantation development is planting density. Planting density is easily adjusted during plantation establishment. Typically, 750–1250 trees/ha are planted to establish a plantation. If trees are planted on a wide spacing, then it takes longer for the crowns of the growing trees to begin to overlap one another. When crowns begin to overlap, sunlight is no longer available to other plants or branches beneath them. Wide spacing can lead to a longer lasting grass-forb-shrub phase during the stand initiation stage of stand development. It can also lead to deep crowns and abundant limbs low on the trees, thereby increasing the number of knots in the wood when it is harvested and reducing its value unless the limbs are pruned. Variable density planting of mixed species represents a more creative way of building horizontal complexity into a stand and can lead to quite variable vertical complexity in the stand. Mixed-species planting or variable-density planting or both can provide heterogeneity in an otherwise homogenous system.

## VEGETATION MANAGEMENT EFFECTS ON HABITAT ELEMENTS

Either chemical or manual vegetation control can influence the heterogeneity of developing stands. Spot control of competing vegetation could lead to a more heterogeneous stand than possible with broadcast application of a treatment. Manual control of many shrub species can lead to a proliferation of sprouts that increase amounts of available browse, but this may lead to an increase in the concentration of phenolic compounds in the leaves and twigs. Perhaps the most profound effect of vegetation management is the influence that such activities have on the future composition and structure of developing stands. Lack of any vegetation management in many northwestern or Gulf coastal sites may lead to stands with a large component of shrubs or hardwoods that will benefit some species of wildlife (Huff and Raley 1991), but interfere with tree growth and stand development.



Intensive vegetation management may lead to a conifer-dominated stand early in stand development with little shrub development unless stand density is manipulated as the stand develops. Hence, the balance between competition control and maintenance of nontimber plants requires careful thought and planning on the part of the silviculturist.

Prescribed burning is often used as a management tool to prepare sites for planting, prepare seedbeds, reduce competing vegetation, or reduce fuels and hence future fire risks. Fire is a natural and frequent event in many forest systems; so prescribed fire would seem to be a natural surrogate for uncontrolled wildfires. The choice of using fire, herbicides, or mechanical tools in forest management, either individually or collectively, is highly dependent on the types of habitat elements that a manager wishes to retain, remove, or develop in a stand. The effects that fire has on forest floor litter and dead wood can lead to adverse effects on forest floor amphibians and mammals (Randall-Parker and Miller 2002, Schurbon and Fauth 2003). On the other hand, forage quality and availability are often enhanced for herbivores following prescribed burning (e.g., Canon et al. 1987, Sparks et al. 1998). Again, as with any other forest management approach, there are species that will likely benefit from the treatment and others that will not likely benefit. The winners and losers in proposed management must be carefully considered prior to initiating a prescription, or plan, for a stand.

### Herbicides and Pesticides

Use of chemicals to control vegetation in forest management is a controversial issue in many parts of the world. Effects of chemical use can be direct (the chemical affects the animal physiologically) or indirect (the chemical affects the habitat that in turn influences animal fitness). Direct effects can be lethal or sublethal, and sublethal effects can take many forms. Sublethal effects can alter an animal's behavior, alter hormonal levels, or disrupt the function of the nervous system, among other consequences. Sublethal effects can lead to a higher likelihood of the animal being killed by a predator, a disease, or a parasite. Specific examples of direct and indirect effects are provided, but the crux of the debate is around the ethics of knowingly killing or harming animals. Application of chemicals in typical forestry operations are infrequent (Christmas tree farms and other intensive plantations are an exception). Typically, herbicide may be sprayed prior to or immediately after planting and then perhaps once or twice to release seedlings until they overtop the competing vegetation. Insecticides are rarely used except in irruptions of populations that defoliate trees or cause tree mortality.

Consider three scenarios.

Scenario 1: If a chemical application kills 100 individuals of one species, say 80% of the population occurring in the stand that was sprayed, then the impact to that population is significant but may be a miniscule percentage of the entire population of that species. Over time the remaining individuals may repopulate the stand if the habitat is adequate.

Scenario 2: A chemical is applied to a stand and no animals are killed and no sublethal effects are expected, but the habitat is changed such that no individuals of this species can be found in the stand after 3 years. Where did they go? Did they all leave? Die? Simply not reproduce?

Scenario 3: No chemicals are applied and succession is allowed to advance such that the shrubs on which the species relies are absent after 10 years and the species can no longer be found in the stand. Where did they go? Did they all leave? Die? Simply not reproduce?

Which of these three scenarios is acceptable to society? Which presents an ethical dilemma? For most people in our society, applying a chemical to a site with a risk that some of the animals may die or be adversely affected physiologically is less acceptable than changing the habitat or simply letting succession cause the species to go locally extinct. This is most likely the case because if other species can be adversely affected by exposure to a chemical then the assumption is that humans may be similarly affected. In most instances, the risk to human health trumps risks to other species.

There has been growing acceptance of integrated pest management as a means of reducing reliance on chemical solutions to pest and weed problems. Although part of the reason is risk to health of humans and other species, part if the rationale is economic. Pimentel (2005) estimated economic and environmental losses due to the application of pesticides in agriculture, home use, and forestry

(which accounts for a small portion of total use) in the United States were: public health, \$1 billion/year; pesticide resistance in pests, \$1.5 billion; crop losses caused by pesticides, \$1.1 billion; bird losses due to pesticides, \$2.2 billion; and ground water contamination, \$2.0 billion.

### Direct Effects

Given the number of species that could be affected by chemical exposure in forests in combination with the number of chemicals that could be used to address control of plants and insect pests, the possible combinations of species  $\times$  chemical is immense. Bernanke and Kohler (2009) provide an excellent overview of potential effects of a suite of environmental contaminants on birds, mammals, reptiles, amphibians, and fish. A few examples of potential effects of chemicals used in forestry practices are provided simply to raise the issue of possible direct effects, but it is important to remember that different species respond to different chemicals in different ways.

Some types of pesticides have the potential to affect developmental processes in larval amphibians or reproduction of adults, though early life stages seem to be particularly vulnerable to some chemicals (Bernanke and Kohler 2009). Although direct mortality was not detected in adult amphibians (McComb et al. 2008), larval individuals can be affected and sublethal effects have been detected. In some cases, exposing larvae to 1 or 2 ppm Roundup® (active ingredient glyphosate) for 43 days in a laboratory resulted in increased mortality, earlier metamorphosis, and smaller-sized adults (Swift and Bell 2011). Swift and Bell (2011) cautioned that these effects may not be seen under typical applications of this chemical in forest management.

Another example is an arsenic-based chemical, MSMA that was widely used in Canada to control bark-beetles, which were then eaten by woodpeckers. MSMA is no longer allowed for such use in Canada, though its use was shown to have the potential to have direct effects on woodpeckers (Morrissey and Elliott 2011). In cases such as this, the prey species for the woodpecker were also adversely affected, so even if there was no direct mortality, reduced prey availability could reduce individual and population fitness (Awkerman et al. 2011).

### Indirect Effects

Management of competing vegetation can significantly affect the availability of certain plant species as food and cover. Herbicide applications that release conifers can temporarily decrease the availability of shrubs for shrub-nesting birds (Morrison and Meslow 1984, Easton and Martin 1998). For small mammals and amphibians, changes in plant species composition and cover as a result of site preparation seems to have a greater effect on animal abundance than spraying of glyphosate herbicide following plantation establishment (Cole et al. 1997, 1998).

How vegetation is controlled has different effects on different species. Santillo et al. (1989) found fewer small mammals on glyphosate herbicide-treated clearcuts for 1–3 years after spraying compared with untreated clearcuts in Maine. Insectivores seemed to be most significantly affected by herbicide spraying in this study probably because of the effect that herbicides had on the vegetation and habitat for forest floor invertebrates. Herbivores also seemed to be adversely affected until vegetation recovered from spraying (Santillo et al. 1989). Santillo et al. (1989) suggested that changes in mammal abundance seemed to be associated with herbicide-induced reductions in invertebrates and plant food and cover. Patches of untreated vegetation within herbicide-treated clearcuts may provide a source of invertebrates and plants for those species adversely affected by herbicide spraying.

Herbicide applications that release conifers can temporarily decrease the availability of browse during early stages of stand development, but they also can increase the availability of browse or herbaceous cover when applied later in stand development. Jones et al. (2009) found that a combination of spraying with imazapyr and prescribed burning opened the hardwood canopy and provided an opportunity for a more diverse herbaceous layer to develop in southern U.S. pine plantations. Similarly, Sullivan and Sullivan (2003) found that plant species diversity was not adversely affected by glyphosate spraying in young conifer plantations in British Columbia (BC). Responses of plant species to herbicides in forest ecosystems differ from responses in agro-ecosystems where

glyphosate is used repeatedly to reduce non-crop vegetation (weeds) in most situations. In forests, some species of birds and small mammals decline temporarily following spraying, while other species increase in abundance. Management for a mosaic of vegetation conditions within forested landscapes should help ameliorate the short-term changes in species composition accompanying vegetation management using herbicides.

Herbicides are also an important tool for control of invasive plant species, allowing recovery of native plant communities that can have positive effects on habitat quality for some species of animals (Miller et al. 2010). In many cases, invasive species may actually increase in dominance when other types of control mechanisms are used, such as manual cutting or fire (some species sprout prolifically or find a suitable seedbed in ash). Herbicides may be the most effective and cost-efficient approach to control (Flory and Clay 2009).

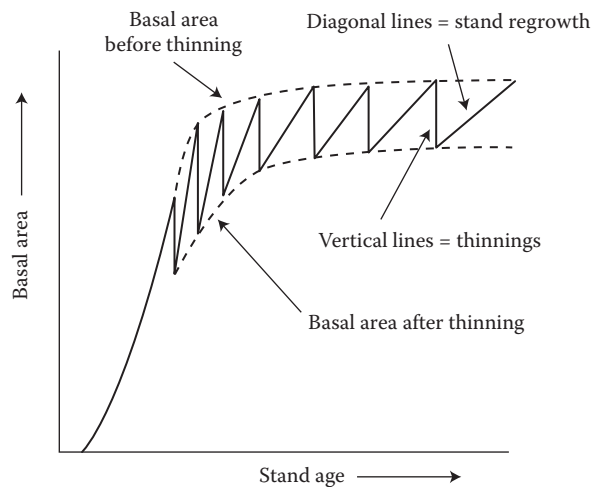
### PRECOMMERCIAL THINNING

Precommercial thinning involves cutting or killing trees to achieve the desired density that promotes the growth of the residual trees and the development of desired stand structure and composition. This type of treatment is done early in stand development before the trees are large enough to be sold for a profit or at least offset the cost of the thinning. This activity is a net cost to the landowner. The value of precommercial thinning is to concentrate growth on the remaining trees and allow them to grow faster. Further it increases the space between the crowns of the trees in the stand and may prolong the stand initiation (grass-forb-shrub) stage of forest development. This type of management can lead to extending the period of stand development that provides habitat for many early successional vertebrate species (Dellasala et al. 1996). Openings of less than 0.1 ha (0.25 acres) in second-growth spruce–hemlock stands in southeast Alaska provide food close to cover for Sitka black-tailed deer and allowed certain bird species to occur in otherwise unsuitable habitat (Dellasala et al. 1996). Other species respond quite differently to precommercial thinning. Indices to snowshoe hare abundance in precommercially thinned stands was about half that in unthinned stands in Maine (Homyack et al. 2007), which in turn has implications for lynx (Vashon et al. 2008).

Timing of the thinning is critical. If the thinning occurs just as crowns are closing but before early successional plants have been shaded out of the stand, then the early successional conditions will be maintained later into stand development. If the thinning is done after crowns have closed and shaded out the grasses, forbs, and shrubs, then the newly created openings in the stand will be filled by those species found in the seed bank in the soil. Some of these species may be different from those species found early in stand development. Precommercial thinning influences the structure and composition of the understory and may consequently influence the vertical and horizontal complexity in a stand. Precommercial thinning can also provide the opportunity to shift the dominant plant species in a stand. Preferential cutting of hardwoods in a mixed species stand can shift stand development more to conifers, or vice versa.

### COMMERCIAL THINNING

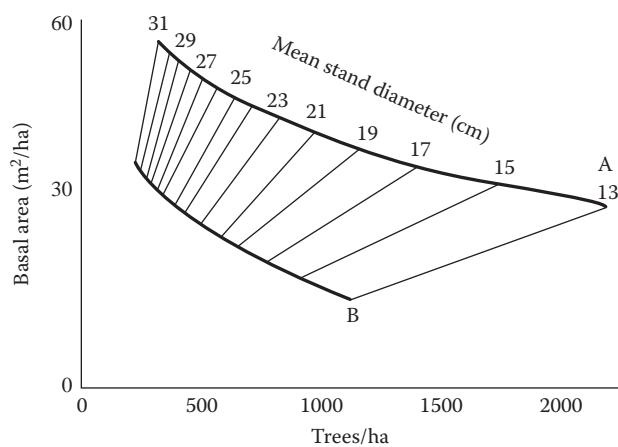
Commercial thinning is a more common intermediate treatment used during stand management. Commercial thinning manipulates stand density by harvesting trees that can be sold and thereby provide some income to at least offset the cost of the stand management treatments. As illustrated in Figure 8.3, basal area declines abruptly as trees are removed during thinning, and then the stand regrows to recover the growing space that was lost by the trees that were removed. It also is important to realize that stands that may begin with 10,000 trees/ha (4000 trees/acre) at year 10 likely will have only 500 trees/ha (200 trees/acre) at age 100. So thinning can occur naturally (self-thinning caused by competition or fire or wind) or by humans. In most forests, however, most plants that establish after a stand-replacement disturbance die in the earliest stages of stand development through competition for resources, regardless of whether humans cut or spray them.



**FIGURE 8.3** Stands increase in basal area over time. Thinning reduces basal area and reduces the number of trees in a stand. The residual trees grow more rapidly and recover basal area, thereby focusing increased growth on fewer, larger trees. (Smith, D.M. et al., *The Practice of Silviculture: Applied Forest Ecology*, 1997, Copyright Wiley-VCH Verlag GmbH & Co. KGaA. Reproduced with permission.)

However, the plants that do survive this period of plant competition may not always be the plants that humans want. Management action may be necessary. Management is not an ecological necessity; it is done to meet human needs and desires.

How many trees should be cut? How many should be left? One way of estimating the effects of thinning on stand growth and structure is to understand the stocking in the stand or the amount of the stand that is covered by trees in relation to the density of the stand. Stocking charts have been developed for even-aged stands in many forests in the world. Figure 8.4 provides an example of a stocking chart for an eastern white pine stand in the United States. To understand this diagram,

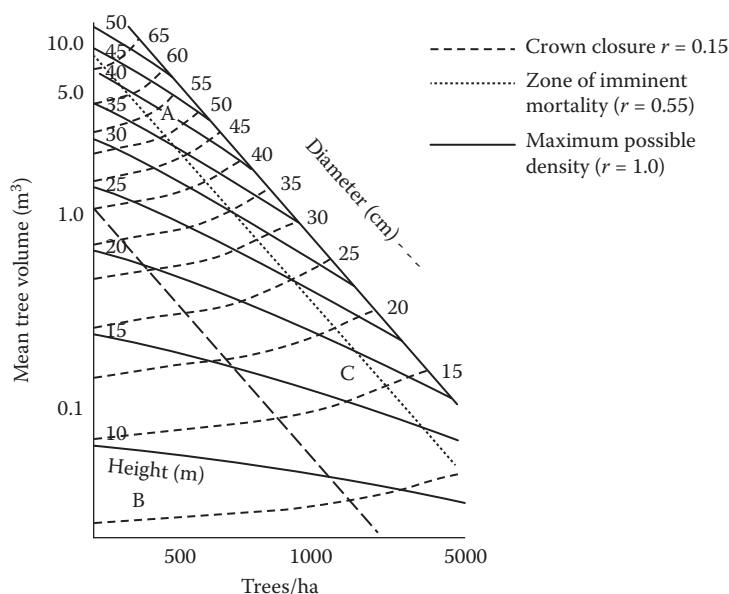


**FIGURE 8.4** Eastern white pine stocking chart. Stands above the A line are considered to be overstocked (growing slowly, trees likely to die from competition). Stands below the B line are under-stocked (not all growing space is being used). Stands between the A and B line are considered to be fully stocked (growing well, all growing space is occupied). (Redrafted from Wendel, G.W. and H.C. Smith. 1990. *Silvics of North America*, Vol. 1, Conifers. USDA For. Serv. Agric. Handb. 654.)

work through an example. On the  $x$ -axis find stand density 200, that is, 200 trees/acre (500 trees/ha). Draw a line vertically (parallel to the  $y$ -axis) from this point. On the  $y$ -axis find 200, that is, 200 ft<sup>2</sup>/acre (46 m<sup>2</sup>/ha) of basal area. This is the area of the stand covered by the boles of growing trees at 1.2 m (4.5 ft) above ground. Draw a horizontal line from 200 ft<sup>2</sup>/acre parallel to the  $x$ -axis. Where these two lines intersect is the stocking level for a stand with 200 trees/acre that occupy 200 ft<sup>2</sup> of basal area per acre. Since most trees in a single-species even-aged stand are approximately the same size, we can also estimate the average diameter of trees where these two lines cross (about 14 in [36 cm] dbh [diameter at breast height] in this case).

There are several things to notice. First, if you follow your vertical line up (increase basal area at the same number of trees per acre), tree diameters increase, up to a point. As trees get larger, eventually all growing space in the stand is occupied by trees and they begin to compete with one another. You can pack only so many trees into a hectare before some start to die. The relationship between basal area and density can also be portrayed as the relative density—that is the density of the stand at a given basal area. For example in Figure 8.5, draw a vertical line up from 1000 trees/ha on the  $x$ -axis. Eventually, that line will cross the upper diagonal line on this chart. This upper line indicates the maximum number of trees at a given diameter that can occur in the stand. The stand cannot add any more basal area (growing trees) at that density unless some trees in the stand die from competition. So as trees in the stand grow, some have to die and stand density will decrease. If natural disturbances or humans cutting trees do not thin the stands, then inter-tree competition leads to tree death within the stand. Trees that die from suppression mortality usually are about half the diameter of the dominant and codominant trees in the stand, which is important to keep in mind when thinking about snag formation from tree competition.

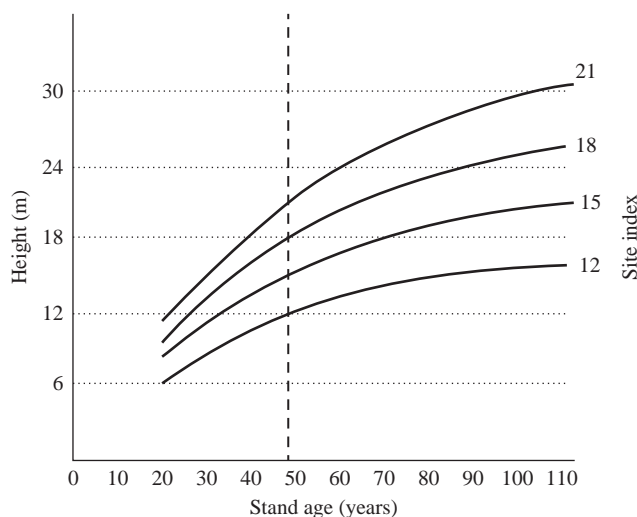
Note that neither of these charts includes anything about tree age. The rate at which trees grow to certain sizes at a given stand density and basal area is dependent on the quality of the site to grow trees. Some sites grow trees very rapidly and other sites grow the same species of tree much more slowly. Silviculturists use the term *site index* to indicate the potential of a site to grow trees in height. Why height and not diameter? Tree diameter growth is highly influenced by stand density.



**FIGURE 8.5** Stand density index diagram for coastal Douglas-fir. Stand density cannot exceed the upper diagonal line as trees compete for growing space. (Adapted and redrafted from Cameron, I.R. 1988. *An Evaluation of the Density Management Diagram for Coastal Douglas-fir*. B.C. Ministry of Forests and lands, Victoria, BC.)

A tree growing among many other trees is competing for resources and so allocates the resources it is able to capture into height growth first and diameter growth second. A tree growing in the open may not grow in height much differently than one growing in a dense stand, but the open-grown tree will have the resources that enable it to grow rapidly in diameter. Consequently, the better indicator of growth potential of a site is height because it is more independent of stand density but dependent on site quality (Figure 8.6). Foresters use the height of the dominant trees in an even-aged stand at a specified age (usually 50 years) as an index to site quality. Figure 8.6 is an example of site index curves for northern red oak in the eastern United States. Stand development, such as diameter growth, volume growth, and other features, will occur more slowly on sites with a lower site index. On high-quality sites, trees grow in diameter faster and stands increase in basal area (i.e., move through a stocking chart) more rapidly than on lower quality sites. The rate at which a stand gains volume and some habitat elements is dependent on the quality of the site.

A stocking chart (Figure 8.4) or density management diagram (Figure 8.5) can be a useful mechanism for biologists and silviculturists to work together to design stands to meet certain needs. If managers wish to produce large snags or logs, it is best to allow the trees in the stand to grow as quickly as possible until the codominant trees reach a size that is desired, and then let competition mortality kill some of those trees. In Figure 8.5, location A in the diagram would be a stand condition where 55-cm trees are competing with one another and some will have to die to allow the rest to grow. Small trees, those about half the diameter of the live trees (25–30 cm in diameter), would begin to die and add dead wood (snags and logs) to the stand. Opening the crowns by thinning also influences habitat quality for those species that find cover and food in tree crowns and in the flush of understory vegetation that might occur following thinning (location B in Figure 8.5). For instance, species that feed among tree crowns, such as Hammond's flycatchers, which perch on a branch and sally out to a space between crowns to catch flying insects, may benefit from thinning (Hagar et al. 1996). Thinning densely stocked conifer stands in landscapes dominated by younger stands enhances habitat suitability for several species of mammals and birds, but some unthinned patches and stands may be retained to provide refugia for bird species that are impacted by thinning (Location C in Figure 8.5, Hayes et al. 2003, Suzuki and Hayes 2003). Variable-density thinning can produce a wide range of tree diameters and greatly influence the production of small snags early in stand development (Carey and Wilson 2001). And some species seem to be adversely affected by thinning. Northern



**FIGURE 8.6** Site index curves for northern red oak in the Lake States. (Redrafted from Gevorkiantz, S.R. 1957. Site index curves for red oak in the Lake States. USDA For. Serv. Tech. Note 495.)



flying squirrels, an important prey species for several species of owls, seem to be negatively affected by thinning for 10 years or more after thinning has occurred (Manning et al. 2012). Thinning is conducted to accelerate development of late forest structure or restore forest functions when fire has been excluded and fuels have built up (Stephens and Alexander 2011). Consequently, when managers plan thinning treatments, the spatial arrangement of thinned and unthinned stands across a landscape over time is critical to maintaining the full suite of species that may be found in managed forests.

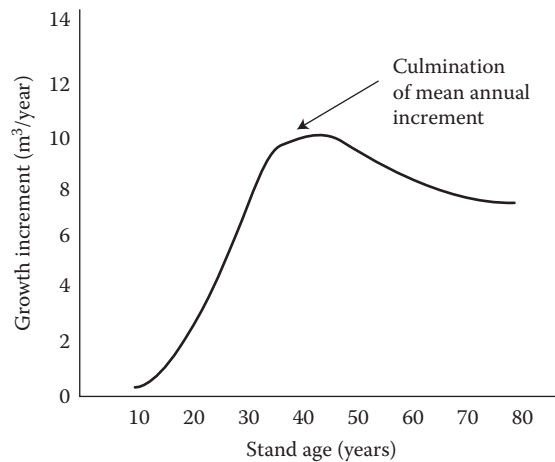
## FERTILIZATION

Fertilization of forest stands has two dominant potential effects on habitat elements: increased diameter growth on the dominant trees and increased nutrient content in browse and forage. Added tree growth can be particularly important from a habitat complexity standpoint if some trees with added growth are retained into the next rotation. The returns in tree growth compared with the investments of fertilizer application are often most apparent on nutrient poor sites.

On nutrient-deficient sites, fertilizer applications can increase tree growth and can also increase forage production. Nutrient poor sites in the pine flatwoods of the Gulf coast are sites where fertilization seems to provide benefit to growing trees for forest products and should benefit forage for herbivores as well (Tiarks and Haywood 1996). The added nutrients that find their way into the leaves and twigs of browse plants and herbaceous forage seem to provide benefit to herbivores in a wide range of forest types. In Scandinavian conifer stands during the winter and summer following fertilization, moose strongly selected fertilized plots over unfertilized sites and hares left more fecal pellets in the fertilized plots than in untreated sites, indicating that they probably were using these fertilized patches more than unfertilized ones (Ball et al. 2000). Also in Scandinavia, fertilized stands of Norway spruce supported 38% more bird species and 21% more individuals than unfertilized stands (Edenius et al. 2011), suggesting that increased nutrient availability changed the stand structure, and perhaps increased arthropods or fruit production allowing effects to ripple through trophic levels to insectivores and fructivores as well as herbivores. Gibbens and Pieper (1962) found that fertilization increased growth and palatability of the shrubs in deer winter range in California resulting in selective thinning and browsing by herbivores. Increased browse production and use by elk has been reported following fertilizer application in Washington (Pierson et al. 1967). Selective browsing by deer on fertilized plants can reduce deer damage to fir seedlings on fertilized areas (Rieck and Jeffrey 1964 in Scotter 1980). The use of fertilizer and herbicides in combination can also benefit ungulate food resources by stimulating plant growth and by making plants more palatable (Carpenter and Williams 1972).

## ROTATION LENGTH: ECOLOGICAL AND ECONOMIC TRADE-OFFS

Even-aged stands typically gain volume over time following a logistic or S-shaped curve. The average volume growth per hectare per year (mean annual increment, MAI) will peak at a point where it is economically most efficient to harvest the stand. This is called the culmination of MAI (Figure 8.7). The stand will have peaked in volume growth and delaying harvest beyond this point means that you are investing money in maintaining a stand that is no longer maximizing a profit. The point in stand development when the manager decides to harvest the stand and regenerate a new stand is the *rotation age*. The *economic rotation age* is often determined by the culmination of MAI or the point at which the stand growth begins to increase at a decreasing rate. This peak will vary depending on the site quality for the stand, the products being managed (because volumes are estimated based on product values), and the interest rate associated with income from the products derived from the stand. The culmination of MAI will occur earlier on high-quality sites (high site index), and for products requiring small tree diameters (e.g., pulp) than on poor-quality sites or for large sawtimber or veneer products. Most industrial forests will harvest stands before the culmination of MAI, and try not to carry too many stands longer than the culmination of MAI unless there is a need to provide larger products to a mill.

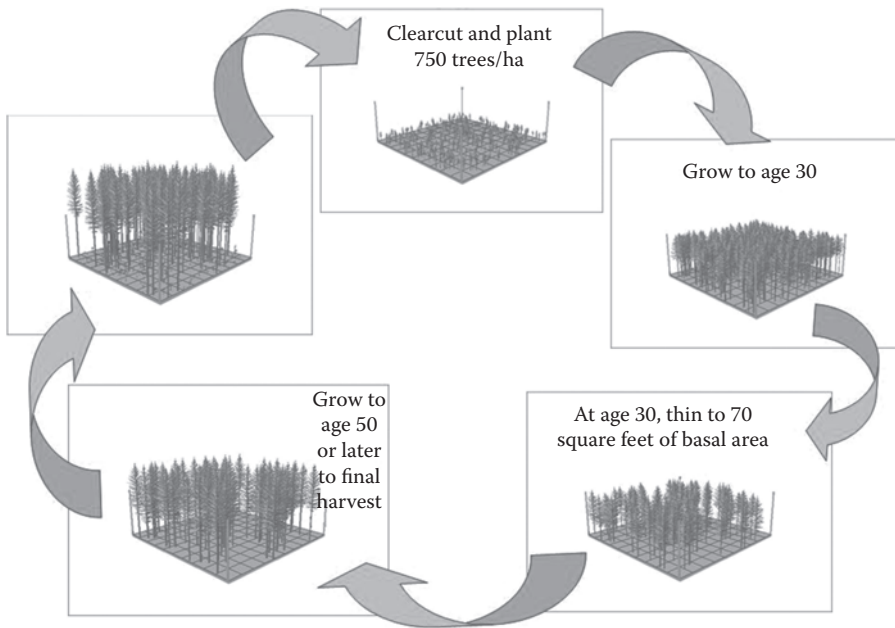


**FIGURE 8.7** Change in MAI with stand age. As MAI begins to decline, stand volume is increasing at a decreasing rate and will reduce net income to the landowner over time. (Adapted and redrawn from Smith, D.M. et al., *The Practice of Silviculture: Applied Forest Ecology*, 1997, Copyright Wiley-VCH Verlag GmbH & Co. KGaA.)

The *ecological rotation age* is the average interval between stand replacement disturbances that are likely to regenerate a stand. There often is a very large difference between an economic rotation and an ecological rotation. Liu et al. (1994) modeled trade-offs between providing maximum economic return and habitat for Bachman's sparrows in southern pine forests. They found that the economic value (land expectation value) peaked at 20 years, but that rotation lengths shorter than 80 years resulted in declines in Bachman's sparrow population size (Liu et al. 1994). Species of plants and animals that typically occur in stands older than the economic rotation age will lose habitat in forests managed to maximize profit. However, the economic rotation age can be extended by thinning and by changing product goals. Further, the use of variable density plantings and thinnings, retention of legacy, and use of mixed species stands can provide many of the elements in young stands that normally might be found in older or unmanaged stands.

## CASE STUDY: DOUGLAS-FIR PLANTATION

This case study provides an example of the difference in the potential structure of even-aged stands managed using approaches that consider habitat complexity. These two stands are from the mid-elevations of the western Cascades in Washington (data and projections based on Landscape Management Systems, McCarter et al. 1998). Traditional management of Douglas-fir plantations involves clearcutting followed by site preparation and vegetation management that prepares a planting site and removes as much competing vegetation as possible. In this example, 750 seedlings/ha (300 trees/acre) were planted and allowed to grow to stand age 30 as an even-aged single-species stand (Figure 8.8). At age 30, there was approximately 28 m<sup>2</sup>/ha (120 ft<sup>2</sup>/acre) of basal area, and 675 trees/ha with an average dbh of almost 23 cm (9 in) (Figure 8.9). At this point, a commercial thin could be implemented removing about 2 MBF (thousand board feet) per acre (5 MBF/ha) by removing those trees that would likely die through competition mortality and concentrating the remaining growth on the residual trees. The simulated thinning represents a reduction in basal area to 16 m<sup>2</sup>/ha (70 ft<sup>2</sup>/acre) and by stand age 50, there would be about 69 MBF/ha (28 MBF/acre) of volume to harvest in 160 trees/ha. The trees would average about 50 cm (20 in) in diameter. If sawtimber sold for \$400/MBF, then the landowner would gross approximately \$27,664/ha (\$11,200/acre) from which harvesting, planting, and site preparation costs must be deducted.

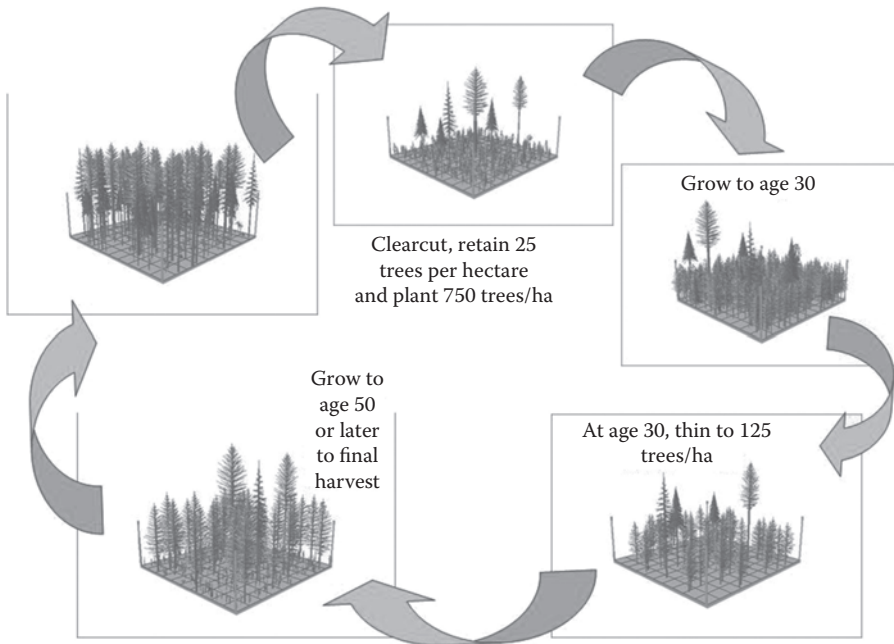


**FIGURE 8.8** Example of plantation establishment and management in a Douglas-fir stand in the western Cascades of Washington. Data from Landscape Management Systems, University of Washington, Seattle. (McCarter, J.M., et al. 1998. *Journal of Forestry*, 96(6):17–23.)

Now consider a stand managed to enhance stand complexity. The stand is clearcut with 25 trees/ha (10 trees/acre) retained from the previous stand and representing five species (Douglas-fir, western hemlock, bigleaf maple, western redcedar, and black cottonwood). The site is then prepared for planting taking care to retain advance regeneration of as many species as possible, but also planting a mixture of tree seedlings representing the above species plus red alder (Figure 8.10, see Figure 8.11 for a similar approach from central Washington). At stand age 30,



**FIGURE 8.9** Plantation of Douglas-fir on forest industry land in western Oregon. Note the uniformity in tree size and the lack of vertical structural complexity in this heavily stocked, even-aged stand.



**FIGURE 8.10** Example of plantation establishment and management in a Douglas-fir stand in the western Cascades of Washington that includes legacy trees, a multispecies planation, and thinning to maintain structural complexity and tree species diversity. Data from Landscape Management Systems, University of Washington, Seattle. (McCarter, J.M. et al. 1998. *Journal of Forestry*, 96(6):17–23.)



**FIGURE 8.11** Seed tree regeneration method implemented in the north central Cascades of Washington. The harvest was designed to retain seed trees of ponderosa pine and western larch, with a legacy of dead trees and other species, leading to increased structural and compositional complexity early in stand development.



**FIGURE 8.12** Green-tree retention stand in western Oregon 15 years after harvesting and planting. Note the increased level of structural complexity provided by regeneration, retained green trees that are still growing in size, and snags that were created following tree harvest.

there are approximately 865 trees/ha (350 trees/acre) representing seven different species, and having a total basal area of approximately 41 m<sup>2</sup>/ha (180 ft<sup>2</sup>/acre) and greater complexity (Figures 8.8 and 8.12). Approximately 7.5 MBF/ha (3 MBF/acre) are then removed in a commercial thin that reduces the stand density to 125 trees/ha (50 trees/acre), and the site is under-planted with a mix of five tree species to develop a second age class. At age 50, there is a broad range of tree diameters, multiple species, and approximately 84 MBF/ha (34 MBF/acre) of timber. If this mixed species sawtimber sold for \$300/MBF, then the landowner would gross approximately \$25,194/ha (\$10,200/acre) from which harvesting, two plantings, and site preparation costs must be deducted. Hence, the bottom line profit would be less, but there still would be a profit and money to pay for developing habitat complexity.

## SUMMARY

Even-aged management typically involves the use of clearcut, seed-tree, or shelterwood regeneration methods to establish a new stand consisting of one dominant tree age. The choice of the regeneration method is dependent on the characteristics of the site and the desired stand composition and structure in the future. Bird species richness and abundance are significantly higher in complex than in structurally simple plantations (Nájera and Simonetti 2009), and presumably other vertebrate taxa would be similarly affected. Stand complexity can be influenced by manipulating the number, size, and arrangement of habitat elements. Retaining legacy from the previous stand, and choices in the intensity and arrangement of site preparation, planting, and thinning activities all influence future stand complexity. The combination of choices made throughout a rotation can lead



to structurally simple or complex stands at rotation age. Hartley (2002) provided some useful and simple guidelines that should be considered if a forest manager wishes to increase complexity in managed even-aged stands:

- Retain dead and living trees in the stand or in retention islands or strips where they will not interfere too badly with other stand management activities. Snag and reserve tree management (e.g., leave strips). These should include mature native trees and/or understory vegetation left unharvested or allowed to regenerate.
- Polycultures should be favored over monocultures by planting multiple crop species and/or leaving some native trees unharvested.
- Native species should generally be favored over exotics.
- Site preparation should favor methods that reflect natural disturbances and conserve dead wood.
- Thin early and frequently to retain horizontal complexity in structure and composition.
- Extend rotations as long as possible.
- Grow some crop trees through two rather than one rotation.

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# 9 Silviculture and Habitat Management

## *Uneven-Aged Systems*

Even-aged management is the most common approach to managing forests for commodity production, but on many of the privately owned forests where a single forest owner might manage 2–4 ha, uneven-aged systems are more consistent with owner goals (Gagnon and Jokela 2010). Where certain characteristics of forest structure and composition are desired while still managing a forest for commodities, uneven-aged management represents a useful and practical approach, regardless of landowner. Although uneven-aged systems may not maximize financial income, this system does have ecological and sociocultural benefits (Axelsson and Angelstam 2011).

### CHARACTERISTICS OF UNEVEN-AGED STANDS

Uneven-aged stands consist of three or more age classes represented in the same stand. Managing a variety of tree ages in the same stand can be challenging because of the potential for large trees to outcompete smaller trees for growing space. Further, in mixed species stands each tree species varies in its tolerance to shade, water, or nutrients so management becomes even more challenging. The advantage to managing stands using an uneven-aged system is that there are trees of various ages and sizes in the same stand at all times. So from the standpoint of providing homogeneity over a large area, or for a small private landowner to always have trees on her property, uneven-aged stands can be an attractive alternative to even-aged stands. Because there should be a variety of tree sizes represented in the same stand, vertical and horizontal complexity and tree size diversity can be high and can provide a diverse set of food and cover resources for many, but not all, species in a region. Further, many old-growth stands have structural characteristics typical of uneven-aged stands, but uneven-aged stands should not be considered a substitute for old-growth. Old-growth stands often have much higher stocking levels than managed uneven-aged stands, and most trees in a managed stand are not allowed to get very old. Old trees, regardless of their size, support some habitat elements, such as tree hollows, lichen communities, and large dead limbs, that can only develop over a long period of time.

In order to maintain an uneven-aged stands, the manager must be sure that there are always enough small, young trees to replace larger trees that are harvested or die over time. Consequently, there are three primary factors defining an uneven-aged stand and the habitat elements therein:

1. Basal area—how much growing space is occupied?
2. Tree density in each diameter class—are there enough small diameter trees to replace the larger ones that are cut?
3. Target tree size—what diameter class represents the largest harvestable trees in the stand?

First it is important to understand when cutting occurs. Uneven-aged management is based on a *cutting cycle* or a period of time between harvests when some trees of all tree diameters are cut. A typical cutting cycle is once every 15–30 years in most North American managed forests.

The following example is for a fully regulated balanced uneven-aged stand—something that rarely exists. But it provides the conceptual basis for approaching uneven-aged management. It may

be easiest to understand how uneven-aged management operates by working backward from the target tree size. Let us assume that the forest manager wants at least 1 tree/ha (1/2.5 acres) that is 76 cm (30 in) in dbh (diameter at breast height) in the stand at all times and that she is using a 15-year cutting cycle. When the manager decides to cut these 76-cm trees, there must be at least 1 tree/ha that is 71 cm (28 in) dbh to grow to be 76 cm during the next 15 years. And there must be at least 1/ha that is 66 cm (26 in) to grow to be 71 cm, and so on. This means that you need at least 1 tree/ha of all size classes down to the smallest size class that is the regeneration you want to establish (Table 9.1). But that would be a perfect world, and we know that unpredictable things happen in forests so we always try to have more trees in each smaller size class. In our example, let us say you want twice as many trees in each successively smaller diameter class, so you may want to have 1 tree/ha that is 76 cm dbh, but 2 that are 71 cm, and 4 that are 66 cm, and so on.

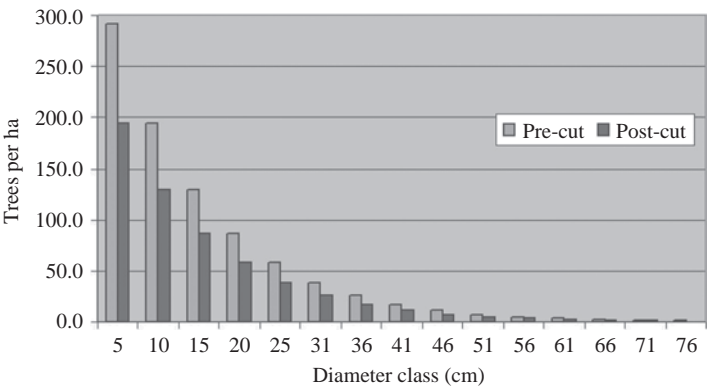
But if you calculate the number of trees that you would need in the smallest diameter class, then you would need to have over 16,000 5-cm (2-in) trees per ha (Table 9.1)! If you could get that many seedlings started per hectare in a stand, their growth would probably be very slow because the stocking would be impossibly high. It simply is impossible to maintain a stand like this. But if we wished to maintain a reasonable stocking level where there is enough room for all trees to grow, then we would want to have 1.5 trees/ha in each successively smaller diameter class in this example.

Note that under ideal circumstances there is a negative exponential distribution of tree diameters in an uneven-aged stand (Figure 9.1). Also that the shape of the curve is a function of the factor by which you multiply the number of trees in one diameter class to get the number in the next smallest diameter class. This is known as the *Q factor*. The higher the Q the steeper the curve (more small trees). The larger the target tree size, the longer or more protracted the curve. But because growing space is limited, setting a large target tree size automatically means that there have to be many fewer small trees to allow space for all trees to grow. Consequently, managers will estimate basal area for a stand and see if the stand is overstocked (trees dying from competition), fully stocked (growing without imminent tree mortality), or understocked (not all growing space is being used). In our previous example, let us assume that 25 m<sup>2</sup>/ha (110 ft<sup>2</sup>/acre) of basal area is a fully stocked stand.

**TABLE 9.1**  
**Examples of Tree Densities by Diameter Classes for Three Diameter**  
**Distributions with a Target Tree Size of 76 cm (30 in)**

dbh (cm)	Trees per ha Q = 2.0	Trees per ha Q = 1.5	Trees per ha Q = 1
76	1.0	1.0	1.0
71	2.0	1.5	1.0
66	4.0	2.3	1.0
61	8.0	3.4	1.0
56	16.0	5.1	1.0
51	32.0	7.6	1.0
46	64.0	11.4	1.0
41	128.0	17.1	1.0
36	256.0	25.6	1.0
31	512.0	38.4	1.0
25	1024.0	57.7	1.0
20	2048.0	86.5	1.0
15	4096.0	129.7	1.0
10	8192.0	194.6	1.0
5	16384.0	291.9	1.0
Basal area (m <sup>2</sup> /ha)	407	25	2.5
	Impossible!	Fully stocked	Understocked





**FIGURE 9.1** Idealized distribution of tree sizes in an uneven-aged stand. Note that trees of all tree size classes are harvested during each cutting cycle.

When this point is reached at the end of a cutting cycle, we may want to reduce the basal area to 16 m<sup>2</sup>/ha (70 ft<sup>2</sup>/acre) to provide more growing space for the remaining trees and allow the remaining trees to grow faster. But we want all trees to grow in diameter, not just the big ones, so we have to provide growing space for all tree size classes. Therefore, a harvest that would reduce the basal area to 16 m<sup>2</sup>/ha would remove some trees from each size class.

After harvesting, the remaining trees will grow into the larger size classes and replace those that were cut. In addition, there is a new influx of regeneration established by creating a *seedbed* or growing site for seedlings and sprouts that will replace the trees in the 5-cm (2-in) dbh class that grew larger. It is important to realize, though, that if the minimum marketable tree size is 31 cm (12 in) dbh, then only those trees 31 cm dbh or larger that are cut can be sold for a profit (Table 9.2).

**TABLE 9.2**  
**Example of Trees Harvested (Bold = Merchantable)**  
**by Diameter Class in an Idealized Uneven-Aged**  
**Stand at the End of a Cutting Cycle**

dbh cm (in)	Trees Cut/ha	Trees Cut/acre
5(2)	97.3	39
10(4)	64.9	26
15(6)	43.2	17
20(8)	28.8	12
25(10)	19.2	8
<b>31(12)</b>	12.8	5
<b>36(14)</b>	8.5	3
<b>41(16)</b>	5.7	2
<b>46(18)</b>	3.8	1.5
<b>51(20)</b>	2.5	1.0
<b>56(22)</b>	1.7	0.7
<b>61(24)</b>	1.1	0.4
<b>66(26)</b>	0.8	0.3
<b>71(28)</b>	0.5	0.2
<b>76(30)</b>	1.0	0.4

*Note:* See Figure 9.1.

Trees <31 cm dbh are cut to provide growing space at a cost to the landowner and effectively represent a precommercial thin. In addition, the stand is harvested more frequently than an even-aged stand might be harvested, increasing harvesting costs and impacting the site more often. Finally, less timber volume is removed during each harvest than in an even-aged stand so the net short-term profit to the landowner may be less, but she will generate a more regular income from the property, which will come at the end of every cutting cycle instead of at the end of a rotation.

The above is an idealized example. Managers can never control tree densities by size classes as accurately as described in this example so there is considerable art involved in managing uneven-aged stands to ensure that the resulting diameter distribution after a harvest approximates a negative exponential distribution. For tree species that are intolerant of shade as they regenerate and grow, the ability to maintain the shape of the diameter distribution (and hence the foliage height profile) becomes even more complicated and dictates the type of regeneration harvest that will be used.

## CONSIDERING THE SITE POTENTIAL

The choice of which regeneration approach to use, what the target tree size should be, and what stocking level to maintain are influenced by many factors. The ability to manage an area using uneven-aged regeneration methods is often constrained by topography. Due to the high cost of harvesting per income from volume harvested using this system, typically ground-based harvesting equipment, such as horses, skidders, and feller-bunchers, is used. Cable, skyline, and helicopter logging often costs more than the value of the timber to be removed and so are cost prohibitive except in very high value stands.

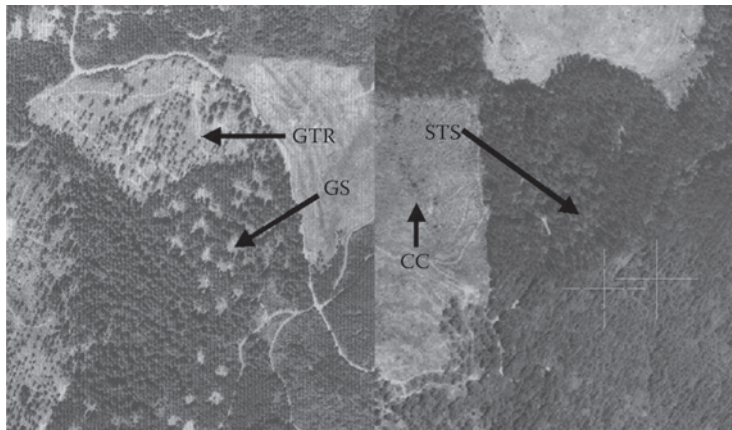
The site index for the tree species being grown will influence the cutting cycle length. Low site index locations grow trees more slowly and extend cutting cycles. As mentioned earlier, the tree species that you wish to manage influences the basal area removed depending on the shade or moisture tolerance of these species. In addition, the presence of competing vegetation such as shrubs or herbs may represent excellent forage resources, but inhibit establishment of regeneration. Consequently, careful consideration must be given to the restrictions that the characteristics of the site place on your ability to use uneven-aged systems to achieve habitat structure and/or timber goals.

## UNEVEN-AGED REGENERATION METHODS

Uneven-aged management usually involves *group selection* or *individual tree selection*. Individual tree selection is usually used with tree species that are moderately to very shade tolerant because it requires the removal of one or a few trees from a location in the stand to create a canopy gap to allow tree regeneration to occur. For many tree species, there is simply not enough light entering the forest floor to allow the regeneration to survive and grow if only one tree crown is removed. For these less shade-tolerant species, a group selection system may be used that involves creating small openings in the stand to allow more light and somewhat larger patches of regeneration to become established. These groups are usually less than one tree height in width but may necessarily exceed that width for very shade-intolerant species. The point at which a large group becomes a small clearcut is somewhat semantic, as is the point at which a small group selection becomes individual tree selection. These uneven-aged regeneration systems cause a fine-scale disturbance so within-stand vertical structure and fine-scale horizontal patchiness are usually high compared with even-aged systems (Figure 9.2).

## NATURAL REGENERATION AND PLANTING OPTIONS

Site preparation in uneven-aged systems may range from none to mechanical scarification or prescribed burning. If advance regeneration is already established in the existing litter layer, then no site preparation is needed. But if the regeneration needs to be established and the plant species requires bare mineral soil for seed germination, then litter layer disturbance may be done during



**FIGURE 9.2** Single-tree selection (STS) and group selection (GS) stands, compared with green-tree retention (GTR) and clearcut (CC) stands, McDonald Forest, Benton County, Oregon. Note the fine scale patchiness in the single-tree selection stand and the coarser level of patchiness in the group selection stand.

harvesting by using the blade on the skidder to expose soil. Alternatively if the remaining trees are reasonably tolerant of fire, a cool burn may be used to expose an ash layer as a seedbed (Salverson et al. 2011). Mechanical scarification of the harvest groups and compaction of the soils along designated skid trails can significantly affect below-ground habitat by reducing the availability of burrow systems to many animal species and restricting the ability of animals to burrow in the compacted soil. Careful harvest planning and use of designated skid trails is essential on many soil types (Kellogg et al. 1996). Alternatively, fires and litter layer disruption can lead to increased sprouting of a wide variety of plant species as potential forage and allow seeds in the soil seedbank to germinate and proliferate. Chemical site preparation is also an option but can be expensive and time consuming because herbicides often must be applied using backpack sprayers.

Artificial regeneration may be established if advance regeneration is not present. Tree species that are somewhat shade intolerant can be regenerated more easily with a group selection than with a single-tree selection. Small patch, group-selection systems or single-tree selection systems that rely on existing advance regeneration or large planting stock may eliminate or significantly reduce the duration of grass-forb-shrub patches in the stand.

## UNEVEN-AGED STAND DEVELOPMENT

During management of uneven-aged stands, the structure is in a continual state of flux. Trees grow until the growing space leads to a fully stocked condition at the end of a cutting cycle, then some trees of all size classes are harvested to produce the growing space needed for regeneration and continued growth of residual trees. In unmanaged uneven-aged forests, growing space occurs from competition mortality or disturbances such as insects, disease, fire, or wind. When competition causes mortality and creates growing space, usually shade-tolerant tree species fill the gap unless the gap is large and disturbs the soil. For instance, in an old-growth forest in Great Smoky Mountains National Park, Tennessee, when large (dbh > 70 cm) shade-tolerant trees die and fall, they are usually replaced by less shade-tolerant species such as yellow birch, yellow poplar, and Fraser magnolia (Barden 1979). Uneven-aged management simply imitates this process and selects certain species to favor during the harvesting process, causing the desired species to dominate the stand.

One uneven-aged approach taken by some forest managers is *diameter-limit cutting*, or cutting all the trees above some minimum diameter, usually the merchantable tree diameter. Although widely used, this approach is often criticized by silviculturists because the harvest leaves small trees (usually with small crowns) and diseased, damaged, or shade-tolerant trees to regenerate the next stand.

So from a timber production standpoint, diameter-limit cutting may not produce consistent long-term yields of products in some forest types. Effects on habitat elements and vertebrates are less clear, however. A study in the eastern United States demonstrated only minor effects on abundances of various bird species using this system (Weakland et al. 2002). Abundances of most songbird species present prior to harvest changed little after the timber removal (Weakland et al. 2002). Two species, the Canada warbler and dark-eyed junco, were more abundant in harvested areas than in an unharvested forest. Stands that were harvested differed from unharvested stands in only a few structural characteristics. Harvested stands had more snags, more trees (8–14.9 cm [3–6 in] dbh), and more down wood (Weakland et al. 2002). Canopy cover over 24 m (80 ft), density of saplings, and the amount of leaf litter decreased after harvesting. Another study modeled three stand management options in southern Indiana. The “do-nothing” management provided the best gray squirrel habitat but the worst economic return; the diameter-limit alternative produced a poorer squirrel habitat but a better short-term financial return; and intensive management provided the highest long-term economic return but produced the poorest squirrel habitat (Brand et al. 1986). Although the impacts of diameter-limit harvesting may be well accepted from a timber management perspective, the effects on habitat elements and vertebrates are highly variable and may not be problematic at low levels of volume removals.

## HABITAT ELEMENTS IN UNEVEN-AGED STANDS

Several factors influence the development of an uneven-aged stand and the resulting habitat elements. Clearly, the tree species composition and the ability of the desired species to regenerate in the stand have the greatest effects on stand development. Mixed species stands can be difficult to manage because of the varying growth rates of the different species, but also can provide the manager with the opportunity to favor some species over others during management. The desired range of basal area can also influence stand development. If a manager wishes to provide growing space for regeneration, browse, soft mast, and hard mast production, then the stocking level must be kept quite low from one cutting cycle to the next, much lower than would be expected under most natural disturbance processes unless disturbances are frequent. Nonetheless, uneven-aged stands seem to support more species of birds typical of unmanaged forests than even-aged stands, at least early in stand development. Chambers et al. (1999) reported that many bird species found in clearcut and green-tree retention stands did not occur in stands managed using small group selection (Figure 9.2). Many of the species using the group selection system were also found in uncut mature forest stands (Chambers and McComb 1997, Chambers et al. 1999, Gram et al. 2001). These patterns likely reflect the distribution of habitat elements in uneven-aged stands. High vertical structural diversity and fine-scale horizontal patchiness tend to be associated with the single-tree selection system (Kenefic and Nyland 2000) (Figure 9.2). Although single-tree selection systems decrease total canopy closure, they maintain high vertical structural diversity and an even distribution of foliage among canopy strata. Single-tree selection regeneration systems can reduce the number of cavity-bearing trees and dead wood but increase browse (McComb and Noble 1980, Kenefic and Nyland 2000). Cutting cycle length, target tree size, and stocking all affect the structure and composition of uneven-aged stands.

## VERTICAL STRUCTURE

Probably the most obvious effect of using uneven-aged approaches is that the vertical structure of the stand is more complex than would typically be found in even-aged systems, particularly in single-species stands. Indeed, Lei et al. (2009) recommended uneven-aged stand management to maintain high structural diversity in spruce-dominated forests. However, the shape of the diameter distribution can have a significant effect on the distribution of foliage in the stand. A stand with a high  $Q$  will have more foliage represented among the smaller trees and less in the larger trees (steeper diameter distribution). A stand with a low  $Q$  (flatter diameter distribution) will have proportionally more foliage in the larger trees. Consequently, depending on the species of animals that

you wish to manage and with which foliage layers they are associated, you may wish to use different diameter distributions to meet the needs of those species.

The plant species composition of the various foliage layers can also have an effect on the responses of vertebrates to this vertical structure. If the lower foliage layers are manipulated to remove shrubs and allow tree regeneration to become established, then those species of birds and mammals that rely on shrubs more heavily than tree seedlings could be adversely affected. Removal of understory vegetation in uneven-aged management could decrease populations of some ground- and shrub-nesting forest interior species of birds (Rodewald and Smith 1998). Liang et al. (2009) suggested that a combination of cutting cycle (10 years), target basal area (14 m<sup>2</sup>/ha), and *q* ratio (1.2) in Douglas-fir–western hemlock stands would maximize the percentage of veneer logs (for timber product value) while also maximizing tree-size diversity and tree-species diversity. Holmes and Pitt (2007) found that a residual stocking of 20 m<sup>2</sup>/ha in northern hardwoods allowed mature forest associated bird species to persist while also providing habitat for some early successional associates. Clearly, specific guidelines to achieve specific goals will vary by forest type and the habitat elements that you may wish to produce.

## HORIZONTAL DIVERSITY

Horizontal diversity or patchiness is high at a small spatial scale especially using group selection approaches. If the groups are sufficiently large then early successional species might colonize them, though they may be too small to be of value to some early seral bird species. King et al. (2001) found gaps served as sinks, not sources for many of these bird species. Chambers et al. (1999) found that small gaps of 0.2 ha (0.5 acres) were not colonized by early successional bird species in western Oregon. As you would expect, the responses of various species to these small gaps varies from one species to the next. Small gaps and single-tree selection systems tend to support a species assemblage more similar to that of a mid- to late-successional forest, especially if snags, logs, hardwoods, and shrubs are allowed to persist. Large gaps (small clearcuts) allow the colonization of some early seral associates.

## FORAGE AND BROWSE

Group selection systems can also provide patches of browse and forage adjacent to cover for ungulates, hares, and other herbivores. The interspersed forage and cover can be an excellent management strategy for these species if the openings are large enough to produce browse of the correct species and quality. Creating small gaps often leads to increased levels of shade in the gaps and reduces the production of browse, but this apparently is a problem only with very small gaps (80–100 m<sup>2</sup>) in northern hardwood stands (Webster and Lorimer 2002). But quantity may not be as important as quality for many herbivores. Gap sizes of 100 m<sup>2</sup> or larger were needed to allow dominance of more palatable browse species (Webster and Lorimer 2002). This may be particularly important where edges of gaps decrease plant growth (York et al. 2004). On the other hand, plants may allocate more energy to growth than to defense under low light conditions, allowing the plants growing in partial shade to be higher quality browse than plants grown in full sunlight (Dudt and Shure 1994). Following selection system harvests to various densities in loblolly-shortleaf pine stands, herbage and browse production were generally related to residual pine basal area and site quality (Wolters et al. 1977). Browse made up about one-fourth of the forage under stands having high residual pine basal area but represented considerably lower proportions in clearings (Wolters et al. 1977). Stands with lower basal areas tend to have higher browse production, denser and higher vertical structure, more woody vine and fern biomass, and higher plant species diversity and richness (Miller et al. 1999).

In addition, Hanley and Barnard (1998) suggested that patches of hardwoods, specifically red alder in conifer forests of southeast Alaska, offer significant food resources to herbivores beyond simply browse. These patches allow more sunlight to the forest floor and provide a diversity of forage species for Sitka black-tailed deer in this region.



## DEAD AND DYING TREES

Maintaining stands at low stocking levels means that competition mortality is kept to a minimum. Snag and fallen log availability in uneven-aged stands is often lower than in unmanaged old-growth stands (Goodburn and Lorimer 1998). If competition mortality is occurring in a stand, then the trees most likely to die are the smaller ones in the stand, and in an uneven-aged system these trees are the regeneration and browse resources. Gronewold et al. (2010) reported that northern hardwood stands maintained at a 11.5 m<sup>2</sup>/ha of basal area had significantly lower amounts of down dead wood, snags, and large overstory trees compared with the stands maintained at 20.7 m<sup>2</sup>/ha. So providing dead wood in uneven-aged stands often requires either managing at high stocking levels or active management through killing trees or retaining patches of forest that are allowed to remain dense while giving up the opportunity to recruit regeneration and browse in those patches. In addition, legacy trees can be retained from one cutting cycle to another to ensure that some of the elements of old trees are present in the managed stand. But remember that these legacy trees often will grow to a size larger than the target tree size and take up growing space that could be occupied by regeneration if they were not retained.

## MAST

Soft mast production in shelterwood stands and clearcuts is often greater than in single-tree selection, group selections, and unharvested stands (Perry et al. 1999), and we would expect that soft mast production increases as gap sizes increase to allow more full sunlight to strike the shrubs. Hard mast production is generally associated with crown size and tree age, and consequently can respond to silvicultural treatments that provide more sunlight to the crowns of mast-producing trees (Perry and Thill 2003). Mast production in many oak species is highly variable from year to year and seems to be heavily influenced by weather and time since the last heavy mast crop. Nonetheless, trees with large crowns should periodically produce an abundant crop of mast. Generally, open crowns are capable of producing many more fruits than closed crowns (Johnson 1994). Larger stem diameters (and consequently larger crowns) also produce greater crops of acorns than smaller diameter stems, so uneven-aged management methods that use large target tree sizes and keep stocking levels near or below crown closure should produce more abundant mast during years of high mast production (Desmarais 1998). Low stocking and widely spaced trees may increase mast production but decrease production of large snags and fallen logs.

As a rule of thumb, the shape of the diameter distribution and the target tree size will influence the ability to provide vertical and horizontal complexity, forage, and mast. If the diameter distribution is steep with very many small trees and only a few large ones, then it will probably function similar to an early to mid-successional even-aged stand for most species; browse availability may be greater in these stands. If the diameter distribution is somewhat flat or if the target tree size is large, with very few small trees and more large trees, then it may function more similarly to a late-successional even-aged stand; hard mast production may be better in these stands. In both cases, uneven-aged regeneration methods cause a more fine-scale of disturbance than even-aged systems, so within-stand vertical structure and fine-scale horizontal patchiness are usually high compared with even-aged systems.

## CHALLENGES TO USING UNEVEN-AGED METHODS

Achieving timber and habitat goals using uneven-aged methods presents a few challenges that should be understood before accepting this technique as a way of meeting these goals. First, if stocking levels are not kept low enough, many species of shade-intolerant plants will likely decline in abundance in the stand. These may be important plant species as food resources for herbivores or valuable timber species. Hence, cutting cycles may need to be more frequent on highly productive sites, or volume removals heavier than one might wish to achieve goals related to vegetative cover.



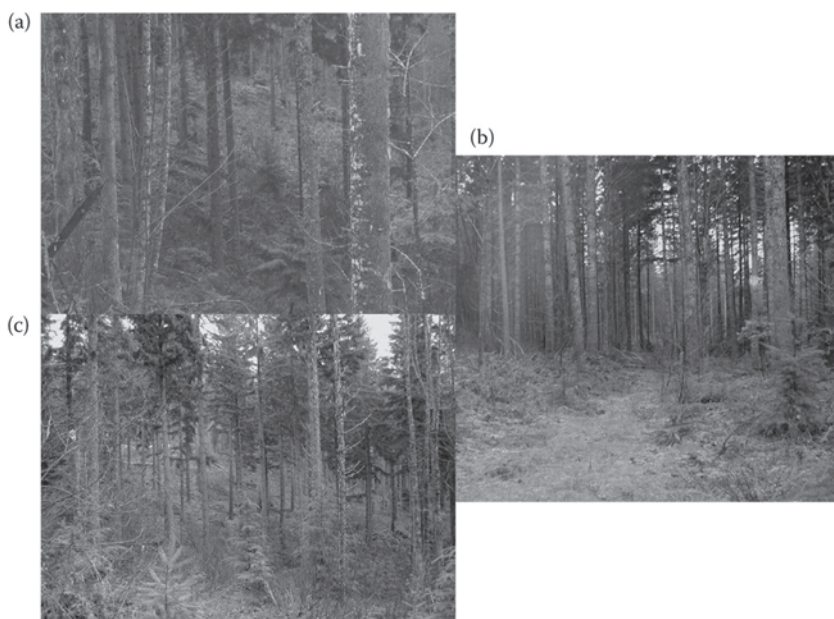
In addition, each entry will require that trees of a wide range of tree diameters (and often tree species) be harvested. These various tree sizes and species have different market values. So it is quite likely that sawtimber, pulpwood, firewood, and perhaps veneer logs could all be removed in one harvest. Ensuring that logs are sorted and that markets are available for each tree size and species can present challenges to the manager. Harvesting these various-sized trees can also be a challenge especially where advance regeneration occurs. Felling large trees onto existing regeneration can damage the smaller trees and reduce the ability to maintain the desired diameter distribution. Use of directional felling and designated skid trails can help to reduce these problems but may increase harvesting costs.

Finally, keeping unmerchantable trees and shrubs in the stand as legacy trees or shrub patches is feasible but must be taken into consideration during each cutting cycle to maintain these structures or plan for their replacement as they age and die. These residual plants also occupy growing space and consequently represent a tradeoff between timber production and habitat availability for desired species.

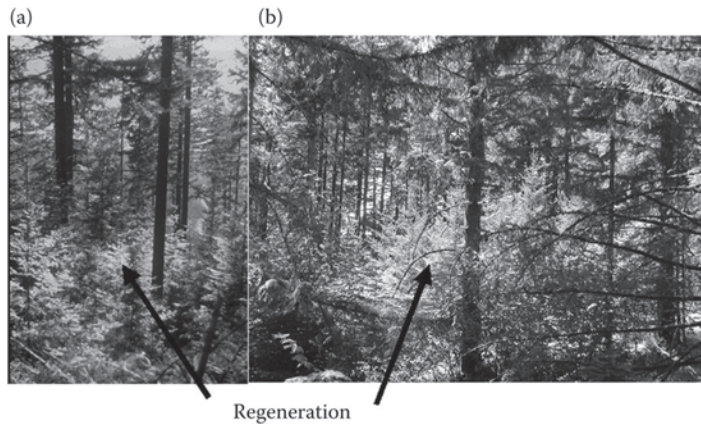
## NONTRADITIONAL MANAGEMENT APPROACHES

The uneven-aged systems described in this chapter and the even-aged systems described in Chapter 8 represent only a few examples of stand management approaches that span a spectrum of possibilities (Figure 9.3). The opportunities to develop stand structure and composition to meet land manager objectives is endless and can be crafted to each site to meet those specific objectives. McComb et al. (1993) used the structure and composition of unmanaged stands that were meeting habitat objectives for late seral species as models for proposed managed stands. This is one approach, though certainly not the only approach, to defining a *desired future condition*—a description of the structure and composition of a stand that you would hope to achieve through active management. Defining the desired future condition or specific goals for the stand is the first step in stand management.

One type of management approach described by McComb et al. (1993) is a many-storied stand that uses small group selection cutting to create a stand that is composed of >3 layers of canopy trees



**FIGURE 9.3** Several silvicultural approaches to increase complexity in 50-year-old Douglas-fir stands in the Oregon Cascades. (a) an unthinned stand, (b) a stand with 0.2-ha gaps, and (c) a heavily thinned stand with shade-tolerant trees and shrubs in the understory.



**FIGURE 9.4** Vertical complexity arising from regeneration in a single-tree selection stand (a) and a group selection stand (b) in a Douglas-fir–grand fir forest type in Oregon.

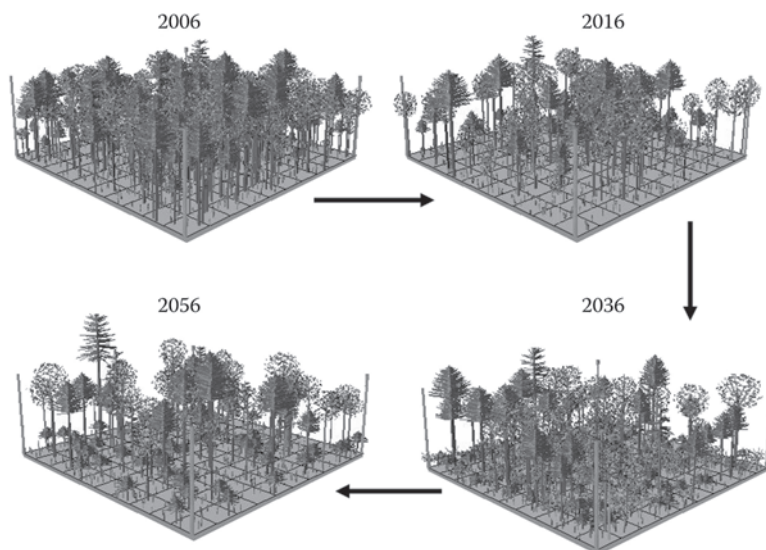
in a mosaic of gaps while retaining large legacy trees and snags in the stand. The approach contains elements of a forest found in and produced by gap-phase forest dynamics and may be applicable to many forest types. The many-storied system is patterned after fine-scale natural disturbances. Cut gaps may have to be larger than most natural canopy gaps to allow successful natural regeneration of shade-intolerant species and to make harvesting more efficient. This system would have high within-stand variability in tree size and vertical complexity. This system might provide acceptable habitat for mature forest species while allowing some small but regular timber removal and as such be attractive for nonindustrial forestland managers.

The choice of which silvicultural system to use is determined by the plant community, site conditions, logging constraints, and species of vertebrates of highest interest. Uneven-aged management strategies that could improve habitat quality for species that inhabit late seral stage conditions include establishing a large target tree size, lengthening cutting cycles, minimizing disturbance to the stand during logging with designated skid trails, harvesting with small-group or single-tree selection systems where they are appropriate, managing for shade-tolerant tree species, and maintaining high-density groups of regeneration (Figure 9.4). Bauhus et al. (2009) analyzed a suite of options for developing old-growth structure of stands using a range of silvicultural options with varying levels of complexity in achieving structural goals. An allocation of dead or large, living trees also would increase habitat quality for many species typical of late seral stages.

Altering the scale or frequency of cutting also might influence habitat quality for forest vertebrates. Imposing a single-tree selection system in a forest with a cutting cycle of 10–15 years and target tree sizes of >50 cm dbh, for example, would result in small, widely scattered openings. On the other extreme, locating 60-ha clearcuts side-by-side within a watershed would create huge areas of early seral stage stands. Colonization of parts of this area by relatively less mobile species would be less likely than colonization by larger, more mobile species. Both the silvicultural strategy employed and its arrangement in context with other stands on the landscape, therefore, can have a tremendous influence on the future abundance and distribution of animals in the landscape.

## CASE STUDY: MANAGING A SMALL PRIVATELY OWNED FOREST

As an example of using uneven-aged management to provide a variety of ecosystem services, we can examine how a family owning 10 ha of forest land in western Massachusetts might approach management. First it is important to recognize that the family has multiple objectives for the forest that include, in order of priority:



**FIGURE 9.5** Example of several cutting cycles in a mixed hardwood pine stand in western Massachusetts. Data are from the UMass-Amherst Cadwell Forest, and projections are from the Landscape Management System. (McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

1. Always having a forest on their land
2. Firewood to help heat their home
3. Periodic income sufficient to cover taxes
4. A multistory forest to support a diversity of nesting songbirds
5. Enough browse and mast production to attract white-tailed deer and ruffed grouse

The stand is a mixed oak–pine forest on glacial till. It is approximately 80 years old and established following farmland abandonment in the early 1900s. There is currently 46 m<sup>2</sup>/ha (200 ft<sup>2</sup>/acre) of basal area and is dominated by northern red oak, red maple, black birch, and eastern white pine, with seven other tree species common in the stand (Figure 9.5). If we reduce the basal area to 16 m<sup>2</sup>/ha (70 ft<sup>2</sup>/acre), then that results in about 27 MBF/ha or 271 MBF removed from the property. At current stumpage values that would be yield about \$27,000. Using a 20-year cutting cycle, we can have another harvest that yields about \$13,000 in 2026. A third harvest 20 years later is a cordwood sale (no sawtimber in cut). During each harvest, 25–50 cords of wood are cut per hectare to provide firewood. At the end of 60 years of management, the stand contains 44 cords of firewood per hectare and 25 MBF of sawtimber per hectare available for future harvests. Openings are sufficient to always have an understory present and large enough to provide browse and soft mast. Red and white oaks average about 30 cm in diameter and should produce regular acorn crops. With careful attention to regeneration of desired tree species, the needs of the landowners should be met for several generations of the family.

## SUMMARY

Uneven-aged stands consist of three or more age classes represented in the same stand (Figure 9.6). Uneven-aged stand structure and the structure and function of habitat elements are governed largely by basal area, tree density in each diameter class, and target tree size. These characteristics are manipulated to achieve the desired negative exponential diameter distribution to meet the goals for a stand. This structure can be achieved using single-tree selection or group selection regeneration



**FIGURE 9.6** Abundant regeneration following single-tree selection in a ponderosa pine mixed-conifer forest type, Ochoco National Forest, Oregon.

methods. In general, vertical complexity in an uneven-aged stand is high, and horizontal complexity is fine scaled. Browse and mast are less abundant than in early stages of even-aged management unless large gaps are made using group selection systems. Because stocking must be kept low to allow trees to grow, recruitment of dead wood is usually minimal, so active management is usually needed to ensure adequate dead wood for desired wildlife species. Alternative management approaches usually define a desired future condition and then adapt even-aged and uneven-aged approaches to meet that goal.

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# 10 Desired Future Conditions

To effectively manage habitat for a species, a group of species, or to contribute to biodiversity conservation, we need goals or targets toward which management will be directed. This may involve a condition that will occur on its own in the absence of active management, or it may require intervention to guide the development of the stand or landscape toward your goal. Describing the structure, composition, and scales of a condition that you think will meet the needs for species on your site is one of the first steps in developing a habitat management plan for a stand, forest, or landscape. Landres et al. (1999) described *desired future conditions* (DFCs) as expressions of ecosystem conditions preferred by stakeholders and managers. Kessler et al. (1992) also referred to an articulation of a DFC as a goal in ecosystem management. This may be a reference condition, or more appropriately it may be a set of reference conditions that currently achieve some desired objectives, or it may be a sequential set of future conditions that achieve different objectives for different species over time. Given inherent uncertainty in achieving goals in the face of stochastic disturbances, ecological pathways, and novel stresses on forest dynamics (e.g., climate change, spread of invasive species), monitoring to assess progress toward a DFC is probably a reasonable strategy for achieving habitat objectives. Adaptability to unexpected outcomes is also important. Joyce et al. (2009) described developing a set of practices that would build resistance and resilience into current ecosystems, while also managing for change in system function and adaptability to new system states. Despite the uncertainties of the future, having a goal or DFC helps to direct management actions while keeping options open for unexpected changes. Goals for habitat are typically set at large scales (regions) and achieved at small scales (stands). Foresters typically develop plans for managing stands that contribute to some overall forest-level goal.

However the DFCs are described, they must be implementable; that is, the site must be capable of producing those conditions. All of the factors described in the previous chapters come into play when considering if current conditions, past actions, and likely future changes will result in achievement of a set of DFCs. Models of forest development under alternative management strategies can help guide development of management plans for a stand or landscape.

## DEVELOPING THE STAND PRESCRIPTION

Foresters write *prescriptions* or silvicultural management plans for stands to achieve a DFC. Personally I find the term “prescription” unfortunate because to the lay public it implies that the stand is unhealthy and needs fixing! Which may or may not be the case. Nonetheless, once the DFC is deemed achievable, the prescription for the current stand can be developed. It is important to have a written prescription that clearly describes the current condition, the DFC, management actions, schedule for anticipated future entries, and a monitoring plan. For many prescriptions the DFC may not be attained for several decades. Consequently, it is important that there is a written record for the rationale behind the prescription. Future managers of this stand may be able to make more informed decisions based on written records for the stand.

Your prescription should include a description of any of the stand summary information (species, basal area, tree diameter distribution, site index, stocking, etc.) that would be needed to develop marking guides. In addition, to achieve habitat objectives, the prescription should contain each of the following sections.

### **SPECIES BACKGROUND**

What plant and animal species goals are intended for this stand? Are individual species or groups of species to be managed? What are their habitat needs? Over what spatial scale (e.g., nest sites, foraging patches, home ranges) must they be met? How do these goals complement goals for adjacent stands?

### **CURRENT STAND CONDITION**

What are the habitat conditions in the stand now for the species that you intend to manage? What factors are limiting habitat suitability for the species? What is the tree and shrub species composition? Site index? Stocking? Existing or possible future regeneration? What are the physical conditions or cultural history that lead to the current condition?

### **DESIRED FUTURE CONDITION**

What are the habitat conditions that you would like to produce? Be specific and describe the plant species composition, size classes, basal area, and any predictions of future stand development that you can develop. Stand growth models such as the forest vegetation simulator (FVS) (Dixon 2003) or forest disturbance and succession models (Shifley et al. 2008) are particularly useful for understanding if your DFC can be met.

### **MANAGEMENT ACTIONS TO ACHIEVE THE DESIRED FUTURE CONDITION**

What will you do to the stand now and over time to achieve your DFCs? How long do you think it will take to achieve them? How long will they last? Is it possible to maintain the DFCs with or without management? How much will it cost?

### **MONITORING PLANS**

What will you measure and how often will you measure to determine if your management plan was implemented correctly and if the actions were successful? How will you decide if you need to change your management plans? For more details on developing monitoring plans see McComb et al. (2010).

### **BUDGET**

What will implementation and monitoring of the plan cost? Regardless of the landowner, cost becomes a factor when implementing a prescription to achieve a goal. For most landowners, there must not be a net loss, and for some landowners there must be a maximum profit. Understanding the products that can be derived from implementing the prescription, both economical and ecological, can help the manager decide if the trade-offs are acceptable.

### **SCHEDULE**

When will each step in the plan be completed? Of course the monitoring of stand development may lead to changes in the schedule, but there should be a plan for when actions will occur.

### **REFERENCES**

Scientific references should be used to support assumptions used in the development of the prescription. Since new information is always becoming available to guide management, it is important to understand why decisions were made at this point in time. As stands develop and plans are altered,

it is important to review the literature yet again. Changes in scientific knowledge can alter the plans developed even a few years ago and can help you interpret your monitoring data.

Once a prescription has been developed, then it can be implemented and monitored over time. If the monitoring data indicate that the stand is developing over time differently from what was predicted in the prescription, then midcourse corrections can be made. By having a prescription for each stand that you manage, and keeping it on file, then future managers can understand how the stand developed and if it is developing as intended to achieve a DFC. If a new DFC should be developed at some time in the future, then the manager at that time can learn from the past efforts. Documentation of plans, actions, and results is critical.

## **CASE STUDY: GROWING RED-COCKADED WOODPECKER HABITAT**

Red-cockaded woodpeckers are listed as threatened under the U.S. Endangered Species Act. Their numbers are a fraction of what they were historically. The decline in numbers and distribution is considered to be largely a product of past timber liquidation and current short-rotation forestry for pulp and small sawtimber production in the southern yellow pine region of the United States. This example prescription is developed for a 7 ha pine stand on the Kisatchie National Forest in Louisiana. The stand data were provided by Jo Ann Smith, Forest Silviculturist on the Kisatchie, but I use them merely as an example of managing this stand to achieve a DFC for red-cockaded woodpeckers. Managers of the Kisatchie quite likely would have a different prescription based on their goals, their personal knowledge of the stand, its history, and its context on the forest. I simply use the data to demonstrate how a prescription could be developed for red-cockaded woodpeckers and several associated species, Bachman's sparrows, and brown-headed nuthatches.

### **SPECIES BACKGROUND AND MANAGEMENT OPTIONS**

Red-cockaded woodpeckers have long been associated with mature southern yellow pine forests containing old trees with red heart disease, a fungal decay softening the heartwood of living pines. Red-cockaded have a broad geographic range throughout the Gulf Coastal Plain and Piedmont north into southern Kentucky. Home range sizes vary from 25 to over 100 ha (Delotelle et al. 1987, Doster and James 1998), being somewhat larger during the time when they are not nesting (Wood et al. 2008). Because our stand is only 7 ha, any plans for habitat improvement must consider the conditions of the surrounding stands, which should be managed to complement the conditions in our 7-ha stand. This is especially important as we consider one pair as part of a larger metapopulation that may need 800–200 ha of habitat (Mitchell et al. 2009).

Within their home range, they defend a somewhat smaller territory around a central nest tree. Home ranges must be large enough to provide resources for young in the nest as well as additional helpers at the nest. Red-cockaded woodpeckers are similar to acorn woodpeckers by having a social structure known as a clan, in which nonreproducing members of the clan assist the reproducing pair with raising young. They feed primarily on insects and fruits and forage on trees >60 years of age and on trees >25 cm dbh (diameter at breast height) more than would be expected by chance (Zwicker and Walters 1999). The nest tree is usually >100 years of age and >40 cm dbh (Conner and O'Halloran 1987). Cavities are excavated in live trees, unlike most other woodpecker species, and they typically make shallow excavations around the cavity entrance into resin wells, causing a flow of sticky resin around the cavity entrance and down the tree. This behavior is presumably to reduce risks of predation by rat snakes (Conner et al. 1998).

The stand around the nest tree typically is dominated by pines (basal area averages 16 m<sup>2</sup>/ha) with a few midstory pines (average = 1.1 m<sup>2</sup>/ha) and very few if any midstory hardwoods (Conner and O'Halloran 1987). James et al. (1997) found that variation in the size, density, and productivity of clans was related to the ground cover composition and the extent of natural pine regeneration, both of which are indirect indicators of local fire history.

Bachman's sparrows also are associated with mature pine woodlands characterized by widely spaced pines, an open midstory, and a dense understory of grasses and forbs (Plentovich et al. 1998). They have territory sizes of 2–3 ha (Dunning 2006). Populations of the Bachman's sparrow began declining in the 1930s, reflected in a reduction in geographic range and local extinctions (Plentovich et al. 1998). Areas suitable for red-cockaded woodpeckers were not always suitable for Bachman's sparrows. Red-cockaded woodpeckers appear to be more tolerant of a hardwood midstory although they do not require a dense cover of grasses and forbs; however, they may be more successful in forests with frequent fires that produce this understory condition. Prescribed burning seems to be a key factor in development and maintenance of the dense herbaceous understory preferred by Bachman's sparrow (Dunning 2006, Jones 2008). Bachman's sparrow populations disappear 4–5 years after a burn (Dunning 2006). In areas managed for red-cockaded woodpeckers, frequent (3–5 year interval) burning early in the growing season appears to be the best way to increase habitat suitability for Bachman's sparrows.

Brown-headed nuthatches are also associated with loblolly–shortleaf pines of the Upper Coastal Plain and the longleaf-slash pines association of the Lower Coastal Plain (Withgott and Smith 1998). They have a territory size of about 3 ha and create nest cavities in pine snags typically >20 cm dbh (Harlow and Gynnn 1983) but most often forage on live pines (Withgott and Smith 1998). This species is most often found in open, mature, old-growth pine forest, particularly where natural fire patterns have been maintained and where snags are present for nesting and roosting (Withgott and Smith 1998). This combination of vegetation characteristics occurs in mature pine forest in which fire has kept understory open and created snags (Withgott and Smith 1998).

On the basis of the life history characteristics of these three species, several management options seem possible. James et al. (2001) suggested that smaller size classes of trees in closed-canopy stands should be thinned, creating patchy openings in the forest that will promote natural pine regeneration. This can be achieved by uneven-aged management such as group selection regeneration methods in conjunction with maintaining a low basal area (Outcalt 2008) or through irregular shelterwoods (Conner et al. 1991). Since longleaf pine was once a dominant species on the Kisatchie and since red-cockaded woodpeckers are often associated with longleaf pine savannah forests, management actions that allow recovery of longleaf into managed stands might also be beneficial to red-cockaded woodpeckers and other species.

Management approaches suggested for recovery of red-cockaded woodpeckers have included both short and long-term strategies (Rudolph et al. 2004). Nearly all populations require immediate attention in the short term, including manipulation of midstory and understory conditions (Rudolph et al. 2004). Management techniques including prescribed fire and mechanical and chemical control of woody vegetation are often used to achieve these needs. In the long term, cost-effective management of red-cockaded woodpecker populations requires a timber management program and prescribed fire regime that will produce and maintain the stand structure characteristic of high-quality nesting and foraging habitat, so that additional intensive management specific to the woodpeckers is no longer necessary (Rudolph et al. 2004). Management that achieves this goal and still allows substantial timber harvest is feasible. With some attention to understory conditions and availability of snags, these approaches would benefit Bachman's sparrows and brown-headed nuthatches as well. Effects on other species probably would be negative for ground-nesting birds (Wilson et al. 1995) and possibly those associated with midstory conditions, though when done at a small scale, negative effects on these other species may be minimal (Lang et al. 2002).

## CURRENT STAND CONDITION

The stand is currently dominated by loblolly and shortleaf pines both in basal area and tree density (Table 10.1). Approximately 8% of the basal area is in hardwoods, primarily oaks. Tree diameters range from 30 to 60 cm (12–24 in) dbh, averaging 38 cm dbh. There are about 200 trees/ha and the stand basal area is approximately 23 m<sup>2</sup>/ha (101 ft<sup>2</sup>/acre). The stand has understory shrub and

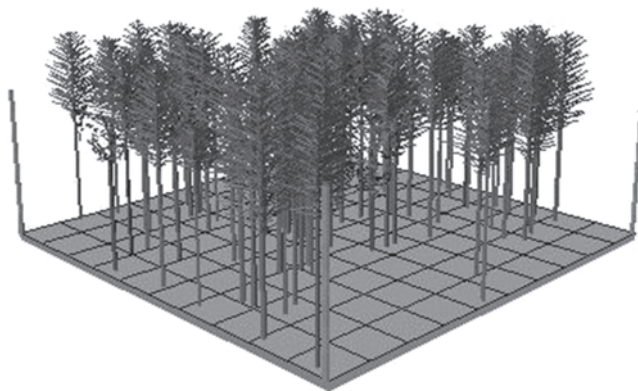
**TABLE 10.1****Summary Statistics for an Example 7-ha Pine Stand from the Kisatchie National Forest, Louisiana, 2005**

Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Loblolly pine	40.6	126	16.5
Shortleaf pine	33.9	54	6.0
Oaks	38.5	15	1.8
Total	38.6	195	23.3

hardwood cover but little regeneration (Figure 10.1) and little grass or herbaceous cover. The presence of hardwoods in the stand suggests that it has not been burned for some time, and that there may not be adequate fine fuels to carry a burn through the stand. Or worse, suspended fuels (needles in shrub crowns) may allow fires to scorch the crowns of the overstory trees. If a thick duff layer exists, then the slow-burning hot fire may kill pines and release hardwoods. The stand currently could serve as foraging habitat for red-cockaded woodpeckers but would not likely be adequate nesting habitat. The stocking is too high, there are hardwoods in the midstory, and the trees are likely not old enough to have heart rot. The stand is quite likely not a habitat for Bachman's sparrows, because there would be insufficient sunlight to support a dense grass-forb condition in the understory. The area could support brown-headed nuthatches now, especially if inter-tree competition has created snags (no snag data were available).

**DESIRED FUTURE CONDITION**

To provide nesting habitat for red-cockaded woodpeckers, brown-headed nuthatches, and Bachman's sparrows, we would need to have a stand in which the dominant trees were >100 years of age (increasing the probability that some had red heart disease), had a basal area closer to 16 m<sup>2</sup>/ha or less to provide grass-forb understory conditions, a few midstory pines, and no hardwoods. Since the area once supported longleaf pine, we would want to reestablish longleaf as a functional component in the stand. We also will want to ensure that tree mortality is likely to occur at some times during stand development to provide snags as potential nesting sites for brown-headed nuthatches. Since pine basal area will be kept somewhat low, and dominant trees well-spaced for Bachman's sparrows, snags may be created either from fire or by killing a few dominant trees per hectare periodically,



**FIGURE 10.1** Schematic of the current stand condition dominated by loblolly and shortleaf pines, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

depending on tree mortality and snag fall rates. Once the DFC was reached we would like to maintain the condition for the foreseeable future to contribute to red-cockaded woodpecker population recovery and maintenance.

MANAGEMENT ACTIONS TO ACHIEVE THE DFC

Although the specifics of management approaches for these species vary from one author to another, the general theme is that stands be managed using either group selection (McConnell 2002) in longleaf systems or irregular shelterwood regeneration methods in loblolly–shortleaf systems (Rudolph and Conner 1996, Hedrick et al. 1998). Use of an irregular shelterwood approach displaces red-cockaded woodpeckers that might use the site for foraging, suggesting that group selection regeneration, planting of longleaf pine, and frequent prescribed fire could be used to manage the site (personal communication, Dr. Robert Mitchell, Jones Center, Georgia). But to illustrate how both even-aged and uneven-aged systems can be used to manage habitat, I will initiate management with an irregular shelterwood. Irregular shelterwood methods retain all or a part of the overwood well into the next rotation and can provide older and larger trees as habitat from one rotation to another. Management practices that reduce litter, maintain relatively low tree and shrub densities, and that encourage the growth of forbs and grasses are recommended for Bachman’s sparrows (Haggerty 1998). Haggerty (1998) suggested that a combination of thinning and burning would contribute to the most suitable habitat for this species. Similarly, brown-headed nuthatches also find suitable habitat in mature pine forests, but with snags present. For this example, I propose initiating management with an irregular shelterwood to remove hardwoods and establish longleaf pine, then managing stand density and hardwood competition through a combination of thinning and prescribed burning. The series of management actions needed to produce a DFC that follows is based on use of the Southern version of the Forest Vegetation Simulator (SN) (Dixon 2003) and the Landscape Management System (McCarter et al. 1998).

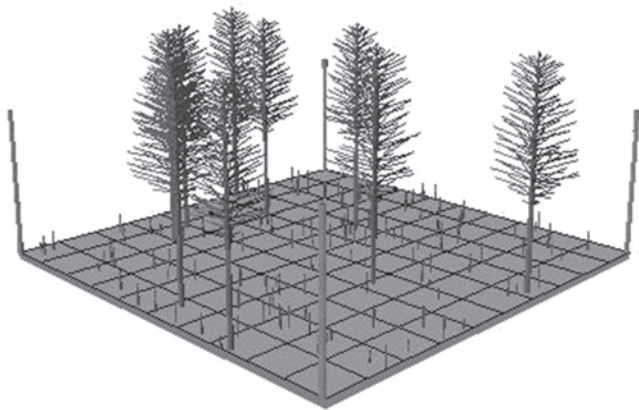
*Year 2015:* We will initiate stand management with an irregular shelterwood retaining 25 pines/ha (8 m<sup>2</sup>/ha, Boyer and Peterson 1983) and cutting all hardwoods. Following harvest, we will prepare the site for planting using a cool winter burn to remove fine fuels and then plant 250 longleaf pine seedlings per ha. Since the overwood is dominated by loblolly and shortleaf pines, I will also assume that natural regeneration of these two species will become established following harvest. There should be no prescribed burning for the first few years after longleaf pine establishment (Boyer and Peterson 1983), and then early spring burns (prior to bird nesting) should be initiated to control hardwoods. Once longleaf regeneration is well established, we can continue with prescribed burning every 2–3 years to control hardwoods, shortleaf pine regeneration, and any brownspot disease that might become established on the longleaf pine seedlings (Haywood 2007). Longleaf persists in a “grass stage” for several years during which it puts on root growth but not shoot growth. Dense long needles protect the terminal bud from fires at this time, and fires remove needles infected with brownspot fungus.

*Year 2025:* By 2025, I would anticipate that the new stand would be much lower in basal area with a high density of pine regeneration (Table 10.2, Figure 10.2). Note that although we thinned

**TABLE 10.2**  
**Summary Statistics for a 7-ha Pine Stand Projected to the Year 2025, 10 Years after Initiating an Irregular Shelterwood, Kisatchie National Forest, Louisiana**

Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Longleaf pine	8.3	250	1.35
Loblolly pine	23.9	71	6.3
Shortleaf pine	6.9	50	0.1
Total	11.2	371	7.9





**FIGURE 10.2** Schematic of the stand condition at year 2025, 10 years after initiation of management, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

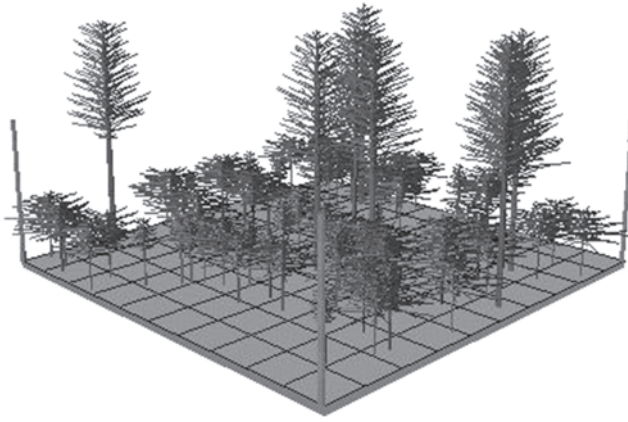
from below, the average dbh declined. This is because the stand is no longer even-aged. By 2025 there are 20 trees/ha that are >56 cm dbh along with many that are small seedlings. Hence average dbh may not be a good indicator of tree sizes in the stand. Also note that the basal area is far below our target. This low basal area is necessary to allow longleaf pine to become established while still retaining some trees that can become potential nest trees in the future. With the initiation of prescribed burning, habitat for Bachman’s sparrows has likely improved tremendously. Finally, note that although we retained 25 trees/ha in the overwood, only 20/ha were predicted to survive the following 10 years. We will assume that 5/ha have become snags, some of which may remain after burning and provide potential nesting habitat for brown-headed nuthatches.

*Year 2035:* With continued prescribed burning, hardwoods should be kept under control and longleaf pine will initiate height growth. The overstory trees now released from inter-tree competition will also continue to grow. Although red-cockaded woodpecker nesting habitat likely has not yet developed, habitat for Bachman’s sparrows and brown-headed nuthatches is available. However, as basal area increases, overstory cover will reduce the production of grasses and forbs following burning. If frequent fire does not allow adequate production of grasses and forbs, then we may need to initiate a light thinning of the overstory by creating snags and a precommercial thin of the seedlings and saplings to ensure continued tree growth while providing growing space for grasses and forbs.

*Year 2045:* If regeneration of longleaf was not adequate, we can create gaps in the stand and replant longleaf adding to the uneven-aged character of the stand. Otherwise we will let the stand grow, with early spring burns every 3–4 years (Table 10.3). Note though that regeneration may again

**TABLE 10.3**  
**Summary Statistics for a 7-ha Pine Stand Projected to the Year 2045, 30 Years after Initiating Management, Kisatchie National Forest, Louisiana**

Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Longleaf pine	12.2	120	1.4
Loblolly pine	37.8	59	9.1
Shortleaf pine	20.8	25	0.3
Total	20.6	203	10.8



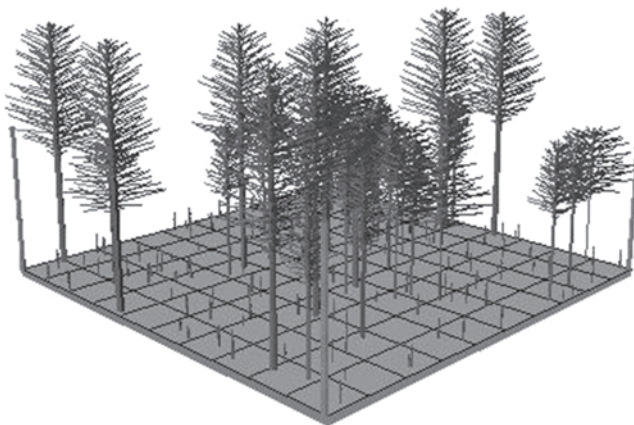
**FIGURE 10.3** Schematic of the stand condition at year 2045, 30 years after initiation of management, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

reduce the availability of grasses and forbs and that some precommercial thinning may be necessary simply to maintain Bachman's sparrow habitat (Figure 10.3).

*Year 2055:* In addition to prescribed burning to reduce hardwoods and to maintain a grass–forb understory, density management is now necessary to ensure that growing space is available for regenerating trees as well as the grass–forb layers. We will initiate a precommercial thin, reducing the density of the regeneration by 50% if necessary to ensure continued stand development.

*Year 2065:* With the precommercial thin in the previous decade and the site maintained with fire, additional regeneration may become established in the understory (Figure 10.4). Stand basal area has increased to nearly 15 m<sup>2</sup>/ha (Table 10.4).

*Year 2075:* We will continue to manipulate the density of all tree sizes and by this time have entered a group selection regeneration system. Group sizes will vary and cutting cycles of 10–15 years will be needed to ensure that all size classes of longleaf pine continue to develop. We also will retain some of the large loblolly pine in the stand as potential nest trees. The stand is now at



**FIGURE 10.4** Schematic of the stand condition at year 2065, 50 years after initiation of management, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

**TABLE 10.4**  
**Summary Statistics for a 7-ha Pine Stand Projected to the Year 2065, 50 Years after Initiating Management, Kisatchie National Forest, Louisiana**

Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Longleaf pine	9.0	274	1.8
Loblolly pine	30.1	108	12.8
Shortleaf pine	6.9	50	0.2
Total	14.0	431	14.9

**TABLE 10.5**  
**Summary Statistics for a 7-ha Pine Stand Projected to the Year 2105, 90 Years after Initiating Management, Kisatchie National Forest, Louisiana**

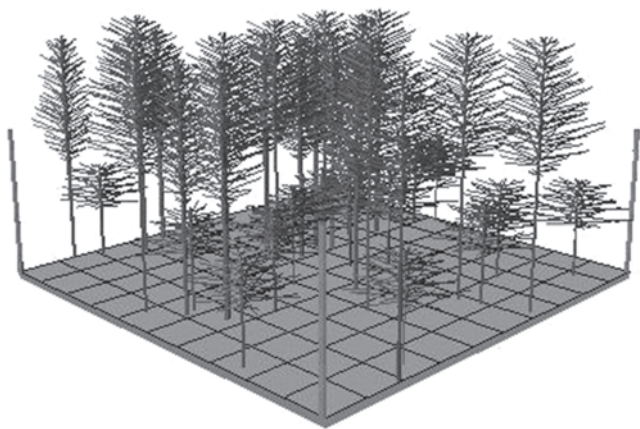
Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Longleaf pine	24.1	58	2.8
Loblolly pine	56.1	25	6.2
Total	33.5	83	9.0

the stage that red-cockaded woodpeckers could likely begin using the stand for nesting if adjacent stands provide adequate foraging habitat.

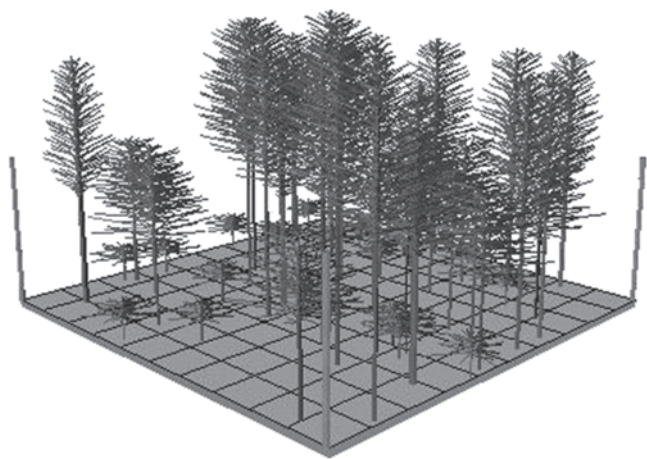
*Years 2085–2105:* Management will continue as needed in a maintenance mode by managing stand density to ensure rapid tree growth, retaining potential nest trees, and burning to maintain a grass–forb understory (Table 10.5, Figure 10.5).

*Year 2115:* Additional density management may or may not be needed at this point. Monitoring data should be used to ensure that habitat elements needed by the three species are persisting.

*Year 2125:* By this time the DFC should be reached or nearly so (Figure 10.6). Monitoring data will help determine if these stand growth projections produced a reasonable schedule toward development of the DFC. Basal area is now just below our target of 16 m<sup>2</sup>/ha (Table 10.6), the stand is 80% longleaf by density, and comprises nearly half the basal area. Loblolly pines are



**FIGURE 10.5** Schematic of the stand condition at year 2105, 90 years after initiation of management, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)



**FIGURE 10.6** Schematic of the stand condition at year 2125, 110 years after initiation of management, Kisatchie National Forest, Louisiana. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

now over 60 cm dbh and over 150 years old. Continued burning has allowed persistence of a grass–forb understory to provide habitat for Bachman’s sparrows. Either through natural mortality or by actively killing trees, loblolly pines can be a source of snags for brown-headed nuthatches. Stand density can be reduced even further by cutting some, but not all, of the large loblolly pines to release longleaf pines.

**MONITORING PLANS**

The above plan is a hypothesis. We cannot be sure that the stand will respond as intended, but on the basis of the species, sizes, and site conditions, the steps outlined above represent a reasonable approximation of how we might achieve our DFC. As we initiate management, however, we also should initiate a monitoring plan to see whether the stand is developing as predicted. Because the stand prescription has a number of steps, including regeneration cuts, thinning, and burning, it is important that each step be implemented correctly and in a timely manner. Implementation monitoring would be conducted to ensure that the plan is implemented correctly. Questions that you might address in your implementation monitoring might include: Were any unmarked trees harvested? Were burns conducted in the correct seasons? Was the fire return interval as prescribed?

In addition to implementing the prescription correctly, we would want to know whether we were effective in achieving both the stand structure as well as function. *Effectiveness monitoring* may take two parts. First we would monitor the survival, growth, and development of the vegetation to

**TABLE 10.6**  
**Summary Statistics for a 7-ha Pine Stand Projected to the Year 2125, 110 Years after Initiating Management, Kisatchie National Forest, Louisiana**

Species	Average dbh (cm)	Trees per ha	Basal Area (m <sup>2</sup> /ha)
Longleaf pine	22.4	101	5.3
Loblolly pine	64.1	25	8.2
Total	30.6	126	13.5

ensure that the habitat structure goals were being met. Regeneration surveys to assess survival can be done at 10–20 randomly established circular plots in the stand for the first 5 years after planting. Once seedlings are established and free to grow, then tree growth and survival should be monitored. Tree growth and survival is typically monitored on continuous forest inventory (CFI) plots and may be a fixed plot or a variable radius (wedge prism) sample. Measuring tree growth and survival every 5–10 years would be adequate to see whether the stand is developing as predicted. These samples will provide data on growth, basal area, density, and species composition, and should include information on snag densities, and understory cover. Understory cover estimates can be used to assess whether hardwoods are being effectively controlled by burning and if the grass–forb cover needed by Bachman's sparrows is developing.

We also want to know if the stand is functioning as we intended. We will want some indication of species use of the stand over time. Conducting a census of red-cockaded woodpecker nest trees in the stand every 5 years can be quite easily done by searching for resin flows on trees. In addition, 3–5 variable circular plots can be established in association with the CFI plots to sample bird density. Plots would be visited 4–6 times during the breeding season during the early morning. All birds seen or heard are recorded at each plot, and the distance from plot center to the bird is estimated. These data not only provide evidence of birds using the plot, but can also estimate densities. If our focal species are using the stand, then more intensive nest searching may be needed to document reproductive success in the stand (and more closely estimate potential fitness). Samples taken every 5 years would allow estimates of trends in use of the stand over time. If structure is not developing as intended or if species are not using the stand as predicted, then the prescription can be revised to increase the probability of producing functional habitat for these species.

## BUDGET

Cost almost always becomes a factor when implementing a prescription to achieve a goal. For most landowners, there must not be a net loss, and for some landowners there must be a maximum profit. Understanding the products that can be derived from implementing the prescription, both economic and ecological, can help the manager decide if the trade-offs are acceptable.

I developed a budget for our prescription based on a number of assumptions:

1. I assumed that pine sawtimber would sell for at least \$200/MBF and pine pulpwood for \$20/cord as stumpage. Stumpage is the value of the wood before it has been cut and is the price paid by a logger for standing trees.
2. That site preparation burning would cost \$425 and that in-stand burning would cost \$175 each time the stand is burned.
3. That a fire line can be established for \$100.
4. That planting stock, planting, and herbicide applications would collectively cost approximately \$250/ha.
5. That labor costs associated with marking tallying, sale oversight, and monitoring cost \$20/h.

On the basis of these assumptions and our prescription, we would anticipate two sales that could be sold as stumpage: an irregular shelterwood in 2015 and produce 49 MBF/ha of pine sawtimber and 66 cords of pulpwood per ha. Group selection and thinning in 2095 would yield similar volumes (49 MBF/ha and 47 cords/ha). Income and expenses are outlined in Table 10.7 and clearly indicate the potential for having incomes exceed expenses while managing habitat for these three animal species. And I did not include interest earned on income over the life of the plan, which would make the plan even more feasible for private landowners. Are profits maximized? Clearly not. But for a nonfederal forest landowner, both habitat objectives and timber income can be realized in this example. For federal managers there is a positive net return on the investment of taxpayer dollars with a net present value of \$69,333 over a 100-year period.



**TABLE 10.7**

**Estimated Income and Expenses Associated with Managing a 7 ha Pine Stand on the Kisatchie National Forest to Improve Habitat for Red-Cockaded Woodpeckers, Bachman's Sparrows, and Brown-Headed Nuthatches**

Decade	Activity	Income	Expenditure
2005	49 MBF/ha sawtimber	\$68,000.00	
	66 cords pine pulpwood/ha	\$9,225.00	
	Tree marking and tally		\$2,700.00
	Site prep burn		\$425.00
	Fire line		\$100.00
	Planting, weed control		\$1,700.00
	Implementation monitoring		\$200.00
	Monitoring regeneration success		\$200.00
	Prescribed burn		\$187.00
2015	Prescribed burn (3)		\$561.00
	Monitoring tree survival and growth		\$200.00
	Monitoring animal use		\$700.00
	Snag creation (if necessary)		\$1,000.00
2025	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2035	Prescribed burn (3)		\$561.00
	Precommercial thin		\$1,700.00
	Snag creation (if necessary)		\$1,000.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2045	Replant longleaf as necessary		\$1,700.00
	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2055	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
	Precommercial thin		\$1,700.00
2065	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2075	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2085	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2095	49 MBF/ha sawtimber	\$68,000.00	
	47 cords/ha of pulpwood	\$6,401.00	
	Tree marking and tally		\$2,700.00
	Implementation monitoring		\$200.00
	Prescribed burn (3)		\$561.00
2105	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
2115	Prescribed burn (3)		\$561.00
	Monitoring tree growth and snag recruitment		\$200.00
	Monitoring animal use		\$700.00
Total		\$151,626.00	\$30,683.00

*Note:* Numbers in parentheses indicate the number of activities during the designated time period.



## SUMMARY

Developing a prescription or plan for a stand or landscape entails articulating a set of goals and objectives. These objectives usually come in the form of describing a DFC, or set of conditions, that will likely produce both stand structure and function in the future. The challenge for the forest manager is to understand the potential of a site to achieve the DFC, the opportunities and constraints imposed by the current stand conditions, and the silvicultural approaches that should logically be applied to guide stand development toward one or more DFCs. Use of stand growth models can aid in visualizing future stand conditions and may help in developing a schedule and budget to help ensure that a DFC is attainable. Finally, since any plan is a hypothesis, monitoring of stand structure and function must be conducted and data used to refine future management actions.

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# 11 Riparian Area Management

Water—a requirement for most if not all life. Free flowing or standing water is indeed required by many species. Many amphibian species need free water during at least part of their life to lay eggs that will hatch and develop into adults before the water evaporates (Figure 11.1). Other amphibians spend all of their lives in water, such as aquatic forms of Pacific giant salamanders. Waterfowl use lakes, ponds, streams, marshes, and swamps as places to feed and raise young. River otters, snapping turtles, and bullfrogs are all aquatic predators that use resources in both aquatic and terrestrial environments (Figure 11.1). Because so many species are associated with free water, the interface between the water and the land, that area known as a *riparian area*, is usually given special consideration during forest management.

Riparian areas are perceived by many ecologists as being particularly important for animal species due to several conditions:

- Free water is available near food and cover for many species.
- Because of the high latent heat of water, riparian and wetland areas can have more humid and stable microclimates than adjacent uplands.
- Edges are often formed between riparian and upslope vegetative communities, enhancing the number of niches for vertebrate species.
- Streams and riparian areas can provide corridors (or barriers) for animals moving across the landscape.
- Net primary production often is higher in riparian and wetland areas and food quantity and quality tends to be higher there for many species.

The availability of free water near the food and cover resources in the terrestrial environment is critical for a large number of vertebrate species. For example, mountain beaver occur near riparian areas (or in very moist forests) because their primitive uretic system requires a humid environment (Schmidt-Nielson and Pfeiffer 1970). White-tailed deer use water as a means of escaping predators in bottomland forests. Elk and deer will often use riparian areas as a cooler environment during warm weather. Because of the high latent heat of water (it takes 1 cal to raise 1 g of water 1°C), riparian and wetland areas can have more stable microclimates than adjacent drier uplands. Some species are able to take advantage of these more stable riparian conditions.

Edges are often formed between riparian and upslope vegetative communities, enhancing the plant species richness and vertical complexity of the forest (Naiman et al. 2000). Horizontal complexity in vegetation is also often high due to the dynamic nature of stream systems, creating canopy gaps and patches of shrubs and trees of various species and sizes, some of which are soft-mast producers (Naiman et al. 2000). This increased complexity provides a wide range of habitat element types, abundances, and distributions, thereby providing greater opportunities for more species to occur than in simpler systems.

Although riparian areas can provide many of the elements needed by some species, they also can provide corridors or barriers for animals moving across the watershed. A water shrew may find a fast moving river to be an excellent corridor, but a short-tailed shrew or Pacific shrew may find the same river to be a significant barrier. Indeed, even for some bird species, riparian areas >50 m wide can become barriers that they are reluctant to cross (Tremblay and St. Clair 2009). Again the function of the stream as a connector or barrier is species specific (Savidge 1973, Henein and Merriam 1990).



**FIGURE 11.1** Species typical of riparian areas clockwise from top left: snapping turtles are found in slow-moving streams and ponds (photo by Mike Jones, with permission), spotted salamanders reproduce in vernal pools in the northeastern United States (photo by Mike Jones, with permission), cottonmouths are an aquatic pit viper of southeastern swamps and marshes, and beavers create ponds used by many other species.

Net primary production can be quite high, especially in slow-moving rivers and side channels, ponds, and lakes. Marshes represent one of the most productive ecosystems on Earth (Waide et al. 1999). Aquatic productivity is dependent on nutrient availability and sunlight in the water column. Nutrient-rich aquatic systems, referred to as *eutrophic systems*, can have very high levels of primary production through algal blooms, submerged vegetation, and emergent vegetation. But there is a price to pay for too much productivity. For instance, enriched streams with high levels of nitrogen and phosphorus from fertilizers can lead to algal blooms, which provide plant food for herbivores, but these plants can also deplete the water of oxygen. As plants photosynthesize, they return oxygen to the water, but in the evening they respire, using oxygen to survive while releasing carbon dioxide. Without adequate oxygen during the night many fish die from oxygen deprivation, reducing not only the fish fauna but also the food resources for their predators. So reducing nutrient loads is good, right? Perhaps, but again it depends on the species. Species that rely on a grazing-based energy acquisition by eating aquatic vegetation (e.g., carp) need nutrient-rich systems, while species that feed on aquatic insects associated with decaying leaves or with trees over the stream (e.g., trout) often do better in nutrient-poor (*oligotrophic*) streams.

## ANIMAL ASSOCIATIONS WITH RIPARIAN AREAS

Animal species richness is often higher in riparian areas than in adjacent upslopes (Knopf et al. 1988), especially in arid environments. In arid environments, the availability of water not only provides the opportunity for evaporative cooling but also supports a richer and more palatable vegetative community than adjacent upslope areas. In moist environments, the importance of riparian areas to many species is much reduced (McGarigal and McComb 1992).

Nonetheless in every forest system in North America, there is a suite of species that requires free water. These species are referred to as *riparian obligates*—you only find them near water. River otters, American dippers, beavers, and wood ducks are examples of riparian obligate species. Other species tend to be found more commonly near water but do not require the water directly for some aspect of their life. Yellow warblers, jumping mice, several species of myotis bats, and bald eagles are examples of *riparian associates*.

One group of riparian obligate species, pond-breeding amphibians, preferentially breed in isolated ponds and wetlands that hold water for only a part of the year. These *vernal pools* provide a predator-free (no fish) environment where the pond breeders can lay eggs and the larvae can feed on decaying vegetation, metamorphose into adults, and leave the pond before the pond dries out (Semlitsch and Bodie 1998). In some years, ponds dry out too soon, stranding the immature amphibians, while in other years the water lasts until after they have metamorphosed. Apparently pond-breeding species that reproduce in vernal pools have a selective advantage over those that use permanent fish-bearing water bodies where they face greater risks of predation. Animals must disperse from the ponds once they have metamorphosed, and there is evidence that at least some species may be adversely affected by forest management near vernal pools. Popescu and Hunter (2011) found that canopy removal and conversion of natural forest to conifer plantations in Maine may affect regional population viability of vernal pool breed amphibians by hindering successful dispersal of juveniles as they leave the vernal pool.

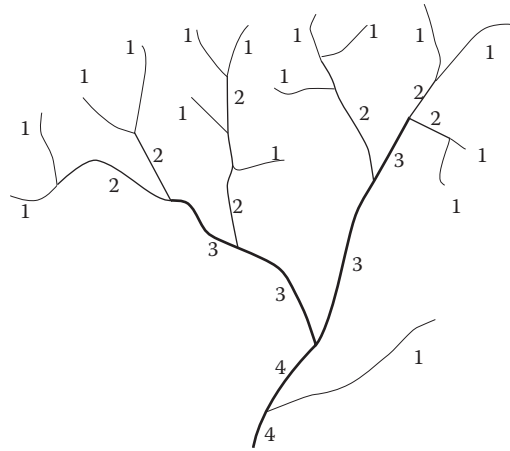
There are also some species that occur in forests that are not associated with riparian areas. They may obtain their water from condensation, rain, or their food. So although riparian areas are an important component of the forested landscape for many species, other upslope parts of the landscape are also important for additional species. Often forest management regulations focus on riparian areas due to the requirements to provide clean water, habitat for fish species, as well as habitat for terrestrial species, but may not consider the condition of the forest for other species some distance away from the riparian area. Such a focus on riparian areas can be to the detriment of other species not associated with riparian areas.

## GRADIENTS WITHIN RIPARIAN ZONES

There are two dominant gradients of habitat elements associated with riparian areas, the intra-riparian and trans-riparian gradients. *Intra-riparian gradients* refer to the continuum of conditions from the headwaters to the confluence with larger water bodies, and eventually with the ocean at an estuary. Riparian areas are hierarchical systems constrained by a *watershed*, the area of the land that captures and routes water down hill (Figure 11.2). The upper reaches of a stream system are usually intermittent streams that flow only following rain or during snowmelt. Long considered somewhat irrelevant to the function of the permanent stream system, intermittent and headwater streams can provide important refugia for many species of amphibians (Sheridan and Olson 2003, Stoddard and Hayes 2005). These areas can also serve as conduits for transport of nutrients, sediment, and pathogens from upslope areas into the stream system (Naiman et al. 2000). If water temperature, sediment loads, or other pollutants are of concern downstream, then these intermittent streams deserve attention, in addition to meeting habitat needs for species associated with these areas (Wigington et al. 2005). Roadside ditches also often serve as conduits of sediments and chemicals into the stream system. Indeed road systems are the primary cause of excessive sediment loads in many forested stream systems (Reid and Dunne 1984).

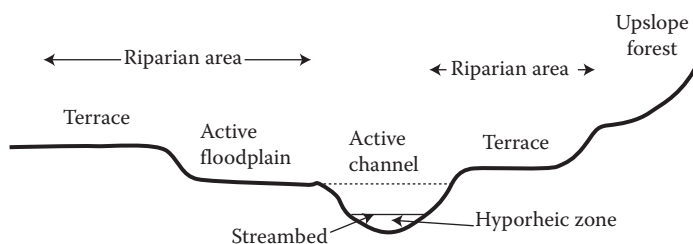
Depending on the geology of the area, the upper portions of mountain watersheds usually support streams that have high gradients and deeply incised channels. The opportunity for the stream to move from side to side and develop a complex floodplain with side channels, pools, riffles, and glides are limited until the gradient declines somewhat and sufficient time has elapsed to lead to stream bank erosion, deposits of sediments, and expansion of the active channel (Figure 11.3). Within the middle part of watersheds, adequate stream volume usually leads to the development of a floodplain





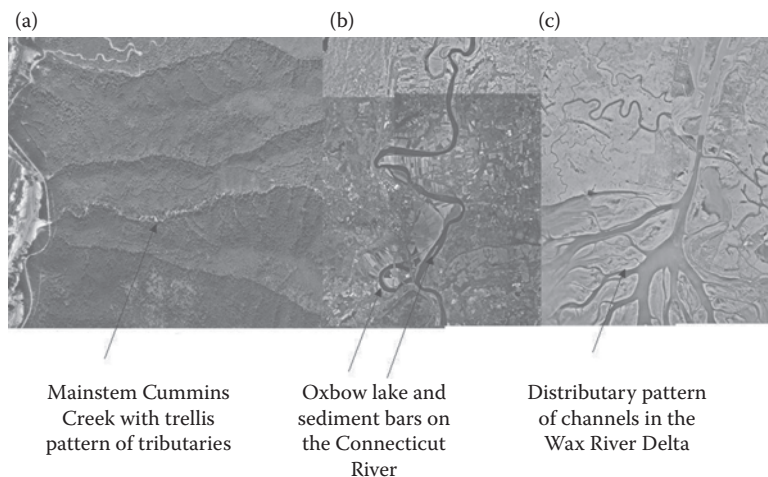
**FIGURE 11.2** Hierarchical system of stream orders along an intra-riparian gradient. Note that two first-order streams form a second-order and two second-order streams form a third order, and so on.

following successive flood events that not only erode stream banks but also deposit sediments from farther upstream. These *alluvial* sediments often form terraces that represent different flood intensities and frequencies (Naiman et al. 2000). Intense floods that carry sediments and wood from the headwaters into the mid-watershed area spill over the banks of the active channel into the active floodplain. As the water slows along the flooded edges, it cannot carry the same sediment load and so it deposits the sediments in this floodplain. Frequent floods such as this allow the development of an active floodplain where tree species such as cottonwoods and willows may become established. Species such as these are well adapted to colonizing the exposed sediments when the water levels recede. They have seeds that are carried on wind and water, deposited on the sediments, and grow rapidly to claim the site. These mid-watershed areas often have braided channels as the stream velocity slows, depositing sediments (Naiman et al. 2000). Depending on the stream volume under high flows, the mid-watershed is also the portion of the stream system where beavers most often build dams, creating a staircase of ponds. The active floodplain also provides an opportunity for the stream channel to begin to meander and create a more complex stream channel system. Side cutting of the channel allows the formation of steep stream banks with bars, or areas of deposition on the opposite side of the stream (Figure 11.4). Steep stream banks provide places for belted kingfishers to nest and the bars provide nesting sites for spotted sandpipers and other shorebird species. Finally, as the stream approaches an estuary, water velocity slows further and is affected by tides causing sediment loads to further decline, forming a delta. These delta conditions can allow the formation of *marshes* (wetlands dominated by nonwoody vegetation) and *swamps* (wetlands dominated by woody vegetation). As the water flows into estuarine conditions, many tree and shrub species cannot tolerate the saline conditions and are replaced by grasses, sedges, and rushes to form a marsh.



**FIGURE 11.3** A schematic of a trans-riparian gradient from streamside to upslope.





**FIGURE 11.4** Three types of stream morphology in Cummins Creek, Coastal Oregon (a) with steep gradient and narrow valley floors; the Connecticut River Valley in Massachusetts (b) with broad floodplains and oxbow lakes; and the Wax River Delta in Bayou Vista Louisiana (c) with high sinuosity and delta formation. Note as floodplain width increases and gradient decreases, sinuosity and braided channels increase, leading to greater riparian complexity. (Images courtesy of TerraServer.com/DigitalGlobe. With permission.)

Swamps that often occur at the lower end of a watershed are frequently flooded forests dominated by species tolerant of being partially submerged for prolonged periods, such as baldcypress and water tupelo in the deep South of the United States. Because of the often nutrient-rich environments associated with these swamps, they tend to be highly productive areas for growth of trees and shrubs and as well as the animals associated with them. Louisiana's Atchafalaya Basin is the largest swamp in the United States at 241,000 ha (595,000 acres). Because this area and others like it along the Mississippi River and other large rivers produce nutrient-rich floodplain soils, many have been cleared for agriculture and more recently for urban expansion, despite the risks of repeated flooding.

Along the intra-riparian gradient, changes in the geomorphology of the watershed give rise to areas of different stream velocities and stream substrates. Continuously rushing water over boulders produces a cascade, which is excellent habitat for Harlequin ducks and tailed frogs. Where erosion-resistant rock or a log partially dams the stream, a pool can form both above and below the obstacle (Figure 11.5). The upstream pool collects water and allows sediment deposits to build up above the obstacle, aggrading the stream channel. As the water flows over the object, it forms a plunge pool. These areas provide places where beaver often initiate a dam (Leidholt-Bruner 1990). Both areas can add to stream channel complexity and hence provide conditions suitable to more species of aquatic and semiaquatic animals.

Large logs falling into a stream or carried downstream from headwaters (often from landslides) can add significantly to channel complexity in many forested areas. Indeed, the transport of wood from the headwaters to the estuary provides opportunities for use by many species from salamanders to otters to salmonids to mollusks along its journey to the sea (Maser et al. 1988). This stream complexity contributes habitat elements and increases the potential number of species that use the variety of conditions that are created. The variability in flood frequency and severity adds complexity such as fine-scaled features of topography, which are strongly related to plant species richness and species composition (Naiman et al. 2000).

*Trans-riparian gradients* refer to the changes in conditions as you move from the edge of the stream into upslope forests, perpendicular to the gradient (Figure 11.3). Along the edge of a stream there often is the opportunity for emergent plants to become established, especially if there is adequate sunlight and the gradient is not too steep. Along the edge of a steep active channel, erosion and



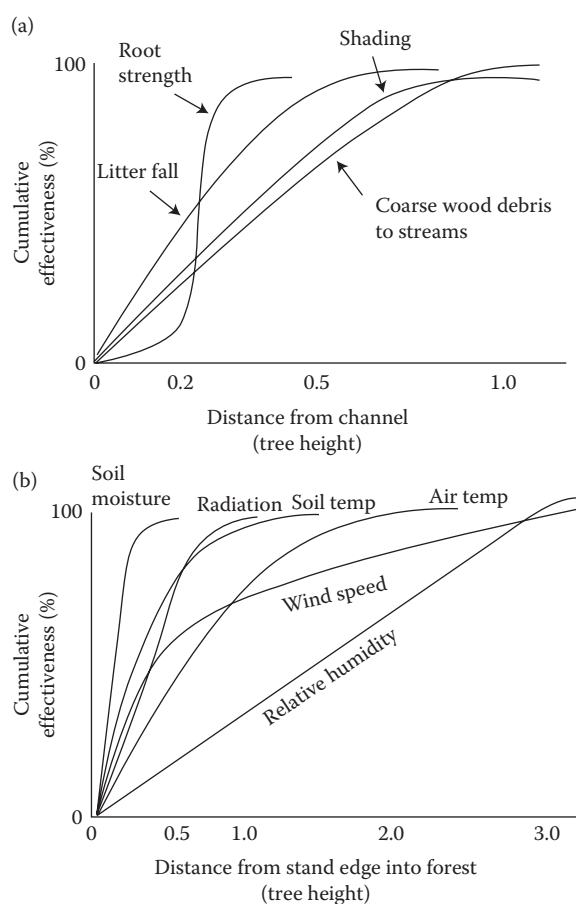
**FIGURE 11.5** Dead wood in a stream in the Berkshires of Massachusetts reduced stream flow and caused sediment deposition upstream and a plunge-pool downstream, adding to channel complexity.

deposition often prevents trees from becoming established, so water-tolerant herbs and occasionally shrubs dominate. Farther upslope, species of trees that are water tolerant begin to dominate in those areas that are flooded less frequently. Species such as red alder, cottonwoods, silver maple, water oak, and pin oak can be found here (depending on what region of North America you are in). As you move farther upslope, these species give way to others that are not so tolerant of saturated soils and more drought tolerant; eventually you are in an upslope tree community. We can see changes in plant species along the moisture gradient all the way to the ridge top. As these plant species change along this trans-riparian gradient, many animal species also change accordingly. In the northwest, we might find Pacific jumping mice and Dunn's salamanders in the floodplains near the stream and not find them farther upslope (McComb et al. 1993). On the other hand, we might find California red-backed voles upslope but rarely along the stream. Many species distribute themselves throughout a watershed in response to the moisture, soils, and vegetation seen along the trans-riparian gradient.

## RIPARIAN FUNCTIONS

The function of a riparian area in providing energy, nutrients, and other resources to animals is heavily influenced by a set of processes linking streams to adjacent forests. Forested streams often produce cool, clean water; so it is not surprising that most public water is supplied from forested watersheds. Forests are the natural filter. Because they are clean, they tend to be low in nutrients. Low nutrient availability can limit the productivity of fish and invertebrates in the stream and hence can limit the abundance of their predators. Indeed, low availability of nitrogen, phosphorus, and carbon in streams of the Pacific Northwest has been suggested as a reason for poor productivity of salmon that once thrived in these streams. Because millions of salmon once spawned and then died in streams, the decomposing bodies provided a rich food source for native fish, including young salmon. Now that salmon runs are at 5% or less of historic levels, this source of nutrients for juvenile salmon is no longer present. In addition, there is evidence that these salmon-derived nutrients influence the function of the entire riparian area (Compton et al. 2006). As a way of giving these streams a boost in nutrients, some fisheries biologists have begun stocking dead salmon to enrich the stream (Compton et al. 2006). Not all forested streams have a natural source of anadromous fish carcasses; so they receive nutrients in other ways. Leaves from trees and shrubs are nutrient rich. Most nutrients in a tree are not stored in wood (except for carbon) but rather in leaves. When

leaves and needles fall into a stream as *allochthonous* material, they decompose and provide an energy source for decay organisms. This material then serves as a source of detrital-based energy in nutrient-poor (oligotrophic) streams. Consequently, having a streamside forest ensures that the leaves that fall into a stream provide an important energy pathway (Figure 11.6). Litter fall into the stream provides a source of nutrients and food for aquatic invertebrates, which in turn are food for vertebrates. Streamside forests also provide root systems along the stream margin, stabilizing the streambanks and minimizing erosion and hence sediment loads. Sediment loads that are too high can cover cobble and gravel substrates important to many species of spawning fish (Kondolf 2000), as well as reducing habitat quality for species of salamanders that occur in well-aerated gravels. When trees die or are carried downstream, the wood increases channel complexity. Large pieces of wood are particularly important in large streams because they tend to move through the system more slowly than smaller logs. When a log falls into a stream, water flowing over and under it increases in velocity. Increased water velocity under the log leads to scour pools and the log forms a source of cover for fish and amphibians, as well as a place for otters to feed, muskrats to mark territories, and salamanders to survive in moist conditions.



**FIGURE 11.6** Hypothesized functional (a) and microclimate (b) relationships from streamside forests. (From FEMAT [Forest Ecosystem Management Assessment Team]. 1993. *Forest Ecosystem Management: An Ecological, Economic, and Social Assessment*. US Forest Service, US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, US Bureau of Land Management, Fish and Wildlife Service, National Park Service, Environmental Protection Agency, Portland, OR.)

Dead wood decay rates in water are usually less than on dry land because aerobic decomposition is decreased in water. Indeed, to store logs for long periods they are often submerged in log ponds prior to milling. Nonetheless, logs will eventually decay or move through the stream system; so a continual supply of logs is needed to ensure that these processes of developing channel complexity are maintained.

Tree canopies also provide shade to water and are especially important during the summer. Direct solar radiation striking a water surface can raise the temperature (Sinokrot and Stefan 1993) and increased temperatures can be deadly for some species. Tailed frogs in the western United States usually only occur in streams with water temperatures of 5–17°C. Many other species of amphibians and fish also are associated with cold water, and even American dip-pers, which feed on invertebrates in cascading mountain streams, are often associated with cold water. But some sunlight should enter streams. Sunlight allows photosynthesis to occur in the stream allowing the formation of periphyton on rocks, as well as other aquatic vegetation to grow (Welnitz and Rinne 1999). Periphyton and decaying plants provide energy to some animal species that support a food web. So a continuous riparian leaf cover, especially conifer cover, may not be ideal for most stream species. The manager often will strive to find a balance between providing enough leaf cover to maintain stream temperature but not so much that it inhibits photosynthesis in aquatic plants throughout too much of the system. The forest structure and composition along riparian areas all contribute to a set of microclimatic conditions that influence habitat quality for some species up to three tree heights from the stream edge, depending on the upslope forest condition (Figure 11.6). If the upslope forest is a recent clearcut or open rangeland, then the advantage of the microclimatic buffering to the riparian area may be critical to allowing some species to persist. On the other hand, if the upslope forest is similar in structure and to the streamside forest, then the stable microclimate can probably be realized over less than three tree heights. Height of mature trees, rather than a fixed distance, is often used as a guideline for maintaining riparian functions because many factors (e.g., shade, dead tree fall) relate more to tree height than to a fixed distance.

## RIPARIAN BUFFERS

Most states and provinces in North America have regulations regarding stream protection (Figure 11.7). Often the rationale for these streamside management areas include protecting water quality (especially temperature and sedimentation) and providing habitat for “fish and wildlife.” But by now you should be asking, which fish and which wildlife? Therein lies the conundrum.

These streamside management areas may be no-harvest zones, or they may allow harvest down to some minimum basal area, or they may be designed to meet specific needs. Typically a linear distance from the active channel is used to delineate the buffer strip width (Figure 11.7). Habitat for selected species of vertebrates will be provided dependent on the width and length of the buffer (riparian management area). Buffers that are narrow will likely only be used by species with small home ranges or those that have naturally linear home ranges (e.g., belted kingfishers and mink). The appropriate width will depend on the species and what it is being used for: a nesting territory, movement corridor, feeding, or resting. There is not a “one size fits all” prescription for these riparian management areas (Figure 11.8).

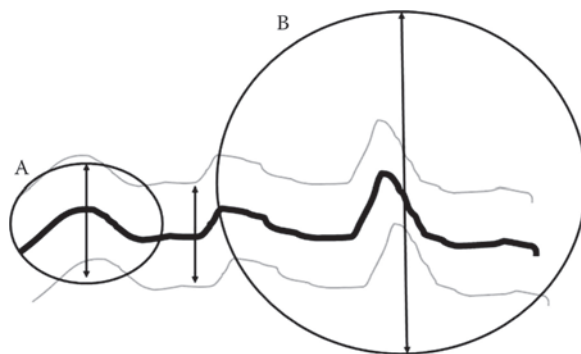
As a first approximation of the necessary width of a riparian buffer for a species consider its home range and the habitat elements that it needs. If it is a riparian associate and has a home range of 20 ha (50 acres), then the radius of that home range is 142 m (470 ft;  $A = \pi r^2$ ). If the species is mobile and the stream does not represent a barrier to movement, then the width of the buffer should be 142 m on each side of the stream (Figure 11.8). If the stream is a barrier to movement, then the width would need to be 2 times as large or 284 m (940 ft) on each side of the stream. Managing for multiple species requires assessing buffer width for each species. The species requiring the largest buffer width sets the width for the others. So the appropriate buffer width for a small amphibian



**FIGURE 11.7** Streamside buffer in a managed Oregon forest. These buffers are required by the OFPA. (Photo by Dave Vesely. With permission.)

with a home range of 0.2 ha would be 14 m (47 ft), correct? Well, maybe. If the species is associated with the riparian area due to the moisture needed for survival (a salamander or frog), then designing a buffer to support a suitable microclimate may be more appropriate. Vesely and McComb (2002) found that buffers probably need to be far wider than would be expected based on what is assumed to be a small home range for these species.

Another aspect of buffer width to consider is the land ownership pattern. Buffer strips may be very wide on some federal ownerships or where public drinking water supply is a primary use of the water. The adjacent private landowner may be required only to provide a buffer of 15 m (50 ft), and be able to harvest half of the basal area from it. The next landowner along the stream may be a dairy farmer who is not required to leave any buffer; so cows are allowed to graze to the stream bank. The ability for the stream to meet goals of the various landowners is inherently compromised by who is upstream of whom. Should the farmer own the upstream parcel and allow cows to use the stream,



**FIGURE 11.8** The width of a riparian management area is different for different species. Species A has a territory size that fits within the proposed buffer; species B has a territory size that would not be accommodated by the buffer.

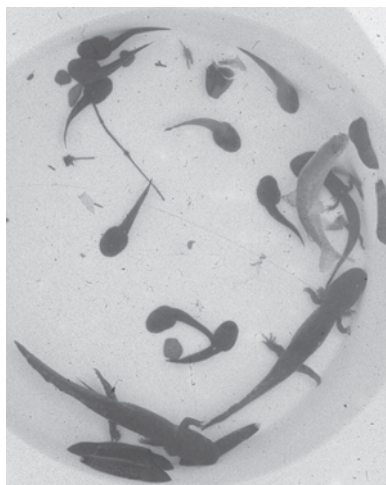


then the south end of a north-facing cow (doing what cows will do) will contribute significantly to poor water quality on the downstream ownerships. It takes a cooperative community to manage clean water in a watershed.

### MANAGING WITHIN STREAMSIDE MANAGEMENT AREAS

Many landowners approach riparian areas as set-asides—areas not to be managed. Regulations certainly restrict management options (Figure 11.9). Within the limitations of what is allowed by law, the principles of managing habitat elements described in the previous chapters also apply to riparian areas. Depending on the species for which you would like to provide habitat in a riparian area, you can manipulate the vegetation to provide or produce those habitat elements. As mentioned many times already, managing for one species or set of species is managing against other species. Over the past 10–20 years in the Pacific Northwest, there has been an effort to restore conifer dominance to many riparian areas so that there is a continual supply of large dead wood to the stream (MacCracken 2002). This dead wood adds to channel complexity and habitat quality for salmonids and other species. At least two things must be kept in mind when trying to achieve this goal. First some species such as white-footed voles (Manning et al. 2003) and several species of myotis bats (Holly Ober, personal communication) select hardwood riparian areas over conifer-dominated riparian areas as places to feed. So some hardwood riparian areas should be retained. Second, dead wood not only comes from the stream side but also from the upslope areas during landslides; so streamside trees are only part of the source of dead wood to these streams. The conclusion drawn from this information is that there is not a single riparian management strategy that will meet the needs for all species, just as there is no stand or landscape management strategy that will meet the needs for all species.

For instance, if you wished to improve soft mast production along streamsides, then providing openings in the canopy would provide sunlight to fruit-producing plants and enhance production of soft and/or hard mast. But too much sunlight can raise water temperature and reduce habitat quality for some species. Balancing these conflicting goals is a social dilemma—what are the goals for the stream system? But when setting goals for riparian areas it is important to consider the context within which the riparian area resides. Is it an agricultural area with abrupt edges, a clearcut (Figure 11.7), a thinned stand, or an old-growth stand? How the riparian area will function as habitat



**FIGURE 11.9** Two species of aquatic amphibians captured in a forest stream in Coastal Oregon, larval forms of Pacific giant salamander and tailed frog tadpoles, in addition to a cutthroat trout. State and federal riparian rules are designed to maintain water quality and protect species such as these. (Photo by Joan Hagar. With permission.)



for a suite of species will be greatly affected by these adjacent conditions. Although the goals for riparian areas may be different than for upland systems, the two systems are connected and should be considered part of a larger landscape or watershed, and not managed in isolation.

Several important principles have been provided by others for maintaining or restoring riparian conditions to meet a variety of riparian goals (Stanford and Ward 1988, 1999, Naiman et al. 2000):

- Restoring biophysical properties of riparian zones improves other natural resource values. Riparian zones allowed to respond to disturbance and regrowth may maintain a high level of complexity in plant species composition and structure used by a variety of animal species.
- Protecting interactions between surface flows and groundwater is essential to aquatic–riparian ecosystem integrity. This is particularly important relative to the *hyporheic zone*, the subsurface saturated sediments along the stream bottom.
- Allowing streams and rivers to migrate laterally is necessary for development of riparian habitat elements. This continual disturbance creates a mosaic of substrates and vegetation used by a wide variety of species.
- Incorporating natural flow regimes in regulated rivers promotes aquatic and riparian diversity and resilience. Many species are well adapted and indeed rely on the variability in flow rates in rivers and streams that have occurred for centuries prior to use of dams and levees.
- Modify human-imposed disturbance regimes to create and maintain a range of habitat conditions in space and time within and among watersheds that reflects the range of conditions to which desired species are well-adapted (Reeves et al. 1995). Humans like stability. Unexpected large-scale disturbances are considered catastrophes by many humans, but it is these events, over entire watersheds, that can influence changes in habitat availability for the full suite of terrestrial and aquatic species found in a region.
- Control invasive species that can simplify vegetation structure and composition. Aquatic and streamside vegetation that is invasive can exclude other *phreatophytic* (water-associated) vegetation from the site decreasing the vegetative structure and composition of the streamside area.

## BEAVERS: THE STREAM MANAGERS?

Busy beavers: they just love to stop water from flowing. Although not all beaver populations build dams (some live in dens in streambanks), most build quite impressive dams that flood large areas for long periods. In many respects, they are the streamside managers or destroyers—much depends on your perspective (see Chapter 1). Beavers cut trees and shrubs and use them to build dams, lodges (where they raise young), and as a source of food when they eat the bark from these plants. The openings that they create in streamside forests provide a flush of early successional plants along streams and provide habitat for a wide variety of early successional associated species (e.g., yellow warblers, jumping mice, and Carolina wrens) (Figure 11.10). Their dams create a *lentic* (lake-like) environment out of a *lotic* (flowing water) system, providing brood habitat for wood ducks and places for pond-breeding amphibians to reproduce (e.g., newts). These changes in vegetation and pool conditions give rise to different assemblages of animal species in the vicinity of beaver ponds than where beaver ponds are not present (Suzuki and McComb 2004). The dams cause ponds to capture sediments and if the pond persists long enough, then it fills in with sediments over time and forms a wet meadow, which also is habitat for a completely different set of organisms. Because of their profound effect on riparian area function and ecological succession, beavers have been proposed as an umbrella species for which management could be focused in riparian areas (Stoffyn-Egli and Willison 2011).

To a forest manager trying to raise a commercial timber species near a stream or trying to develop forested riparian conditions for other species, beaver can be a problem (Bhat et al. 1993).



**FIGURE 11.10** Beavers flood bottomland forests, creating snags, meadows and pools, markedly changing the composition of the riparian animal community in a stream reach.

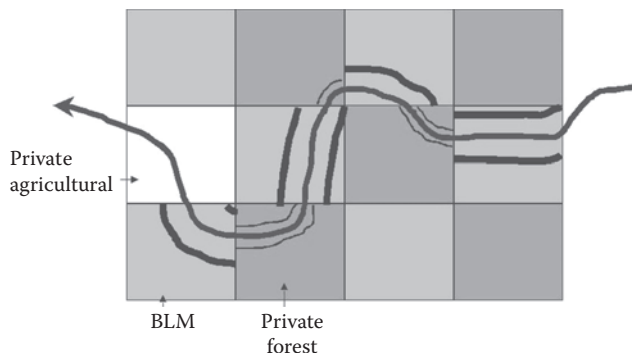
They preferentially cut hardwoods over conifers and small trees over large ones (Basey et al. 1988). But they eventually cut nearly all trees around their pond. So planted seedlings are often cut, and mature trees may be cut as well, especially those close to the stream. In addition, their dam leads to flooding of low-lying areas causing tree death. And because culverts are easy places to block and flood a large area upstream of the culvert, damage to culverts and roads also becomes problematic (Jensen et al. 2001). Use of beaver deceivers, devices to allow water to flow through a dam, can help to reduce the damage caused by flooding (Nolte et al. 2001). Tree cutting, however, can only be reduced if trees are protected by wire mesh—an expensive proposition to a forest manager.

So at what point do beavers become a problem vs. a natural part of riparian area dynamics? Again that is a social decision. Since beavers do not respect property lines, decisions made by one landowner clearly influence the riparian conditions of her neighbor.

### **CASE STUDY: RIPARIAN AREA MANAGEMENT IN A PATCHWORK OWNERSHIP**

To illustrate the conundrum associated with riparian area management, consider the pattern of riparian management areas along streams in western Oregon where the Bureau of Land Management (BLM) manages public forest land. During 1866, Congress created the Oregon and California railroad lands as alternating square miles of land in western Oregon and providing an incentive for the railroads to build infrastructure into the region. Congress revested or pulled these lands back into public ownership in 1916 and eventually gave the BLM the responsibility for managing them, mandating that a portion of the timber receipts go to the counties to support schools. The BLM now manages a checkerboard of lands across much of western Oregon. Intervening lands, many of which are forested but some of which are agricultural, are privately owned.

For years, the BLM followed the Oregon regulations with regards to streamside protection, but when concerns arose regarding endangerment of spotted owls, marbled murrelets, and coho salmon, a new management strategy for all federal lands in the region emerged as the Northwest Forest Plan (NWFP) (FEMAT [Forest Ecosystem Management Assessment Team] 1993). A stream passing across this landscape has three predominant land uses: federal forests designed to provide habitat for late-seral species, state and private forest lands that are largely timber producers, or private



**FIGURE 11.11** Schematic of possible riparian management area widths along a stream passing through three types of land ownerships.

agricultural lands for crop and livestock production. Each land use has its own standards and guidelines for riparian management area (Figure 11.11).

Under the NWFP, buffers along nonfish-bearing streams are one site-tree height in width, and along fish-bearing streams they are two site-tree heights in width. A site tree is the height of the dominant trees in the region—45 m (150 ft) in many of these watersheds. So, in this example, buffers on fish-bearing streams would be 90 m (300 ft wide) and on nonfish-bearing streams they would be 45 m. These buffers could be adjusted up or down depending on the results of a watershed analysis, a process where the functions of the riparian area are more completely considered before additional management is allowed (Montgomery et al. 1995).

On private forested lands, the Oregon Forest Practices Act (OFPA) prescribes streamside management areas that vary depending on stream width and whether they are fish-bearing streams (Figure 11.11). Typical OFPA buffers might be 8–33 m (25–100 ft) in width, with some timber removal allowed within them if canopy cover and basal area guidelines are met. On private agricultural lands, buffers or other management actions are not required, unless the stream is listed as “impaired” under the Clean Water Act (Hill and Blair 2005). Impairment may be caused by many things, most often by increased temperature, sediment, nutrients, or pesticides compared with standards set by state environmental quality offices.

Now consider a stream passing through a set of these ownerships (Figure 11.11). What imprint does land ownership have on the ability of a stream to maintain certain functions? If the stream passes through agricultural land causing the water to become too warm and the stream reach to be listed as impaired, and that warm water flows through an adjacent forest owned by someone else, then the downstream neighbor inherits the problem. And we all live downstream. We see similar inconsistencies in streamside protection in forested regions across the country. Is there a more thoughtful design? Clearly there is, but a more thoughtful design makes regulation of policies very difficult. The challenge is to provide incentives to encourage a more thoughtful and cooperative streamside protection strategy that allows streams and other water bodies to meet habitat, clean water, recreation, and other goals.

## SUMMARY

Riparian areas are adjacent to and influenced by a body of water, typically a stream, lake, pond, or wetland. Some species are riparian obligates, those that only occur in or near water and the associated streamside conditions. Other species may not require water but are riparian associates and are found in riparian areas more frequently than in upslope areas. The distribution of habitat elements is influenced by the intra-riparian gradient, from the headwaters to the ocean, and the trans-riparian

gradient, from the streamside to the ridge top. Contributions of matter and energy to stream systems is often a function of the distance from the stream edge upslope several tree heights. Delineating and managing streamside areas as habitat for a species can be constrained by regulations or influenced by desired ecosystem services and functions. Buffer strip width appropriate to meet a species needs is dependent not only on the home range size of the species being managed, but also on the energy and microclimatic conditions needed by the species. Buffer regulations are highly variable, and current state and federal policies in the United States can lead to highly inconsistent riparian area management strategies across a mixed-ownership landscape.

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# 12 Dead Wood Management

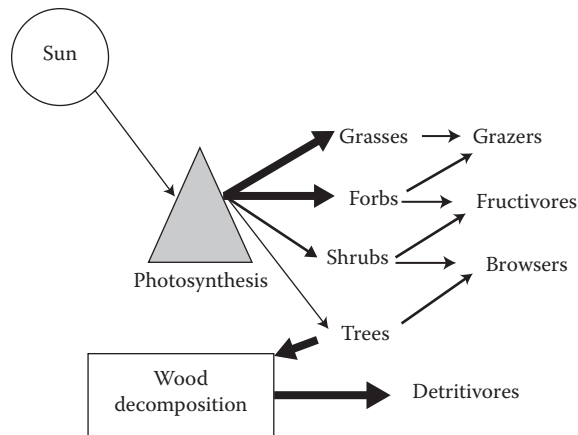
Dead trees are the losers in density-dependent competition and a product of forest disturbance and disease. Considered by many to be a waste of wood fiber and a fire hazard, dead wood provides habitat for many animal species, nursery sites for germination of plants, and pathways for energy in a cellulose-based environment (Harmon et al. 1986). A large western redcedar may live to be 300 years old, and then may take another 300 years or more to decay (Embry 1963). Throughout its life, and after its death, a tree can play a role in contributing to habitat quality for a succession of organisms (Maser et al. 1979). Consider the pathway of energy following a natural disturbance that creates an early successional forest (Figure 12.1). Photosynthesis leads to allocation of energy to leaves, fruits, tree boles, and roots. In later stages of forest succession, most forest energy is stored in cellulose, and cellulose must be broken down into simpler molecules to allow the stored energy to become available to other organisms. Following an intense disturbance, that cellulose is abundant and can be decomposed to provide energy to other life forms. This process is the primary mechanism allowing energy flow through trophic levels in detrital-based systems. Cellulose is also the primary source of stored carbon in forest systems. Carbon is slowly released as CO<sub>2</sub> during wood decomposition (Harmon et al. 1986). The decaying wood is also associated with nitrogen-fixing bacteria, which may contribute to the soil nitrogen, thereby influencing soil fertility in some forest types (Sollins et al. 1987).

The fungi and invertebrates responsible for decomposing and fragmenting the wood become the basis for energy flow into other organisms. The organisms responsible for decomposition can differ markedly between aquatic and terrestrial systems, often leading to slower rates of decay in submerged wood versus wood exposed to air. Further, dead wood can affect the function of terrestrial and aquatic systems. Dead wood adds complexity to forest floors, increasing ground-surface and below-ground heterogeneity. Trees and snags that fall into streams can have significant impacts on sediment deposition and scouring within the channel, leading to a more complex channel structure than would be present without these logs (Bisson et al. 1987).

When a tree dies it may: (1) remain standing in some cases for decades, (2) be uprooted by wind, or (3) progressively break into pieces from damage or decay (Putz et al. 1983, Tyrrell and Crow 1994) (Figure 12.2). As trees die and decay, the species that can use the tree change as well. Further, changes in the structure of forest through ecological succession influence the function of the dead and dying trees. Many cavity-nesting species rely on dead and dying wood as a source for their nest or roost cavity. In New England forests, 41 species of birds and mammals use standing trees with decay present (DeGraaf and Shigo 1985). Scott et al. (1977) listed 85 species of cavity-nesting birds (CNB) that occur in North American forests. Forest management decisions have become increasingly based on habitat relationships of animals dependent on dead wood in forests around the world. Often these relationships are summarized for large functional groups of species, such as primary and secondary cavity users, and log users.

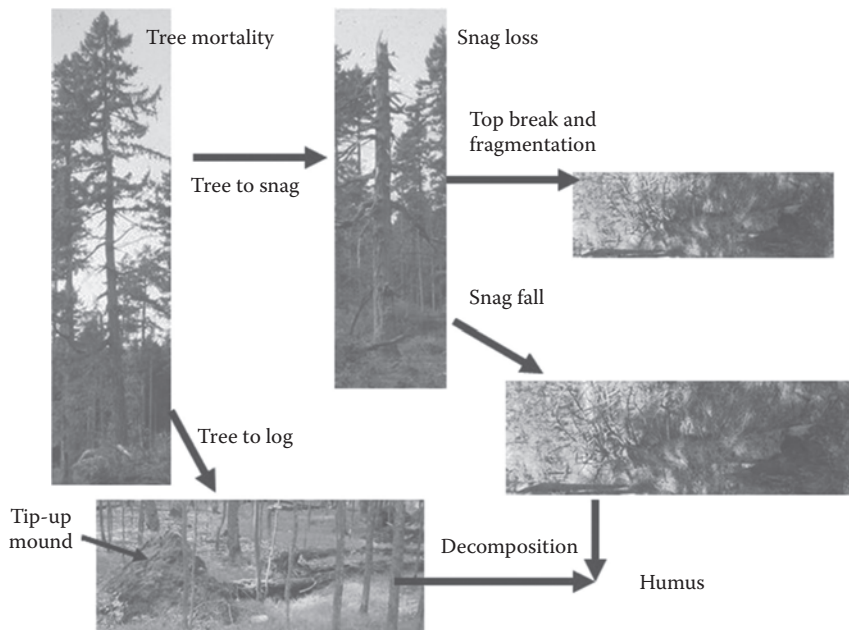
## PRIMARY CAVITY EXCAVATORS

Up to 40% of the bird species in North American forests are cavity nesters (Scott et al. 1977, Evans and Conner 1979). In many forest systems, primary CNB (species such as woodpeckers that excavate their own cavities) play a key role by providing cavities that are used by secondary cavity nesters (species that use cavities excavated by primary cavity nesters or natural cavities created by decay). Much attention has been given to management of primary cavity nesters based on the



**FIGURE 12.1** Early in succession following a natural disturbance, energy is transferred to higher trophic levels through both grazing-based systems and detrital systems due to the high levels of dead wood at this successional stage.

assumption that if they are present and excavating cavities then secondary cavity nesters will have the habitat that they need to survive (Neitro et al. 1985). Although some species of primary cavity nesters can excavate cavities in living wood, most excavate cavities in either dead wood or through live wood into decaying heart wood (Conner et al. 1976). Because most hardwoods and some conifers compartmentalize heart rot (Shigo 1984), excavation through sapwood into softened heartwood may allow organisms to create cavities in tree sections that are only 2–3 times the diameter of a bird's body. However, in many conifers and some hardwoods, decay of sapwood must occur to a sufficient depth toward the heartwood to allow excavation of the sapwood alone (Miller and Miller



**FIGURE 12.2** The fate of trees, snags, and logs in forests. Live trees can die and become snags, fragment and fall and become logs, or fall directly and become logs. Each step in the process provides habitat for a different suite of animals.

1980). For instance, pileated woodpeckers may excavate a cavity in a tree of only 55 cm (22 in.) in diameter in eastern hardwood forests of the United States (Evans and Conner 1979) but often select much larger conifer snags for nesting in the Pacific northwest of the United States (Nelson 1988). Generally, snags or dead limbs <10 cm (4 in.) in diameter are of little or no value as nest sites for primary cavity-nesting vertebrates. Small pieces of dead wood may become important feeding substrates for some species, but foraging probably is more energy efficient on larger stems than on smaller ones, leading to selection of large stems for foraging by most species (Brawn et al. 1982, Weikel and Hayes 1999).

Most species of primary CNB use only one nest cavity per year, although a few species may use different cavities if they raise more than one brood of young in a year (Bent 1939). The excavation of a cavity is a required part of the nesting ritual for most primary cavity-nesting species (Nilsson 1984). Additional cavities often are created and used by CNB as roost and rest sites (Bent 1939). A pair of CNB may use 1–10 or more cavities within a territory for nesting and roosting each year. For instance, species such as acorn woodpeckers and red-cockaded woodpeckers have nesting clans that include helpers to help raise the young (Lennartz and Harlow 1979, Neitro et al. 1985). Roost sites must also be available for the breeding pair as well as the helpers. Consequently, primary cavity excavators create many cavities in a pair's nesting territory over time.

Many species of primary CNB feed on wood-boring insect larvae and so require dead wood as a foraging substrate within a territory (Otvos and Stark 1985). Consequently, there must be a continual replacement of feeding sites as well as nest sites within territories to allow them to remain occupied. Other species, such as common flickers, feed primarily on insects found on the ground or in understory vegetation; dead substrates are not as important as foraging sites for these species (Brawn et al. 1982). In summary, the need for dead trees or limbs as feeding sites varies considerably among different species of primary CNB occupying any given tract of forest.

## SECONDARY CAVITY USERS

Secondary cavity nesters can be conveniently placed into one of two groups: (1) obligate cavity users (those species that must have a cavity for nesting or breeding) and (2) opportunistic cavity users (those that use cavities but do not require them). There are many species in the second group ranging from invertebrates to black bears (McComb and Lindenmayer 1999) that opportunistically use dead or dying trees as cover, but we will focus on obligate cavity users.

Secondary cavity nesters can use cavities created by primary cavity nesters or cavities created by wood decay following damage to a tree. Trees that sustain physical damage from wind or fire often become infected with fungal decay (Shigo 1965). The death of branches by self-pruning, incomplete branch shedding, and wound occlusion, or mechanical damage usually provide avenues for decay microbes to enter live trees. Compartmentalization of decay can lead to isolated columns of decay, commonly producing a cavity (Shigo 1984). If the tree remains alive, then compartmentalization of the wound may allow cavity formation, or subsequent healing may preclude development of a cavity (Sedgwick and Knopf 1991). Tree cavities provide a very secure and microclimatically stable den, nest, or roost site (McComb and Noble 1981b).

The number of cavities used by an individual varies widely among species. Some secondary CNB change nest sites between broods presumably to avoid parasite burdens (Mason 1944); some mammals also move among den sites in response to high ectoparasite loads (Muul 1968). For example, house wrens and bluebirds may use 1–3 nest cavities each year and defend each from other species. Cavity-using mammals also tend to use many den sites. In North America, northern flying squirrels use multiple cavities as well as external nests within their home range (Martin 1994). Some species use communal nest and roost sites. Swifts and bats may roost communally, with hundreds of individuals occupying one site.

There are many more species of secondary cavity nesters than of primary cavity nesters, and each species has its own requirements for the type of cavity or roost site used (Balda 1973). Long-legged

bats and brown creepers use spaces behind loose bark on snags (Scott et al. 1977, Ormsbee and McComb 1998). Species such as wood ducks have more specific requirements and occupy large cavities usually near water (Lowney and Hill 1989).

Cavities may be particularly important roost sites during the winter for species in temperate climates (Haftorn 1988). Energy savings of cavity-roosting species can be significant where ambient temperatures drop below freezing over long winter nights (Weigl and Osgood 1974).

## LOG USERS

Logs are used by many species of vertebrates and invertebrates as cover (e.g., red-backed voles), foraging sites (e.g., shrews and moles), and sites for attracting mates (e.g., ruffed grouse). Logs in streams provide cover for fish and influence the scouring and deposition of sediments in streams thereby increasing stream complexity for many fish species (Bisson et al. 1987). In terrestrial environments, the interior of hollow logs, or the spaces beneath a log, provide a stable and often moist microenvironment that is especially important to the survival of some species of amphibians and reptiles (deMaynadier and Hunter 1995). Other species use the space between the bark and the wood (e.g., scarlet kingsnakes) and some use the interior of well-decayed logs (e.g., clouded salamanders, Stelmock and Harestad 1979).

Log size dictates the area or volume of space available to be occupied (Maser et al. 1979). Logs smaller than 10 cm (4 in.) in diameter are probably of little value to most vertebrates; large logs seem to be used by more species than small ones. Moreover, large logs persist longer than small logs. Decay status also affects log use by organisms. Few species are capable of using undecayed logs (e.g., ruffed grouse, Figure 12.3); most use well-decayed logs (e.g., clouded salamanders and California red-backed voles). Ideally, the habitat requirements of each species must be considered when deciding where logs should be retained and what log characteristics are sufficient to meet their needs. Obviously, with species representing a range of organism sizes from microbes, mites, and tardigrades to salamanders, fishers, and bears, managing the spatial distribution of logs must



**FIGURE 12.3** Ruffed grouse use logs in dense patches of forest as drumming sites where males attract females during the spring. Ermine use hollow logs as den sites. (Photos by Michele Woodford. With permission.)

consider a wide range of home range sizes. Realistically, the needs of most species will probably best be met if large logs are retained in clumps of various sizes ranging in numbers that are representative of the range of conditions one might expect following a natural disturbance (Landres et al. 1999).

There are several key attributes of logs that influence their value to vertebrates: piece size and condition (decay stage), biomass or areal cover, and the successional stage in which it occurs. Piece size can be important to vertebrates for a number of reasons. Large-diameter logs provide more cover per piece than small-diameter logs. Western red-backed voles select large logs as cover (Hayes and Cross 1987), and logs provide cover and a source of fungi for food for southern red-backed voles (Buckmaster et al. 1996). A wide variety of other species also reportedly use logs as cover: shrews, weasels, mink, and northern river otters, among many others (Maser et al. 1981). Further, long logs provide more connectivity across the forest floor than short logs. Connectivity throughout a home range theoretically can influence animal fitness because an individual can remain under cover during movements, thereby reducing the risk of predation while also possibly providing microclimatic advantages to the organism.

The distribution of log sizes in a forest generally reflects the site quality for tree growth, stage of stand development, and sources of mortality that in the past have led to tree death. Trees dying from suppression mortality are typically 50% the diameter (but similar in length) of dominant and codominant trees in the stand (McComb and Lindenmayer 1999). Because large-diameter pieces take longer to fully decay than small-diameter pieces, large piece sizes may last longer as functional habitat for more species. The desired size class distribution for the suite of species being managed in a stand or landscape should be determined by the species requiring the largest piece size.

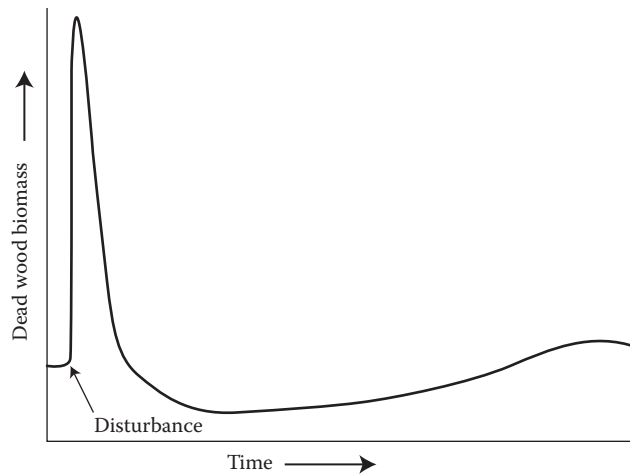
The areal cover or biomass of logs may influence the function of the wood as cover to some mammal and amphibian species (McComb 2003). The physical structure of the log is also important to some species. Maser et al. (1979) described stages of log decay that are similar to that used to describe snag decay stages. Each stage of decomposition can provide different resources to a suite of organisms (Maser and Trappe 1984). Early in the decay process, sloughing bark and infestation by bark beetles, carpenter ants, and termites provide food and cover resources for small mammals, bears, and woodpeckers (Maser and Trappe 1984, Torgersen and Bull 1995). Once the wood has softened and fragmented, vertebrates can begin to excavate the wood to extract insects and/or build nests. Red-backed voles and shrews use very decayed logs as nest sites (Zeiner et al. 1990, Tallmon and Mills 1994) that provide cryptic, dry, and thermally stable environments for their young. Eventually, the structural integrity of the log is so severely compromised by the fungal infection that the log loses value as a potential nest site or feeding site.

Some species such as woodrats, foxes, black bears, skunks, and ermine also use hollow logs as dens (Figure 12.3). Hollow trees form because a column of decay develops following top breakage that extends up and down the bole of the tree from the wound (Shigo 1984). Logs become hollow only after a hollow tree falls to the ground. Recruiting hollow logs into managed stands requires the identification and retention of injured and decaying trees, allowing them to grow to sufficient size or to decay to an acceptable extent, then allowing or promoting their death. Black bears use hollow logs averaging 106 cm (42 in.) in diameter for winter denning in British Columbia (BC) (Davis 1996), so recruitment of potential den sites for bears may take centuries. It is apparent that logs can function as a habitat element for many species in all successional stages of forests in North America.

## PATTERNS OF DEAD WOOD FOLLOWING DISTURBANCE

Two processes contribute to dead wood recruitment in a stand over time: (1) the number of trees dying increases rapidly shortly after stand establishment, then declines in a negative exponential manner through the period of “self-thinning” (Oliver and Larson 1996), and (2) the biomass of dead wood increases immediately after an intense disturbance (unless biomass is removed during logging), declines slowly over time, then recovers as large trees die late in stand development (Spies





**FIGURE 12.4** Generalized pattern of changes in dead wood biomass over time following a natural forest disturbance that creates a pulse of dead wood followed decompositional losses and a slow recovery from tree mortality.

et al. 1988, Figure 12.4). Dead wood biomass accumulates when inputs of dead wood are greater than decomposition losses. Inputs (suppression mortality or exogenous disturbance) and losses (decomposition or fire) interact to produce a “U”-shaped trend in dead wood biomass over time seen in forest types throughout North America (Gore and Patterson 1986, Spies et al. 1988, Van Lear and Waldrop 1994, D’Amato 2007) (Figure 12.4).

Natural old forests contain high volumes of large pieces of dead wood, but not to the level found following intense disturbances such as fires or hurricanes. Infrequent but severe disturbances create pulses of dead wood (Spies et al. 1988). High levels of dead wood produced following a disturbance also may represent a fuel source for subsequent fires in fire-prone systems (Spies et al. 1988). Fear of recurring fire led to salvage logging and snag removal several decades ago in the Pacific Northwest of the United States (McWilliams 1940). Now managers often try to recruit dead wood to stands that were salvaged in past years.

### CHANGES IN DEAD WOOD OVER TIME

Dead wood changes over time through decomposition (Miller and Miller 1980). When a tree dies, fungal decay usually begins. Fungal decay facilitates wood fragmentation when combined with the activities of invertebrates, such as termites (Atkinson et al. 1992). Tree mortality and wood decomposition rates interact to dictate dead wood biomass on a site. The size and species composition of the live trees influence the potential dead wood production on the site. Hardwood forests generally have less dead wood than conifer forests (Harmon et al. 1986, Harmon and Hua 1991). Eastern hardwood forests may support 11–220 m<sup>3</sup>/ha of dead wood (Tyrrell and Crow 1994, D’Amato 2007), but western coniferous forests may have 376–1421 m<sup>3</sup>/ha of dead wood (Huff 1984). Variability in the amounts and distribution of both standing and fallen dead wood is considerable (Everett et al. 1999). Indeed, managing dead wood to reflect variability among sites over a landscape may be a more meaningful approach than mandating a minimum retention level in managed stands or trying to manage for individual species (Everett et al. 1999).

The size of the dead wood influences the rate of decomposition and its value to organisms. Large pieces of dead wood provide habitat for a large number of species in various seral stages. These large remnant snags and logs can last for centuries before becoming an unrecognizable part of the forest humus (Tyrrell and Crow 1994). Trees that die and remain standing provide habitat as snags.



**TABLE 12.1**

**Comparison of Decay Constants (*k*) among Tree Species in Various Parts of North America (Listed from Slowest to Fastest Decay Rates)**

<b>Taxon</b>	<b>Location</b>	<b><i>k</i> (per Year)<sup>a</sup></b>	<b>Citation</b>
Douglas-fir	Oregon	0.005–0.10	Harmon and Hua (1991)
Douglas-fir	Oregon	0.0063	Means et al. (1985)
Balsam fir	New Hampshire	0.011	Lambert et al. (1980)
Western hemlock	Oregon	0.012	Grier (1978)
Western hemlock	Oregon	0.016–0.018	Harmon and Hua (1991)
Mixed oaks	Indiana	0.018	MacMillan (1988)
Western hemlock	Oregon	0.021	Graham (1982)
Eastern hemlock	Wisconsin	0.021	Tyrrell and Crow (1994)
Red spruce	New Hampshire	0.033	Foster and Lang (1982)
Jack pine	Minnesota	0.042	Alban and Pastor (1993)
Mixed maples	Indiana	0.045	MacMillan (1988)
Red pine	Minnesota	0.055	Alban and Pastor (1993)
White spruce	Minnesota	0.071	Alban and Pastor (1993)
Trembling aspen	Minnesota	0.080	Alban and Pastor (1993)
Mixed hardwoods	New Hampshire	0.096	Arthur et al. (1993)
Mixed hardwoods	Tennessee	0.110	Onega and Eickmeier (1991)

<sup>a</sup>  $k$  = a decay rate constant when calculating decay rates as  $D_t = D_0 e^{-kt}$  where  $D$  = wood density,  $t$  = time (years).

Fall rates of live trees and snags vary among tree species (McComb and Lindenmayer 1999). Ten-year fall rates (the proportion of trees expected to fall in a 10-year period) for pine and fir snags in the western United States and many hardwoods in the eastern United States exceed 50% (Morrison and Raphael 1993, Wilson and McComb 2005). Fall rates of large-diameter Douglas-fir snags may be <20% per decade (Cline et al. 1980).

The combination of a tree's size and the variability among species in their resistance to decay leads to considerable variation among trees in rate of decay and fragmentation (Harmon et al. 1986). As fragmentation of the tree bole advances, the diameter and height of snags (or length of logs) decreases (Figure 12.2). Tree species and size also influence other characteristics of dead and dying trees. Decomposition rates are generally described using decay rate constants (Olson 1963):

$D_t = D_0 e^{-kt}$  where  $D$  = wood density,  $t$  = time (years), and  $k$  = a decay rate constant. Decay rates vary among species, with conifers generally being more decay resistant than hardwoods (Harmon and Hua 1991, Table 12.1). As decay proceeds within a bole of wood, the bole becomes subject to fragmentation (Harmon et al. 1986, Tyrrell and Crow 1994). Consequently, the dead wood biomass on a site at any one time will be dependent on a number of factors. These include site quality and tree species composition, the disturbance regime for the site, and the climatic factors that influence tree growth and decomposition (Muller and Liu 1991).

## DEAD WOOD DURING STAND DEVELOPMENT

Stand establishment following a disturbance often results in over 1000 tree seedlings per hectare in some western U.S. coniferous stands and over 40,000 seedlings per ha in eastern hardwood stands. In plantations, stand density is controlled. In both cases, as stands develop, inter-tree competition results in mortality among those trees that are intolerant of shade or drought (Oliver and Larson 1996). It is common to see over 90% of the trees in a stand die during the first few decades following natural stand establishment. High-density stands produce many small dead stems early in stand

development. Competition mortality commences later in stand development in lower density stands, allowing trees to grow rapidly for many years prior to competition (Figure 8.5A). Thinning that reduces stand density may benefit the production of dead wood of large sizes later in stand development. Manipulating stand density allows the manager to influence the size and numbers of dead trees throughout even-aged stand development, and manipulating stocking rates in uneven-aged stands can produce similar results.

## MANAGEMENT OF TREE CAVITIES AND DEAD WOOD

There are two general approaches to dead wood and tree-cavity management, and they represent two complementary philosophies. The first is based on the concept of a historical range of variability (HRV) or the range conditions produced through natural disturbances over an area (Figure 12.5). So a manager might ask, “Do the levels of dead wood biomass, piece size, and condition over large areas fall within the range of conditions to which the species should be adapted, the range that might be represented under natural disturbances (HRV)?” If the answer is “no,” then you might question which species (if any) might be at risk based on this departure from the HRV, and if management actions should be taken to address these risks. If impacted species and processes are adequately addressed elsewhere in the landscape, then allowing some stands or landscapes to fall outside the HRV may be an acceptable risk. If, however, the addition of another stand or landscape to areas that already fall outside the HRV means that there is a likelihood of cumulative risks on species or processes over space and time, then the manager may wish to take actions that contribute to goals related to the HRV (Table 12.2, Landres et al. 1999).

Alternatively, the manager can assess functional relationships between animals and dead wood and manage for these conditions as part of a desired future condition (McComb and Lindenmayer 1999, Mellen et al. 2005). These functional relationships are not clear for most species, but they can be hypothesized and tested in an adaptive management approach. Indeed, some habitat relationships models already include estimates of the abundance of dead wood as a component contributing to habitat quality for a species (e.g., Allen 1983). The compilations of these relationships for desired species in future landscapes would dictate the dead wood goals (Mellen et al. 2005) (Figures 12.6 and 12.7).

Regardless of the dead wood management approach chosen, managers should identify high-priority sites for dead wood management. Intensively managed plantations might fall within this group because they have dead wood and tree-cavity levels that often fall outside of the HRV (Butts and McComb 2000). Modest inputs of dead wood to these stands may make a greater impact on



**FIGURE 12.5** Expected range of variability of old-growth forests in the Oregon Coast Range over the Oregon Coast Range. As seral stages change in abundance over time, so do patterns of dead wood associated with them. (Wimberly, M.C. et al. Simulating the historical variability in the amount of old forests in the Oregon Coast Range. *Conservation Biology*. 2000. 14:167–180. Copyright Wiley-VCH Verlag GmbH & Co. KGaA. Reproduced with permission.)

**TABLE 12.2**  
**Likely Ranges of Dead Wood (CWD) among Variability among Forest Age Classes in the Oregon Coast Range Under the Historical Range of Variability**

Age Class (years)	Coast Range (%)	Coast Range (ha × 100,000)	Number of Patches	CWD Range (m³/ha)
0–30	4–11	0.9–2.5	1–4	376–1421 <sup>a</sup>
31–80	6–19	1.4–4.3	1–6	163–305 <sup>b</sup>
80–200	15–45	3.4–10.1	2–14	93–165 <sup>b</sup>
>200	25–75	5.6–16.9	4–24	219–324 <sup>b</sup>

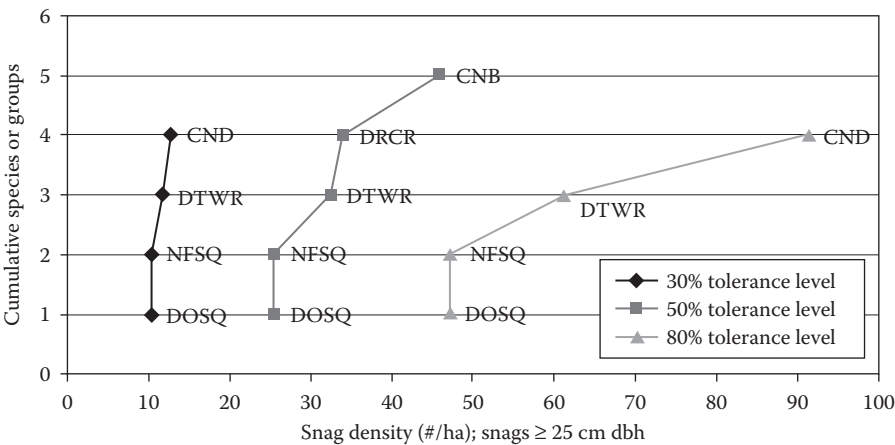
Source: Based on Wimberly et al. 2000. *Conservation Biology* 14:167–180; McComb, W.C. 2003. *Mammal Community Dynamics: Management and Conservation in the Coniferous Forests of Western North America*. Pages 567–586. Cambridge University Press, Cambridge, England.

<sup>a</sup> From Huff (1984).  
<sup>b</sup> From Spies and Franklin (1991).

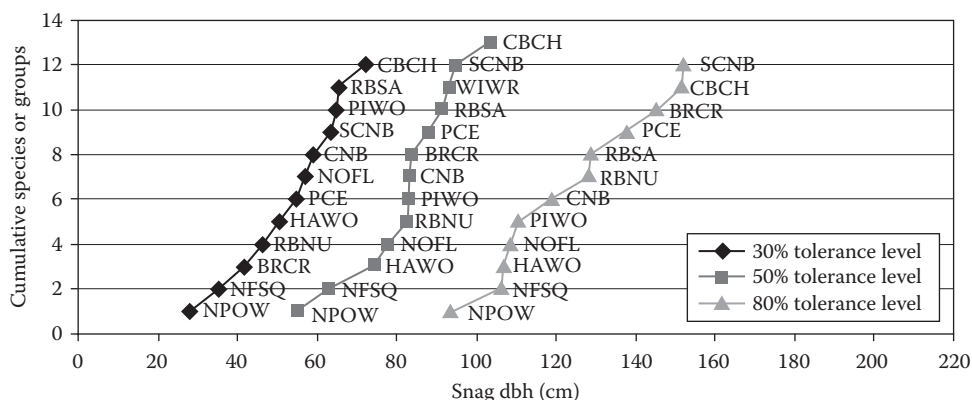
animal habitat and/or ecological processes than a similar treatment in stands that already contain dead wood.

The steps in the management process that I recommend are

1. Inventory dead wood at the desired scale at time 0 (see Harmon and Sexton (1996) and Bull et al. (1990) for inventory techniques). The chosen spatial scale should be biologically meaningful to the species of interest.
2. Compare dead wood levels to the HRV estimates for the region and/or compare estimates to your habitat goals for species. In western U.S. forests, DECAID can provide a useful reference for comparison (Mellen et al. 2005) (Figures 12.6 and 12.7).



**FIGURE 12.6** Cumulative species curves of snags (number per hectare) supporting species use of areas for nesting, roosting, and occurrence for 30%, 50%, and 80% tolerance levels, Westside Lowland Conifer-Hardwood habitat type in the small tree structural class. (From Mellen, K. et al. 2005. *DecaID, the Decayed Wood Advisor for Managing Snags, Partially Dead Trees, and Down Wood for Biodiversity in Forests of Washington and Oregon*. Version 2.0. USDA For. Serv. Pac. Northwest Res. Sta. and USDI Fish and Wildl. Serv., Oregon State Office, Portland, OR.) CNB = cavity-nesting birds.



**FIGURE 12.7** Cumulative species curves of snag dbh (cm) supporting species use of areas for nesting and denning for 30%, 50%, and 80% tolerance levels, Westside Lowland Conifer–Hardwood habitat type in the small tree structural class. (From Mellen, K. et al. 2005. *DecaAID, the Decayed Wood Advisor for Managing Snags, Partially Dead Trees, and Down Wood for Biodiversity in Forests of Washington and Oregon*. Version 2.0. USDA For. Serv. Pac. Northwest Res. Sta. and USDI Fish and Wildl. Serv., Oregon State Office, Portland, OR.) CNB = cavity-nesting birds.

3. Conduct this analysis across your planning area and prioritize stands for dead wood management based on the risk of not meeting future dead wood goals.
4. Beginning with the highest priority stands, determine if there are trees of sufficient size that could be felled or killed now to fulfill the snag or log goals or that could be retained into the future to replace those snags that fall over time.
5. If trees in the current stand are not appropriate for meeting dead wood goals, then silvicultural actions should be considered to achieve goals. Thinning from below to allow dominant and codominant trees to grow more rapidly may be preferable to allowing an overstocked stand to grow slowly and to contribute small amounts of dead wood to the stand.
6. Monitor species of highest concern prior to and following active management and assess population changes. Given the long-term nature of wood decay and the habitat functions that develop throughout decay processes, monitoring may need to occur periodically for decades.
7. Assess monitoring results and decide if changes should be made to the dead wood goals for the area.

Many states and provinces have either regulations, or standards (you shall do them) and guidelines (you should do them) regarding dead wood retention levels. Several factors come into play when regulating dead wood levels in managed stands. First, the minimum level of the range chosen for regulation is usually the level that managers will strive to retain in stands. Providing one dead wood level in all managed stands homogenizes that condition over managed landscapes. Although current dead wood guidelines could be rewritten to ensure that dead wood levels fall *within* the HRV, it is much more difficult to develop regulations that will lead to dead wood levels that *represent* the HRV for the region. Incentives such as dead wood credits provided to landowners by local, state, or federal agencies may allow better representation of the HRV in dead wood conditions across landscapes than mandating it by law (McComb 2003).

Clearly, such management actions will require a commitment of time and money to providing dead wood. Costs can be modest if management is for one or a few species, but much higher if dead wood is managed to represent goals for multiple species or the HRV. Dead wood guidelines should be scale dependent, however. Dead wood biomass among many stands should collectively

contribute to landscape goals. Landscapes should also represent variability in dead wood levels, but collectively contribute to regional goals.

A delay in dead wood management in a stand with low levels of dead wood now may result in a gap in dead wood availability in the future. Certainly a few stands with low dead wood levels in an area with otherwise high levels may be relatively unimportant, unless overall dead wood levels decline over time and no action is taken now to ensure that advanced decay class (class 5) logs will occur in the stands 50 years from now.

### LIVE CAVITY-TREE MANAGEMENT IN MANAGED STANDS

Managing forests to achieve goals for secondary cavity nesters can partially be achieved by managing habitat for primary cavity nesters, but not entirely (Figure 12.8). Clearly, there are some secondary cavity users that are larger in body mass than the largest primary cavity nesters. There are more species of secondary cavity nesters than primary cavity nesters. Based on nest box studies, they can occur at much higher densities than primary cavity nesters. Providing natural cavities can be an important supplement to the cavities created by primary cavity nesters.

Nest boxes are one alternative to providing natural cavities, but nest boxes are expensive to build and maintain and they are likely to last only a fraction of the time that a natural cavity would last in a live tree (McComb and Lindenmayer 1999). Nonetheless, nest box programs have been very successful for some species such as bluebirds and wood ducks. Nest boxes are widely used to increase nesting and roosting site availability for a number of species, and the proportion of nest boxes used by animals can be higher than use of natural cavities for many species (McComb and Noble 1981a). However, maintenance costs for nest boxes are high, microclimates are less stable than natural cavities (McComb and Noble 1981b), and primary cavity nesters rarely use them unless they are filled with a substance that can be excavated. Nest boxes should only be considered a temporary solution to a shortage of nest cavities and one that can only be used in a relatively small area for a small number of species.

Managing natural cavity abundance in forests is a bit more challenging than managing dead wood because they are more difficult to inventory and the rate of gain and loss in a forest is very slow, and somewhat unpredictable. Estimating cavity abundance is difficult. Sampling trees for



**FIGURE 12.8** Dead limbs on live hardwoods and cavities in live hardwoods both contribute to cavity resources for secondary cavity nesters. Hence providing some hardwoods in conifer stands can add to snags as a source of cavities for cavity-nesting species. Natural cavities also provide nest sites for species such as barred owls (owlet shown here), which cannot use cavities created by primary cavity nesters.





**FIGURE 12.9** Leaving snags and logs must be balanced with worker safety and fuels management in managed stands to meet the needs for species such as woodpeckers (nest cavities on left) and alligator lizards (right).

cavities often is complicated by inadequate access to or visibility of cavities in standing trees. Cavities judged to be suitable from the ground may not be useable by a given species (Healy et al. 1989). Typically, sampling for cavities is conducted during the leafless period in temperate climates if hardwoods are present in the stand. The size and number of plots used to sample for cavities will be largely a function of the density and among-plot variability in cavity density. To adequately predict the prevalence of trees with cavities, a very large number of plots may be required to sample cavity abundance (Healy et al. 1989). DeGraaf and Shigo (1985) provided guidelines for managing natural cavities in eastern U.S. forests.

Predicting cavity availability in a stand from tree size and species information is even more problematic. Cavity occurrence in a tree is a function of tree size and tree age, as well as the often highly stochastic disturbance factors that initiate cavity formation. Nonetheless, it seems that there are relationships that can be developed for some hardwood species in North America (McComb et al. 1986, Allen and Corn 1990). In general, large-diameter trees with some past injury are more cavity-prone than small-diameter trees that lack obvious signs of past injury (Figure 12.9). Assumptions made regarding the processes of cavity formation, such as the continued role of insects and fire, must be monitored carefully throughout prescription development and implementation in order to ensure that cavities will be available over time in a stand.

## DEAD WOOD RETENTION AND HARVEST SYSTEM CONSIDERATIONS

Due to the logistics of harvesting around dead and green trees reserved from harvest, snags and replacement green trees often are left in clumps between cable corridors or between skid trails, and soft snags are left opportunistically between the clumps. But clumping snags can have adverse effects on snag use. Location matters. In clumps, a territorial individual can exclude other individuals of the same species from a clump. If the same number of snags were distributed at a spacing consistent with the territory size of the species being managed, then snag use can be optimized. Where human safety issues occur, then some balance must be achieved between the optimum distribution for animal use of snags and reducing risk to forest workers. In the United States, harvest operations must be coordinated with retention of snags, logs, and cavity trees to avoid interference with harvest systems (e.g., skid trails and cable corridors) and to ensure worker safety during the operations (Hope and McComb 1994) (Figure 12.9). In the United States, the Occupational Health and Safety Administration (OSHA) places restrictions on loggers working around dead or dying trees limiting options when managing dead wood in stands.

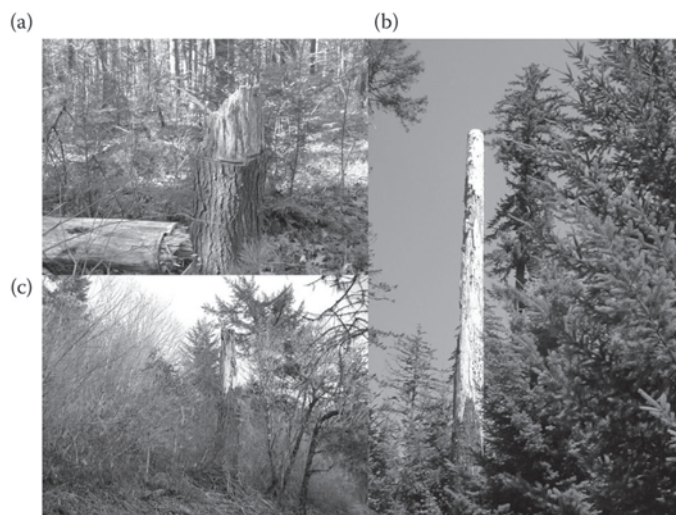


### CREATING SNAGS AND LOGS FOR WILDLIFE

The goals for dead wood abundance in a stand should be compared with the levels of dead and dying trees predicted to occur in the stand over time. These estimates can be developed using a forest growth model that includes a tree mortality function (e.g., Forest Vegetation Simulator (FVS), Dixon 2003). Once we know how many trees are likely to die each decade and what size they are likely to be, then we can predict additions of dead wood over time. We can also estimate dead wood loss through decomposition (Table 12.1) or snag fall rates. If the predicted recruitment of dead wood does not balance the losses to meet or maintain dead wood goals, then some trees can be killed to meet the goals.

The process for deciding which trees to retain as replacement snags during management activities have largely been driven by tree species, tree size, and costs associated with forgoing timber value (Washington DNR et al. 1992). Generally, large trees with some timber defect have the potential to provide tree cavities and dead and dying wood (Healy et al. 1989). In intensively managed stands, defective or diseased trees may be thinned early in stand development. In these stands, dominant and codominant trees may provide habitat for cavity-using species early in the rotation if some of these large trees are retained and killed (Bull and Partridge 1986) or injected with fungal spores (Parks et al. 1995). Indeed, thinning can accelerate tree diameter growth tremendously in some forest types, providing an opportunity to kill some large trees much sooner than would occur in the absence of management.

There is a range of methods available for killing trees to produce snags or cavity trees for vertebrates (Bull and Partridge 1986). Topping the trees with a chain saw or explosives is effective for both Douglas-fir and ponderosa pine (Bull and Partridge 1986; Chambers et al. 1997, Figure 12.10). Herbicides also have been shown to be an effective method for killing trees that are then used by primary cavity nesters (McComb and Rumsey 1983). Girdling, although potentially effective, may be less cost effective than other techniques simply because trees often break at the point of girdling, creating short snags of limited value to some species (Figure 12.10). Hardwoods have been killed to increase invertebrate food resources for woodpeckers in Europe (Aulen 1991), but live hardwoods may be used by more species for a longer period of time than dead hardwoods. Killing trees as habitat management for selected vertebrates must be done based on needs for primary cavity excavators



**FIGURE 12.10** Snags created by girdling often will break at the point where the girdling occurred (a), whereas topping trees create a longer lasting snag (b), and one more typical of a snag that develops following natural death and decay (c).

and the potential for subsequent use of these cavities by secondary users. Generally, killing trees as a remedial measure is most appropriate in managed conifer forests.

Other techniques are available, but rarely used. Wood-decaying fungi have been experimentally injected into live trees to create a pocket of rot that can be excavated by cavity nesters at some later date (Parks et al. 1995). Artificial cavities also have been created by excavating holes in live trees in eastern hardwood forests (Carey and Sanderson 1981), and cavity inserts have been used to create artificial nest sites for red-cockaded woodpeckers in pine trees without heart rot.

## MONITORING CAVITY TREES, SNAGS, AND LOGS

Most goals for dead wood management in managed forests are based on a number of assumptions. These include, but are not limited to: estimates of the number of snags required by each individual or breeding pair; distribution of trees, snags, and logs within territories; estimates of fall rates and decay rates of snags; and persistence of populations that may become isolated over time. Monitoring of management effectiveness becomes a key part of the management process, especially given the uncertainties associated with requirements for each species, stand projection estimates, and estimates of snag decay and fall rates. Effective management of dead wood habitat will require consideration of not only the primary cavity nesters (Neitro et al. 1985), but also foraging and nesting sites for those secondary cavity nesters that do not use nest sites abandoned by the primary cavity nesters (e.g., bats, wood ducks, and some invertebrates, Figure 12.8). Secondary cavity nesters are generally dependent on the activities of primary cavity nesters and on cavities formed by wood decay processes. Consequently, secondary cavity nesters may be better candidates to monitor the effects of forest management on dead-wood-dependent species.

## CASE STUDY: MANAGING DEAD WOOD IN OREGON FORESTS

To illustrate the process of managing dead wood in a managed stand and the effects of biofuels management on dead-wood-dependent species, let us consider a 100-year-old stand in the Oregon Coast Range.

1. *Inventory dead wood at the desired scale now.* The stand is fully stocked at 75 m<sup>2</sup> of basal area per ha (250 ft<sup>2</sup>/acre) and is dominated by Douglas-fir with minor components of grand fir, western hemlock, bigleaf maple, and red alder. A stand exam (a systematic or random sample of trees and habitat elements in the stand) revealed an estimate of 10 snags/ha (4 snags/acre) >76 cm dbh (diameter at breast height) (30 in. dbh) and 7/ha (3/acre) are larger than 80 cm dbh (32 in. dbh). The remainder of dead trees in the stand (12/ha; 5/acre) are <5 cm dbh (2 in. dbh). There are 106 trees/ha (43 trees/acre) >76 cm dbh (30 in. dbh).
2. *Compare dead wood levels to your goals.* Using the Dead Wood Adviser (DECAID) developed by Mellen et al. (2005), we chose to manage for snag levels that represent a 50% tolerance level, or a likelihood of providing ecosystem functions intermediate between management providing primarily ecosystem function goals (80% tolerance level) and providing primarily timber production (30% tolerance level). DECAID uses empirical relations from dead wood–species relationships to develop these curves (Figures 12.6 and 12.7). Since we intend to use a clearcut regeneration system with legacy to regenerate the stand yet provide habitat elements, we selected the early successional condition of the Westside Lowland Conifer Hardwood habitat type to best represent the stand that will result from our management (Mellen et al. 2005). Our species goal is to manage to provide snags at a level that will meet the needs for CNB as a group at the 50% tolerance level, or approximately 42 snags/ha >25 cm dbh (17/acre >10 in. dbh) (Figure 12.6). But not all cavity nesters can use such small snags; so we also need to set snag size goals. DECAID indicates that 80-cm dbh snags (32 in. dbh) are needed to meet the size goals (Figure 12.7); so at least some of the

- snags in the stand should be greater than this diameter. Hence we want >42 snags/ha that are >25 cm dbh, and as many as possible of these should be >80 cm dbh.
3. *Prioritize stands for dead wood management based on the risk of not meeting future dead wood or species goals.* Since there are only 10 snags/ha (4 snags/acre) in the stand now and we want to have 42 snags/ha (17/acre) following harvest, this stand becomes a high priority stand for increasing dead wood availability.
  4. *Determine if there are trees of sufficient size that could be felled or killed now to fulfill the snag or log goals.* Since we have 106 trees/ha >76 cm dbh then there are sufficient live trees that can be killed to provide these snags.
  5. If trees in the current stand are not appropriate for meeting dead wood goals, then silvicultural actions should be considered to achieve goals, including retaining trees for future snags, and killing trees to create snags and logs. We know that we will need more than the simply 43 snags/ha (17/acre) goal because some snags will fall during the future development of the stand. But how many more? We may choose to mark and retain 50 trees/ha (20/acre) of which we will kill 32/ha to supplement the 10/ha that are on the site now to meet our goal. The remaining reserved trees (18/ha) are retained as live trees as a future source of dead wood, if needed later in stand development. The stand is then harvested and, after harvest, 32 of the 50 retained trees per ha (13 of the 20 retained trees per acre) are killed by topping the tree (cost = \$50/tree, or \$1600/ha; \$650/acre). The tops will be left on the site to add to the log availability. Because the largest trees in the stand were retained as logs for species requiring dead wood on the forest floor, 108 MBF/ha (44 MBF/acre) were harvested and 141 MBF/ha (57 MBF/acre) were retained and killed. If Douglas-fir sold for \$500/MBF, then the gross timber receipts would be \$53,340/ha (\$22,000/acre); \$69,160/ha (\$28,000/acre) would have been allocated as timber value foregone to create dead wood.
  6. *Monitor species of highest concern prior to and following active management and assess if populations decline.* Following harvest and snag creation, we will monitor snag fall rates and populations of CNB every 5 years until the stand moves into another vegetative structural condition (Mellen et al. 2005). Projections of snag loss using the Snag Recruitment Simulator (SRS, Marcot 1992) suggests that of the 42 snags/ha (17 snags/acre) available immediately after harvest, there would be 37/ha (15/acre) available after 10 years, 35/ha (14/acre) after 20 years, 20/ha (8/acre) after 30 years, 10/ha (4/acre) after 40 years, and 5/ha (2/acre) after 50 years due to snag decomposition, decay, and subsequent fall. Of course these fallen snags add to the log biomass available for other species, but substrates for cavity nesters would decline considerably during the first 50 years of stand development due to snag fall. Monitoring data collected over the 50 years would allow the managers over that time to assess if these projections were correct and if additional trees should be killed.
  7. *Assess monitoring results and decide if changes should be made to the dead wood goals for the area.* Recall that we retained 50 trees/ha (20/acre) and only killed 32/ha (13/acre) of them after harvest. So there are still 18 trees/ha (7 trees/acre) carried into the new stand that have grown for 50 years and that could be killed if monitoring indicated that more snags were needed. These retention trees represent the insurance policy for dead wood in the stand so that future goals can be met as the created snags fall over (Figure 12.10).

Approaches such as this that also consider the cavity tree resources and logs on the forest floor can be used to help ensure that the species and ecological processes associated with dead and decaying wood are maintained in stands and across landscapes. But it should be apparent that these activities come at a financial cost, sometimes a significant cost, to a landowner. On public lands, where timber profits are not a goal, such a dead wood recruitment and maintenance strategy is clearly feasible, but on private lands different goals and approaches may be necessary.

## SUMMARY

Forest management activities that influence the frequency, severity, and pattern of disturbances in forest systems can have marked effects on the abundance of cavities and dead and dying trees in the system. Dead and dying trees function differently in each stage of forest succession and the trees themselves progress through a succession of decay stages. Decay stages provide opportunities for use by vertebrates, with a different suite of species selecting each decay stage. If forest managers wish to maintain these functions in managed forests, then they must actively manage the dead wood resource.

Live trees with decay are especially important to animals in hardwood forests. Standing and fallen dead trees are particularly important in conifer forests. Integration of management of dead, dying, and decayed trees in forest management will be key in any management prescription designed to balance biodiversity conservation with commodity production. Delay in initiating active management can have long-term implications because of the time needed to both recruit large trees and for the large wood to decay to a stage suitable for certain organisms. There are seven steps to managing habitat for species that depend on cavities, snags, or logs.

1. Inventory dead wood at the desired scale now.
2. Compare dead wood levels to your goals.
3. Prioritize stands for dead wood management based on the risk of not meeting future dead wood or species goals.
4. Determine if there are trees of sufficient size that could be felled or killed now to fulfill the snag or log goals.
5. If trees in the current stand are not appropriate for meeting dead wood goals, then silvicultural actions should be considered to achieve goals, including retaining trees for future snags, and killing trees to create snags and logs.
6. Monitor species of highest concern prior to and following active management and assess if populations decline.
7. Assess monitoring results and decide if changes should be made to the dead wood goals for the area.

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# 13 Managing Fire in Forests

Smoky Bear says, “Only YOU can prevent Forest Fires!” But, is Smoky correct? Should you prevent forest fires? Let them burn? Prescribe fire for your forest? The debate over control of fire in forests is, pardon the pun, indeed hot. Fires have occurred in many forest ecosystems for millennia, usually started by lightning. Humans have used fire in forests for millennia as well, to encourage berry and other fruit and vegetable production, to move game animals so that they were easier to kill, and presumably to make movement through forests easier. Now more and more people are building their homes in forests, each wanting to live in a place where they can enjoy the solitude and beauty of forests around them each day, so fire becomes a very significant risk to life and to homes. As our population grows and our economy grows, the expansion of the urban forest fringe increases. We need merely look to examples in Oakland, California; Sydney, New South Wales, Australia; and the Front Range of the Rocky Mountains in Colorado to see examples of fires that have led to human death and property destruction. So, is Smoky correct? Should we prevent forest fires? Perhaps in some places, but forests have burned and will burn again when there is sufficient fuel, the weather is correct, and there is a source of ignition. Why? Despite Smoky’s best efforts the forest area burned in the United States has increased steadily over the years (Stephens and Ruth 2005).

There are several conditions that interact to increase not only the probability of a fire occurring but also the size of a fire once it starts. First, conditions that cause lightning vary from year to year, but on average probably have not changed significantly over the past 60 years (Stephens 2005). As indicated earlier, the number of people living in and around forests has increased and human-caused ignitions, either by arson or by accidents have increased substantially in the western and southeastern United States (Stephens 2005). Second, because Smoky has been somewhat effective in keeping fires out of many forests, fuels have begun to build up in some forest systems. Without periodic fire in many dry forests, tree densities increase, some trees experience greater moisture stress due to competition for water, and we can see tree death through competition mortality or through insect outbreaks related to tree stress. All of these activities add fuel to the forest, so that when conditions are dry and an ignition source is provided (lightning, a match, a spark from a chainsaw), then the fire can spread rapidly (especially if there is any wind) and extend over huge areas very quickly. In addition, because the fuel loads are now often higher than they have ever been historically, the severity, or effect on the ecosystem, of the fires can be unusually high (Miller et al. 2009). The best conditions for fire spread are hot, dry, windy weather. As climate change produces more variable and extreme weather patterns, we can expect to see more extreme conditions that support the spread of fire in some (but not all) forests. Indeed, Westerling et al. (2006) found that there is a relationship between earlier and warmer spring and summer temperatures, and the frequency of large wildfires. So what can we do? Fuel loads have increased in many places, climate is changing, more people are providing more sources of ignition, and epidemics of insects are killing more and more trees. We cannot let fires burn uncontrolled everywhere, but we cannot control many wildfires under current conditions once they begin. Given the uncertainty associated with what future conditions might be, strategically located active management of some forests to increase their resistance or resilience to fire may be our only reasonable approach to managing this situation (Millar et al. 2007).

## EFFECTS OF FIRE ON HABITAT ELEMENTS AND SUCCESSION

Is fire beneficial or destructive to habitat elements important to forest dwelling vertebrates? The answer of course is “Yes,” but of course it depends on the species. The severity of the fire as

indicated by the destruction of forest biomass, such as trees, snags, logs, leaf litter, foliage, shrubs, and herbs, can have a significant effect on the structure and composition of habitat elements. A very intense fire moving slowly through an area with high levels of fuel can remove nearly all of the fine fuels and some of the large fuels and leave the forest floor barren of any leaf litter. On the other hand, a rapidly moving ground fire in which fuel moisture levels are high may kill some small plants and seedlings of trees and shrubs and leave an ash layer on top of the leaf litter, but otherwise not substantially change the structure or composition of the stand. Both of these conditions may occur in the same fire, the former more likely on an exposed southwest-facing slope and the latter along a riparian area. The range of fire intensities, severities, frequencies, and sizes influence the successional pathways throughout the burned area.

### **SPECIES, FUELS, FIRE FREQUENCY, AND SEVERITY**

Some species of trees are adapted to fire. Species such as jack pine and lodgepole pine have serotinous cones that open following a fire, releasing their seeds onto the ash layer that was created by the fire, and the germinating seeds produce another stand of these species. Longleaf pine seedlings persist in a "grass" stage for several years, developing deep root systems while they survive ground fires before erupting in rapid height growth and developing thick bark to protect the cambium from subsequent fires. Similarly, mature ponderosa pine and Douglas-fir may have bark that is 5–10 cm thick, thereby insulating the cambium from fire effects. Many hardwood species such as mountain laurel, manzanita, eucalyptus, and other species that have flammable foliage are able to sprout back and dominate a site following a fire. A fire of a certain intensity (energy per unit area per unit time) may completely kill nearly all trees (e.g., lodgepole pine), the aboveground portion of trees (e.g., oaks), or no mature trees (e.g., ponderosa pine), resulting in different levels of severity.

Fuel loads influence the effects of fire on trees in a stand. In many eastern deciduous forests, leaf litter accumulates for decades. Litter depth on the uphill side of trees where it is trapped by the tree bole can be much higher than elsewhere on the hill slope. Consequently, when a ground fire occurs, fire burns more intensively and for a longer period in the collected leaf litter, resulting in fire scars and subsequently basal hollows on the uphill side of many trees. Taken to a greater extreme, where insects have killed trees in a stand and there are both fine fuels (e.g., twigs) and large fuels (tree boles), fire can be hotter and last longer, having a more profound effect on the system.

### **CREATION AND LOSS OF DEAD WOOD**

Depending on the intensity, a fire can kill trees and create snags and fallen dead wood, and it can simultaneously consume dead wood as the stand burns. The net effects of a fire, dead wood created minus dead wood lost, is critical to understanding the long-term dynamics of dead wood in forests subjected to fires.

Wildfire creates a pulse of snags, and then snag density declines as snags fall and they are not recruited into the stand until large trees die as the stand matures (Passovoy and Fulé 2006). Following a fire there is an economic incentive for the landowner to harvest any trees killed by the fire through salvage logging. In western coniferous stands, Monsanto and Agee (2008) reported that stands that had been salvage logged had lower log biomass than unsalvaged units, except for the most recently burned site, where salvaged stands had higher log biomass; moist sites had higher log biomass than dry sites (Monsanto and Agee 2008).

The effects of fire on dead wood in managed stands following prescribed burning can be quite different than following a wildfire. Horton and Mannan (1988) reported a 45% net decrease in snags during the first year following a prescribed fire, and they also reported a 42% reduction in the number and volume of ponderosa pine logs following the prescribed fire. With regularly occurring prescribed burning, we would expect that the recruitment and consumption of dead wood would begin to fluctuate around a mean dead wood volume, but it is not clear where that level is for many forests.

## EFFECTS ON SPECIES COMPOSITION

As fire changes in frequency or intensity, the species of plants and animals that can occupy a site will change as well. Following an intense wildfire, plant species adapted to colonizing new sites will do well, as will those that can resprout following the fire. Because some species present before the fire can persist through resprouting or seed germination and a new set of species colonizes the site, species richness and abundance of all vascular plant species tend to be higher in burned than nearby unburned areas (Crawford et al. 2001). The more intense the fire, the more different the plant community is following the fire (Morrison 2002). Unfortunately, exotic species, including invasive species, may be more likely to dominate a site if the fire is intense (Crawford et al. 2001).

Plant communities can also be affected by the size and shape of a burned area. Plant species composition can exhibit a significant change across burn edges that can last for 30 years or more in some forests (Coop et al. 2010). Similarly, tree species that establish following a burn can change with elevation and distance from the edge of the burn to the interior of the burn, but overall species richness can be highest at the center of a burned area because of the increased diversity in grasses and forbs (Coop et al. 2010).

Changes in climate may influence the types of changes that we see in plant communities both through direct effects on temperature and precipitation in a region but also through the effects that the changed climate has on fire frequency and intensity. Cushman et al. (2011) suggested that climate-driven changes to disturbance regimes may have larger effects than fire suppression or fuel reduction treatments. If the suggestion by Cushman et al. (2011) is correct, then management activities may not have much of an effect on the trajectory of landscape change. Because of the overwhelming changes to landscapes from fires, changes to habitat structure and composition for selected species may be much more affected by climate-driven changes in fire characteristics than by local management activities (Cushman et al. 2011). Nonetheless, management of fuel is becoming widespread and could have short-term effects on some species through alteration of vegetation structure and composition and reduction in large fuels (i.e., dead wood).

## CHANGING FIRE RISK THROUGH MANAGEMENT

Although Cushman et al. (2011) make a strong case for climate-driven changes in vegetation communities and future fire regimes, there are many local efforts that are designed to mitigate adverse fire effects before a fire occurs. In general, these treatments are designed to remove fine and coarse fuels in an economical manner to reduce the risk of fire occurrence, or lessen fire intensity. Following a fire, salvage logging has long been proposed as a mechanism for reducing the risk of a reburn of the same site by removing fuels. There is a considerable controversy over the fact that indeed salvage logging may actually increase the likelihood of a reburn (Thompson et al. 2007). Although driven largely by economic returns, salvage logging has occurred for decades in an effort to capture the economic value of fire-killed trees before the wood begins to decay. Conceivably this action would reduce fuels that would otherwise contribute to a reburn of the area, but in fact some areas that are salvage logged reburn at a higher intensity than areas that were not salvage logged (Thompson et al. 2007). More recently silvicultural activities have focused well on reducing fuels in stands prior to a fire. By strategically reducing fuels and then, where appropriate, reintroducing prescribed fire, the risk of ignition or intense fire may be reduced, at least temporarily.

### SALVAGE LOGGING

Salvage logging focuses on removal of trees killed by a disturbance, often fire or wind, which are of a size and species that can be sold to a mill before decay has reduced the value of the wood. Typically, large snags, logs, and residual green trees are more economically valued than small ones, and of course, large trees, snags, and logs are also used by more species of animals than small ones.

So it is not surprising that salvage logging, especially on public lands, has met with opposition and controversy (DellaSala et al. 2006). Consequently, we would expect that, unless sufficient levels of snags and logs were retained during salvage logging, those species of cavity nesters and others that rely on dead wood would be adversely affected by this activity. Indeed, for those species that respond negatively to salvage logging, even moderate salvage intensity had a noticeable effect on populations in the study conducted by Cahall and Hayes (2009). Some areas retained as unlogged burned forests seemed to be important in maintaining areas with high habitat suitability for dead-wood-dependent species (Cahall and Hayes 2009).

D'Amato et al. (2011) also compared the effects of salvaging timber from burned areas and areas with wind-thrown trees, and they reported that impacts on structural legacies persist well into the future. Unless plans are made and implemented that protect these legacy features, ecosystem recovery may be adversely affected (D'Amato et al. 2011). Further, the severity of a subsequent fire (a reburn) in the area of their study in the Lake States of the United States may be higher where salvage logging has occurred due to exposure of bare mineral soil prior to the subsequent fire and adverse effects on soil structure following the subsequent fire.

Diverse early successional conditions that result from disturbances such as fires and windthrow are one of the rarest habitat types in many forests that are managed for commercial values (Swanson et al. 2011). Consequently, we can actively recruit these sorts of conditions through innovative stand replacement management (Franklin and Johnson 2011), or we can allow natural disturbances with limited to no post disturbance intervention to provide these conditions. Or both. Should salvage logging of burned trees be allowed on public lands? That is a question that only has a social or political answer. Depending on the weight society wishes to give economic return from public lands compared to retention of ecological values, we arrive at different conclusions (Eklund et al. 2009). When landowners decide to salvage log but still retain habitat for selected species that require dead wood, they can refer to the information in Chapter 12 for guidance on setting dead wood goals.

## EFFECTS OF FUELS TREATMENTS ON HABITAT ELEMENTS

Managers of forests, especially along the rural–wildland interface where property values are at stake, resort to aggressive removal and management of fine and coarse forest fuels. This is particularly common in parts of the world with hot dry summers, such as the western United States. The issue is significant because fire has been controlled far longer than the typical return interval for fires in the forest type. Consequently, fuels such as dead needles, twigs, and small trees have accumulated to levels that are outside the range of natural variability for these forests. Further, because fire has been restricted from some of these forests, tree and shrub species that are fire intolerant, but that have flammable foliage, begin to dominate. This makes it very risky, if not impossible, to reintroduce prescribed fire to the forest without first controlling the fuel levels.

Typically, a manager would thin the stand so that the remaining trees are widely spaced and crowns are not close enough to allow a flame to move from one crown to another and the retained trees are resistant to fire, that is, those having a large size and thick bark. Further, the crowns of the retained trees should begin well above the flame height, preventing the formation of a crown fire if a fire starts in the forest. This means that most trees that are cut will be small, fire intolerant, and in many cases dead or dying. In other words, the trees that would be cut will largely be of low economic value. Consequently, either the slash is piled and then burned during the wet season (winter or rainy season), or it is chopped into biofuel and shipped to boilers or other facilities that can convert the wood biomass to energy.

Once the fuels have been effectively reduced and removed, then a prescribed burn may be introduced to the stand to periodically remove the fine fuels and reduce the level of tree regeneration or shrub cover that would otherwise develop. Under the correct weather conditions and with sufficient fine fuels to carry the fire through the stand, the ideal fire would move rapidly through the stand consuming fine ground fuels, but with flame heights that do not reach the tree crowns.



Thinning and prescribed burning will reduce fuel loads at least temporarily so that treated stands are more resilient to high-intensity wildfire (Stephens et al. 2012). Generally, there are few adverse effects of such management actions, though short-term changes in dead wood dynamics is clearly one, but many of the ecosystem components (vegetation, soils, wildlife, bark beetles, carbon sequestration) exhibit very subtle effects to these treatments (Stephens et al. 2012). Indeed, some treatments can be beneficial for some species of animals. Thinning and prescribed burning can provide better foraging opportunities for elk during spring, while unburned stands provide better forage during summer (Long et al. 2008a); so a patchwork of burned and unburned stands may be beneficial for elk forage throughout the growing season. In the same study, mule deer were not so selective of burned or unburned stands (Long et al. 2008b).

### USE OF PRESCRIBED FIRE IN MANAGED FORESTS

Once prescribed fire can be reintroduced safely to a forest, then repeated burns are necessary to maintain desired fuel loads. Making fire a periodic low-intensity disturbance in a forest in many cases is consistent with the disturbance regimes to which species have evolved or adapted. Fire in longleaf pine stands can create conditions that are often suitable for nesting red-cockaded woodpeckers. Even species of old-growth forests, such as the California spotted owl, seem to tolerate low to moderate severity fires, which have been historically common in the Sierra Nevada mountains of California (Roberts et al. 2011). Managed fires that emulate the historic fire regime of these forests may maintain habitat for California spotted owls and red-cockaded woodpeckers and also reduce the risk of an intense fire that would remove habitat elements important to these species (Roberts et al. 2011).

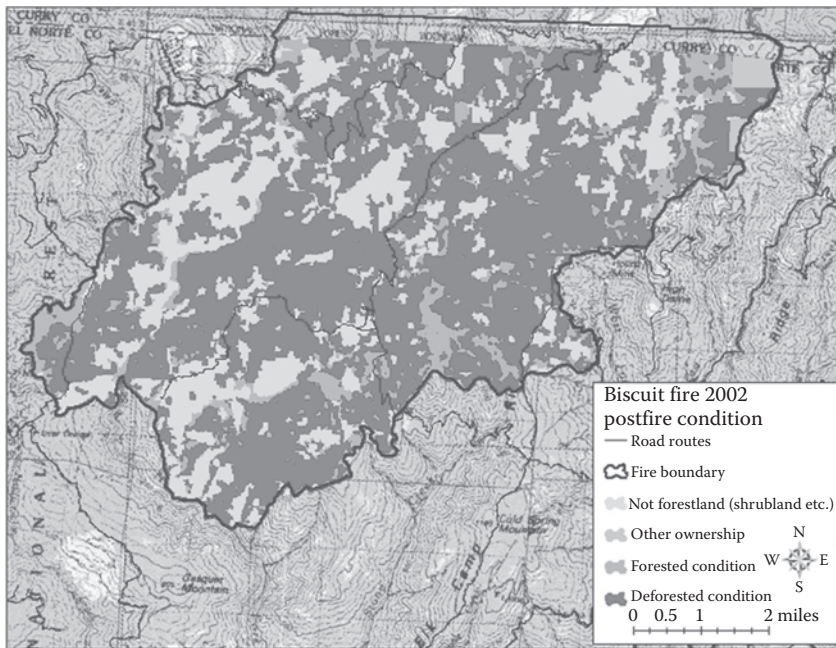
It should be apparent by now that doing the same thing everywhere is never a good strategy for managing forests to conserve biodiversity. Thinning and prescribed fires may mimic low to moderate-severity fires in a forest, but low/moderate-severity fire is not a substitute for high-severity fire, which may produce habitat elements important to another suite of species (Fontaine and Kennedy 2012). Fontaine and Kennedy (2012) advocated for including high-severity fires in fire-prone forests in order to be more consistent with historic fire regimes and contribute to conservation of regional biodiversity. Such an approach would suggest that managers let some wildfires burn where they are not endangering humans. Indeed some land managers have begun to recognize the need to adopt a let burn policy on some public lands (Bradshaw 2010).

### CASE STUDY: TO SALVAGE LOG OR NOT? A SCIENTIFIC DEBATE WITH A SOCIAL SOLUTION

Five separate fires were ignited by lightning in the Siskiyou National Forest over a 3-day period beginning on July 13, 2002. The five fires merged to form the Biscuit Fire, which burned almost 500,000 acres in southern Oregon and Northern California and cost over \$150 million to extinguish (U.S. Government Accounting Office [U.S. GAO] 2004) (Figure 13.1). Following the fire there was significant pressure to salvage some of the burned timber, despite protests from individuals concerned about the long-term effects of salvaging on the Siskiyou forest ecosystem. Due to the unusual size, intensity, and social controversy associated with the burn, a number of studies were conducted that investigated the response of the forest to the burn and to salvage logging following the burn (e.g., Thompson et al. 2007).

In 2006, five authors published a paper in *Science*, arguably the most prestigious science journal in the world, entitled “Post-Wildfire Logging Hinders Regeneration and Increases Fire Risk” (Donato et al. 2006a). A graduate student at that time, Dan Donato was the senior author, and here is the abstract from that paper:

We present data from a study of early conifer regeneration and fuel loads after the 2002 Biscuit Fire, Oregon, USA, with and without postfire logging. Natural conifer regeneration was abundant after the



**FIGURE 13.1** Boundaries and pattern of the Biscuit Fire, 2002. (From <http://www.fs.fed.us/r5/rsl/projects/postfirecondition/2002/>.)

high-severity fire. Postfire logging reduced median regeneration density by 71%, significantly increased downed woody fuels, and thus increased short-term fire risk. Additional reduction of fuels is necessary for effective mitigation of fire risk. Postfire logging can be counterproductive to the goals of forest regeneration and fuel reduction.

The paper ignited a firestorm of its own in the region. Almost immediately after the publication of Donato's paper, seven other scientists sent a critique of the work to the editor of *Science*, stating, "Donato et al. (*Brevia*, 20 January 2006, p. 352) concluded that logging after wildfire kills natural regeneration and increases fire risk. We argue that their paper lacks adequate context and supporting information to be clearly interpreted by scientists, resource managers, policy-makers, and the public" (Newton et al. 2006). Donato et al. (2006b) defended their work in a response, but the debate did not end there. A U.S. Congressman also commented on the lack of statistical rigor in the study and the press fueled the debate even further with newspaper articles, news spots on television, and articles in news publications such as *Evergreen Magazine*, the *Klamath Forest Alliance*, *High Country News*, and others. A number of arguments ensued, such as allegations that the original authors did not follow protocols on authorship and release of data, and counter arguments by the authors that their academic freedom to express their interpretation of the data was being attacked. The work was even questioned in the political arena when Representatives Greg Walden and Brian Baird questioned Donato in a public meeting. The controversy affected the perception of the role of science in solving problems in a unique way. Data collected, analyzed, interpreted, and published were challenged in the scientific community and in the political arena while the public looked on asking questions about the role of science in societal decisions. Two sets of scientists had different views on an issue. So which side was correct? What was the resolution? The issue continued to brew among the scientists for some time, but in reality the decision about whether to salvage log following a wildfire is not one made by scientists. Donato et al.'s (2006) findings are simply one more piece of information that can inform a decision made either by land managers or by the general public who

must take into consideration all available information, weigh its limitations and strengths, and then decide what is in the best interest of society now and for many years in the future. It is truly unfortunate that those who challenged Donato initially, and the politicians and others who were subsequently involved, could not recognize the value of the Donato et al. (2006) data to inform decisions, and consequently the role of science in contributing to thoughtful decision making. Their actions instead decreased the credibility of science in the eyes of many members of the general public and will make it that much more difficult for society to trust the weight of scientific evidence in an issue.

## SUMMARY

Fire is a natural part of many forest ecosystems. It can create diverse early successional conditions and shape the plant and animal species composition over a landscape for decades into the future. Fires kill trees but create snags and logs. When conditions are correct, with sufficient continuous and dry fuel and an ignition source, fires can creep across the forest floor or race through fine fuels or climb into the tree crowns. Attempts to reduce the number and extent of fires over many parts of the world has resulted; in some forests, with an unusually high accumulation of fuels, when fire does occur, the fire intensity is unusually high. Whether to prevent forest fires, extinguish them, attempt to control them, set prescribed burns, or let wildfires burn are decisions that result from social pressures through political systems. Each decision has a set of ecological consequences affecting the structure and composition of the forest and the availability of habitat elements for species.

Additional decisions must be made after a fire has occurred. If the fire was unusually intense, then are ecosystem restoration efforts necessary? Or will the forest recover to form another forest? Or be transformed into a chaparral? If most trees are killed, can some be cut and sold for economic gain? If so, then how many and what sizes and species? All of these questions are being asked following wildfires, especially those on public lands. Decisions can be guided by science, but even scientists do not always agree on the appropriate actions and associated risks. Public debate, policy development, and monitoring of the effects of policy to guide revisions are all steps underway in issues that deal with fire in forests.

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# 14 Urban Forests and Habitat Elements

New York, New Orleans, Boston, Auckland, Seattle. These are just few of the world's cities that developed in what were once forested landscapes. But now, that forest is gone and it has been replaced with roads, buildings, utility systems, and parking lots. Habitat lost, right? Well, yes, lost for some species, but not all. And with a bit of planning and forethought, even cities can provide habitat for a wide range of species, and even some of the same species that occupied the site before it became a city. Some habitat elements are provided serendipitously as a result of city development and others must be planned into the city system to ensure that they are effective, not ecological traps, and are sustainable. This chapter will focus on how an urban forester and a biologist can think collaboratively about providing habitat elements in the urban forest.

## DEFINING URBAN FORESTS ALONG AN URBAN–RURAL CONTINUUM

Over half of the world's population lives in cities or in associated suburban areas. Less than half are now living in rural environments, a marked shift over the past 100 years. The problem is that the boundaries of cities are continually expanding as populations rise, and what was once rural is now suburb, and what was once suburb is now urban. A gradient of urban development radiates from an urban center and the boundaries of that gradient may be continually changing. So, not only are species adapted to more rural forest and farm habitat types displaced, but the probability that they will wander into urban areas from rural areas is quite high for many species. The further from the rural environments, the less likely that dispersers would be encountered, but regular reports of bears in suburbia make it clear that encounters between humans and animals we may consider associated with extensive forests are increasing. Indeed, these urban–rural interfaces are at the front line of those responsible for resolving human–wildlife interactions. Moose jams and moose–auto fatalities are all too common now in urbanizing parts of the northeastern United States.

The impact of human development on a landscape varies from minimal in many wilderness areas, to greater where forests are managed for economic gains but on long rotations, to greater on commercial timberlands, to greater where forests are cleared for agriculture, and still greater where agricultural and forested lands are converted into houses and roads, and finally into urban centers where the built environment dominates. Simply, human population density, though related to urban environments, is not always the best indicator of urban impacts on habitat elements. Building height and density, impervious surface percentage, and road densities may be better indicators of impacts on animals, and on the people who live there. Many suburbanites travel to rural areas to see wildlife, while others, who are unable or unlikely to travel or may be inner city residents, may seek parks and other open spaces to relax. Encountering wildlife may not be a goal for many urbanites.

Many people who live in cities or suburbs enjoy seeing animals in their parks or backyards, while others view them with fear or disdain. A raccoon sprawled on a branch on a summer afternoon, sleeping, may attract some urbanites with cameras, others to call the police concerned that it is rabid, and another calling animal control convinced that the animal has been the one raiding the trashcan and making a mess. Reactions to other species such as snakes, bats, coyotes, and cougars are even more extreme. So, although we can provide habitat elements in cities that will encourage



species to cohabitate with us, is that what the residents want? Will they tolerate these species? Managing habitat in urban forests must engage the residents to know what they would like, and it must be adaptable because the residents of a community change their values over time. Once the hard social work is done to understand the goals and limits of tolerance for a community, then, the easy work begins in providing the habitat elements.

## HOW SOME SPECIES INTERPRET THE “BUILT ENVIRONMENT?”

There are some species that do surprisingly well in cities. Peregrine falcons nest on ledges on cliffs, and forage out from that nest site to capture birds and small mammals. To a peregrine, skyscrapers and tall bridges are a sea of ledges on human-made cliffs. And all these pigeons (rock doves) and squirrels that people love to feed are scrumptious. What more could a peregrine want? Well, perhaps a few less cars and fewer bioaccumulated toxins (Park et al. 2011), but for the most part, peregrines in cities are doing quite well (Cade et al. 1996). Chimney swifts and Vaux's swifts are equally as adaptable to using masonry chimneys as nest sites (Rioux et al. 2010). Where did chimney swifts nest before there were chimneys? Hollow trees no doubt. White storks in Europe select chimneys and house roofs for nest sites too (at least until electrical utility poles were erected), but still occasionally nest in large flat-topped trees and snags. These species are taking advantage of serendipitous similarities between what humans build and the habitat elements that they used before humans built tall buildings and chimneys.

When humans build cities and suburbs, they bring with them the plants and animals that they enjoy having around them, including those species from other ecosystems that can become invasive. Many plants used in the landscaping industry have escaped and now create problems as invasive plants. Import of species that could be invasive is more carefully monitored now, but we are living with a legacy of imported plants and animals—European starlings and House finches in northeastern cities, pythons in Florida (Reed 2005), Monk parakeets in the eastern United States (Simberloff 2003), and many more. Some of these invasive exotic species become food for native species, but too often native species are displaced by the exotics, and there are extreme cases of this displacement in many parts of Australia where placental mammals were introduced and had devastating effects on mid-sized native marsupials. Unfortunately, cities become the focal points of spread for many of these invasive species or nonnative species that natives are not adapted to use.

Parking lots, roads, sidewalks, railroad tracks, rooftops, and compacted soil all preclude the growth of plants (unless they are designed to), and as such, form a maze of potential barriers to animal movement in a city. In addition to the risk of being run over by a car, any animal crossing an impervious surface has no cover from a predator such as a free-ranging pet dog or cat, peregrine, coyote, or red-tailed hawk. Hence, many patches of vegetation that could be managed to provide habitat elements are isolated in this maze of concrete and asphalt, unless active efforts are made to connect the patches. Tunnels under roads, riparian areas with greenway strips, and even wildlife overpasses are all possible, but expensive or impossible to retrofit into an urban area (Glista et al. 2009). The stepping stones of multiple backyards may be the most likely means of connecting larger patches for some species, and backyard habitat management is becoming increasingly popular (Palmer 2004). But for species that are inhibited by curbs, fences, and other barriers, such as salamanders and large mammals, corridor connections may be the most effective (Angold et al. 2006). Wildlife corridors, managed with appropriate habitat elements needed by the focal species, can also be urban greenways and simultaneously provide walking paths, bikeways, and other recreational opportunities (Teng et al. 2011). Urban foresters and wildlife biologists will need to team up with landscape architects and urban planners to design these multipurpose greenways. In doing so, and with an accompanying educational effort, more people in urban environments can be in contact with more wildlife species and raise the level of awareness around habitat management and species conservation.



## FINDING SPACES FOR HABITAT MANAGEMENT

Greenways and wildlife corridors connect patches of habitat. But habitat for which species? Urban parks come in many forms, from those that are primarily used for athletic events, picnicking, and are highly manicured, to those that maintain a more full complement of native plants and other habitat elements important to native species. Spreading shade trees distributed over a manicured lawn may provide a habitat for mourning doves and gray squirrels, but not for species associated with shrubs, leaf litter, or dead wood. Parks containing a shrub layer, dead wood such as snags and logs, hollow trees, and few invasive plants are most likely to support a diverse animal community. These patches, if maintained as primarily native vegetation, can also act as an island from which plants and animals can disperse should a “new” habitat develop as a result of urban renewal and brownfields reclamation.

Habitat patches can also be found in some unexpected places. Green and brown roofs can provide a habitat for some species while also mediating stormflow and adding insulation to a building. Green and brown roofs may be important sites for invertebrate conservation and some species of nesting birds (Baumann 2006, Kadas 2006). Species richness in spider and beetle populations on green roofs is associated with plant species richness (Gedge and Kadas 2004). These findings have led some scientists to begin discussions with engineers to design green roofs to maximize biodiversity (Oberndorfer et al. 2007). Of course, there are limits to the types of plants that could be grown on some roofs, but increasing the diversity of native plant species including grasses, forbs, and shrubs, could substantially aid as a habitat for birds and invertebrates.

## HABITAT ELEMENTS LIMITING SPECIES IN URBAN AND SUBURBAN SETTINGS

Even where there are opportunities to manage habitat elements in cities, some elements present logistical problems for urban biologists and foresters. Because humans are such a dominant part of the urban ecosystem, human safety becomes a primary consideration when designing urban areas for recreation and habitat. Dead wood, burrows, and shrubs are habitat elements that present special challenges.

*Dead wood.* Dead limbs on both living and dead trees are viewed by urban foresters as potential hazards to humans and their property, and consequently, whenever there is a chance that some or all of a dead, dying, or decayed tree or limb has a chance of falling on someone or their property, then it is removed. Of course, if the dead wood is removed before it has fallen, then there is no opportunity to recruit fallen logs into the urban environment. The challenge for urban foresters and wildlife biologists is to find places in the urban environment where the risk to people and property is low and dead wood resources can be maintained or created. For instance, if a tree has died and it is within 30 feet of a sidewalk, then topping the tree at 15-feet high would provide a short snag that could then be used by woodpeckers and secondary cavity nesters without significant risk to people or property. Indeed, I have colleagues living in suburbia who have created short snags that will not fall on their houses and that are used by primary and secondary cavity nesters, bats, and raptors as perches.

Similarly, there has been little effort to recruit fallen logs into urban environments. Usually, hazard trees are cut, small pieces are chipped, and large pieces are removed to landfills or other dump sites. Placement of these logs in parks, greenways, along streams, and in backyards can provide a habitat for reptiles, amphibians, and invertebrates (Guderyahn et al. 2010), probably at no greater cost than traditional disposal approaches.

Although the role of arborists in creating or removing habitat elements has been known for some time (Dunster 1998), very little is known about the use of dead wood in urban environments (Blewett and Marzluff 2005). As urbanization continues across our planet, there is an increasing need to understand the possibilities for providing key habitat elements within the matrix of the built environment.

*Burrows.* Although very little work has been done to understand the potential for supporting burrowing animals in urban and suburban environments, we can surmise that two factors would restrict the availability of burrow sites for mid-sized burrowing species: creation of expansive impervious

surfaces and compaction of the remaining soil due to human and vehicular traffic. Indeed, increases in the impervious surface area that increases stormwater runoff can be expected to also have removed the potential den sites for burrowing animals (Arnold and Gibbons 1996). By 2004, the impervious surface area in the lower 48 states of the United States was enough to equal the land area of the state of Ohio (Elvidge et al. 2004). Add to that compaction in parks, playgrounds, and other areas with human and vehicular impact that few areas may remain for species such as woodchucks, burrowing owls, and gopher tortoises to live.

*Shrubs.* The shrub layer is important for many species of birds and mammals that either nest in the shrubs, use shrubs for cover, or eat the fruit, leaves, or twigs of the plants. In many urban areas, exotic shrubs can tend to dominate, but maintenance of native shrubs may be advantageous, at least for some species (Savard et al. 2000, Hostetler and Main 2010). But shrubs present a problem in a park. Many park managers will systematically remove shrubs and other cover, for humans, along trails and sidewalks. Unfortunately, due to the risk of muggings, rapes, and attacks, shrubs are not allowed in areas where people might be vulnerable to other people. Consequently, the management of a safe zone along walkways and paths may be needed, but shrubs could be allowed outside the safe zone. In Europe, habitat islands have been proposed where shrubs are a key component of the island (Fernández-Juricic and Jokimäki 2001), but connectivity among islands becomes challenging, especially once the built environment forms the matrix condition within which habitat islands are embedded. In most cities, we are retrofitting the habitat into an already-built environment. Only where urbanization is affecting rural environments along the urban fringe can we proactively plan islands and corridors. Within the existing city infrastructure, only a few possibilities remain for creation of new islands and corridors. One is to recover former industrial sites that are no longer used (brownfields reclamation) or incorporate habitat management into abandoned railways as part of rails to trail projects. Both approaches require a commitment of money and management to achieve habitat goals. Given the potential for environmental contamination of brownfields, additional concerns remain about the impacts on animals using these islands, especially if phytoremediation (using plants to uptake and hold contaminants) is used as a technique for sequestering contaminants (Bañuelos et al. 2002). Should contaminants accumulate in plants designed to take up contaminants and then animals feed on those plants, then we could easily create a biological trap for species using the newly created island. Again, little research has been done into this potential effect, though work on insects suggests that there could also be effects on herbivores and insectivores using these sites.

## URBAN STREAMS AND WETLANDS

Urban streams and associated wetlands provide an opportunity to create connections and habitat patches for many species in urban and suburban environments. Unfortunately, many urban streams have been altered markedly as a result of years of changes occurring in the urban environment. First, and foremost, many urban streams are constrained or channelized to ensure that they do not erode land that is now part of the urban area. This may be as marked as concrete channels or may be large rocks along the bank (rip rap). No longer are many streams in urban areas allowed to meander and create channel complexity important to many aquatic species. In addition, due to the increase in impervious substrates in urban environments, any contaminants on those impervious surfaces are washed into the stream unless measures are taken to clean the water first. And because rainwater falling on impervious surfaces moves directly into the stormwater system and is not absorbed into soils, stream levels can fluctuate wildly following rains. In many cases, streams have become simply channels to move water away from the city as rapidly as possible rather than functional stream and riparian systems (Paul and Meyer 2001).

Restoration of degraded urban streams is a significant challenge. Site-specific local changes to the stream structure or riparian area are unlikely to be effective because the scale of impacts is basin wide. Consequently, changes to the drainage system must be approached in a more holistic manner (Walsh et al. 2005). This will involve multiple landowners with multiple objectives, as well as newly

designed holding areas for stormwater that mediate peak flows in the stream. Palmer et al. (2005) proposed five guidelines for assessing the success of stream restoration:

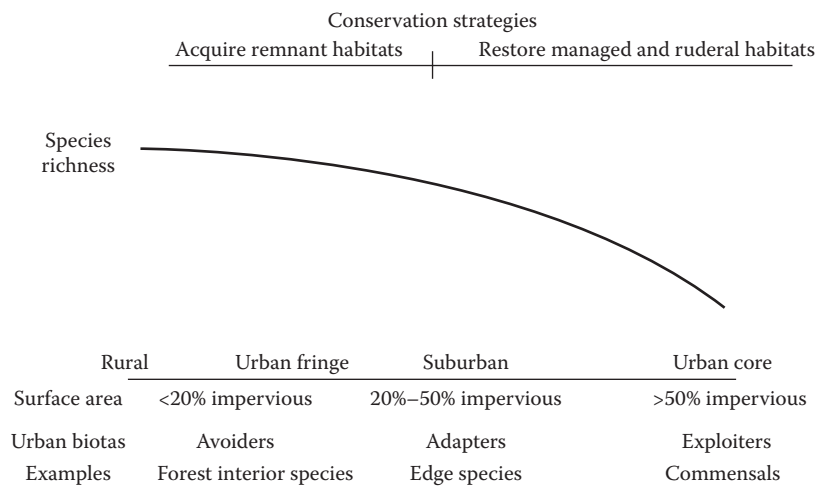
- 1. The design of an ecological river restoration project should be based on a specified guiding image of a more dynamic, healthy river that could exist at the site.
- 2. The river’s ecological condition must be measurably improved.
- 3. The river system must be more self-sustaining and resilient to external perturbations.
- 4. No lasting harm should be inflicted on the ecosystem.
- 5. Pre- and post-assessment must be completed and data must be made publicly available.

In addition, Jansson et al. (2005) proposed that the restoration plan is based on a conceptual model of a functional ecosystem agreed upon by all stakeholders.

Achieving success as indicated by these guidelines over the watershed is a daunting task that requires the managers to meet social demands with ecological goals. Including both riparian and in-stream conditions in the design of urban watersheds adds complexity to the problem but also adds significant value as a habitat and connections for many species of animals. Any plan will have constraints; cities will not tolerate flooding. Some landowners will prefer a manicured streamside, while others will want a wilder environment. Space is at a premium; so, unless the municipality buys lands for greenways, landowners may or may not be inclined to participate. An ecological restoration venture becomes a joint effort among sociologists, biologists, and politicians to achieve long-term ecological and social goals (Pickett et al. 2001).

URBAN EXPANSION, WETLANDS, AND MITIGATION

Regardless of how much planning is conducted and how much attention is given to habitat needs across an expanding city, urban growth replaces the rural habitat with impervious surfaces and buildings. The habitat for many species will be lost (McKinney 2002). More than 5% of the surface area of the United States is urban or suburban land use (USCB 2001). Further, McKinney (2006) suggests that because cities, regardless of where they are in the world, have commonalities in design and growth and tend to homogenize the biota associated with them (Figure 14.1). Consequently, McKinney (2006) argues that we should focus our efforts on native species, although that too can



**FIGURE 14.1** Generalized pattern of species diversity as a function of urbanization. (McKinney, M.L., Urbanization, biodiversity, and conservation, *BioScience*, 2002, by permission of Oxford University Press.)

be problematic unless we are sure that our management actions are contributing to increasing, and not declining, population fitness.

Documenting the rate of loss of a habitat for a wide range of species is important so that species that will need particular attention can be identified. The use of species-specific habitat models is most appropriate in that case, although the number of species that could be affected can be daunting to urban planners and conservation biologists. Consequently, Hasse and Lathrop (2003) suggested more generalized land resource indicators that could be used to monitor the conversion of land taken to urban development: (1) density of new urbanization; (2) loss of prime farmland; (3) loss of natural wetlands; (4) loss of core forest habitat; and (5) increase of the impervious surface. One of these, natural wetlands, has received considerable attention. Under Section 404 of the U.S. Clean Water Act, the United States has established a no-net loss of wetlands policy. Each year, approximately 47,000 acres of wetland mitigation are required to offset 21,000 acres lost (Kihslinger 2008). When developers are permitted to destroy a wetland, they must pay for restoration or creation of wetlands to offset the loss. More recently, developers can purchase habitat credits from habitat banks that are established as new wetlands for the specific purpose of establishing a market for those developers in need of credits. Semlitsch (2008) has extended this idea to that of conservation banks where credits can be purchased for the restored or created habitat designed to replace that which has been lost. Such an approach considers not simply the acreage of wetlands mitigated but also the context for that wetland that is more likely to make it functional for more species. Fox and Nino-Murcia (2005) reported that there were 35 official conservation banks covering nearly 16,000 ha in 2003, which provided a habitat for at least 22 species listed under the U.S. Endangered Species Act. Nearly all conservation banks were established based on monetary return, with most of for-profit banks that are breaking even or making money (Fox and Nino-Murcia 2005); in 2003, credit prices ranged from \$3000 to \$125,000/0.4 ha (1 acre). The approach of conservation banking has also been adopted in other countries (Briggs et al. 2009).

## MANAGING TREES, PARKS, AND FORESTS IN URBAN SETTINGS

Vegetation in most parts of most urban areas must be managed. Liability issues arise to landowners and public safety offices if certain actions are not taken. For instance, if a tree owner has knowledge of a hazard, he or she is required to address the danger posed by the tree (Mortimer and Kane 2004). Similarly, municipal park managers must do everything that is reasonable to protect people using bikes and trails from the risk of falling trees, limbs, or people who might attack other people. Consequently, many areas in urban environments will lack dense shrubs, or dead trees, or hollow trees simply due to the liability that landowners face by having them present. Different landowners are more or less willing to accept potential risks; so, few areas removed from human or vehicular traffic become few areas in a city where these habitat elements can be retained or recruited. Geographic information systems (GISs) provide a means of identifying these potential areas for habitat management. By buffering on all roads, sidewalks, buildings, and trails, the remaining patches have the potential as sites for management. Even after they are identified, it will be the decision of the landowner as to the risk that she or he is willing to incur.

What constitutes a tree that presents a hazard? Many aspects of a tree contribute to its potential to cause harm. Dead limbs certainly can be hazardous, but are also an evidence of decay, lean, height, and species, to name a few. A trained arborist can identify trees based on a set of characteristics (Matheny and Clark 1994), and given the liabilities involved, homeowners often contact arborists for an assessment of risk. Once the landowner is apprised of the risk, then they must conduct their own risk assessment and decide if the risks justify the expense of tree removal or pruning (Ellison 2007). Knowledge of habitat benefits could influence a landowner's decision in some cases.

Even when areas are found where habitat elements can be provided, we need a better understanding of the function of the habitat patch. Avoiding creation of habitat sinks in urban areas should be considered. Creating habitat elements and patches and having species use the patches in urban

areas does not necessarily mean that the population is sustainable. For high-priority species, some evidence of reproductive rates and survival is needed to ensure that individuals who would otherwise be successful in rural patches are not attracted to urban patches simply to be unsuccessful. Indeed, the risk to many species of being killed from cats, dogs, rats, and vehicles is much higher in urban and suburban areas than in other habitat types. Should the reproductive or survival rates for species be very low in these urban patches, then the patches could be ecological traps, attracting animals to what may seem to be suitable habitat, but the population fitness is so low that over time, the population declines. Ideally, large patches connected by wide corridors may prevent ecological traps from occurring, but both these conditions are extremely rare, and very expensive in cities.

## SUMMARY

Urban and suburban areas are often simply written off as habitat loss, but through serendipity more often than planning, some species do find a suitable habitat in developed areas. Active planning in cities to provide the habitat elements important to species that can use urban forests is rare but can aid in adding to the diversity of organisms sharing urban environments with humans. By integrating habitat elements into planning for urban parks, greenways, riparian areas, and new suburban developments, a habitat can be provided for a wide range of forest-associated species. Particular attention to those habitat elements that are often removed for safety purposes, such as dead wood and shrubs, can cause urban forests to meet the needs for a broader range of species than we typically find. Further, the use of green roofs, planting native trees and shrubs, and promotion of backyard habitat plans can add an additional urban habitat for shrub- and tree-associated species. Active management of urban areas for wildlife conservation is an area that has received very little attention and research is desperately needed.

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# 15 Landscape Structure and Composition

## LANDSCAPES

In art, they are paintings of places, usually from a distance, a mosaic of forms interwoven to portray a recognizable place familiar and pleasing to the eye. That is the function of the painting. Ecological landscapes also can be viewed as a mosaic. They have structure, as portrayed by the pattern of the patches comprising the landscape. They have composition, as portrayed by the types of patches on the landscape. They have function, as defined by the resources of concern—in our case, patches of habitat for a species or collection of species. Forested landscapes, as habitat for vertebrates, are patterns of habitat patches distributed over forest-dominated areas, with each patch representing a set of resources used or needed by the species. The pattern of habitat patches across a landscape differs from one species to another. We, as humans, tend to view forested landscapes as collections of forest stands, a mosaic of units of vegetation that is relatively uniform within stands, but heterogeneous among them. We do that because, to us, those stands represent different resources, such as economic value, places to hike, or different aesthetics. For some other vertebrates, the stands that we see also represent patches of different values, habitat values. But not for all species. Take the shrub-associated Swainson's thrush that is widely distributed across North America (Figure 15.1). What type of mosaic of patches would a Swainson's thrush see? The thrushes' search image (what they seek) for a territory, likely includes patches of shrubs with varying quality for foraging on insects in the leaf litter and foliage, as well as nest sites low in the shrubs. The overstory vegetation seems to have little, if any, influence on what Swainson's thrushes see as habitat patches, except through the effect it has on shrub development. Old-growth, stand establishment, or understory reinitiation conditions are not really as important to this species as the shrub cover present in the stand. If we were to develop a very simple map of habitat with white patches being habitat and black being unusable, then we would map patches of shrubs of a certain cover and height for Swainson's thrushes, and these patches might be quite unrelated to tree species' composition or structure.

Mapping habitat for northern flying squirrels would focus more on snags, tree species, size and canopy cover, and not so much on shrubs. And, to make things even more complicated, consider habitat for cottontail rabbits that feed in grassy areas but use dense shrubs as cover. They live on the edge between these two types of vegetation, and the acceptable habitat (white) on our map would be a linear strip extending perhaps 30 m into the grassy area and 20 m into the shrubs. Everything else would be black. Three species, three landscape patterns, all in the same forest. Managing stands in this forest could influence the patterns of habitat for each of these species, depending on the types of vegetation and other features provided in each stand.

We know that habitat varies in its quality from place to place and over time. At the very least, we might consider sources and sinks for each species, but even these represent a gradient of habitat quality (as expressed through individual fitness). It is more appropriate to think of a mosaic of habitat as represented by shades of gray, or perhaps, more appropriately, a collage of habitat values, with edges that are more or less blurry. A spectrum of resources painted across an area for each species. If we are concerned about providing habitat for 100 species in our forest, then there are 100 landscapes in our forest, each with its own collage of habitat. For each species, management decisions made at places in our forest represent the building blocks of a habitat mosaic for that species that we manipulate through our forest management activities. So the structure, composition, and



**FIGURE 15.1** Swainson's thrushes select patches of shrubs as habitat across a landscape mosaic. (Photo from the North Cascades National Park, USDI National Park Service.)

function of a landscape depend entirely on the species or other values that we wish to produce from that landscape.

We can take this one step further. Dr. Sam Cushman and his colleagues questioned the value of patches when defining landscapes as habitat for a particular species (Cushman et al. 2010b). Not only did they find that different people define patches inconsistently, but that the resources of value to a species are not homogenous within a patch. Sam and others suggest that we think of landscapes as gradients of resource availability, and not as a collage of patches, each representing a certain level of habitat quality. Gradient-based analysis of landscapes that represent habitat for individual species is clearly the future of landscape ecology (Cushman et al. 2010a). Indeed, new gradient-based measures of the landscape have been developed (McGarigal et al. 2009). But most work to date has been based on patches, so the remainder of the chapter focuses on patch-based approaches. Just keep in mind that you may need to think of landscapes as a series of gradients of habitat quality, rather than patches.

## DEFINING THE LANDSCAPE

There are several attributes of a landscape that must be defined when considering habitat function for a species across a landscape: Grain, extent, and context. The *grain* represents the smallest unit of space that we (or the species of interest) use and identify. Recall the hierarchical selection of habitat discussed in Chapter 3. Each species scales its habitat differently at each of four (or more) levels of habitat selection (Johnson 1980). Depending on the scale of habitat selection we wish to understand, when managing the species, the smallest unit of space (that we would need to consider) varies. For many species, the grain might best be an individual tree or shrub, where resources are available: browse plants for a herbivore; fruit from a yaupon shrub; or blackgum trees where dens are available for squirrels. But considering the spatial pattern of all trees and shrubs in a forest is impractical

when developing a management plan. Rather, we typically consider the patches within which these resources occur (however, see the gradient-based approach suggested by Sam Cushman, provided earlier). The way in which we define patches on the ground, or on a map, defines a grain, which is some minimum size that is both meaningful to the species of interest and practical to consider during management. For example, we might define patches of shrubs of various species, 0.5 ha or larger, as resource patches for white-tailed deer. Or we could simply characterize each stand in our forest as having certain levels of food or cover resources within the stand, without explicitly considering where in the stand these resources occur. From a management standpoint, we would just know that some stands have more high-quality browse than other stands.

Now let us assume that the forest landscape that we are managing is 20,000 ha; this would define an *extent*, the area over which we are managing resources. Alternatively, we could define the extent as the population of deer that we are managing and the area over which they occur. If we are managing the 20,000 ha landscape with a grain size of 0.5 ha, then we would need to keep track of up to 40,000 patches of browse across the landscape over time. We can do that with the aid of computers and geographic information systems (GIS); but from a forest management standpoint, such a fine level of information may not be practical to monitor or manage over time. Instead, resource patches such as these are often considered as part of a stand prescription. Prescriptions for stands could (and often should) include the variability and spatial arrangements of habitat elements within a stand, so that fine-scale patterns across landscapes are inherently considered, even if they are not mapped.

If we have a 20,000-ha landscape that we are managing, then it resides in a larger area, an area that we are not managing but which affects the function of our landscape. The *context* for the landscape can have a significant effect on what can be achieved through management of the landscape. This is analogous to management of stands in our forest. Manipulating the habitat elements in a stand *may* be effective in managing habitat for a species, depending on the resources available in the surrounding stands. Managing resources across a landscape *may* be effective, depending on the resources available in the surrounding landscapes. Managing pileated woodpeckers in Central Park, in New York City, is highly constrained by the surrounding urban context. Managing them in a small watershed of similar size in the Catskill Mountains of upstate New York may present far different constraints and opportunities in that context. Manipulating the resources among the patches represented by the grain, over the extent as constrained by the context, is the essence of landscape management for various species.

So how large is a landscape? By now you should know that the answer is, "It depends!" From a functional standpoint, there is no ideal upper or lower bound that describes a landscape. A decaying log on the forest floor represents a landscape with a mosaic of fungal colonies, bryophytes, bacterial colonies, and tree regeneration. The grain is the patch of fungus or the patch of tree seedlings. The extent is the log. The context is the forest in which it resides. Alternatively, the area extending from intermountain rangelands in North America to the pampas of Argentina represents a landscape of appropriate size for Swainson's hawks. They nest and feed in the northern hemisphere and spend the winter in South America (actually Argentina's summer is our winter; so they are never in winter!). Patches representing feeding, nesting, and roosting sites in both hemispheres and along the migratory pathway, represent a landscape used as the basis for management of this species.

From a practical standpoint, landscapes are usually considered areas of hundreds to thousands of hectares, areas where consideration of a meaningful grain does not become overwhelming to map and manage over the extent of the landscape. One characteristic of a landscape that is particularly important when managing animals, especially mobile animals, is that the extent applies to all owners within the landscape. Recall that in the United States, wildlife are public resources. And most are somewhat mobile. And they do not respect property lines or political boundaries. So a landscape that you define when managing a set of resources, that includes multiple owners, must not be restricted to just ownership. Habitat among all owners is what the animals will respond to (Spies et al. 2007). Too often, large forest owners restrict consideration of impacts of their management on

a species to those impacts occurring on their lands, not the multiownership landscape of habitat to which the population responds.

### HABITAT QUALITY AT THE LANDSCAPE SCALE

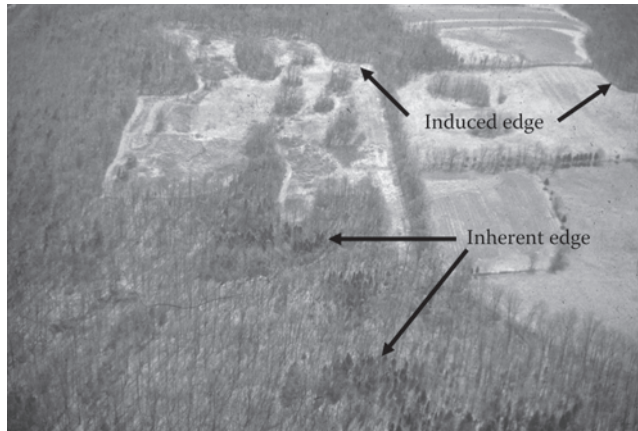
Recall that habitat quality is a function of individual and population fitness. Habitat elements distributed among patches on a landscape change over time. They change as plants grow, soil erodes, and disturbances occur. The quality of the habitat for a species also changes among patches over time. We can direct those changes, to at least some degree, by management actions. Over landscapes, we may wish to increase, maintain, or reduce (depending on our goals), the fitness of the individuals in the population of concern. This is usually done in concert with forest management activities, as well as natural disturbances and subsequent plant regrowth. Consequently, we manipulate habitat elements within stands, and other management patches on the landscape, to influence the direction of the cumulative changes in these resources, over time.

Depending on how a species scales its environment, we might expect a population response from these actions. Populations do not respond solely to changes in availability of habitat elements. Disease, competitors, predators, parasites, and other density-dependent factors, which may or may not be mediated by habitat elements, can influence population responses. Altering habitat simply provides the potential for populations to respond in certain ways. There is always uncertainty in what responses we will see. Disturbances occur but are usually unpredictable. Climate is changing. Invasive species are competing with native populations. Consequently, tracking the vital rates of the population and understanding the long-term changes in birth rates, death rates, and *lambda* (the population parameter used to define if a population is increasing or declining) allow us to understand the consequences of management actions over large areas. These population parameters are usually used to define habitat quality across a landscape, which reflects the cumulative value of habitat among patches across the landscape. But the location of the patches matters. If a species needs 40% of its home range in high-quality food patches to achieve high levels of fitness, and you manage to provide 40% of a landscape in these patch types, then the food patches must be distributed so that each of multiple potential home ranges are distributed appropriately. Locating all high-quality patches on one side of a landscape would leave the other side of no value to this species.

### LIVING ON THE EDGE

A feature of landscapes that has an inordinate effect on habitat quality for many species, is the boundary between patches. Edges between patches, as well as other linear features such as roads, rivers, and ridge-tops, all influence habitat quality for various species. Even though these linear features occupy only a very small part of a landscape, they can have a disproportionate effect on the function of a landscape for a species. Depending on the species and the type of edge, edges can be beneficial, detrimental, or have no effect whatsoever. Certain types of edges *are* habitat for species such as cottontail rabbits, but other species are adversely affected by certain types of edges. Edges are usually characterized as induced or inherent (Figure 15.2). *Induced edges* are those that occur between two patch types of different successional condition, are usually caused by a disturbance, and represent structural differences between the patches. Edges between early and late successional forests, forests and utility right-of-ways, or forests and agricultural lands, are examples of induced edges, though each of these types of edge functions differently, depending on the species. *Inherent edges* are those formed because the composition of the forests differs between the patches. Although this may ultimately be the result of disturbance, it is less a structural interface, and more a floristic interface.

Induced edges between forests and agricultural lands are ideal conditions for brown-headed cowbirds to nest (Figure 15.3). Cowbirds fed among bison herds historically and now feed among cattle; cowbirds eat insects disturbed by the grazing animals. They reproduce by laying their eggs



**FIGURE 15.2** Induced edges reflect structural differences between two patch types, while inherent edges represent floristic differences between patches.

in the nests of other birds, a reproductive strategy called *brood parasitism*. Once the female is fertilized, she flies into the forests adjacent to the grazing lands and seeks hosts for her eggs. These hosts are often *neotropical migratory birds* that generally produce one *brood* (cohort of young birds) per year. Once the female finds a host nest, she removes the host's eggs and lays her egg in their place. Should the host continue to lay eggs, then the host eggs hatch after the cowbird egg has hatched. The nestling cowbird grows quickly and the host parent raises it as its own. Being larger and more aggressive, it tends to get most of the food provided by the host parents, and the host nestlings often die as a result of competition for food. Consequently, the cowbird has a high probability of fledging (successfully leaving the nest), and the host nestlings are unlikely to fledge. Host species within a



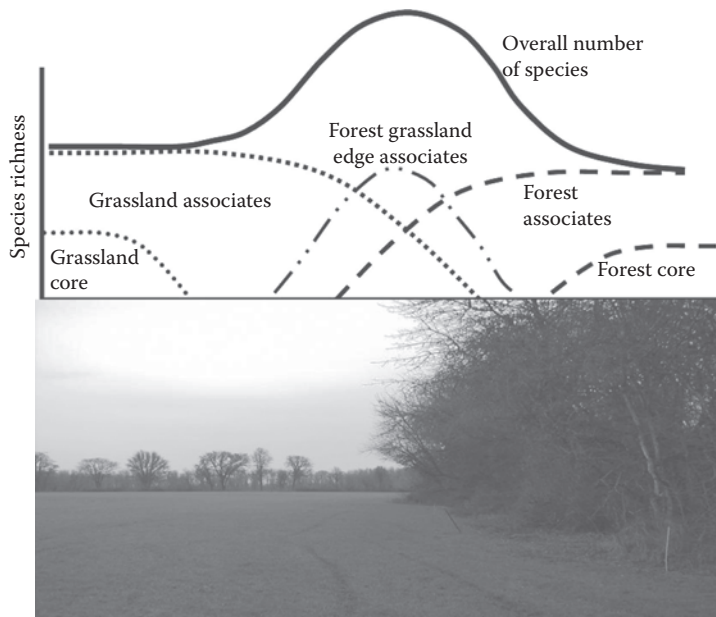
**FIGURE 15.3** Brown-headed cowbirds (large nestling in this photo) are brood parasites that can reduce nest success of other bird species, often neotropical migrants, when they nest near edges. (Photo provided by Dr. Jim Rivers. With permission.)



few hundred meters of agricultural or grassland edges often have depressed levels of nest success, due to cowbird parasitism. They can also experience higher levels of predation along edges than in the interior of the forest because raccoons, rat snakes, opossums, and deer mice also forage near edges. So creating these edges between grasslands and forests can be source habitats for brown-headed cowbirds, cottontail rabbits, and rat snakes, but sinks for neotropical migrant birds, that experience high levels of brood parasitism and predation along the edges.

Species can be classified based on their associations with edges. *Edge associates* are those species that find the best quality habitat where there is access to required resources in two or more vegetation patch types. There are also *edge specialists*, those such as cottontail rabbits that are likely to only occur where these edges exist. And there are some species that are found along edges but do not require the edge; rather, their home ranges simply abut the edge. And there are *forest interior species*, those that avoid edges and use the interior or core of a patch. Managing these groups together on one landscape often results in a high diversity of species along edges (specialists, associates, and those that occur in each of the adjoining patch types). You tend to find more species along edges than in the interior of forests or grasslands (Figure 15.4). Since Aldo Leopold's book *Game Management* was first published in 1933, managers have often focused on providing edges as a way of increasing populations of some game species or increasing species' diversity across a landscape. As in all other management choices, some species are winners and others are losers. Increasing edges can be to the detriment of forest-interior species.

In addition to increased levels of predation and parasitism along edges, some species can be either positively or adversely affected by edges, due to changes in the microclimate, probability of natural disturbances, vegetation structure, and a suite of human effects. Microclimatic changes along induced edges are highly influenced by edge contrast, or the degree of difference in forest structure between the two patches. If a forest adjoins an open area, then the edge of that forest receives more sunlight (depending on aspect and slope), is drier, receives more wind, and relative humidity is likely to fluctuate more throughout the day. These conditions can affect habitat quality



**FIGURE 15.4** Species richness is often high along edges due to the contributions of species to edges from both adjoining patches as well as edge specialist species. Note though that patch interior species avoid these edges.



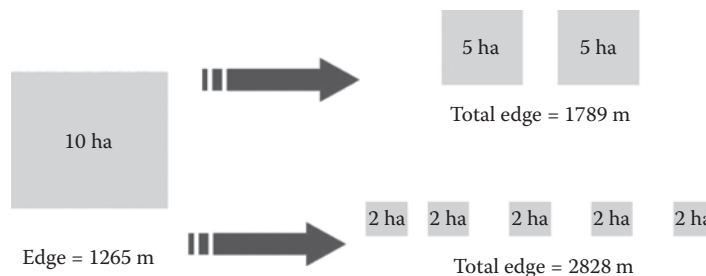
for some species directly, especially reptiles and amphibians (and other ectotherms), and can result in changes in vegetation structure. Time since edge creation is often an important factor influencing the structure of an edge. An edge formed by a clearcut decreases in contrast with the adjacent forest as the trees in the clearcut grow. The edge in the forest that receives more sunlight after the clearcut occurs, will also fill in with shrubs, trees, and forbs, creating a “softer,” less abrupt, edge.

Disturbances also can be influenced by the presence of an edge. Fuels for fires that differ between patches can allow intense fire to affect the edges of patches that otherwise would not be so severely affected. Abrupt edges in wind-prone sites can lead to continuous windthrow of the trees along the edge, resulting in accumulations of dead wood (or timber salvage opportunities). Consequently, disturbance intensity can be unusually high along edges, leading to changes in the structure and function of the recovering forest following the disturbance.

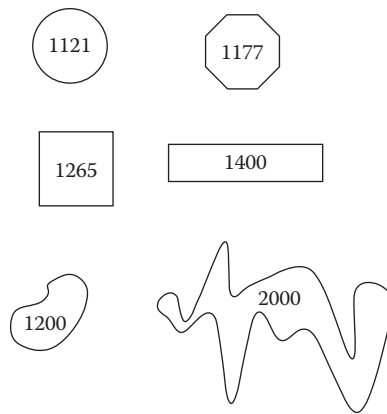
Humans not only create edges, they use them. Hunters often will follow edges to find game animals. Roads and fences often occur in conjunction with edges. Consequently, domesticated animals tend to use the edges as well so grazing pressure can be higher there. Free-ranging dogs and cats can have a significant impact on native animals, and they often hunt near edges. The geographic ranges of nearly all mid-sized marsupials in Australia have been greatly reduced due to predation by feral cats and foxes. The Wildlife Society produced a position paper on this topic in 2006; excerpts follow: “The estimated numbers of pet cats in urban and rural regions of the United States have grown ... to nearly 65 million in 2000. ...domestic cats are a significant factor in the mortality of small mammals, birds, reptiles, and amphibians. ...Effects of cat predation are most pronounced in island settings (both actual islands and islands of habitat), where prey populations are already low or stressed by other factors, or in natural areas where cat colonies are established. Competition with native predators, disease implications for wildlife populations, and pet owners’ attitudes toward wildlife and wildlife management also are important issues.” Houses in forests perforate—if not fragment—the forest, proliferating edges. Many of these houses support nonnative plants and animals so the proliferation of effects of these nonnative species on native species is further exacerbated. Indeed, many landowners have planted exotic plants specifically to attract wildlife, which may have long-lasting adverse effects on native plant species.

## EDGE GEOMETRY

The effect of an interface between patches on habitat quality for a species is often dependent on the amount of edge occurring within an animal’s home range. Size matters—both the size of the patches constituting the edge, as well as in the length of the edge between patches. Consider the patches in Figure 15.5. A 10 ha square patch has 1265 m of edge. But two 5-ha patches have 1789 m of edge, and five 2 ha patches have 2828 m of edge. So minimizing edge for forest interior species can be achieved by having fewer large patches, and maximizing edge for edge specialists can be achieved by having many smaller patches.



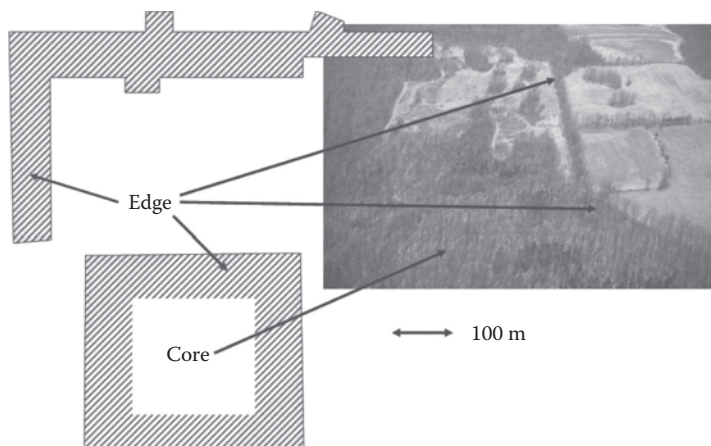
**FIGURE 15.5** Decreasing patch size but keeping total area constant increases the total amount of edge. Increasing patch size per habitat area increases core.



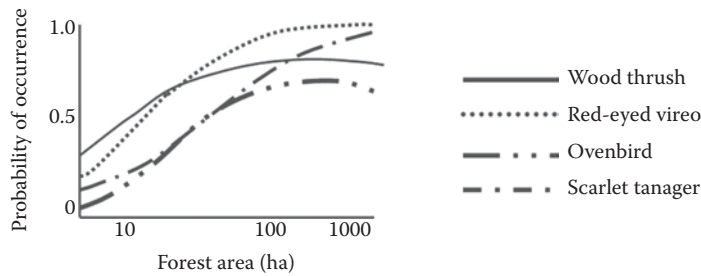
**FIGURE 15.6** The more that a regular shape diverges from a circle the greater the edge per unit area and the less core that is available.

*Edge density* (edge length per unit area) can also be influenced by patch shape. Consider the 10 ha patches that all have the same area as shown in Figure 15.6. The more that a patch diverges in shape from a circle, the greater the edge that is represented per unit area. Patches shaped like amoebas have more edges than regular shapes. So when identifying stand boundaries, stands that are large and approximate a circle or hexagon in shape, will minimize edge and maximize *core* (interior) conditions. Patches that are small and irregularly shaped will minimize core and maximize edge. Core conditions are usually identified by considering the area of a patch some distance away from an edge, say 100 m. It should be clear by now that the distance away from an edge that constitutes a functional core varies from one species to another. For some species it might be 5 m, and for others 500 m. For species that are particularly edge-sensitive, the amount of core area in small or irregularly shaped patches can easily become zero (Figure 15.7). Of course, if we think about landscapes as gradients and not patches, then there is no need to think about a band or buffer on an edge, but rather a gradient of habitat quality across the edge.

The arrangement of patches on landscape can also influence edge conditions. Consider a set of patches in the pattern of a chessboard. If all the white squares are acceptable as habitat and the black ones are not, then edges are maximized. Now place all the white squares on one side of the



**FIGURE 15.7** Depending on the functional width of an edge, then some patches can have no core area. Size, shape, and edge width dictate the amount of core.

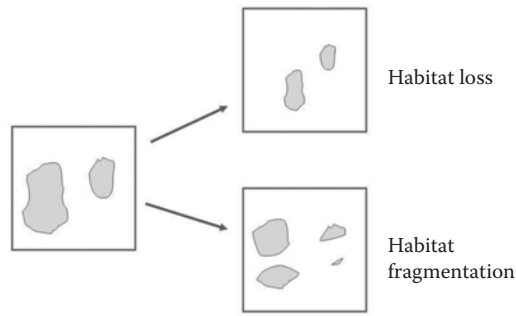


**FIGURE 15.8** As the area of a forest patch increases, the probability of detecting forest interior bird species increases. For some species, very large patches are needed before there is a high probability of occurrence of these species. (Adapted and redrafted from Robbins, C.S., D.K. Dawson, and B.A. Dowell. 1989. *Wildlife Monographs* 103:1–34.)

chessboard and the black squares on the other side. By blocking patches of similar habitat quality you can increase overall patch size and minimize edge area. Pattern can also influence edge contrast. If a decision is made to minimize edges by systematically harvesting adjacent stands, using a system for clearcut regeneration across a watershed, then the contrast between recent clearcuts and various ages of regenerating clearcuts will be less abrupt. Alternatively, if we wanted to maximize induced edges using this system, we would disperse clearcuts throughout a watershed in a “staggered setting” approach, a technique commonly used on National Forests in the United States for years to improve edge conditions for game species and distribute the clearcut disturbances. The decisions of how to make large patches or small patches during management can be based on the area required by various species or by the area impacted by various disturbances. Even small disturbances can have significant effects on some forest interior species, because the area requirements for the species extend far beyond the territory size of the species, due to adverse edge effects. Consider four species of forest-interior birds from eastern United States (Figure 15.8). For some species, the patch area must be thousands of ha before there is a high degree of confidence that the species will be found in the patch. Since the territory sizes for these species are in the range of 1–10 ha, demographic processes associated with avoidance of edges and associations with core conditions are assumed to be largely responsible for these species being associated with such large contiguous forest patches. At least that is the case in landscapes where forests are interspersed among agricultural lands such as in eastern United States (Robbins et al. 1989). There seem to be less strong associations with large patch sizes for small neotropical birds in forested landscapes where older forests are interspersed with various ages of younger forests (Welsh and Healey 1993, McGarigal and McComb 1995). Agricultural lands provide a very stable *matrix* within which forested patches are embedded and may represent a context for the forests, which results in significant adverse edge effects that extend into the forested patches. In forested landscapes interspersed with various aged forest stands, these edge effects do not seem so significant for many forest-interior species, probably because stands are dynamic. Edge contrast can be ameliorated by rapid tree growth on some sites. On high-quality sites, forest structure and composition changes rapidly, minimizing the length of time that high contrast edges occur on the landscape.

## HABITAT FRAGMENTATION

*Habitat fragmentation* is a landscape-level process in which a specific habitat is progressively subdivided into smaller, geometrically more complex, and more isolated fragments as a result of natural and/or human causes. It involves changes in landscape composition, structure, and function at many scales and is overlain on a mosaic created by changing landforms and natural disturbances. Habitat fragmentation is related to but different from habitat loss (Figure 15.9). Habitat can

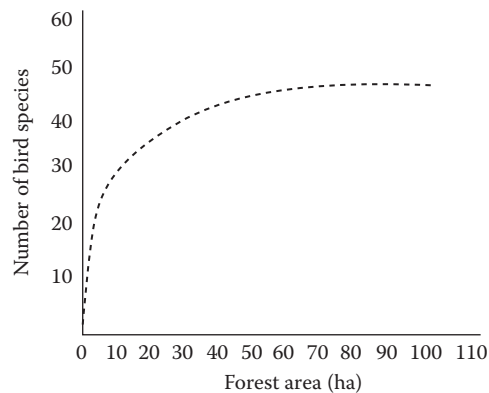


**FIGURE 15.9** Habitat loss does not necessarily increase the isolation or geometry of patches, while fragmentation is a process that increases patch isolation and exacerbates edge effects. (Redrafted and adapted from Fahrig, L. 1999. *Forest Fragmentation: Wildlife and Management Implications*, Pages 87–95. Brill Press, Netherlands.)

be lost because of natural disturbances, human actions, or ecological succession. Loss can occur in a manner that erodes the edges of a large patch, such that habitat area declines, but the patch is still intact and it is not dissected into smaller, more isolated pieces. Alternatively, a large patch can be dissected by a road, utility right-of-way, or clearcut, and then another, and then another, until there are many smaller patches on the landscape and not just one large patch. Habitat area has declined, but, more importantly, habitat patches have become subdivided and more isolated from one another. This latter process is termed fragmentation; the effects of this process are clearly specific to a species. Fragmenting habitat for ovenbirds may be caused by reducing the size and connectedness of mature eastern hardwood forests. Fragmenting habitat for bobolinks may be caused by allowing forests to encroach upon grasslands. And fragmentation for both species may be caused by expanding housing developments and roads. The process of fragmentation is species-specific, though it is often generalized in the popular literature. Forest fragmentation: How often have you seen that term? But fragmenting a forest can be good, bad, or indifferent, depending on the structure and composition of the forest and the species with which you are concerned. Indeed, some forests are fragmented from natural disturbances on a regular basis and others rarely so. The effects of the loss of habitat, separate from the fragmentation of habitat, must be considered carefully to understand the potential impacts on animal and plant species (Wilson et al. 2009). There also is evidence of threshold effects in habitat loss, and isolation for some species of birds, suggesting that changing habitat area or isolation does not change habitat quality in a linear manner (Betts and Villard 2009, Betts et al. 2010).

### HABITAT AREA: SPECIES–AREA RELATIONSHIPS

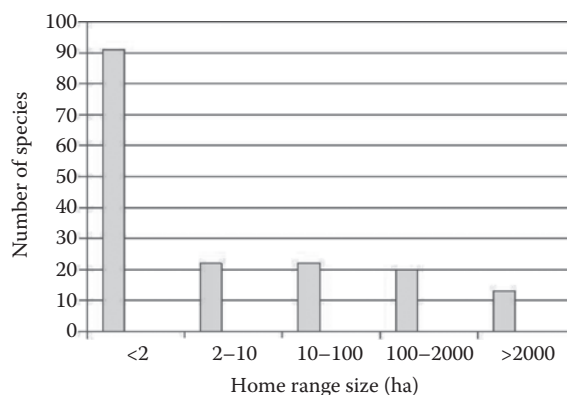
Forest fragmentation can occur from natural disturbances or human activities. Indeed, we would expect that many, if not all, species should be adapted to the frequency, severity, and size of natural disturbances in the landscapes where they occur. We often use this assumption as the basis for designing human disturbances in forests to minimize the risks to species, both known and unknown, through our management actions (Landres et al. 1999). Simply from the standpoint of supporting species richness (number of species in an area), we know that large patches of forest (or grassland) support more species than small patches of forest. All other things being equal, large oceanic islands support more species than small islands. There is typically an asymptotic relationship between area and species number (Figure 15.10). Hence the conventional wisdom, when considering identification of reserves designed to meet the needs of many species, is that large areas—areas where an asymptote in species-richness is reached—represent appropriate areas to “capture” the most species in a region. This approach to conservation of biodiversity is dependent



**FIGURE 15.10** Example of a typical species-area relationship for forest birds among patches in a landscape. Species richness becomes asymptotic at large patch sizes.

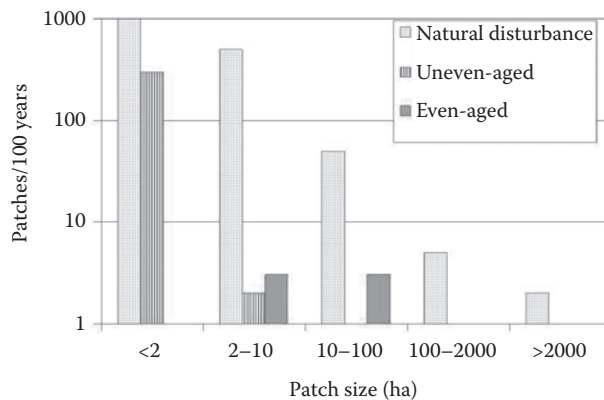
on two spatial scaling properties: (1) the range of spatial scales represented following disturbance and regrowth across a landscape, and (2) the range of spatial scales represented by home ranges and populations of organisms occurring on a landscape.

As we have discussed already, natural disturbances of a particular severity typically occur over a range of sizes within a region or forest type. Less frequent disturbances tend to be larger and more severe. The combination of forest regrowth and disturbance produces a range, or domain, of spatial scales that are associated with the severity and frequency of a disturbance. The exact point within this domain, that any point on the landscape might experience, is dependent on many factors and is not deterministic but rather probabilistic. There is some probability that a disturbance will be of a particular size or severity, but we cannot predict exactly what it will be or when it will occur with certainty. Rather, it is this domain of sizes that we can predict, and it is the domain of sizes to which most species in the region should be reasonably well adapted to using. How those species react to the severity and size of the disturbance is largely dependent on the spatial domains represented by the suite of species occupying an area. For instance, if we look at the frequency of occurrence of species' home range sizes among all species in a region (Figure 15.11), it is typically a negative exponential distribution, with many species occupying very small portions of a landscape, and few species occupying large portions of a landscape. The spatial domain of the cumulative species



**FIGURE 15.11** Frequency distribution of home range sizes for species in the northeastern United States. (Summarized from DeGraaf, R.M. and M. Yamasaki. 2001. *New England Wildlife: Habitat, Natural History, and Distribution*. University Press of New England, Hanover, NH, 496 pp.)

area requirements defines a domain of spatial scales that would need to be represented across large areas to meet the needs of the full suite of species. The domain of disturbance sizes in a landscape provides the template upon which the species can or cannot find the habitat areas and arrangements that meet the cumulative needs of the group of species. The interface between the spatial domains inherent in a landscape (from disturbances and regrowth) and the spatial domain of the animal community (from home ranges of multiple species) cause some landscapes to be better able to support a more full range of species than others. Now consider human impacts on landscape pattern and composition. As land-use developments, such as roads, housing, forest management, and agriculture, change the pattern of forests across a landscape, the domain of spatial scales represented in a forested landscape changes and the domain is often narrowed. Forest management occurs over a narrower range of spatial scales (fewer very small patches, fewer very big patches, Figure 15.12) than natural disturbances. The spatial scaling properties of the animals that use a landscape pattern that is constrained by land use may change somewhat in response to these new landscape configurations, but unless a species is preadapted to this new narrower range of conditions, it may find that its habitat availability has declined. This reduction in habitat for some species is caused by having a narrower range of patch sizes and arrangements than occurred prior to modern human intervention (Figure 15.12). Small patches may be less common because humans tend to homogenize stand conditions, making them easier to manage and producing uniform products within and among stands, unless we take actions to make stands heterogeneous (see Chapters 7, 8, and 9). Large patches are also less likely. Because of social constraints, we are unlikely to manage an entire 10,000-ha watershed using even-aged management to replicate a wildfire, but some species are well adapted to that forest condition at that spatial scale (especially if a complex early seral condition is provided). We often are managing forests and the animals that occur in them over a limited spatial domain. Those species adapted to conditions (stand types, edge densities, legacy material, etc.) occurring at smaller and larger spatial scales than those represented by management can be considered those most likely to be at risk of having decreased habitat quality (Figure 15.12). If we can identify those species, then we can take management actions to address their needs. Species needing large, complex, early successional patches, for instance, may have their needs met if even-aged systems with legacy retention are employed by clustering harvest units over time that collectively contribute to overall large patches, leaving late successional forests, also, in large patches. For those species that require small patches, then variable density planting and thinning or group selection systems using a variety of group sizes may help to address their needs. These silvicultural and harvest considerations must



**FIGURE 15.12** In this range of patch sizes one might expect over a 10,000 ha landscape under natural disturbance regimes and the range of conditions represented by even-aged and uneven-aged forest management approaches. Species associated with patch sizes under-represented by management may face risk unless the range of spatial scales of management is increased or reserves of various sizes are employed within which natural disturbances are allowed to occur.



be planned to ensure that a broader spatial representation is achieved over a landscape than would be achieved simply by maximizing timber profits. That is, planning to represent a broad spatial domain may come at an economic cost when managing a landscape because moving the landscape into an appropriate spatial domain may mean cutting stands earlier or later than would be done to maximize profits in order to achieve the desired patterns and reduce the risk of some species losing habitat availability.

Just as there are spatial domains of disturbance and regrowth, there also are temporal domains of disturbance and regrowth, and there are temporal domains of population demographics. Some species produce several generations in one year, and others produce several generations in decades or centuries. Consequently, the rate of change in forest structure and composition over a landscape, as well as the spatial patterns, can affect species differently. For instance, species adapted to prolonged forest recovery following a disturbance that retains significant legacy from the previous stand may find diminished habitat quality where forest managers are striving to minimize legacy retention and maximize the rate at which growing space is occupied by economically valuable trees. Vegetation management and density management can shorten the period of a diverse, early successional condition and reduce the opportunity for species to find and use this recently disturbed area. Further, intensive early stand management can minimize the time available for them to produce sufficient offspring to colonize new areas that grow into habitat elsewhere on the landscape over time. Similarly, species associated with post-rotation-aged forests may have limited opportunities to produce young. This is particularly a problem with *k*-selected species, which are species that reproduce late in life, produce few young, and reproduce infrequently. Long-lived species, such as tortoises and parrots, require some level of habitat stability over long time periods in order to meet their reproduction and survival requirements. If forest management or forest land use changes the duration of a particular seral stage, then habitat availability for species associated with habitat elements found in those stages can be reduced. Of particular concern are those species that are not only long-lived but also need to produce multiple generations to ensure population viability within a region.

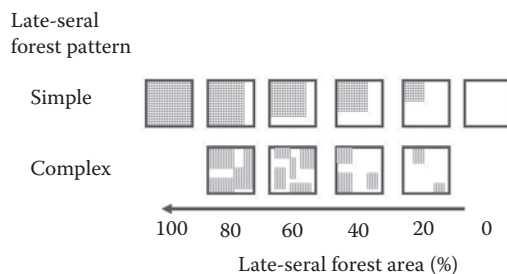
One way of reducing the impact of changing the temporal domain of scale is to provide legacies from one forest condition to the next over large areas and allow some habitat elements to be represented over large areas over time (e.g., dead wood in managed forests), thereby buffering the diminished domains of scale represented among forest age classes. Attending to the needs of species that are long-lived or have multi-generational requirements to maintain viability may represent a cost to the resource manager not only in silvicultural activities and harvest planning, but also in a commitment to legacies of structures and patch types across managed landscapes. It is the intersecting domains of space and time in landscape composition and structure, which interface with the spatial and temporal domains of a collection of species occurring on the landscape, that must be coordinated to minimize the probability that some species will be placed at risk. This approach greatly complicates harvest planning but it is possible to find harvest-planning solutions to spatial problems such as these (Bettinger et al. 2001).

## **CASE STUDY: HABITAT AREA OR PATTERN?**

In the early 1990s, Dr. Kevin McGarigal conducted a study that addressed the relative role of forest area vs. forest pattern in structuring bird communities in the Oregon Coast Range (McGarigal and McComb 1995). Subsequent work by Dr. Karl Martin also addressed associations with mammal and amphibian communities on these sites (Martin and McComb 2002, 2003). The work was set in the dynamic forests of the Oregon Coast Range and followed work by Robbins et al. (1989), Temple (1986), and others, who identified a number of area-sensitive bird species in eastern hardwood forests that seemed to be adversely affected by processes of forest fragmentation resulting from agricultural land uses and urbanization. Few studies had addressed fragmentation effects on vertebrates in managed forest landscapes at that time, and many of the findings regarding the effects

of fragmentation in eastern hardwood forests were being used to design management strategies for forests in western coniferous forests. McGarigal and Martin wanted to see whether these same trends were consistent in forests that were fragmented by forest management, while controlling for habitat loss. The experimental design was to sample three replicates of 10 sub-basin (250–300 ha) conditions representing two dominant gradients in late seral forests: 0%–100% late-seral forest area with minimal fragmentation (remaining forest was in large blocks) and also sub-basins with a high level of late-seral forest fragmentation (remaining forest in many blocks) (Figure 15.13). But forests are not structured as simply as is represented in this experimental design. Patch shapes and arrangements vary considerably from one sub-basin to another so indicators of landscape structure and function were needed. Consequently, McGarigal and Marks (1995) developed an analytical computer program called FRAGSTATS to provide a huge number of landscape metrics to describe many aspects of patch conditions, sizes, shapes, connectedness, edge conditions, and core areas, among others. Using these metrics in their analyses allowed them to understand whether the animals were associated more with the composition of the landscapes (how much late-seral forest was available), or if pattern mattered. On the basis of over 100,000 observations of birds, mammals, and amphibians, several patterns emerged. For many bird species, habitat area matters. Of 15 species of birds that selected late-seral forests (based on use and availability), 11 were associated more with the *area* of late-seral forest than with its *configuration*; (more habitat was better than less). These included species such as brown creepers, pileated woodpeckers, and varied thrushes (McGarigal and McComb 1995). Five species of amphibians also were associated more with the area of a forest condition than its configuration, such as the southern torrent salamander (Martin and McComb 2003). And six species of mammals were more associated with patch area than configuration, including California red-backed voles, deer mice, creeping voles, Pacific shrews, and Pacific jumping mice (Martin and McComb 2002). None of the 14 mammal species captured were negatively associated with edges (Martin and McComb 2002).

But some species were clearly associated with configuration. Most species that exhibited an association with fragmented forests were more abundant in fragmented than unfragmented forests, unlike what was seen in many eastern hardwood forest studies. Species such as olive-sided flycatchers, deer mice, and Pacific jumping mice were positively associated with edges. Very few species were associated with a core area >100 m from an edge than with the edges; Pacific giant salamander was one of these core-associated species. As you would expect, there are species that likely would benefit from fragmentation of late seral western coniferous forests—and those that likely would not. The fact that landscape pattern (fragmented forests) seemed to be only modestly associated with the abundance of many of these species, and in most cases was more associated with fragmented than unfragmented forests, may seem counterintuitive but remember that these forests are dynamic. Harvested stands in these studies were regrowing. Plantations that had regrown even 20 years had reduced edge contrast and provided a matrix condition through which many species could once



**FIGURE 15.13** Experimental design used to assess the contributions of late-seral forest area and pattern to bird, mammal, and amphibian species in the Oregon Coast Range. (Redrafted from McGarigal, K. and W.C. McComb. 1995. *Ecol. Monogr.* 65:236–260. With permission from the Ecological Society of America.)

again disperse. Indeed, in landscapes where the matrix is static (e.g., agricultural lands) and the forest patches are isolated, the results of McGarigal and Martin might have been much different. It is also important to point out that McGarigal now is developing landscape metrics that measure gradients in landscapes rather than patches (McGarigal et al. 2009). If he were to be designing the project now, patch delineation would likely give way to gradient quantification (Cushman et al. 2010).

Further, animals dispersing across fragmented landscapes face different challenges in landscapes with little habitat available than when even marginal habitat is available between high quality patches. Despite the results reported above, there is reason to consider pattern effects on populations, especially under conditions of low habitat area (With 1999). That is the subject of our next chapter.

## SUMMARY

Forested landscapes have structure, composition, and function. Function for each species varies, depending on how habitat patches are defined on the landscape and how they are connected across it. The scaling properties of a landscape are defined by the grain (the smallest unit of space that we find useful), the extent, or the outer boundaries of the landscape under consideration, and the context—the condition of the area surrounding the extent. Animals integrate patches of varying habitat quality across the complex mosaics of landscapes to meet their habitat needs. As landscape structure, composition, and function change, so will populations and the fitness of the species using the landscape.

Edges between successional stages (induced), or plant communities (inherent), represent zones in a landscape where the richness of animal species can be high, but adverse effects on forest-interior species are exacerbated. Predation, brood parasitism, spread of invasives, and altered microclimates along edges can lead to depressed fitness for forest-interior species.

Fragmentation is the process of breaking habitat patches into smaller, more complex, and more isolated pieces. This process is different from habitat loss, where habitat reduction may or may not be accompanied by increased isolation. In dynamic forest systems, where the matrix condition among habitat patches is continually changing, habitat loss is often a more serious issue than fragmentation. But once fragmentation has led to significant isolation of the remaining patches, then fragmentation effects may become apparent, especially if the matrix is inhospitable to the organism of interest.

Finally, it is important to realize that the conceptual framework and analytical approaches are changing when addressing habitat across landscapes. Scientists are now viewing landscapes as a set of gradients and seeing that associated measures of complexity, called surface metrics, may more accurately describe the complexity in landscape composition and pattern.

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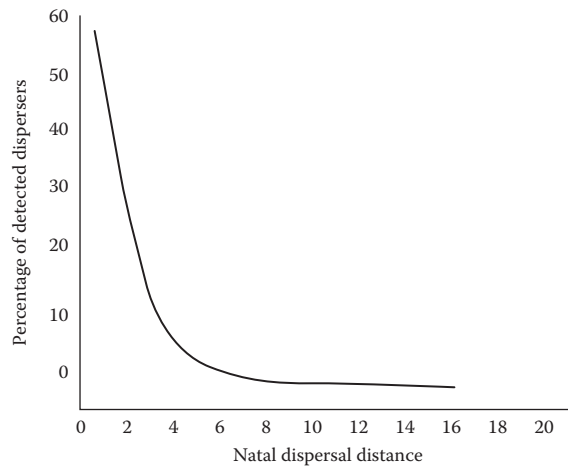
# 16 Landscape Connections

Think for a moment about where you were over the past 24 h. Draw a map and on it place all the locations of those places where you spent more than 5 min in the past 24 h. Now draw a straight line from one place to the other in the order they were visited. Do these straight lines represent your path from one place to another? Probably not. Most likely you used roads, sidewalks, doors, and hallways to get to where you were going. These connections from place to place are similar in some respects to the connections used from patch to patch for some species of animals, but not all. Consider how you would use a compass or global positioning system (GPS) to find your way through an unfamiliar forest. You take a bearing, know a distance, and then walk through whatever is between you and your goal (within reason), more or less in a straight line. Some of the walking through closed canopy forests will be easy, and other places with dense shrubs will be very difficult. So rather than using a connection, you walk in a straight line with intervening patches representing various “risks” to your walking ability. These two types of movement are similar to the types of dispersal that other animal species experience. Some use corridors or connections across a landscape and others disperse across the *matrix* or the intervening conditions between patches of suitable habitat. And the matrix poses various risks to survival during dispersal depending on its habitat quality. This is analogous to a differentially permeable membrane in which some molecules can pass through the membrane easily and others cannot. For dispersing animals, matrix conditions represent differentially permeable conditions for moving organisms. Some intervening conditions allow the animal to move easily during dispersal, others present significant risks to survival (Martin and McComb 2003, Popescu and Hunter 2011).

## DISPERSAL

Animal movements within a home range often follow paths that are used repeatedly. Deer, elk, and moose, for instance, develop well-worn paths through forests when moving to and from cover and food. Even birds repeatedly use somewhat predictable flight paths in their daily movements (Bélisle and Desrochers 2003). But when an animal is displaced from its home range by its parents (natal dispersal), by dominant individuals, or by a disturbance that alters the quality of habitat in the home range, then the animal disperses in search of a location to establish a new home range. Obviously an animal would not want to disperse any farther than needed, or it is expending energy unnecessarily and reducing its probability of survival. On the other hand, it should not settle into a sub-optimal home range if a better home range means spending only a bit more energy. How is it that animals find these potential home ranges and decide to settle in one?

Dispersal capabilities vary widely among species, likely being quite limited and more affected by boundaries for some species considered dispersal specialists (e.g., red tree voles; Hayes 1996) and quite extensive for others considered dispersal generalists (e.g., Pacific jumping mice) (Gannon 1988, Hansson 1991). Dispersal, especially natal dispersal, is a process that is critical to the growth and spread of populations, stability of metapopulations, recolonization of vacated habitat, and flow of genes and bodies among sub-populations. Body mass and trophic level seem to influence natal dispersal distances in mammals; large species disperse farther than small ones and carnivores disperse farther than herbivores and omnivores (Sutherland et al. 2000). Similar relationships have been documented for carnivorous bird species but not for herbivorous or omnivorous birds (Sutherland et al. 2000). Bowman et al. (2002) found that dispersal distances are often associated with the home range size of the animal more than with the body mass of the animal, but recent work by Whitmee



**FIGURE 16.1** When dispersal distances are scaled to the median dispersal distance for the organism (in this case 29 species of mammals), a negative exponential pattern emerges. (Redrafted from Sutherland, G.D. et al. 2000. *Conservation Ecology* 4:16. With permission.)

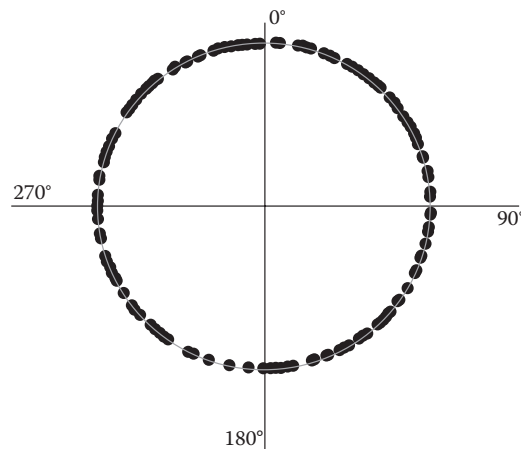
and Orme (2013) found that home range size, geographic range size, and body mass best predicted natal dispersal distances in mammals. Birds and mammals tend to have a dispersal pattern that follows a negative exponential curve; that is, most animals disperse a small distance (relative to the median distance dispersed by the species) and few individuals disperse long distances (Figure 16.1). For many species dispersal distances decline as the number of home-range diameters increase (Bowman et al. 2002). This is an intuitively appealing way to consider dispersal because animals cruise around their home ranges regularly. To move another home range diameter would not seem to be too energetically costly, but as dispersal continues over multiple home-range diameters, the ability of the organism to expend that energy and still survive comes into question quickly. It may be more energy-efficient for an animal to settle into a suboptimal patch and survive there than to continue to disperse and risk the chance of dying by dispersing through low-quality patches.

So what conditions are necessary for successful dispersal across a landscape to allow recolonization of a patch or introduction of new alleles into a population? The probability of successful dispersal across a landscape can be considered a function of the probability of three interacting probabilities: encounter, survival and continuing. To examine the concepts behind successful dispersal, let's first consider a simple conceptual model, and then we will complicate it with reality.

### UNDERSTANDING THE PROBABILITY OF SUCCESSFUL DISPERSAL

There are three dominant effects that determine if an individual dispersing from one patch is likely to reach another patch where it can find the resources to survive. Let's consider the patch from which it is leaving the *source patch* and the patch into which it could settle the *target patch*. First, an animal must travel in a direction that causes it to encounter a target patch. Many animals do not seem to disperse in a given direction, but rather travel in random directions (Figure 16.2). Although the actual path that an animal takes may appear to be like a pinball in a pinball machine, when it settles it will likely be in some random direction unless some totally inhospitable condition (i.e., the ocean for most terrestrial vertebrates) exists in one direction (Figure 16.3). Consequently, the probability that a dispersing animal will travel in a direction that would lead it to even encounter a target patch is a function of the angle formed from the center of the source patch to the outermost edges of the target patch (Probability of encounter =  $P_e$ , Figure 16.3). In our case, if the angle was  $36^\circ$  then the probability of an animal encountering the target from the source would be 0.10% or a 10%

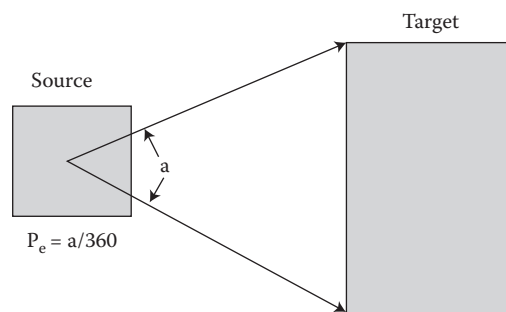




**FIGURE 16.2** Dispersal directions of 151 juvenile male white-tailed deer in Pennsylvania. (From Long, E. S. et al. 2010. *Journal of Wildlife Management* 74:1242–1249.)

chance. Another way of looking at this is that for every 100 animals dispersing at random from the source, approximately 10 would be expected to encounter the target. Now consider what happens to that angle if the target patch were closer to the source—the angle increases and the probability of encountering the target increases. Increase the size of the target and the probability of encounter increases. Increase the number of targets in various directions and the probability of encounter increases. So, from this very simple example, we can see that if we want to maintain a metapopulation through dispersal, then more and larger target patches close to the source patch will increase the likelihood of maintaining the population. But dispersing animals not only have to encounter the target, they need to survive the trip.

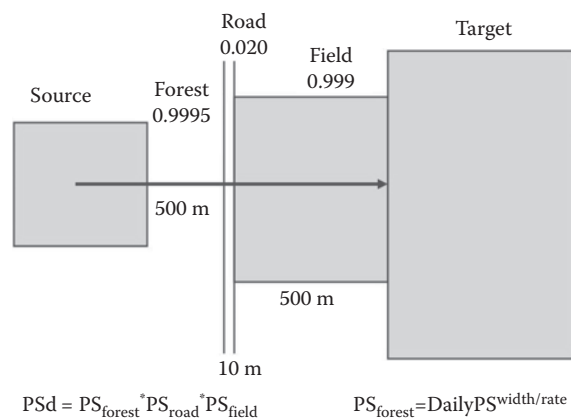
The probability that an animal will survive a dispersal event is dependent on its fitness as it moves from one patch type to another. Recall that fitness is related to the probability of surviving and reproducing in a patch. When dispersing, reproduction is usually not a concern, but survival is a requirement. Consequently, if we can understand what the probability of survival might be for a species in each intervening patch as it moves across a landscape, we can predict the likelihood that it will survive its dispersal trip. One way of measuring this survival is using a time-specific probability of survival, such as its daily probability of survival. Do you know what your daily probability of survival is? I assure you that the life insurance companies have estimated this for you! It is not



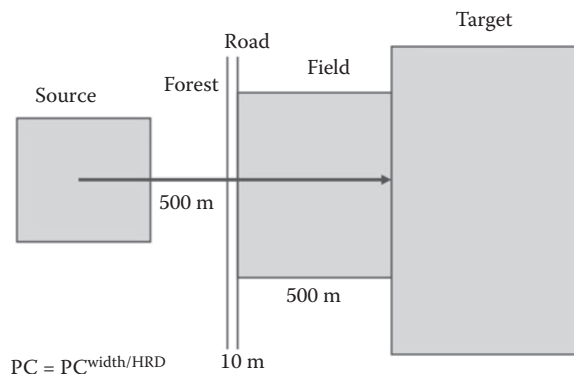
**FIGURE 16.3** Conceptual diagram illustrating the probability of an animal encountering a target patch if dispersal is random. As the target increases in size or decreases in distance from the source, the probability of encounter increases.

100%, unfortunately; you just never know...; but it is high. Say your daily probability of survival is 99.9%, then you would have an annual probability of survival of  $0.999^{365} = 0.694$  or a 69.4% probability of living a year. Not great odds! Now consider how that might change if you were living in Antarctica for that year, or Detroit, Michigan (the most dangerous city in the United States in 2012) vs. living in Plano, Texas (the safest city in the United States in 2012). Where you are matters, and for dispersing individuals with no prior knowledge of the resources available in intervening patches, the probability of survival can fluctuate considerably along its trip. But the overall probability that an animal would survive a dispersal event is not only related to the time-specific difference in survival probabilities among the patches along its route, it is also a function of the time it spends in each patch. Consequently, the movement rate through each intervening patch must be considered. So the probability of survival in a patch is  $PS_x = PSD^d$ , where  $PS_x$  = probability of survival in a patch  $x$ ;  $PSD$  = daily probability of survival in patch  $x$ , and  $d$  = the number of days spent in patch  $x$  (Figure 16.4). For a dispersing individual,  $d$  will be dependent on the rate of movement (distance per day), and the distance traveled in the patch. If the species dispersing is a salamander and patch  $x$  is a closed canopy forest that would protect it from dessication, then it could probably spend quite a long time in patch  $x$  and survive (assuming it can find food). But if patch  $x$  is an interstate highway, then the salamander can run as fast as it can and spend as little time on the highway as possible and still face a very low probability of survival. But even in this case, that probability is not zero. Send enough salamanders across the highway and a few will make it (humans have also used this approach—this was the basis for a common military strategy for invasion during wars!). Nonetheless, these patches with an inherently very low probability of survival are considered *barriers*; those with high probabilities of survival can be considered *corridors*. But it is most useful to think of these intervening patches as having differential permeability or resistance to moving organisms. It is the product of the probabilities associated with all intervening patch types that yield an overall probability that the animal will survive the dispersal event (Figure 16.4). Species that are more mobile and have higher movement rates will realize higher survival rates in low-quality patches than less mobile species. Consequently, forest managers concerned about maintaining regional biodiversity tend to be concerned with providing habitat for low-mobility species such as salamanders, frogs, and small mammals (Spies et al. 2007).

There is still one other factor that must be considered when understanding how likely it is that an animal might travel to a target from the source. Most animals do not disperse more than one home-range diameter from their natal home range. Only a few go two home-range diameters, and fewer go three home-range diameters. The farther the distance between the source and a target patch, the



**FIGURE 16.4** Conceptual framework for understanding the likelihood of an animal surviving a dispersal trip through various patch types, each with its associated own time-specific probability of survival. Barriers are those patch types with very low daily probabilities of survival.

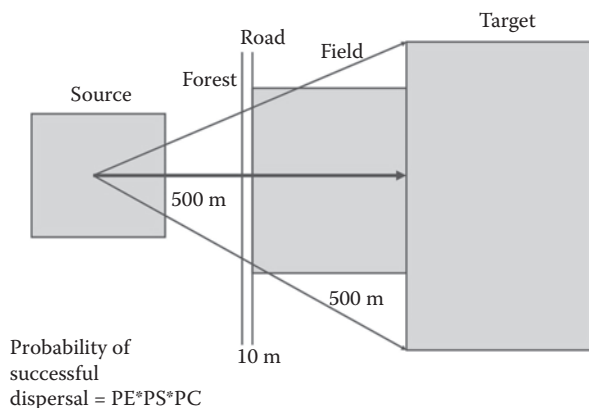


**FIGURE 16.5** Conceptual framework for understanding the likelihood of an animal continuing a dispersal trip through various patch types, each with its associated own time-specific probability of survival. Settling into a patch with low fitness may be a reasonable survival strategy but may not allow a metapopulation structure to persist.

greater the likelihood that the animal will settle and perhaps settle in a low-quality habitat, perhaps even a sink. Consequently, the probability of continuing the dispersal (1-probability of settling) also influences the likelihood that an animal will make it to the target (Figure 16.5).

If you put these three factors together (the product of the probabilities) to understand the likelihood that a specific patch on the landscape might be colonized by individuals from another patch, the likelihood becomes overwhelmingly small as the distance between patches increases, as the size of the target patches decreases, and as the intervening habitat becomes more inhospitable (resistance increases) (Figure 16.6). Some of the problems associated with low survival probabilities can be overcome by the sheer numbers of dispersers. For example, consider that the number of successful dispersers among patches is a function of the cumulative probability of survival per individual among all patches crossed, animal density in the source patch, source patch size, and the percent of the population that are dispersers, then:

$NSD = PS * AD * AREA * PROP$  where,  $NSD$  = number of successful dispersers,  $PS$  = cumulative probability of survival across the patch mosaic,  $AD$  = animal density per unit area,  $AREA$  = source habitat area, and  $PROP$  = the proportion of the population that are dispersers. For species with high



**FIGURE 16.6** Conceptual framework that combines the probability of encounter, the probability of survival and the probability of continuing during a dispersal event from a source to a target. The overall probability associated with an individual dispersing to the target is the product of these probabilities.

reproductive rates, a low probability of successful dispersal may not be such a huge obstacle because of the sheer numbers of dispersers—it is just that most dispersers die. Plants use this strategy quite often. But for species that have low reproductive rates, are rather immobile, occur at low densities, and may have daily survival probabilities that vary considerably among patch types, isolation can be a significant problem. These are often the species of most concern to conservation biologists wishing to conserve regional biodiversity. So what can be done about this problem? We can manage our forests to influence the connectivity among the habitat patches on a landscape for a species.

## CONNECTIVITY AND GAP-CROSSING ABILITY

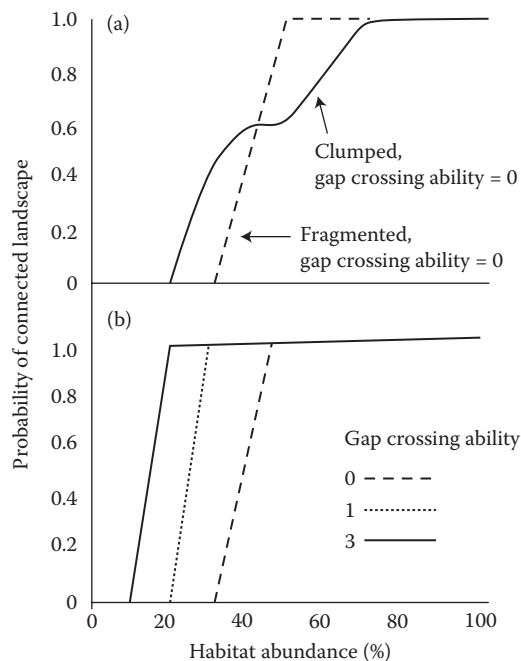
*Connectivity* refers to the degree to which the landscape facilitates or impedes movement among habitat patches and, therefore, the permeability or resistance of the landscape to dispersing individuals, as measured by their survival and by passing genes on to the subsequent generations. Connectivity is largely a function of the size and arrangement of disjunct patches (i.e., area and isolation effects), the permeability of the intervening patches, and/or the physical connections among habitat patches using corridors.

Where a barrier occurs on a landscape impeding the movement of dispersing organisms in a particular direction, a corridor may facilitate movement across this barrier. Overpasses and underpasses built specifically for animals crossing highways have been quite successful in many areas (Jackson and Griffin 2000).

In theory, some of the negative effects of isolation can be mitigated by identifying and maintaining fixed or dynamic connections across the landscape. Connectivity can be managed as either *continuity*, the physical connectivity of habitat, or *connectedness*, the functional connectivity of habitat. Corridors represent one type of connection that can be designed for species with a poor ability to cross gaps. Maintaining a more hospitable matrix condition between patches is an alternative to corridors. In either case, it may be helpful to think of the demographic basis for developing connections. Using the life history characteristics of species can allow us to understand which species are relatively more at risk from isolation effects compared to others. Maintaining connectivity for species that show adverse effects from isolation (e.g., California red-backed voles) may be a much higher management priority (Mills 1995) than species that disperse across complex landscapes more freely (e.g., deer mice).

What ultimately influences the connectivity of the landscape, from the organism's perspective, is the scale and pattern of movement relative to the scale and pattern of intervening patches (With 1999). The size, number, and distribution of habitat patches influence the physical connectivity of habitat across the landscape and are the primary determinants of connectivity. So under what conditions does connectivity become an issue? Clearly, if most of a landscape is a high-quality habitat for a species, then there should be little concern about connectivity. But if only 1% of a landscape is high-quality habitat in many tiny patches, then connectivity may be a huge concern. Determining the point at which a reduction in habitat area leads to a disconnected habitat would seem to be an important piece of information when managing species in complex landscapes. Dr. Kimberly With developed a model to at least conceptually understand the relationship between habitat area and connectivity (With 1999).

With (1999) developed a simple landscape structure in which black squares were habitat and white squares were not habitat for a species and then used a simple percolation model (testing connectedness from one side of the landscape to another) to assess at what point(s) we would expect to see habitat connectivity disappear from a landscape. Habitat loss and fragmentation were simulated by removing habitat at random in one set of simulations and in clumps in another set of simulations (Figure 16.7a). As With (1999) decreased habitat area in the randomly fragmented landscapes, she observed a threshold at 50% habitat area, where connectivity declined abruptly and fell to zero connectivity when approximately 30% of the habitat area remained (Figure 16.7a). The probability of



**FIGURE 16.7** Simulated effects of landscape fragmentation in two patterns on the probability of maintaining a connection across a landscape (a) as affected by the species gap-crossing ability (b). (Redrafted and adapted from With, K.A. 1999. *Forest Fragmentation: Wildlife and Management Implications*, Pages 97–115. Brill Press, Netherlands.)

having a connected landscape in clumped fragmentation landscapes began to decline slowly when less than 65% of the habitat remained and fell to zero at about 20% of the habitat remaining. The latter finding was due to the random chances of losing a connection vs. maintaining a connection being highly influenced by the pattern of clumps of habitat on the landscape, with clumps more likely to be either connected or disconnected compared to random fragmentation. Consequently, for species that require a connection across a landscape, connectivity would seem to become an issue when about 30%–50% of the landscape remains as habitat; when habitat area drops below 30%, then maintaining or restoring connectivity is an important consideration for the species.

Some species do not need a complete connection to move across a landscape. Some species are quite well-adapted to crossing inhospitable areas that represent gaps in the connection across the landscape, while others are not (Bélisle and Desrochers. 2003). Some species can cross wider gaps (e.g., goshawks) than others (e.g., clouded salamanders). With (1999) also examined the potential effect of gap-crossing ability on these thresholds. As you would expect, the better able a species is at crossing a gap in habitat across a landscape, the greater the level of fragmentation of its habitat it can tolerate and still be able to move across the landscape (Figure 16.7b). For species with the ability to cross three blocks of non-habitat, the threshold for a disconnected landscape fell to 10%–15%. From this work, several management implications emerge. First, although habitat area is the landscape feature most associated with animal abundance across complex landscapes, connectivity emerges as an important feature when habitat area has declined to 30%–50%, or less, depending on the species gap-crossing ability. Second, when habitat area falls to or below this threshold, planning will be needed to either provide corridors or a matrix condition that is more permeable to the species.

What is missing from the previous example of the interrelationship of habitat area and connectivity is the concept of differential permeability of habitat patches across a landscape. The permeability

of the matrix is not black or white, all or nothing; it is indeed shades of gray in its permeability to dispersing organisms. Permeability influences the degree to which the two patches are isolated or not.

### UNDERSTANDING PHYSICAL AND GENETIC CONNECTIVITY

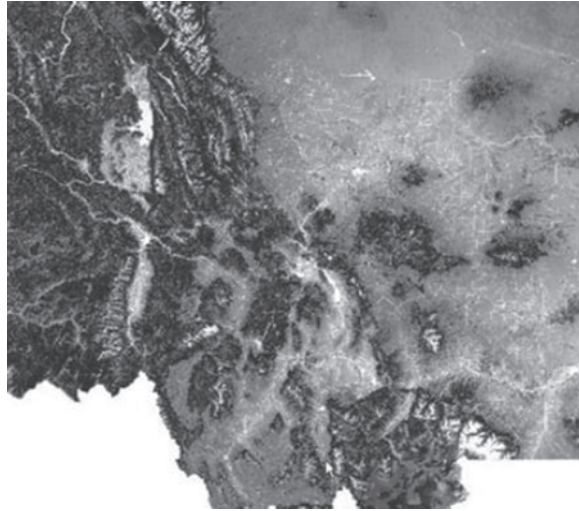
The inverse of permeability to movement across landscapes is resistance to movement across a landscape. There have been recent advances to understanding movement of organisms across complex landscapes using a gradient rather than a patch-based approach. These approaches are based on the complex set of factors at any one point or pixel on a map having the characteristics that make it more or less resistant to movement by an organism. These resistant kernel approaches allow us to begin to understand how animals would be most likely to move across the landscape along pathways of least resistance. For instance, if a herd of caribou were making their way across a boreal forest on their way to summer calving grounds and they encountered the Alaskan oil pipeline (Figure 16.8), would they change their movement patterns or would they be oblivious to it? One can easily imagine that for species associated with shrubs, a regenerating clearcut would have low resistance to movement but a dense plantation may have high resistance, a river may have even higher resistance, and a highway could have even higher resistance, such that a map of resistance would illustrate the potential routes that animals would be most likely to follow (Figure 16.9, Cushman et al. 2006).

As far as we know, animals don't read maps. So, how can we begin to understand if our perception of resistance to movement as portrayed by a model is accurate or even reasonable? If dispersing animals not only find new patches but also find mates in those new patches, then they should leave a trail of their presence through their genes. Recent technologies allow us to examine patterns of species and individuals across landscapes and use genetic characteristics to infer relatedness and flow of genes from one population to the next. Two types of DNA are typically used to address different types of questions. DNA contained in the mitochondria of a cell is inherited from an individual's mother and does not change very much from one generation to another (Morin et al. 2010). The DNA in your mitochondria is very similar to the mitochondrial DNA found by your predecessors from many generations ago. DNA found in the nucleus of a cell however is inherited from both the mother and father. Nuclear DNA contains segments of the DNA strands that are highly variable from one individual to another, and these segments are called microsatellites (Morin et al. 2010). Hence, mitochondrial DNA (mtDNA) is useful for separating one species from another; nuclear DNA is useful for separating among individuals within a species.



**FIGURE 16.8** The oil pipeline traversing Alaska provides a stark contrast to the surrounding forest. It may act as a corridor for some species, a barrier for others, or be differentially permeable to others.





**FIGURE 16.9** Example landscape resistance map for American black bear in an area of the U.S. northern Rocky Mountains encompassing Montana and northern Idaho. Dark areas are low resistance for movement, while light areas are high resistance for movement. (The map was developed by Cushman, S.A. et al. 2006. *The American Naturalist* 168:486–499.)

Say that you find some hair, and the hair has a piece of skin attached to it. If we examined the mtDNA and nuclear DNA found in the skin, we can learn two different things. We can compare the mitochondrial DNA to known samples to determine which species the hair is from. Once we know that, we can compare the nuclear DNA to other samples from that species to determine if the hair you found is related to individuals from other populations. But, a tiny bit of skin may not contain enough DNA to do both of these analyses effectively. Population geneticists amplify the DNA in that tiny sample using polymerase chain reaction (PCR) techniques that make multiple copies of the DNA or microsatellites of the DNA (Morin et al. 2010).

Tracing the trail of genes across a landscape is made possible by identifying segments of a genome that contain a series of base pairs (known as a microsatellite array) which vary from individual to individual, but the variation is less for those closely related to one another than for those that are unrelated (Cushman et al. 2006, Morin et al. 2010). Finding that microsatellite array in a genome can be a challenge unless the series of base pairs on either side of the array are stable and do not change from one individual to another over time. These stable series of base pairs (primers) allow identification of a marker that can help identify an individual and relatedness to other individuals (Morin et al. 2010). This approach is particularly valuable because even small traces of DNA that contain the array can be duplicated using PCR.

If we have two samples from either side of a mountain range (a potential barrier) and the nuclear DNA indicates that the two samples are related, then we can infer that perhaps the mountain is not a barrier; but, if we have 20 samples from each side of the mountain range and none are closely related, can we infer that there is no gene flow? We can only say that we were not able to detect a flow of genes between these populations and that the mountains may be representing a barrier to movement for this species. But we cannot be sure that it is a complete barrier (Cushman et al. 2006). Either we may simply not have detected relatedness, or animals are moving between populations but not interbreeding.

Consequently, genetic analyses can be useful in understanding connectivity if we detect relatedness or it can be used to assess the realism of a spatial model of resistance to movement across complex landscapes (Cushman et al. 2006). For instance, we can develop a model allowing us to

infer resistance to movement to a set of landscape patches and then “allow” animals to disperse from one patch to another. Our model should be able to estimate the likelihood that an individual would survive, encounter, and continue to a target patch. If that likelihood were very low, but we then followed up by sampling individuals from the source and target patches and found that individuals in the two patches were related, then we would conclude that our estimates of resistance for one or more patches were incorrect. So, we can use genetic techniques to refine models of animal movement and allow us to infer the effects of changes in landscape pattern, or composition, on the likelihood that a species would be isolated or connected.

## MANAGEMENT APPROACHES TO CONNECTIVITY

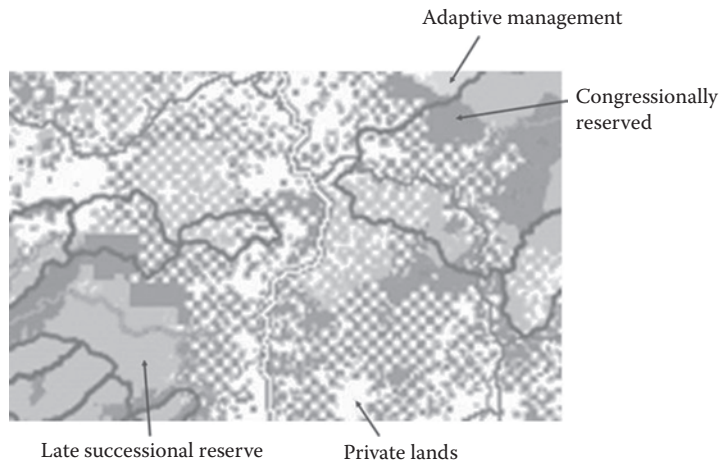
What is the best way to increase connectivity within a landscape? Unfortunately, little information is available on the gap-crossing abilities of most forest-associated species (Bélisle and Desrochers 2003). Further, gap-crossing abilities are species-specific. And dispersal is often seasonal. Cold wet winters and hot dry summers may represent periods of movement facilitation or reduction for various species.

When the area of habitat for a species falls below 30%–50% of the landscape (With 1999), then managers should begin to think about identifying connections across the landscape to reduce risks associated with isolation if the intervening matrix is indeed isolating the remaining patches of habitat. One approach is to purposefully manage a portion of the matrix to be very permeable for a species by developing a corridor between the patches. Corridors require an organism to follow a “path” through the otherwise inhospitable matrix if it is to have a high probability of moving from one patch to another. Connections may be in the form of static or dynamic corridors connecting specific patches on the landscape. *Static corridors* are those that are identified on a map and on the ground and purposefully managed over time to maintain connectivity in that location. *Dynamic corridors* “float” across the landscape over time, such that different portions of the intervening matrix are managed to provide connectivity at all times; it is just that the specific stands that provide the connection, change over time. This approach provides much more flexibility for the land manager to manage more stands for multiple values (Sessions et al. 1998).

Alternatively, the matrix condition as a whole could be managed to be made more permeable to dispersing organisms (*matrix management*). Such an approach does not require the animal to follow a path, but rather to move at random through the matrix. For species that are able to cross gaps, placing small patches of good-quality habitat close to one another between the two larger high-quality patches creates a *stepping stone* approach to connectivity. All of these techniques allow otherwise isolated patches to be connected through planning or patch management. In managed forests, this is usually achieved through harvest planning and silvicultural decisions (Bettinger et al. 2001).

## CASE STUDY: MATRIX MANAGEMENT FOR A WIDE-RANGING SPECIES

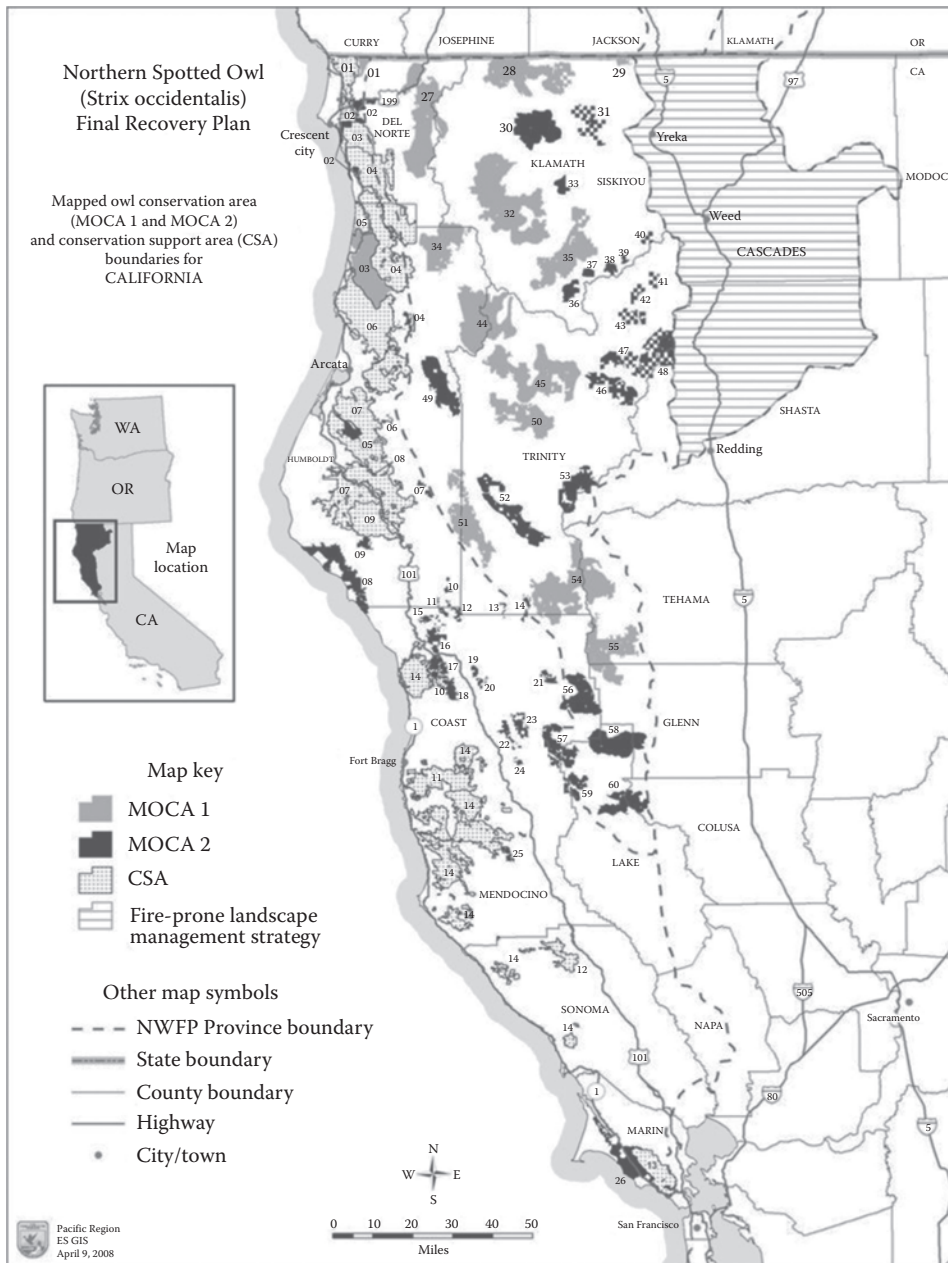
The northern spotted owl is listed by the U.S. Fish and Wildlife Service as a threatened species under the Endangered Species Act. As such, in efforts to allow the species to recover, it has received more attention than probably any other species in the United States in the past few decades. Generally, northern spotted owls are considered a species associated with forests with certain structural features of old-growth forests (Carey et al. 1990) (although they also occur in younger forests when these features are present). The Northwest Forest Plan (NWFP) was developed to ensure habitat for this and many other species associated with old forests and to allow the northern spotted owl populations to recover (FEMAT 1993). The habitat conservation strategy developed for the northern spotted owl identified large blocks of forest on federal lands that established a metapopulation structure for the species (Thomas et al. 1990). These areas were termed late-successional reserves (LSRs) (Figure 16.10). The LSRs largely fall on U.S. Forest Service lands because of the size and distribution of remaining spotted owl habitat, as well as habitat for marbled murrelets and other late-successional associates. Under the Northwest Forest Plan, areas between the LSRs



**FIGURE 16.10** Land allocation pattern under the northwest forest plan. Notice the checkerboard pattern of Bureau of Land Management lands providing stepping stone connections among LSRs as well as the matrix lands. Not shown are extensive riparian reserves designed to enhance connectivity across the landscape. (Map from [FEMAT] Forest Ecosystem Management Assessment Team. 1993. *Forest Ecosystem Management: An Ecological, Economic, and Social Assessment*. USDA For. Serv., USDC National Oceanic and Atmos. Admin., National Marine Fish. Serv., USDI Bur. Land Manage., Fish and Wildl. Serv., National Park Serv., and US Environ. Protect. Agency. Portland, OR.)

are connected through a system of corridors falling largely along riparian management areas that extend 50–100 m from stream-sides (to provide habitat for salmonid fish as well as connecting LSRs for terrestrial late-successional associated species). The plan also used a matrix-management approach, proposing management of the intervening matrix in a way that allows stands to develop with the structural complexity needed to support dispersing spotted owls as the stands mature. In addition, forests on Bureau of Land Management (BLM) lands, which fall largely on alternating sections (one section = one square mile), provide a stepping stone connection among LSRs (Figure 16.10). Consequently, much of the basis for habitat management to recover this species is underpinned by providing landscape connections among LSRs. These connections utilize matrix management, stepping stones, and corridors to achieve landscape connectivity for northern spotted owls and other species across the northwest forested landscape.

This plan has been in place with little modification for over 20 years, so how is the northern spotted owl doing? Not so well in many parts of its range. Populations are declining in nine of 13 demographic study areas (Anthony 2004, 2006). Declines in three areas over the 23-year period have been 40%–60%. The reasons for a failure of many populations to recover is complicated and seem to be related to competition with barred owls that arrived in the northwest about 20 years ago; barred owl populations have expanded significantly, perhaps at the expense of the spotted owl populations. Weather patterns have changed, with winters in some areas being colder and wetter and some summers being hotter and drier. It is not clear if this is a result of climate change or if this is simply random fluctuation. Hotter and drier summers have resulted in large stand-replacing fires in some areas. West Nile Virus has affected bird species in many parts of North America, and it is not known if it is having an adverse effect on populations or it simply is the result of reserve areas that are too small or too poorly connected. Although past and current loss in habitat and competition with barred owls are considered the greatest threats to recovery, most likely it is a multitude of factors that come together to prevent populations from recovering. The newest strategy to encourage recovery for the species enhances the habitat area for spotted owls in the northwest beyond what was allocated in the NWFP (USFWS 2008) and is expected to take 30 years and cost US\$489 million (USFWS 2008) (Figure 16.11).



**FIGURE 16.11** Land allocation under the 2008 Spotted Owl Recovery Plan. (Adapted from U.S. Fish and Wildlife Service. 2008. *Final Recovery Plan for the Northern Spotted Owl, *Strix occidentalis caurina**. U.S. Fish and Wildlife Service, Portland, Oregon. 142 pp.)

## SUMMARY

Although habitat area seems to be the most important feature affecting the occurrence and abundance of vertebrates on managed forest landscapes, at low levels of habitat availability, connectivity can become important. It is at these low levels of habitat availability that landscape pattern would be expected to impact the abundance and distribution of vertebrates. Connectivity mitigates some of the



adverse effects of low levels of habitat area, providing individuals within a population the habitat they need to disperse among patches. Connectivity provides opportunities for exchange of genetic information and the ability to repopulate otherwise isolated areas as habitat improves. Connectivity is effective if dispersal is adequate to meet these population goals. Successful dispersal is influenced by the likelihood that an animal will encounter a patch, survive the dispersal event, and continue dispersal before settling into a patch regardless of its quality. Providing static or dynamic corridors, stepping stones, or undertaking matrix management can facilitate the likelihood of successful dispersal.

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# 17 Managing Woodlands in Agricultural Environments

Clearing forest land for agriculture is a practice that has been ongoing for millennia. The process initially begins as a landowner enters a forest and clears enough land to meet the needs of a family. But when considering a landscape within which this occurs, the forest is still the matrix condition perforated by agriculture, at least initially. New settlers arrive, or the existing settlers realize that not only can they provide food for themselves but they can also sell excess food to others, so deforestation continues. Eventually, agricultural fields and pastures become the matrix condition with isolated patches of forest, or woodlots, remaining scattered throughout the landscape. If the landowner finds value in the remaining woodlots, then they are maintained, but if not, then they too are cleared and forest is lost until agricultural production is abandoned on that site. Much of New England is forest that was once agricultural fields or pastures, as evidenced by stone walls now criss-crossing a mature forest that has grown up over the past 70–150 years.

Much of the early research into the effects of forest fragmentation on a wide range of species was based on clearing forest land for agriculture until isolated woodlots remain within an agricultural matrix (Robbins et al. 1989, Villard et al. 1999). Hence, we know that many species associated with forests in the eastern United States are adversely affected by conversion to agriculture that leaves remaining woodlands in small patches that are disconnected from one another. The process continues to this day in the temperate and tropical regions of the world. Conversion to agriculture is a very broad sweeping term. What kind of agriculture—grazing, crops, livestock, etc.? Does it make a difference how the forest is converted and for what purpose? Belanger and Grenier (2002) reported changes in eastern Canada that have been documented in many other studies: As the area of land in agriculture increases, the number of woodlots increases but woodlot size decreases. They also reported that fragmentation increased along a gradient from dairy farms to intensive cash-crop agriculture. It would appear that the type of agriculture does influence the landscape structure and quite likely has effects on the species of animals and plants that can remain on the landscape. For some species, such as area-sensitive bird species (e.g., ovenbird), such changes in the configuration of the landscape can lead to regional declines in habitat and populations. For others, such as some bat species, fragmenting forests can lead to higher numbers of individuals (Ethier and Fahrig 2011). The purpose of this chapter is not to revisit the effects of forest fragmentation, but rather to ask what changes may be made to how we manage woodlots and the surrounding agricultural lands to alter habitat quality for selected species.

Management of woodlots in agricultural systems can be challenging, depending on the goals of the landowner. Nearly all woodlots are on private lands, so unless the landowner has another source of income, the land is the economic engine for the owner. Economics plays a large role in decisions (Arnold 1987). Small family farms may be in business to provide sufficient income on an annual basis to meet the collective family needs, which, at the very least, may be subsistence, or in highly productive areas, may include aesthetic and recreational goals. Industrial farms, on the other hand, may be less inclined to retain woodlots since profit margins drive many decisions, but even under these circumstances macroeconomics (world food prices), regulatory considerations (e.g., clean water act), or diversifying the portfolio with multiple uses (e.g., crop production and hunting leases), may result in woodlots being retained or even created on private lands. Alexander (2012) suggested that although the agricultural lands on small farms are intensively managed, small farm

owners were more likely to retain woodlands than large farms. Further, the small farms studied by Alexander (2012) provided more jobs and allowed greater public accessibility than industrial farms—two important social benefits. But Alexander's (2012) results may not be applicable to all farm systems. Lovell et al. (2010) reported that larger farms in Vermont had more forest cover arranged in a more complex configuration than small farms, but farmers in that area may have benefited economically from retention of forest lands more than farmers in other areas.

## VALUE OF WOODLOTS TO LANDOWNERS

Many landowners do find value in retaining woodlots, and other landowners integrate forests into their agricultural production using an approach called agroforestry (Lawrence et al. 2010). Woodlots provide firewood, a place to hunt, or simply enjoy forest wildlife, protect a wetland that otherwise is unsuitable for crop production, protect a field from wind erosion, or may provide food (e.g., acorns) and shade for livestock.

Farmers in Vermont derived firewood, timber, maple sugar, edible fruits and nuts, and wood crafts from their forest lands—direct economic benefits from woodlot retention (Lovell et al. 2010).

Alternatively, woodlots may also provide habitat for species of animals that can reduce crop yields. Damage to corn and other crops by raccoons (Beasley and Rhodes 2008), squirrels, deer, turkeys, and bears can be exacerbated if forest cover is adjacent to crops. Species that are associated with grasslands may be less abundant near forested edges, and some of these may damage crops (Delattre et al. 2009) while others may be desirable (Ribic et al. 2009). As with all situations involving habitat management, there are tradeoffs. Not all species will be benefited by any one management strategy. Sanderson et al. (2009) found that although farms with a high proportion of grassland and woody edge are likely to provide habitat for many of Poland's farmland bird species, not all are benefited; species-specific management would be needed for species that avoid woodland edges.

## EDGE EFFECTS IN AGRICULTURAL SETTINGS

Predation, brood parasitism, and nest destruction are concerns for many species that use forest edges in agricultural landscapes (Donovan et al. 1997). These adverse effects on birds in forest interiors are described in more detail in Chapter 15. Grassland species also can be adversely affected by fragmentation of large expanses of grasslands simply by planting trees. Although tree planting may increase overall bird diversity and abundance in a grassland or agricultural landscape, birds associated with grasslands, including species of conservation concern, may be adversely affected. To reduce these adverse effects on these area-sensitive grassland species, Reino et al. (2009) suggested clustering plantations into fewer larger patches to reduce the proliferation of edges that may have adverse effects on these area-sensitive grassland species.

## HABITAT ELEMENTS IN WOODLOTS

All of the habitat elements described in previous chapters apply to management of woodlots as well. A few elements are particularly vulnerable in agricultural systems, however. If a farmer decides to allow cattle to graze through woodlots, then shrub cover and tree regeneration will be compromised, and the vertical complexity of the woodlot will be simplified. Harvest of trees for firewood or timber will reduce stand density and reduce the likelihood of competition mortality, thereby potentially reducing the number of snags and logs in these woodlots. Hence, these two activities—alone or in combination—may reduce the capacity of a woodlot to support shrub-nesting or browsing species as well as cavity nesters and log-users.

The entire farm operation can also influence how a woodlot functions as habitat. For instance, if invasive species are allowed to colonize from adjacent farm fields and are not controlled, then the species composition of the woodlot can be changed considerably. Although often more of an issue

for woodlots in urban areas, the occurrence of non-native plants is frequently higher along woodlot edges in agricultural areas (Brothers and Spingam 1992). Further, birds and mammals inhabiting woodlots disperse seeds from adjacent agricultural areas into woodlots and may further exacerbate invasive species issues, especially where windbreaks and fencerows that contain invasive species are retained (Harvey 2000). These linear features in the landscape provide an avenue for dispersal from woodlot to woodlot. If invasive species are concerns, then farmers will have to be vigilant in controlling unwanted species and encouraging native species. Indeed, farmers, just as any other land manager, need to develop clear goals for their woodlots and manage the woodlot and surrounding landscape to achieve those goals (Haslem and Bennett 2008). For instance, maintaining woodlots to produce veneer oak lumber would likely entail a group selection regeneration system, with subsequent protection of the woodlot from grazing and browsing, to ensure oak regeneration (Kelty et al. 2003). A woodlot managed for maple sugar production might see single-tree selection systems used to focus growth on those trees with large crowns for continued sap production (Kenefic and Nyland 1999).

### **Fencerows and Shelterbelts**

Linear features that are often peculiar to farmlands are windbreaks and fencerows. Fencerows often include shrubs and trees that are the result of either wind dispersal, or—more often—dispersal of seeds by birds and mammals. Birds perch on fences, defecate, and the seeds in the feces germinate. In temperate regions, it is quite common to find fencerows dominated by cherries, roses, blackberries, barberries and other fruit-producing shrubs, usually the direct result of bird-mediated dispersal. Once established, then wind-dispersed seeds can accumulate along these fencerows. Fencerow width is restricted by active farm management, such as plowing, grazing of animals, mowing, or harvest of crops. Nonetheless, fencerows do occupy space that could be used to grow crops; and on large industrial farms, fencerows are often removed to make farming operations more efficient and profitable. In some areas, removal of fencerows, especially large and floristically diverse fencerows, can eliminate habitat for some species of small mammals on farms (Silva and Prince 2008).

Shelterbelts are strips of trees purposefully planted between fields to reduce wind speed across open fields and reduce soil erosion by wind. These strips of trees add structural and compositional complexity to the farm mosaic and are used by many species of vertebrates. They also form forest edges that can have an adverse effect on grassland interior birds (Helzer et al. 1999). Conversely, they can form corridors for potential dispersal of species associated with woody vegetation (Haas 1995). Their use by any one species of animal is largely influenced by the species of trees planted, the shrubs which develop, and the management of the adjacent fields. A dense strip of conifers adjacent to a crop field may be used heavily in the winter as a roost site for over-wintering birds, whereas the same strip planted to deciduous trees may be more heavily used in the summer as a nest site for edge-associated species. Birds were found more likely to use fencerows for nesting in Michigan if they were wide, tall, near old-fields, and contained shrubs (Shalaway 1985). But of course these are the very characteristics that take up additional growing space for crops. Farmers are faced with economic tradeoffs between crop production and habitat when considering shelterbelt and fencerow management.

### **LOSS OF FORESTS IN INDUSTRIAL AGRICULTURAL SETTINGS**

The proliferation of industrial agriculture in meeting the food needs of a growing human population is associated with many environmental concerns (Horrigan et al. 2002). Industrial agriculture treats farms as factories for commodities to be produced as economically as possible, typically over large areas. The rate of forest loss for this purpose is staggering in some parts of the world. In the Brazilian Amazon alone, the rate of forest conversion from 1978 to 2000 ranged from 1.4 to 2 million hectares per year (Lawrence et al. 2001).

Loss of forest land is not simply due to industrial agriculture. Small family farms, which often include a woodlot, already have their lands managed for food production, but the tenure of those lands in the United States is in question due to changes in demographics and costs associated with inheritance taxes that make retaining the farm in the family more difficult. The easy thing for many heirs to do is to sell the family farm to an industrial farm or to a developer who will build homes; so the farm and the woodlot are lost.

The adverse environmental effects of industrial agriculture, including loss of some of the most productive forests in the world, gave rise to more attention to sustainable agricultural systems, and conversion of small farms to housing has led to social changes in the United States. Organic farming, community and urban farms, farmers markets, restaurants specializing in locally grown foods, and gardening grew in popularity over the past few decades, at least partly in response to the environmental concerns associated with industrial agriculture and loss of farmland. But, recently, a new agricultural product has emerged that will likely increase land area in industrial agriculture and perhaps even bring lands that were abandoned in the past back into agricultural production: biofuels. The demand for fuel as well as food and fiber will exert increasing pressure on agricultural lands and increase the likelihood that small farms will be consolidated and that forests will be converted to biofuel production. Ethanol and biodiesel production is increasing; in one year, ethanol production increased by 1 billion gallons and biodiesel production increased from 75 million to 250 million gallons in the United States alone (Altieri 2009). Unless consideration is given to balancing long-term ecological effects of production with increased demands, we can expect to see environmental issues arising from industrial agricultural production of food, fiber, and fuel (Altieri 2009).

## **FIELD AND FARM MANAGEMENT THAT INFLUENCES WOODLOT FUNCTION**

How or whether species of animals will use a woodlot are functions of both the woodlot structure and composition, as well as the types of management activities that occur adjacent to the woodlot. Many species using woodlots also use the surrounding fields as foraging sites. Deer, raccoons, squirrels, quail, and many species of woodland birds and mammals will venture into fields to feed on seeds and insects along woodland edges. The availability of seeds, insects, and other food resources to these species in adjacent fields is influenced by the manner in which the field is managed. It is the juxtaposition of the woodlot and its ability to provide habitat elements with the field that can provide food resources for species able to use the woodlot.

### **ANNUAL CROP MANAGEMENT**

When managing annual crops such as corn, soybeans, wheat, and most other grains that we rely on for our food staples, the fields are typically planted in the spring and harvested in the fall. To prepare the field for planting, the field is often plowed and disked prior to planting so as to create a good seed-bed and also to help to remove competing weeds. Plowing can be done after the crop is harvested in the previous fall, which is when the fields are often less muddy and more likely to allow access to machinery, or in the spring when, following snow-melt, plowing may be delayed due to mud. However, by delaying plowing until spring any waste grain left on the field following harvest remains on the field through the winter and is available to animals living in the adjacent woodlots. Spring plowing, though often more difficult, is preferable if providing food to woodlot-associated species is a goal.

Many farmers are moving toward no-till approaches to soil management. Rather than plowing and disking prior to planting, the next crop is planted through the stubble from the previous year, and either a preemergent herbicide is applied to prevent competition from weeds or some type of mulch is applied (as is most often the case with organic farming). For large farms, use of mulch may be time- and cost-prohibitive so herbicides are typically used, and this may raise concerns about

effects on some species, especially amphibians (Relyea 2012). The advantage to this approach is that waste grain again remains on the site for a longer period of time and provides food. Further, the machinery makes fewer passes over the land each year, and the plant material remaining after harvest can decompose on the site, adding an organic layer to the soil.

## GRAZING MANAGEMENT

When the focus of a farm is production of dairy products, wool, or meat, then, typically, there are both crop management issues (hay and silage for the animals) and grazing area management issues to consider, adjacent to woodlots. Just as forests provide habitat for different species of animals based on its horizontal and vertical structure and composition, grasslands function in a very similar manner. If pastures adjacent to woodlots are grazed continually and at a high intensity, then the structure of the pasture will be simple, homogenous, and dominated by those species of plants that the livestock eats the least. This may be an ideal situation for American robins and killdeer, but not necessarily for other species. Alternatively, if livestock are rotated among pastures and pastures are allowed to recover from grazing pressure, then the structure will be more complex, both vertically as well as (often) horizontally, offering opportunities for use by other species, such as common flickers and accipiter hawks, along woodland edges. Rotational grazing also increases the complexity of the landscape as a whole by providing fields in various stages of recovery.

Several factors can be manipulated to influence the structure and composition of the pastures. Grazing intensity, or the number of livestock per unit area, can be controlled. Historically, this was managed using Animal Unit Months (AUMs), or the forage eaten by a cow and a calf in one month. Pastures with different capacity for net primary production have different allowable AUMs. The frequency with which livestock are rotated from one field to another influences the impact and recovery time on any one field. The type of livestock can also play a role, with sheep, goats, and horses (which have incisors on the upper and lower jaws) more likely to affect roots of plants than cattle (which lack incisors on the upper jaw). Finally, the species of grasses and forbs planted in the pastures can play an important role in determining field structure and composition. If forbs such as clover, alfalfa, and other broad-leafed plants are available, then use of the field by quail may be more likely than if these species were absent. Use of cool-season grasses (species that put on most of their growth in the fall and spring) vs. warm season grasses (species that grow most rapidly in mid-summer) influence the seasonal structure of the pastures.

Managing fields for forage production, by harvesting and storing hay as food for livestock through the winter, presents challenges for some grassland birds and opportunities for species associated with woodlots. When hay is cut prior to July 15 in temperate regions of North America, some grassland birds, such as bobolinks and meadowlarks, may still have young in the nest. Delaying harvest allows these young birds to fledge prior to harvest of hay, and, depending on the types of grasses, can have minimal effects on forage quality (Nocera et al. 2005, 2007).

## LAND SPARING

Setting aside farmland as fallow for a year or more can provide additional benefits to species in adjacent woodlots (Quinn et al. 2012). The Conservation Reserve Program in the United States was designed to set some farmlands aside for 10–15 years and use other soil conservation practices to reduce the rate of soil erosion from intensively managed regions of the country. By managing farms with a portion set aside to control erosion, maintaining riparian buffers, using a cropping system that reduces or eliminates tillage, and vegetated ditches, the complexity of an otherwise homogenous industrial farm can be enhanced, and the diversity of species that can occur on the farm increases. All of these actions cost money; however, if farmers can see the long-term economic advantages to soil conservation measures while also realizing benefits to various species of wildlife, then they are more likely to continue to manage woodlots, fallow fields, and vegetated waterways.



Since entire textbooks have been written dealing with management of wildlife in agricultural settings, the above management options only scratch the surface of the options that are available and the effects that can be seen on species of animals using adjacent woodlots.

## **SPECIALTY CROPS AND AGROFORESTRY**

Specialty crops, especially woody perennial crops, may be the focus of a farm, or they may be incorporated into other aspects of the farming operation. Incorporation of woody plants into a farm operation, or agroforestry, can provide benefits for some species that associate with both trees and agricultural lands. Johnson and Beck (1988) and Jose (2009) suggested that there are several advantages to employing agroforestry over traditional farming practices, when conservation of biodiversity is a goal:

1. Agroforestry may provide habitat for species that can tolerate disturbance but maintains essential habitat elements.
2. Agroforestry may preserve germplasm of sensitive species that would otherwise be lost.
3. Agroforestry helps to reduce the rates of conversion of natural forests by providing an alternative to traditional agricultural systems that would simply clear the forest.
4. Agroforestry provides connectivity by creating corridors between habitat remnants that may contribute to the integrity of these remnants as functional pieces of a metapopulation.
5. Agroforestry helps conserve biological diversity by providing other ecosystem services, such as erosion control and water recharge, thereby preventing the degradation and loss of surrounding habitat.
6. Agroforestry practices also provide more habitat elements than traditional agriculture by increasing structural and compositional plant diversity on the landscape.

Incorporation of woody crops, windbreaks, fencerows, and riparian buffers offers the only woody habitat for animals in many agriculture-dominated landscapes (Johnson and Beck 1988, Jose 2009). But even retention of individual trees in pastures, field edges, or areas that are of low crop productivity has benefits. DeMars et al. (2010) referred to remnant trees in agricultural settings as keystone features that played a disproportionate role in contributing to biodiversity in farmlands. Retention of existing trees and recruitment of replacement trees should be considered, if farms wish to support as diverse a set of species as possible. However, scattered trees in extensive grasslands may reduce habitat quality for species that require extensive grasslands. These scattered trees may allow predators and brood parasites to more successfully locate nests in the grasslands. Agroforestry has been commonly practiced in some tropical countries, and the benefits of including woody vegetation in tropical farms have been recorded in some areas. Bhaghat et al. (2008) found that more mobile species, such as some species of bats and other mammals, as well as birds, used agroforests while less mobile organisms, such as some species of plants, were less likely to use agroforests. Regardless of tropical or temperate settings, adding trees to otherwise homogenous agricultural settings, even in small amounts, can add significantly to the diversity of animal species that can be supported on the farm (McComb et al. 2005).

## **ORGANIC VERSUS TRADITIONAL AGRICULTURE**

Organic farming has grown in popularity in the past few decades and continues to be an important component of agricultural landscapes, especially in proximity to population centers. Organic farming systems generally support a richer community of plants and animals than conventional systems although that may be dependent on how the traditional farm is managed. Further, the context or the conditions that surround the organic or traditional farm play a significant role in contributing to biodiversity in these agricultural areas (Gomiero et al. 2011). Each species responds to agricultural



practices at different spatial scales, so it is the complexity over a landscape, which may be influenced by multiple farmers, that contributes to species observed on any one farm (Gabriel et al. 2010). Consequently, the surrounding landscape may either enhance or reduce the effects of organic farming in supporting higher levels of biodiversity than traditional farming (Smith et al. 2010, Winqvist et al. 2012). In homogeneous landscapes, organic farms can support more species and more individuals of seed-eating birds than conventional farms, but this difference was not detected in more complex landscapes by Dänhardt et al. (2010). Batáry et al. (2010) reported similar results, and suggested that incorporating fencerows and organic farming into otherwise homogenous landscapes could support higher levels of biodiversity than homogenous farms alone.

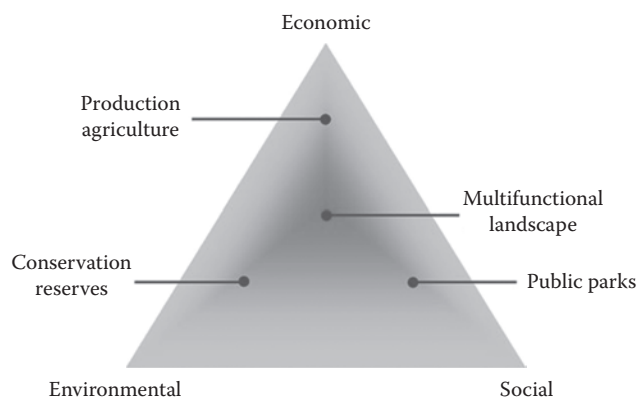
Why do we see higher levels of biodiversity in homogenous landscapes in which organic farming methods are used? Organic farms tend to be more complex systems than traditional farms, and as such, provide more habitat elements for more species than traditional farms (Norton et al. 2009). Smith et al. (2011) reported positive effects of organic farming on biodiversity in 62 of 82 studies they reviewed and concluded that this was due to factors that tended to increase the complexity of the area:

- Avoidance of pesticides and inorganic fertilizers
- Rotation of crops that include grasses and forbs, and mixed crops
- Raising a variety of crops
- Permanent pastures and fencerows
- Restricted use of manure
- Including a variety of livestock species

But having a high level of biodiversity does not mean that all species would be supported. There will be winners and losers with any management approach. If we wish to support as many species as possible, then we need to begin thinking about how to arrange agricultural landscapes, which include woodlots, fencerows, riparian areas, windbreaks, and organic farms, to best meet the needs for the most species. Thinking about all of these landscape elements as part of a multifunctional landscape may be a promising approach.

## MULTIFUNCTIONAL LANDSCAPES

Multifunctional agricultural landscapes attempt to find a balance between maximizing economic, ecological, and social values (Figure 17.1). Imagine a 2000 ha landscape that includes a farm and



**FIGURE 17.1** Conceptual framework for development of multifunctional agricultural landscapes. (From Lovell, S.T., and D.M. Johnston. 2009. *Ecology and Society* 14(1):44. With permission.)

woodlots. In an attempt to meet the needs of the broadest suite of species possible, we would look for the following landscape elements:

- Consolidation of pastures and grasslands to meet the needs for grassland interior species
- Rotational grazing, and fields with cool and warm season grasses, and varying levels of forbs to increase grassland heterogeneity
- Small crop fields managed using no-till or organic farming practices, with fields bordered by fencerows and shelterbelts
- Scattered woodlots of various sizes and in different successional stages, each containing habitat elements important to a suite of species, such as diverse tree and shrubs species, dead wood, and hollow trees
- Fencerows, shelterbelts, and riparian buffer strips connecting with one another and with the woodlots
- An extensive tract of unbroken forest, which is also connected to the fencerows, shelterbelts, and riparian buffers, for forest interior species

Such an approach would view agriculture as a part of an integrated system that extends beyond production of food and fiber to contribute to social and ecological values as well (Renting et al. 2009). But the devil is in the details: How would we best arrange all of these elements in one landscape to achieve multiple goals and meet the needs of as many species as possible? Allocating land-use over space and time must balance economic, social, and ecological goals but when done correctly can achieve desirable results at a fraction of the cost of traditional “manage vs. reserve” binary systems (Reyers et al. 2012). Clearly, forests, woodlots, shelterbelts, and riparian buffers are integral parts of such a system, and they contribute to meeting the habitat needs for many more species than would be present without these features being present.

## **CERTIFICATION OF AGRICULTURAL LANDS FOR WILDLIFE: THE ROLE OF TREES**

Third-party certification of agroforestry and crop farms as sustainable and biodiversity-friendly is a growing trend. Certification has been proposed or implemented for crops such as coffee, rubber, cocoa, and others. Third-party certification will be discussed in more detail in a later chapter, specifically with regard to forest management and effects on biodiversity, but parallel approaches are proposed for agricultural systems as well. For instance, Food Alliance in Oregon is an organization that provides third-party certification for social and environmental responsibility in agriculture (Food Alliance 2012). Food Alliance has developed standards for wildlife habitat conservation on farms that includes continuing education, farm biodiversity plan development, and threat reduction to biodiversity. In addition, it includes many of the habitat elements of forests, as well as the landscape elements described in this chapter, in its assessment of the degree to which a farm is likely to contribute to biodiversity conservation. All in all, seven criteria are identified and rated from 1 (low) to 4 (high). There are many reasons why a farmer may wish to become certified, especially if the crops grown are specialty or high-value crops, in which a biodiversity-friendly certification may lead to a higher demand or market value. But some farmers may simply wish to “do the right thing” as a part of their farm management, because they hold a conservation ethic that they honor, while still turning a profit on the farm.

## **SUMMARY**

Loss of forest land to agriculture to meet demands for food, fiber, and fuel is a significant concern in many parts of the world. But there is increasing awareness that farming need not all be high-impact, high-input, industrial farming. Planned as multifunctional landscapes, farm operations can

retain some of the biodiversity present on the site prior to conversion, or recover aspects of biodiversity lost centuries ago. Woodlots and other woody vegetation in agricultural systems can have a disproportionate influence on supporting a diverse assemblage of plants and animals. Incorporation of fencerows, woodlots, windbreaks, riparian buffers, and scattered trees, strategically located in a farmland mosaic, can provide habitat for a richer suite of plants and animals than homogenous farm operations. There are efforts afoot to reward farmers for biodiversity-friendly practices through awards of third-party certification.

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# 18 Approaches to Biodiversity Conservation

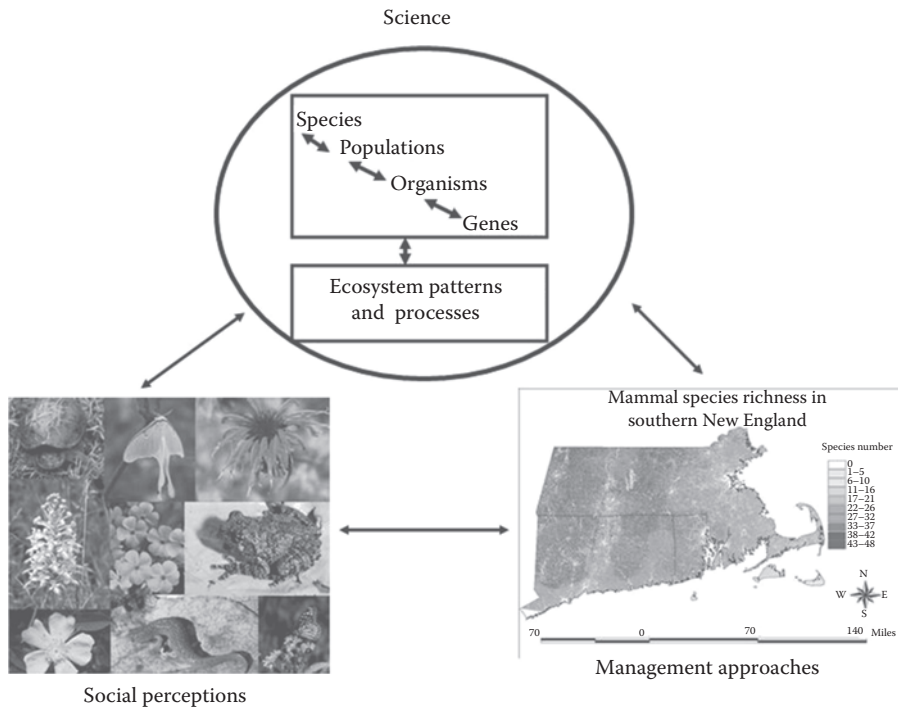
So far, we have focused on habitat management for individual species. For some forest wildlife goals, that is an appropriate approach. Oftentimes, especially on public lands, conservation of the full suite of living organisms present on a site, on an ownership, or in a watershed, is an objective, while also meeting other societal objectives such as potable drinking water, recreation, aesthetics and timber production. By now, you must be asking, “How in the world can we possibly manage forests to conserve the hundreds if not thousands of species that occur within a forest with one owner, let alone multiple owners?” Using a species-by-species approach is clearly untenable. But biodiversity is continuing to decline despite widespread efforts at conservation (Rands et al. 2010).

Logging of forest lands is viewed by many as being incompatible with maintenance of biodiversity. Indeed, unsustainable or illegal logging can have a long-lasting adverse effect on conservation of biodiversity (Rands et al. 2010). To mitigate these effects, Rands et al. (2010) suggested managing biodiversity as a public good, integrating biodiversity goals into public and private decision making, and developing conditions that allow implementation of biodiversity conservation policies. Goals are typically set at large spatial and temporal scales and achieved through multiple local actions. These actions are designed to minimize risk of losing a species while considering uncertainty in our decision-making process (Noon et al. 2009, Schultz et al. 2013). Monitoring of focal species and species of conservation concern is a key part of the biodiversity conservation strategies proposed over United States National Forests, in order to lessen the risk of losing species locally or regionally (Schultz 2013).

## WHAT IS BIODIVERSITY?

Scientists define biodiversity as the genes, organisms, populations, and species of an area, and the ecosystem processes supporting them (Figure 18.1). Key principles that are often included in the definition of biodiversity are those of ecosystem structure, composition, and function, occurring at various scales of space and time. Most nonscientists view biodiversity as the collage of species, and many equate biodiversity with those species that are rare and wild. Clearly, for scientists and managers to be effective in meeting the expectations that society has for conserving biodiversity, the collage of species must be addressed. Indeed, some of the most challenging aspects of biodiversity conservation are in deciding how to understand ecosystem complexity and the uncertainty of implementing management while protecting both known and unknown species. Due to the complexity of the problem, communicating approaches to the public is challenging. There is a triad of biodiversity perceptions, biodiversity concepts, and biodiversity assessments (Figure 18.1) viewed by the public, scientists, and managers, respectively, that must be interconnected if we are to successfully address the biodiversity issues. Successful conservation of biodiversity must involve the public and adequately meet public expectations.

Species are usually considered the primary currency of biodiversity conservation. But even conservation of species presents challenges. Rare, threatened, and endangered species garner much attention politically, and species that are hunted or are aesthetically appealing (e.g., songbirds, wildflowers) are often used as focal species or as special interest species when making biodiversity decisions. These are just examples of species that could or should be considered during management.



**FIGURE 18.1** The scientific concept of biodiversity is a set of processes and conditions that interact to reflect the breadth of life on the planet. Biological complexity is often perceived as a collage of life by nonscientists. (Bottom left; photos by Dr. James Petranka, with permission). Scientific concepts can be used to conserve the collage of life by developing maps of species richness for various groups of organisms. (Bottom right, from Southern New England Gap Program, UMass-Amherst) or by using coarse filter/fine filter approaches (see text).

Mora et al. (2011) estimated that there are 8.7 million species on Earth, of which 6.5 million are terrestrial or found in freshwater systems. Less than 2 million species have been described to date. An approach often proposed, when conserving such a vast array of species, is to use patterns of occurrence for one taxonomic group (e.g., birds) to protect habitat for other taxa (e.g., mammals or amphibians); but this approach does not work very well in many systems (Flather et al. 1997). Of the taxa explored as surrogates for conservation planning, birds, plants, and mammals, seem to hold some promise, but only in certain biomes and only over large areas (Lewandowski et al. 2010, Larsen et al. 2012). There are clear challenges to ensuring that we do not lose biodiversity across the Earth at a rate significantly different from what would be expected if technologically advanced humans did not have such a profound effect on the Earth's resources.

Laikre et al. (2009) found that few nations have biodiversity conservation plans that explicitly consider conservation of genetic diversity, and those nations that did consider genetic diversity were those with a high standard of living. Planners and managers usually assume that genes will be successfully conserved among individuals within a species if we can ensure the long-term viability of populations throughout the geographic range of each species. Policies or actions that eliminate a species from part of its geographic range are assumed to reduce the genetic diversity of the species and increase the risk that the species would be less able to tolerate perturbations to its habitat in the future ultimately leading to extinction (Lomolino and Channell 1995). Two rules of thumb predominate when considering how best to conserve genetic diversity, especially in the face of uncertainty brought on by climate change: (1) maintain large populations and (2) maintain interconnectedness of subpopulations throughout the geographic range of the species (Hendry et al. 2011, Sgrò et al. 2011). We certainly see geographic variation in phenotypes (what an individual looks like), diets,



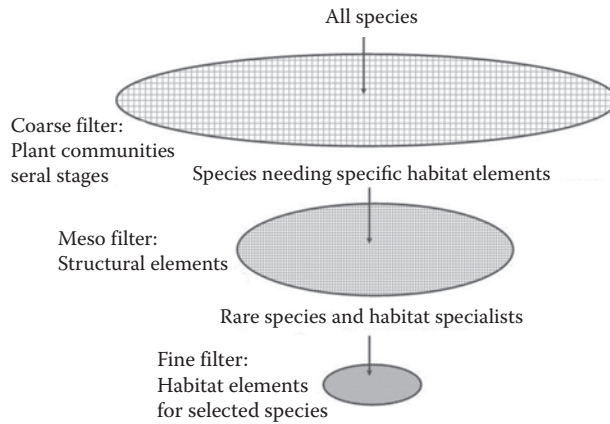
habitat selection, and home range sizes, within many species of vertebrates across their geographic range. A reasonable assumption is that these differences reflect some evolutionary advantage to the species in those places. Very rarely have these assumptions been tested (Lomolino and Channell 1995). But populations fluctuate in size and connections come and go, depending on disturbances and regrowth, so following a precautionary principle of conservation (including active management where and when it is needed) in deference to consumption may be a reasonable strategy for conserving genetic diversity within and among populations.

## SETTING BIODIVERSITY GOALS

Most biodiversity objectives reflect the paraphrased text of Aldo Leopold: "...the first rule of intelligent tinkering is to save all the pieces." Indeed, the pieces are the genes, organisms, populations, species, and supporting ecosystem processes (Figure 18.1). These are the very things that are implicitly part of the integrated filter approach to biodiversity conservation. The key word in this quote is "all" and the phrase begs the question, "How much of each?" The answer, obviously, is "Enough!" Saving all the pieces is a noble goal. Indeed, it is a rule of thumb for people who care about seeing the collage of life on this planet persist for future generations to enjoy. But these people are only part of society. Indeed, in some societies, cultures, and places, this group may well be in the minority. Or society may embrace the noble goal of saving all the pieces, but it may follow that by asking how much is enough? And at what price will it be provided? Take for instance the recovery of wild stocks of salmon in the Pacific northwest of the United States. Years of research indicate that there probably are some key factors all working together to cause wild salmon stocks to be at less than 10% of historic levels. If society truly wants salmon to recover to historic levels, then: (1) remove some or all dams to improve passage, (2) do not mix wild genetic stocks with hatchery fish genes, (3) reduce or eliminate sport and commercial fishing, (4) restore freshwater conditions to be acceptable for spawning, and (5) allow all spawning fish to enter the stream and die to provide stream nutrients (Compton et al. 2006). Remove a source of hydropower? Increase electricity bills? Use coal or nuclear fuels for electricity? Do not allow salmon harvest? Will society agree to this? Not likely. And this is in a wealthy society. Consider the overgrazing situation in the dry tropical forests of South America that has led to desertification. Tell the campesino to stop grazing for a few years to allow the rangeland to recover (and it would), and he and his family will starve. So he is not likely to stop grazing. Successful efforts at conserving biodiversity are often more likely to be successful if a social-ecological-systems approach is taken (Ban et al. 2013). Setting biodiversity goals must consider the genetic resources, the species, the ecological processes, and the goals and objectives of the people affected by a decision. And because goals will change over time, plans designed to meet goals now must be adaptable to allow future goals to be met.

## HOW DO WE CONCEPTUALIZE "BIODIVERSITY" TO BE ABLE TO CONSERVE IT?

Given the complexity associated with biodiversity and recognizing that it is a resource that society values, what is a scientist, manager, planner, or decision maker to do to ensure that biodiversity is conserved for future generations? How can we hope to understand and consider the needs for all species in a planning area? Generally, a tiered approach to decision making that considers the needs of some species explicitly is used, but this assumes that the needs of others will be met through a more generalized strategy of habitat protection and/or management. So scientists simplify the problem by taking a logical step-wise approach, albeit with significant assumptions. The *filter approach* is often used as a basis for reducing the risk of losing a species from an ecosystem (Hunter 1999, Zenner et al. 2010) (Figure 18.2). In this approach, three management strategies, termed "filters," are used. These "filters" are analogous to management filters designed to "catch" species in each hierarchy of management approaches, and minimize the risk of losing species. The three filters are



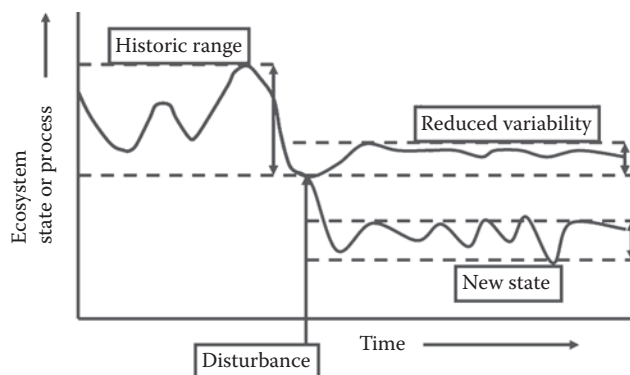
**FIGURE 18.2** Coarse filter goals are met using vegetative types and successional stages that are likely to meet the needs for many species in a planning area. Some species require specific habitat elements provided within a meso-filter. For those that are not likely to be met using this approach, a fine filter (single-species) analysis is conducted.

coarse, meso and fine, each with a set of assumptions about how the combination of these three types of filters can be employed to “capture” species in a management strategy.

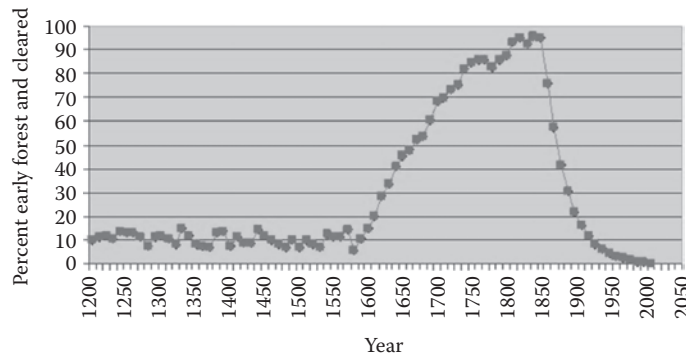
### COARSE-FILTER APPROACHES

The coarse filter is applied to the landscape by describing the distribution of biophysical classes (e.g., vegetation classes, slope classes, stream classes, etc.) that occur in an area of concern and documenting the arrangement and connectivity of these biophysical classes across the landscape. These current conditions may then be projected into the future under various alternative management assumptions or compared to past conditions to see how much they have changed over time. The current and possible future conditions are often compared to some reference condition(s). Recently, that comparison has quite often been to the historical range of variability (HRV) in one or more ecosystem indicators (Landres et al. 1999, Keane et al. 2009) (Figure 18.3).

It is important to understand that, when using the HRV as a reference condition, the objective is *not* to return to a condition that once occurred in the past, but rather to consider the range of



**FIGURE 18.3** Use of the historical range of variability allows managers to consider the implications of future conditions following a disturbance on reducing the variability in a system or creating an entirely new state of conditions in an ecosystem.



**FIGURE 18.4** Generalized changes in open and early-successional forest conditions following European colonization in New England. While increases in this ecosystem state provided habitat for many species others were lost. Now, due to much lower availability of this condition, species associated with this condition are at risk.

conditions that species likely encountered in the past and the process that led to those conditions. Biologists often assume that the species persisted within these ranges of conditions and processes. The more the current and likely future conditions depart from the HRV, the greater the *risk* that genes or species may be lost from the system. For instance, consider the likely distribution of one ecosystem indicator, open and early successional conditions, in New England (Figure 18.4). As European humans introduced new technology and approaches to land management, preEuropean fluctuations in this indicator began to change following forest clearing and, eventually, farm-land establishment and subsequent abandonment (Foster 1992, DeGraaf and Yamasaki 2001). The departure from the historical range of variability was significant with most forest land converted to open land in much of southern New England. What was the risk to biodiversity of this departure? Global extinction of a few species, most notably passenger pigeons. Also, the likely loss of species that we had not identified by the time they were lost and regional extinction of forest-associated species, such as fisher, moose, white-tailed deer, black bear (Figure 18.5), wild turkey, and beaver. As the forest returned and the amount of open land declined, these latter species have occupied the area once again from forests to the north. But now the amount of open grassland and early successional



**FIGURE 18.5** Black bears were extirpated from southern New England during the 1800s and did not return to the area for over 100 years. Now they are common and can cause damage to homes, property, and crops in the region. (Photo by Karl J. Martin. With permission.)

conditions is very low, probably lower than it was historically, and we now see species that are considered at risk because we have less of these conditions than we did historically. Bobolinks, eastern meadowlarks, and chestnut-sided warblers are all considered species associated with these open conditions and are species of concern in the region (Vickery et al. 1992) (Figure 18.6).

The organisms and values that currently occur in the forests of a region have persisted through centuries of natural- and human-induced disturbances. Some did not survive past deforestation. Some may only now be recovering. Considering the plan within the context of the historic range of variability is one way to assess the risk of losing species when attempting to achieve a desired future condition (DFC) for a landscape. Consider the likely representation of early-, mid-, and late-seral forests during the early 1600s in the New England region. Proportions of these successional stages are inherently variable annually, decade to decade, and century to century, as hurricanes, ice storms, floods, fires, and other disturbances occurred with varying levels of frequency and severity. But these proportions clearly now depart from historic ranges. Early-seral shrub stage and grassland conditions are now poorly represented in the northeastern landscape. Due to past forest clearing, only a very small proportion of the landscape is now in late-seral condition. Consequently, species associated with any of these conditions may be considered at greater risk than species associated with mid-successional conditions.

Further, it is important to think of the range of variability that once occurred, not just as seral condition, but also plant community representation. What should be a reasonable proportion of representation of vegetation types across the landscape to reflect the conditions with which these species have evolved? Due to recent land uses, some plant communities are very rare. Prairie communities in the Great Plains and in the Willamette Valley of Oregon have been reduced to a tiny fraction of what they were historically. Old-growth forests in New England are less than 1% of historic conditions. Should we restore these systems to historic levels? Can we? Although we may set restoration as a goal, achievement of that goal may not always be possible. But we may wish to focus on recovery of a greater proportion of these types to minimize further losses of species from the region.

Finally, it is important to consider the size and arrangement of the patch types that would be created across the landscape. Within the context of actively managed forests, harvest planning has a direct bearing on the arrangement of plant communities and seral stages across a landscape over time. The range of patch sizes in managed forests often departs from the distribution of patch sizes created by natural disturbances. In many cases, most patches should be of small size, fewer of larger size, and



**FIGURE 18.6** Early successional forest is being created on Massachusetts Wildlife Management Area lands and privately owned lands using landowner incentives program (LIP) funds in response to regional declines in species associated with shrublands and grasslands.

very few very large patches (of course, small and large are relative to the sizes represented over time, following disturbances and regrowth). Size matters to some species and so does location. Tailoring patch sizes and arrangements to plant communities, based on the effects of natural disturbances within the constraints set by society, may mean that there is less risk of losing species. Creating edges, small or large patches, and connectivity can all influence the species assemblage occurring in a landscape over time. Connectivity may be especially important in the face of climate change (Nunez et al. 2013).

Now consider which plant communities and successional stages are currently underrepresented in the ecoregion in which your landscape occurs and which of these might be underrepresented in the future following a particular course of management. Of those that are underrepresented in the ecoregion, which could be represented in the landscape that you are managing? Given the contribution of your management actions to improving regional representation of underrepresented types, what should be the desired future representation of these types in your forest? Answering these questions will help to design a coarse-filter strategy for a landscape and can help to guide harvest planning in forested landscapes into the future.

The departure of key ecosystem indicators, especially ecosystem processes, from historical conditions under which the species persisted can be useful in understanding if species are likely to be at risk of local or global extinction. But what indicators do you choose when making this coarse-filter assessment? Whitman and Hagan (2003) evaluated over 2000 biodiversity indicators and “simplified” this list to 137 indicator groups. Even this is an overwhelming number of indicators for managers to address, so Whitman and Hagan (2003) proposed that decision makers and scientists should work together with managers to identify indicators relevant to the values identified by stakeholders. Once the group of indicators is identified, then considering historical conditions may be problematic or not even particularly useful in some instances. If current human values differ markedly from the historical range of conditions and processes, then simply using the indicator without a historical reference may still provide a context for setting coarse-filter goals. There is a tendency to use indicators that are associated with current rare communities or conditions, and there is certainly a political justification for doing so. However, care must be taken in selecting indicators that may reflect rarity, isolation, or high values in the future as well (Hauer et al. 2010).

## **MESO-FILTER APPROACHES**

Although many species may receive adequate protection using a coarse-filter management strategy, some species require certain structural elements that must be present in plant communities and seral stages to ensure that they will likely persist in the management area (Crous et al. 2013). Hence, a meso-filter approach that considers the sizes, distribution, and abundance of structural elements, such as snags, logs, hollow trees, and the other elements presented in Chapter 4, are distributed across the landscape over time at a range of spatial scales (Hunter 2004). These structures are often inventoried and managed at the stand level, but it is the distribution of these habitat elements among stands across a landscape that will influence habitat quality for many species, especially those having home ranges exceeding a stand in size. How many of these elements are needed? Again, approximating a range of conditions that would be expected following historical disturbances and regrowth provides one context for estimating the numbers, especially where habitat relationships studies have not been conducted for a wide range of species. Where data are available, then data-driven habitat relationships can influence decisions regarding how much, what size, and where to provide these elements. An excellent example is the use of DecAID to guide management of dead wood across a landscape based on existing habitat relationships data (Mellen et al. 2002).

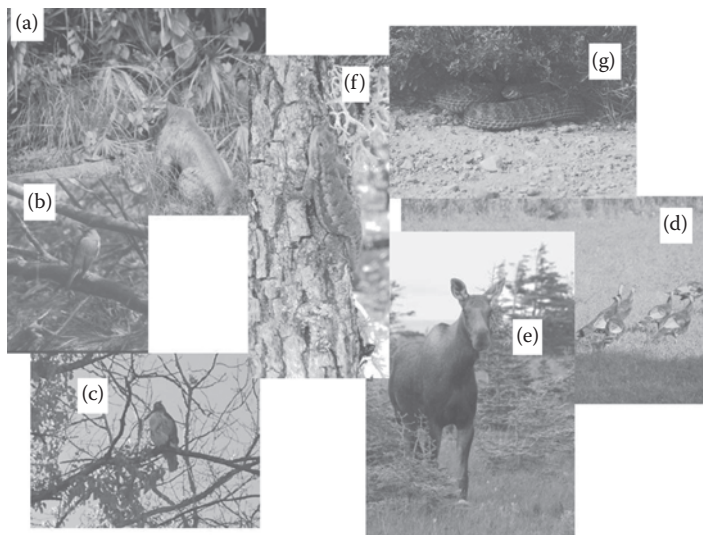
## **FINE-FILTER APPROACHES**

But the combination of coarse- and meso-filter management strategies may not provide suitable habitat for all of the species in a landscape. Some species are simply rare enough, have low reproductive



rates, have large territories, or have been adversely affected by habitat loss (or other factors) that their populations are low and they require special attention. Consequently, a “fine filter” is constructed that maintains the coarse-filter structure and the meso-filter elements but takes special management actions to conserve the set of species identified for fine-filter consideration. Those species that may need to be considered more carefully to ensure that their needs are met in the coarse-filter approach, might include those based on the following criteria (Figure 18.7):

1. *Risk*: Species that are rare or already at risk of declining in abundance so as to become locally or regionally extinct, or are already designated as threatened, or endangered through a regulatory status, or which might become rare in the future.
2. *Narrow niche breadth*: Species that are restricted to specific successional stages, especially those that are or may become uncommon or disconnected in the future (e.g., due to climate change or land use) or are sensitive to environmental gradients, such as moisture gradients or elevational gradients.
3. *Ecological function*: Keystone species are those whose effects on one or more critical ecological processes or on biological diversity are much greater than would be predicted from their abundance or biomass (Aubry and Raley 2002). Also *link species* that play critical roles in the transfer of matter and energy across trophic levels or provide a critical link for energy transfer in complex food webs (e.g., insectivorous birds) (Cohen 1984).
4. *Management focal species*: Umbrella species, which, because of their large area requirements or use of multiple habitats encompass the habitat requirements of many other species, or species that are representative of certain conditions that are now or are likely to be uncommon in the future on the landscape (Lambeck 1997).
5. *Economic importance*: Game species from which local economies and stakeholder groups derive benefit from hunting, or species that inflict costs on forest owners and managers.



**FIGURE 18.7** Examples of species selected as foci for fine-filter analyses based on (a) risk (Florida panther photo from USDI USFWS digital library), (b) narrow niche breadth (many species of neotropical migrant birds such as this hermit thrush are elective of vegetative structural stages), (c) ecological function (raptors such as this red-tailed hawk play a key role in energy transfer among trophic levels), (d) umbrella species (such as wild turkeys, which have large area requirements and use multiple vegetation conditions), (e) species of high economic importance (such as moose), (f) species for which we have limited data or knowledge (such as many species of reptiles, including this western fence lizard), and (g) species that have high public interest due to risks associated with them (such as this prairie rattlesnake).



6. *Cryptic Species*: Those species for which we have limited data or knowledge and need to be explicitly considered during management, often using expert advice.
7. *Public/regulatory interest*: Species in which society has expressed interest because of media events (e.g., rattlesnakes), public policy (e.g., migratory birds), or human health concerns (e.g., carriers of west Nile virus).

These criteria can help managers narrow the list of species that must be explicitly considered in a fine-filter approach. Once identified, then the specific habitat elements needed in appropriate arrangements, sizes, and numbers can be provided in the forest to ensure that these species have their habitat requirements met. Suring et al. (2011) developed a process for implementing a fine-filter strategy on public lands in the northwestern United States. Their process includes identification of species of conservation concern, description of habitats and other important ecological factors, grouping species, selection of focal species, development of focal species assessment models, development of conservation strategies, and designing monitoring, and adaptive management plans (Suring et al. 2011). Occasionally, fine-filter species are grouped into guilds, or an indicator species is used to represent the habitat needs of other species. *Guilds* are groups of species that share common resources such as cavity-nesting birds, bark-foraging birds, or forest-floor insectivores. *Indicator species* are species that are assumed to be surrogates for other species having similar resource needs. In the early ecological literature, however, indicator species were used to indicate certain environmental conditions (e.g., water pollution). Indicator species in forest management are used quite differently. For instance, pileated woodpeckers are often used as a management indicator species for cavity-nesting birds (Landres et al. 1988). But use of both guilds and indicator species as convenient ways to manage for multiple species is fraught with problems. Because each species has its own set of habitat requirements, no species can ever be a perfect surrogate for another. Indeed, tests of individual species responses to forest management indicate that although several species may belong to the same guild, they each respond differently to a forest management treatment (Mannan et al. 1984). Lindenmayer and Likens (2011) outlined a rigorous set of tests that would be needed to assess if indicator species are a reasonable approach to managing habitat for multiple species; rarely have these rigorous tests been conducted. Consequently, it is important to consider management strategies that focus on species, habitat elements, or broad vegetative conditions and not seek “short-cuts” that may lead to misleading management strategies and increase the level of uncertainty in meeting biodiversity goals.

## CHALLENGES TO MANAGING BIODIVERSITY

The filter strategy is based on many assumptions. But using this technique, several factors will likely influence the degree to which protection of biodiversity will be effective. The spatial scale over which the decision is made, its context, and the level of spatial detail used in defining the desired future condition (DFC) for the landscape, and the management approaches used to achieve that condition, all contribute to effective management. Similarly, the temporal framework within which the decision is made is critical. Will the DFC meet the concerns of constituents now? Ten years from now? 100 years? What is the appropriate timeframe? All decisions are couched within a number of factors associated with the uncertainty of ecological and sociological processes. How do we effectively consider uncertainties so that the decisions made are effective, yet still reflect the resilience and adaptability needed to address uncertainty? Each of the following factors must be considered in detail.

### SPATIAL SCALE

Land ownership implies a certain level of commitment to part of the Earth, and that commitment is expressed through the accumulation of individual landowner behaviors over space and time. It

would seem obvious that one landowner making a decision to manage for cavity-nesting birds in a stand on her land is easy. Just leave a certain number of trees or snags of certain sizes, and the goal is reached. Or is it? How will the actions of her neighbors influence the likelihood that these and other biodiversity objectives will be met on her land? And how will her actions influence the achievement of her neighbor's goals to provide a corridor for migrating elk? Can she trust her federal neighbors to follow through on their commitments to follow their plans, even as government policies change? Will her private neighbors sell their land? Subdivide it? Will the state impose restrictions on private land management that inhibit her ability to achieve her goals on her land and those of her neighbors? Will an NGO (nongovernment organization) intervene to offer a conservation easement and purchase development rights? All of these questions, driven by social values, are played out on the patchwork quilt of the landscape occupied by landowners and their neighbors (Figure 18.8). Effective decisions must consider this spatial context for the property or properties being managed or reserved.

Landscape management goals often are formed based on a larger regional plan at a large spatial scale (e.g., the Northwest Forest Plan) and are implemented through cumulative actions made at small spatial scales over time (e.g., stand prescriptions and forest plans). The policy guides the actions (e.g., how many wildlife trees to leave in a clearcut, how wide should a riparian buffer strip be), but decisions must be made locally to determine where and often how these should occur.

Not only must the sociopolitical framework be considered when making biodiversity conservation decisions, but the species and ecological processes must also be considered. Large territory and home-range sizes of some species, combined with the need to ensure that an adequate number of individuals of each must be maintained, may dictate the appropriate spatial scale over which planning should occur. How large an area do we need to consider in order to make effective decisions that include habitat and connections among populations for marbled salamanders? Northern goshawks? Wolverines? Where you draw the line taxonomically in your assessments and decisions will influence the spatial scale associated with the planning and decision-making process. Similarly, the dominant ecological processes that might influence the outcome of a conservation plan should also be considered (Huber et al. 2010). Wildfires, insects, disease, wind, ice, and climate change, all have ranges of frequencies, sizes, and intensities associated with various locations on this Earth. Some managers choose to manage spatial scales such that these natural disturbances are “captured” within the spatial extent of the landscape being managed (Poiani et al. 2000).



**FIGURE 18.8** Management of a 300 ha forest in western Massachusetts is heavily influenced by the goals, objectives, and actions of many adjacent landowners, illustrated here by all tax lots within a few kilometers of the forest.

TIME

Stakeholders in the outcome of a landscape management plan often view effective timeframes as days, weeks, maybe years, and sometimes decades. We all at least try to plan for our financial security throughout our lives so we are used to thinking in multiple decades. Most people want to leave a legacy of their values to the next generation. We humans have a more difficult time thinking in terms of multiple lifetimes. Many Native American cultures view sustainability as seven generations (Hansen 2011), although such a view may not be common in other cultures. Yet, some plans made to achieve biodiversity goals may not be fully realized for many decades. The recovery of nesting habitat for northern spotted owls may take 200 years in many locations and even longer for nesting habitat for marbled murrelets (Spies et al. 2007). In these examples, a decision was made to designate a part of the landscape that contains many square kilometers of young plantations as late successional reserves. Many stakeholders are unable to understand how recent decisions are effectively leading to the intended goal. It is important to not just consider human lifetimes when considering the appropriate temporal scale for projecting likely effects of biodiversity decisions. It is equally as important to understand the effects of decisions relative to the multiple generations of key species affected by the decisions. Consider long-lived species such as box turtles and Puerto Rican parrots. These are such long-lived species (40 years or more) that by the time declines in populations are detected, the options for recovery may be very limited. Similarly, recovery, when it is possible, may take multiple generations for these species—hundreds of years. Consequently, the appropriate timeframe for considering effectiveness of biodiversity management over large landscapes is driven by the interface of several key actions:

- 1. Human schedules for implementing the plan
- 2. The inherent rates of growth and disturbance affecting the vegetative (and occasionally the physical) components of the environment
- 3. The potential and realized rate of population growth for key species
- 4. The rate of movement and colonization of habitat for key species

Consequently, there is not a standard appropriate timeframe associated with all biodiversity management plans. Consider the potential for documenting responses of organisms to management actions. The species in Table 18.1 are ranked by their potential longevity in years. This would be the maximum time needed for a complete turnover in a generation. The number of generations is then portrayed for 40-, 100-, and 200-year rotations in a forest managed using even-aged systems. If a species can reproduce multiple generations on a site before it is harvested, then any single individual

**TABLE 18.1**  
**Approximate Longevity in Years and as Expressed in Number of Generations per Rotation under Even-Aged Management for Six Species with Very Different Life Histories**

Species	Longevity (Years)	Rotation Length (Years)		
		40	100	200
Short-tailed shrew	3	13	33	67
Winter wren	4	10	25	50
Spruce grouse	5	8	20	40
Red-cockaded woodpecker	16	3	6	13
Great-horned owl	27	1	4	8
Box turtle	50	1	2	4

faces less of chance of being displaced to different home ranges. Both the temporal scale associated with the reproductive capacity of the species and the frequency of habitat displacing disturbance must be considered over a spatial scale large enough to allow species to move to available habitat as disturbances displace the species.

Clearly, natural disturbances have probabilities of occurring at a range of frequencies that could lead to displacement for these species. As rotation lengths depart from the historical range of natural disturbance frequencies, the risk to long-lived species (which often have low reproductive rates) increases. In addition, because they are so long-lived, changes in populations can be subtle, making it difficult to detect population declines. Conversely, documenting recovery of these species can also be difficult, requiring long periods of population monitoring.

There are at least two dominant additional factors that must be considered when making biodiversity decisions over large multiowner areas. Land tenure can influence achievement of goals, particularly if the parcels being sold or inherited change owners having one set of core values to another. Rotation lengths also influence the time that a forest will be suitable for a set of species. An understanding of these transition probabilities can be particularly important in understanding the likely trajectory of landscape change over the planning period (Spies et al. 2007).

## UNCERTAINTY

One of the greatest uncertainties facing conservation biologists and land planners is development of conservation and management strategies with incomplete information about the suite of species under consideration. Based on past research, we know enough about the habitat used by many species to at least develop reasonable management plans. However, for some species, we know nearly nothing, and then there are all of the species not described that we have yet to discover.

Consequently, it is important that if biodiversity conservation is a primary goal for a management plan, then a reasonable course of action is to follow the precautionary principle and err on the side of conservation rather than resource extraction. This approach places the burden of proof on managers to demonstrate that there is minimal risk to conservation of biodiversity when extracting resources. Then, once the plan is developed, we can monitor using techniques that will add to the information available to make decisions (Schultz et al. 2013). Surveying and managing, formal monitoring protocols, and moving the system toward the HRV while monitoring key resources are approaches that can improve knowledge and reduce uncertainty over time.

Land management planners must define the spatial and temporal grain (finest level of information needed), extent (outer bounds of the planning problem), and context (surrounding landscape conditions). Once the spatial and temporal context for biodiversity decisions has been decided (perhaps one of the most critical first decisions), planners suddenly find themselves faced with a number of uncertainties that influence the likely effectiveness of their plans. One key uncertainty is the continued social commitment to biodiversity values. Societal values change, and planning must be adaptable to that change (Figure 18.9). Values placed on deer in the United States have evolved from largely utilitarian, to protection, to recreation, to nuisance, to public health concerns. How will values change for northern spotted owls? Townsend's big-eared bats? Burying beetles? We tend to think of goals and objectives as being relatively stable. But they will change, and biodiversity protection plans should reflect an ability to adapt to new values.

There also are a number of biophysical uncertainties: fires, floods, invasive plants and animals, disease, and global climate change, to name a few. For many of these factors we have information that can help us understand probabilities of occurrence of these biophysical factors over time at varying spatial scales. Hence the uncertainty, or likelihood, of these events or effects can be quantified. In so doing, we can also assign risks associated with these events. Many forest "health" issues are framed within this risk assessment paradigm. For instance, an "unhealthy" forest is often considered one with a higher than expected chance of wildfire, disease, or insect irruption, and the ensuing ecological and social effects can be predicted. Further, these risks can be expressed as a



**FIGURE 18.9** Prescribed burning was used to maintain grasslands and grass–shrub savannas in pitch pine forests of Massachusetts to maintain conditions important to many species of vertebrates and invertebrates. Over time, public values may shift due to air quality, smoke, and risk of fire escapement, placing systems like these at greater risk.

departure from the historical range of variability (HRV). But in many systems we cannot return many attributes to fall within the HRV. For instance, in the northeastern United States there are no longer passenger pigeons, American chestnuts, and (as far as we know) wolves. Atlantic salmon are effectively gone and society will not likely pay for their restoration to historical levels. Although some aspects of the northeastern hardwood forests and associated streams have recovered, some aspects will never recover. The past may help inform decisions, but the past cannot define a goal for the future; too much has changed. Even the ability for the past to inform decisions is weakened as we see systems develop in ways that they never have developed in the past. Uncertainties proliferate regarding how systems might develop and how aspects of biodiversity might respond, so we become increasingly unable to use history as a guide to development of desired future conditions.

Finally, there are political uncertainties. Although political decisions are (usually) an outcome of societal values, our political system, and those of other countries, can result in decisions being made that result in significant constraints (or freedom) to achieve biodiversity goals. Honoring the Kyoto Agreement, responses to the events of 9/11/2001, economic recovery, and going to war are decisions made by a few individuals that affected many individuals. Once made, these decisions have significant effects on the certainty with which biodiversity decisions will truly achieve the intended objectives.

Further, changes in policy such as modifications to the Endangered Species Act, Clean Water Act, the National Forest Management Act, and others, are not only likely, they are inevitable, given the changes in society and politicians that we can expect over the next seven generations. Even now we are seeing a society that is becoming more and more divided on many issues important to maintaining a balanced and functional social system. As philosophical beliefs of our elected and appointed officials wax and wane, so will the degree to which policies provide the legal framework for biodiversity planning and decision making.

## SUMMARY

Biodiversity is represented by the genes, organisms, populations, and species of an area, and the ecosystem processes supporting them. Contemporary approaches to biodiversity conservation



typically take a filter approach. Coarse-filter strategies use the proportional representation of plant communities, successional stages and other classes of biophysical conditions as a guide to providing habitat for most species most of the time. Meso-filter strategies further consider the habitat elements present within these biophysical classes in a way that “captures” even more species. Fine-filter strategies consider the needs for a few individual species that are of high social importance. The approach provides the context within which harvest planning can be developed that would minimize risk to biodiversity while allowing resource extraction. This approach is based on numerous assumptions and is influenced greatly by the choice of spatial and temporal scales over which it is employed. Further, uncertainty in the effectiveness of the strategies often leads managers to follow the precautionary principle when conservation of biodiversity is a primary goal for land management.

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# 19 Landscape Management Plans

Plans. You probably have some for the weekend ... for the semester ... for your life. The more complicated the plans and the longer the timeframe, the less certain that you can be that you will realize your plans. There is uncertainty, but to move forward without a plan, to manage haphazardly, probably will not allow you to reach a goal or a set of goals either for your life or for biodiversity.

## ESTABLISHING GOALS

In order for society to achieve biodiversity goals, the filter approach (Chapter 18) or some other comprehensive approach must be described in a plan that is then implemented and monitored to ensure that the risk of losing species from a region is minimized. Plans must have goals. Goals are developed by the landowner, land manager, and/or stakeholders who have a vested interest in the future of the land. Goals are a reflection of the desires and actions of stakeholders, such as affected publics and nongovernmental organizations (NGOs). These goals usually reflect the priorities for ecosystem structure and composition, plant communities, and individual species, as defined in Chapter 18, and also by the economic, aesthetic, or cultural goals for the region. Although involving interested publics in plan development is imperative, under contentious circumstances, the initial investment in stakeholder involvement can be significant. In the long run, it is usually worth the effort.

*Regulatory Goals:* Although private forest landowners may have goals for their forests that are not driven primarily by biodiversity protection, they must still abide by federal and state laws (see Chapter 24). Some goals are prescribed in federal, state, and occasionally local policies. Probably the most powerful environmental protection law in the United States is the Endangered Species Act (ESA). The ESA is a species-by-species approach to addressing species at risk of being eliminated from all—or a significant portion—of its geographic range. Once listed, a species recovery plan must be developed, implemented, and monitored to provide the basis for recovery and removal from the list. Habitat restoration to allow species to recover from risk of extinction represents one type of fine-filter goal that can be combined with other fine-filter goals. Coarse-, meso-, and fine-filter strategies combine to minimize adverse effects of human actions on biodiversity loss (see previous chapter).

Landowners must consider the effects of their actions on protected species, especially those protected under the ESA. Private landowners, corporations, state or local governments, or other nonfederal landowners who wish to conduct activities on their land, that might incidentally harm (or “take”) wildlife species listed as endangered or threatened under ESA, must first obtain an *incidental take permit* from the United States Fish and Wildlife Service. To obtain a permit, the applicant must develop a *Habitat Conservation Plan* (HCP) designed to offset any harmful effects that the proposed activity might have on the species. An approved HCP allows management to proceed, while continuing to promote the conservation of the species of concern. This approach is allowed under the “No Surprises” regulation in the ESA and provides assurances to landowners participating in these efforts that they will be allowed to proceed with management after an HCP has been approved without a “surprise” that new regulations or other restrictions will be imposed on them.

The “incidental take permit” allows the managers to manage forests as described under an HCP that has been approved by the regulatory agency (usually the United States Fish and Wildlife Service). In this instance, *take* does not mean directly killing the animal but, rather, means removal of *critical habitat* for the species, at least in the near term. The agency must designate critical habitat following peer review by scientists (Greenwald 2012). Court battles have resulted from the

debate over the issue of removal of critical habitat constituting “take,” leading to judicial decisions that have broad implications (e.g., the Sweet Home Decisions). The outcome of those battles was the HCP approach that is consistent with the required Recovery Plan for the listed species. Specific decisions regarding what constitutes “take” of species through habitat modification, often requires that site-specific decisions be made to address particular issues (e.g., timber sales). Species Recovery Plans and HCPs require an understanding of not only the effects of management actions on the target species over large multi-ownership areas, but also the effects on other species, including people, both regionally and locally.

HCPs are often used to ensure that both the landowner(s) and regulatory agencies are in agreement about the goals and objectives for the land, the resulting level of “take,” and the likely effects of “take” on the portion of the population covered by the plan. Landscape management plans may or may not be HCPs, depending on the goals of the landowner and the presence or absence of federally listed species. The United States Forest Service has long used harvest-planning models, wildlife–habitat relationship(s) models, and other tools to assist with development of forest plans, required under the United States National Forest Management Act.

States also address complex planning problems using landscape management plans. For instance, habitat issues on the Elliott State Forest in Oregon prompted the Oregon Department of Forestry to develop a landscape management plan that interfaces with an HCP to meet the habitat requirements for northern spotted owls, marbled murrelets, and other species, while also considering economic effects on local communities (Oregon Department of Forestry 2006). Many states have natural heritage programs and lists of sensitive species at risk of being lost from the state or province. Some, such as Massachusetts, have their own state lists of endangered species requiring landowners to take particular actions to avoid habitat damage for state-listed species. Because land managers often are faced with more than one species that is of regulatory concern, multispecies management plans and HCPs are becoming more common.

*Nonregulatory goals:* Goals usually do not stop with species of conservation concern. Game species, ecological keystone species, and others may be the focus of a particular landscape, and clearly, human interests must also be accommodated. Protection of cultural sites, recreation areas, aesthetics, and economic income must all be balanced with the habitat management approaches chosen by the planners and managers. Without a formalized plan in place, the myriad possible effects of management on all of these values can be overwhelming to those managing a landscape. Without a plan, the risk of taking an action that has long-lasting adverse consequences is likely to increase.

Nongovernment organizations (NGOs) also are using landscape management plans to aid in large-scale planning efforts. The Nature Conservancy uses models of ecoregion structure and composition, in combination with principles of landscape ecology, to identify areas of potential high priority for protection or recovery (Poiani et al 2000, Groves et al. 2002). More species-specific groups, such as the Ruffed Grouse Society and the Wild Turkey Federation, may employ landscape management plans at times to facilitate management on public and private lands (Yahner 1984, Ferguson et al. 2002). In so doing, these groups are ensuring, to the degree possible, that the needs of those species of most concern to their constituents are met, though they also freely recognize the need to consider many other species, ecosystem services, and social values as well.

Goals reflecting conservation of biodiversity may also be driven by economics. Forest industries now often seek green certification (see Chapter 23) as a way of assuring that forest practices are sustainable, including measures taken to conserve biodiversity.

Large-scale ecoregional assessments are often used as the context for landscape management plans that address biodiversity protection, among other social values. Often, these assessments rely on landscape management plans as the mechanism to implement the regional plan. Strategies for biodiversity protection are often established at regional scales, which guide the development of landscape plans, which guide the development of stand prescriptions. Regional strategies are realized by implementing stand prescriptions over landscapes and implementing landscape plans over regions. Guidance comes from large spatial scales, and implementation is cumulative over small

spatial scales. Although one can argue the details of effectiveness and subsequent use of the information, efforts such as the Northwest Forest Plan (FEMAT 1993), the Columbia Basin Ecoregional Assessment (Wisdom et al. 1999), the Willamette Valley Alternative Futures analysis (Hulse et al. 2002), and the Sagebrush Ecoregional Assessment (Knick and Rotenberry 2000, Connelly et al. 2004), all contributed to a foundation or framework within which more local decisions could be made to contribute to broader goals and objectives. Each of these assessments has relied on landscape management plans to achieve long-term goals.

## CURRENT CONDITIONS

Classification of vegetation into plant communities and successional stages provides the basis for assessing coarse-filter goals and subsequent identification of under- or over-represented communities on the landscape. This information, and an estimate of conditions that are likely to occur on the ownerships outside the boundaries of the landscape (the context) being managed, can help to guide the articulation of the desired future condition.

Stand maps for each ownership of the forest (cover type and age), in combination with information on roads, streams, underlying geology, known locations of sensitive species, and culturally important sites, can provide the basis for describing the current conditions for many ecological conditions and social values. This information clearly should include an assessment of the distribution and connectedness of plant communities and seral stages, the levels of certain habitat elements associated with focal species, and availability of habitat for each focal species. Collectively, this information is used to describe the *current condition* of the area and is the basis for development of a desired future condition for the landscape.

Once the coarse-filter goals have been established and the species of concern have been identified, then the specific habitat elements needed by each species can be assessed over space and projected over time to understand how management alternatives might lead to changes in habitat quality for these species. The particular elements related to reproduction and foraging, including the spatial requirements, connectivity, and other attributes associated with habitat quality for each species, should be identified and structured in a way that allows large-scale assessment of habitat for each species (McComb et al. 2002, Spies et al. 2007). In some cases, population viability analyses may need to be conducted, a topic that we will cover in detail in Chapter 21. With the use of geographic information systems (GISs), remotely sensed data interfaced with ground plots, and LIDAR images, rapid assessments of current habitat availability for a suite of species is possible. In addition, these tools can help the planners design a desired future condition (DFC) that meets the coarse-, meso-, and fine-filter goals.

## DESIRED FUTURE CONDITIONS

Given the list of values generated by stakeholders, a number of questions should be addressed as you develop a description of the desired future condition:

- Are there plant communities or seral stages that are underrepresented on the landscape now, as determined by comparison of current proportions to proportions represented in some reference condition (often the historical range of variability, if appropriate)?
- Are levels of specific habitat elements, within and among stands, sufficient to minimize a risk of losing species from the landscape? These levels may also be compared to a reference condition or to known habitat relationships (e.g., to DecAID, Mellen-McLean et al. 2009).
- What pattern of habitat availability for each focal species would best contribute to the regional habitat for the species? This is especially important for regulated species. How much latitude do you have in managing habitat for these species without risk to populations inside and outside of the landscape?

- Are there species or values that you would like to favor on the landscape in the future that currently may not be provided for now?
- What species or species groups provide the greatest opportunity for contributing to local or regional populations?
- Are there any species or values that are sensitive to land-use that would likely be eliminated from the area in the future, if current management practices continued?
- Could these species or values be accommodated wholly or in part on your landscape?

Once the questions listed above are addressed, they become the basis for developing a Desired Future Condition (DFC) for each ownership that collectively achieves the DFC for the landscape. Just as no single stand condition will meet the needs of all wildlife species or values, neither will a single DFC for a landscape meet the needs of all species and values. There are always tradeoffs to assess. Habitat for some species will decline over time, while habitat for others will increase. Careful planning of stand treatments over space and time will move a landscape toward a desired future condition in an attempt to meet multiple goals (Bettinger et al. 2001, Öhman et al. 2011).

Often, it is useful to consider a set of DFCs, in which one DFC leads to another over time, to achieve a series of objectives over both space and time. This approach also allows the planner to think about how each DFC can be achieved, while still providing a high likelihood that future DFCs can occur later in landscape development, even in the face of uncertain events such as natural disturbances or social change.

Life history information, habitat suitability models, or habitat relationship(s) models can help to ensure that these conditions are provided over sufficient areas with sufficient connectivity (see Chapter 16). Bettinger et al. (2001) and Öhman et al. (2011) demonstrated how harvest planning can be integrated with habitat goals for selected species to achieve multiple objectives in an economically efficient manner. Once these conditions have been described, then one or more DFCs for the landscape can be articulated by describing in writing what you want and by mapping these conditions to ensure that they are indeed feasible, given other constraints and opportunities (Bettinger et al. 2001, Oliver et al. 2011). These maps and descriptions of the desired future conditions should be developed in cooperation with stakeholders and vetted by all stakeholders affected by plan implementation.

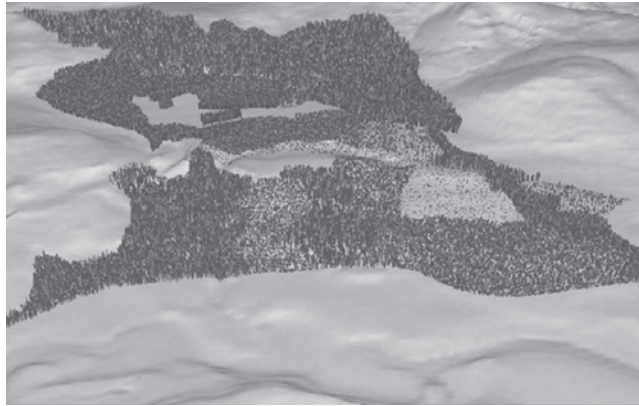
## **PATHWAYS TO DESIRED FUTURE CONDITIONS**

Once you know what the current conditions are and have described the DFCs, then it is important to understand how you will implement the management in a manner that will likely reach the DFC. It is important, at the very least, to develop a set of maps that clearly indicate how you will plan to move the forest from its current condition to the desired future condition through harvest planning, reserves, and other management approaches. There are several possible tools that you can use to project the future conditions of the forest. Interfacing stand growth models (e.g., Forest Vegetation Simulator, Teck et al. 1996) with GIS tools can allow simulation of landscape change over time (Figure 19.1). The Landscape Management System developed at the University of Washington and Yale University provides many opportunities for understanding changes in forest structure and composition over time and has been linked to habitat models for several species that allow an understanding of habitat changes over time (Marzluff et al. 2002, Oliver et al. 2011). These sorts of tools help you to understand if there is, or is not, a clear and achievable path from the current condition to each of the desired future conditions and allow comparisons with past conditions and likely achievement of goals (Figure 19.2).

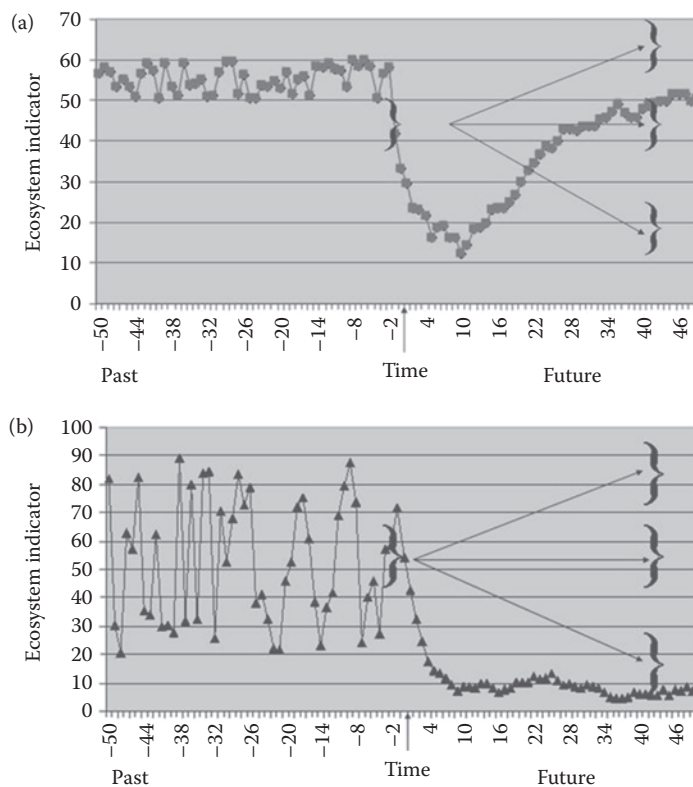
## **DEVELOPING THE LANDSCAPE MANAGEMENT PLAN**

With knowledge of current conditions, what conditions you would like to achieve, and how you might be able to change the landscape to achieve your goals, you are now ready to write a landscape





**FIGURE 19.1** Example of one-time step in forest landscape pattern. Information on each stand and information among stands allows planners to understand if a desired future condition will likely be met. (From McGaughey, R.J. 2001. Using data-driven visual simulations to support forest operations planning. Pages 173–179 in *Proceedings of the First International Precision Forestry Cooperative Symposium*. Precision Forestry, Seattle, Washington; Landscape Management Systems; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)



**FIGURE 19.2** By understanding historical changes in ecological indicators such as habitat availability through present (time = 0) and likely future changes in the indicators, we can begin to see if future conditions are or are not likely to meet societal goals (brackets) for the indicator. Note that the trajectory (a) indicates a recovery following a lag period while (b) indicates a new indicator state in the future.

management plan. You may write several plans, each feasible, but with different emphases in different plans (including a “no action” plan), so that stakeholders, shareholders, and constituents, can understand possible alternatives, and choose among them.

### **POLICY GUIDELINES FOR HABITAT CONSERVATION PLANS**

When a landscape management plan also is a Habitat Conservation Plan, as defined under section 10 of the ESA, then certain guidelines apply when developing the plan. Paraphrasing from Oregon Department of Forestry 2006, four tasks must be completed to determine the impacts that are likely to result from the proposed taking of a federally listed species:

1. Delineate the boundaries of the plan area.
2. Collect and synthesize the biological data for all species covered by the HCP.
3. Identify the proposed activities that are likely to result in incidental ‘take.’
4. Quantify anticipated ‘take’ levels.

Usually, an impact assessment must be developed, that meets requirements for an Environmental Assessment under the National Environmental Policy Act (NEPA) as well as Section 10 of the ESA. This assessment includes techniques that will be used to monitor, minimize, and mitigate impacts on the listed species, the funding available to implement the plan, and the procedures to deal with uncertainty. Section 10(a) of the ESA requires that permit issuance does not ‘appreciably reduce the likelihood of the survival and recovery of the species in the wild.’ (Oregon Department of Forestry, 2006). The ESA does not require that HCPs result in the recovery of species covered under such plans. Since most landowners manage only a portion of the geographic range of a species, they may contribute to population change, but do not directly control it. Each HCP must contain measures that result in impacts that are consistent with the long-term survival of the listed species. Long-term survival includes the maintenance of genetically and demographically viable, well-distributed populations throughout the geographic range of each listed species. Actions that will meet the intent of long-term survival, will vary from one species to another, and must be approved by the regulatory agency.

Until recently, the effects of climate change were not explicitly considered when developing HCPs. Bernazzani et al. (2012) recognized the need to include climate change effects on the likelihood of achieving HCP goals, and offered the following guidelines when developing an HCP:

1. Identify species at-risk from climate change.
2. Explore new strategies for reserve design.
3. Increase emphasis on corridors, linkages, and connectivity.
4. Develop anticipatory adaptation measures.
5. Manage for diversity.
6. Consider assisted migration.
7. Include climate change in scenarios of water management.
8. Develop future-oriented management actions.
9. Increase linkages between the conservation strategy and adaptive management/monitoring programs.

It should be apparent from this list of recommendations that, in many instances, active management, in concert with reserve design, will be needed to meet HCP objectives for many species.

The effectiveness of the HCP can only be assessed based on monitoring information. Section 10(a) of the ESA requires that an HCP describe the specific monitoring measures the applicant will conduct to ensure that the plan is being implemented as described, and is effective in achieving its goals. Such an approach also helps both the land manager and the regulatory agency to deal with

unforeseen changes in habitat or populations, such as those that might result from climate change (Bernazzani et al. 2012). This approach also lends itself well to an *adaptive management* strategy where monitoring data may allow managers and regulators to agree on mid-course changes in the plan to better meet the goals of both parties (see Chapter 22).

The HCP should consider alternative management strategies, including at least one that would not usually result in ‘take’ (usually a ‘no-action’ alternative, although in some cases no action could result in ‘take’ as well), and the reasons why that alternative is not selected. The no-action alternative may not be selected for economic reasons, or for ecological reasons, if disturbance, climate change, disease, or other exogenous factors are likely to affect the target population(s). The regulatory agency may also place additional requirements on the land managers, including an implementation agreement, which is a legal contract that describes the responsibilities of all HCP participants.

### GENERAL STRUCTURE OF THE LANDSCAPE MANAGEMENT PLAN

A landscape management plan can be structured in many ways similar to a stand management plan or prescription. It is important to have a written plan that provides a general schedule of predicted activities and treatments because people charged with implementing the plan will likely change over time, especially on multiowner landscapes. Management actions and the schedule may need to be modified somewhat, as new information becomes available through monitoring and additional research, so the plan should be viewed as a strategic plan (a general strategy for achieving goals), and not a tactical plan (a specific set of actions to be taken in certain places at certain times). A harvest plan that may result from a landscape management plan would be an example of a tactical plan. The plan should include:

*Context:* What is the historical, spatial, and temporal context for the forest? What opportunities and restrictions exist that influence development of a landscape plan? What policies drive current actions on the property? What are the physical, biological, and social resources that must be considered? This information can include the goals and purposes for having a plan, and the process used to arrive at the goals (e.g., public meetings and stakeholder involvement).

*Current Forest Condition:* What are the current conditions with regard to areas of vegetative communities, seral stages, and stand sizes? What conditions exist for those species or resources that you would consider as part of your fine-filter approach? To comply with National Environmental Policy Act (NEPA) requirements, this section should include a description of the affected environment. This section should include at the very least:

1. The current state of vegetation (plant communities and seral stages)
2. Species of plants and animals likely to be affected by proposed treatments
3. Habitat requirements of threatened or endangered species or other focal species of concern to stakeholders
4. Location and extent of wetlands or other sensitive ecosystems
5. The underlying geology and soils and the potential of those conditions to maintain the current species assemblages
6. Current land-use patterns within the planning area and the influence of patterns outside of the area on conditions in the planning area
7. Air and water quality issues apparent now, especially if they are contributing to impaired watersheds and airsheds, under the Clean Water Act, and the Clean Air Act, respectively. If any impairments are noted, then remedial measures that are underway should be described (e.g., Total Maximum Daily Load restrictions under the Clean Water Act)
8. Significant cultural resources, especially those that are protected under antiquities acts. Any ongoing management designed to restore or maintain these resources should be described

*Desired Future Conditions:* All of the factors listed above should be addressed in the DFC as components of the affected environment. What are the conditions that you would like to produce over time for each of several alternative plans? One of the alternatives should be the “no-action” alternative that can be compared to other alternatives. Often a *preferred alternative* is also described and is one that the developers of the plan find most desirable. Providing alternatives allows stakeholders the opportunity to comment on the alternatives and to compare the costs and benefits of various alternatives. This approach is required if the plan involves federal lands or impacts federal resources (e.g., endangered species) in the United States, and, therefore, must follow NEPA policies. Specifically, the DFCs in each alternative should include

1. Coarse-Filter Goals and Objectives: What are the general goals and objectives with regard to plant community representation, seral stages, and stand sizes? What is the rationale for these goals (based on the context above)? Was the Historical Range of Variability or some other reference condition used, and if so, how was it chosen?
2. Meso-Filter Goals and Objectives: Standards and guidelines should be included to provide the manager with the numbers, levels, and distributions of habitat elements that should be provided in the seral stages within each plant community. A rationale for these goals should be provided.
3. Fine-Filter Goals and Objectives: What species are considered explicitly in the management strategy? How do these goals complement coarse- and meso-filter goals? A rationale for selecting these species should be provided.

*Management Actions to Achieve Desired Future Conditions:* What will you do to achieve your desired future conditions? How long do you think it will take to achieve them? How long will they last? How much will it cost? The answers to these questions can best be answered following projections of the landscape forward in time, to understand if there are realistic pathways of forest development to allow DFCs to be realized over space and time.

*Monitoring Plans:* What will you measure and how often will you measure to determine whether your management plan was implemented correctly, and whether the actions were effective? How will you decide if you need to change your management plans? What thresholds must be reached before you make a change? See Chapter 22 for more guidance on developing monitoring plans.

*Budget:* What will implementation and monitoring of the plan cost? Where will the funds come from? In the event of a budget shortfall, what contingency plans are in place?

*Schedule:* What management actions are scheduled to occur during each decade for the next few decades? How will these actions be moving the forest toward the desired future condition? Remember that a plan such as this should be revisited and revised periodically—often every 5–10 years—to allow incorporation of information gained through monitoring and through publications in the scientific literature.

*References:* Use references from refereed journals (journals that require that papers published have been reviewed by peers, and that only the very highest quality papers are published) as much as possible to support any assumptions that were made in articulating the desired future condition. Contemporary approaches to finding information often involves searching the web, using a readily available search engine, but information from unknown sources should be viewed with caution—it is far too easy to find incorrect information. Search engines that search the refereed literature such as Google Scholar can be quite helpful, but be sure to check the original literature before citing it.

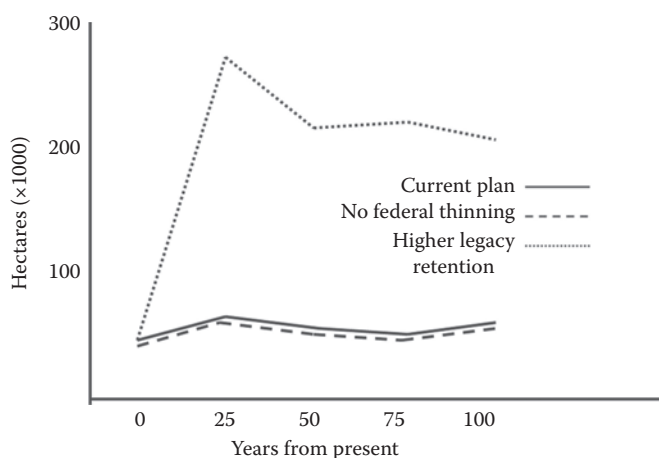
## CONSIDERING ALTERNATIVE PLANS

When considering alternatives to large complex plans, such as the Northwest Forest Plan (NWFP) the landscape dynamics and resource outputs from a large area must be analyzed. The NWFP was developed to address biodiversity concerns on public lands over the geographic range of the

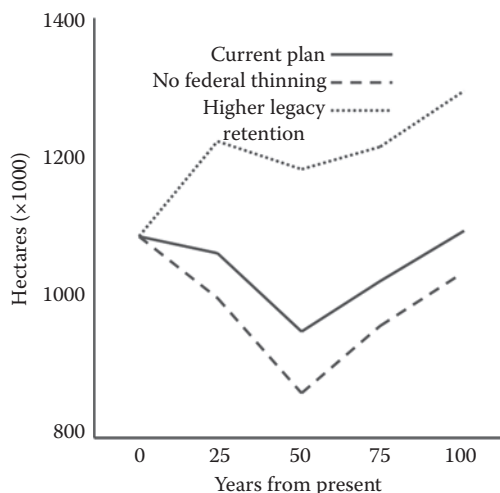
northern spotted owl. Species that fell into a high-risk fine-filter group that needed particular attention to ensure that the plan met their needs, included spotted owls, marbled murrelets, red tree voles, and many others. Indeed, over 1000 species were assessed to understand the risks of implementing one of 11 land management options under the NWFP. The species-by-species assessments constituted the fine-filter assessment designed to ensure that those species that may not be captured by the coarse-filter planning strategy (moving more forest into later successional stages) would still be addressed and protected. But additional questions arise with a planning process like this. As older forests are allowed to develop, young forests decline in abundance, and so do the species associated with younger forests. In particular, young forests that are structurally or compositionally complex (as would occur after a natural disturbance) may decline to an even greater degree, as timber production is intensified on the remaining land-base. Hence, focal species that are known to be associated with particular conditions (snags in young forests, hardwoods, shrubs, etc.) in early seral stages are selected as examples of how a species might respond to changes in forest conditions (Betts et al. 2010, Swanson et al. 2010). By projecting habitat availability for these species forward in time, we can see how their habitat might change under current and alternative policies. Examples for early successional snag associates (western bluebirds) and high-contrast forest edge associates (olive-sided flycatchers) are provided in Figures 19.3 and 19.4. In these cases, we can see that current policies either maintain or increase habitat for the species over time, but that some policies are likely to be better than others in providing these conditions. Inferences can be made to other species that also might be associated with these conditions. It is important to remember, though, that these are not indicator species. Since each species has its own habitat requirements, the responses of one species will never accurately reflect the responses of other species.

### FINDING SOLUTIONS TO LAND MANAGEMENT PLANNING PROBLEMS

With any forest consisting of multiple stands, there are several acceptable plans (plans that will meet goals at varying costs), and a number of unacceptable plans (plans that will not meet goals or will only succeed at unacceptable costs). Usually, there is more than one way of managing a landscape to achieve the desired future conditions. If each of ten managers designed a management plan to achieve one set of DFCs, the ten plans probably would all be different in some ways. So, given these



**FIGURE 19.3** Projected response of western bluebirds to the Northwest Forest Plan and two alternative policies, Oregon Coast Range. Note that two alternatives had the same likely influence on habitat availability over time, and produced identical projections. (Redrafted from Spies, T.A. et al. 2007. *Ecological Applications* 17:48–65. With permission from the Ecological Society of America.)



**FIGURE 19.4** Projected response of olive-sided flycatchers to the Northwest Forest Plan and two alternative policies, Oregon Coast Range. (Redrafted from Spies, T.A. et al. 2007. *Ecological Applications* 17:48–65. With permission from the Ecological Society of America.)

multiple alternative plans, how can we find the best plan? Well, the best plan depends on who is doing the judging. About all we can usually hope for is a plan that is mutually acceptable to a group of stakeholders.

One way of finding not only an acceptable solution, but also a good one, is to develop a number of plans that are all considered acceptable. Each solution can then be ranked from highest to lowest for any number of values that the manager (or society) places on the landscape. Those plans with the lowest cumulative ranks are those that are likely to be better than those with higher total ranks (Table 19.1). Given a clearly articulated and agreed upon set of goals and constraints, harvest planning software can generate a suite of plans that meet goals and fall within constraints, and these can then be evaluated by stakeholders. This approach also has the value of being able to map the projected changes in landscape structure and composition over time, allowing stakeholders a better understanding of how the landscape might appear in the future (Oliver et al. 2011). Care must be taken not to overwhelm stakeholders with too much information; information overload can result in stakeholders not being able to make clear choices among alternatives (Robson et al. 2010).

**TABLE 19.1**

**Example of a Process of Prioritizing Values Associated with Four Resource Values in Each of Five Alternative Plans**

Plan	Turkey	Timber	Aesthetics	Trout	Sum
A	3	2	3	3	11
B	4	4	1	1	10
C	5	1	4	5	15
D	1	3	2	2	8
E	2	5	5	4	16

*Note:* If the stakeholders included turkey hunters, forest products industry, hikers, and fly-fishers, and each group ranked each alternative (1 = best, 5 = worst), then the sum of the ranks can provide an idea of which plan represents the best compromise (lowest sum = plan D). Note that weights are not assigned to alternatives in this example.



I provide a very simple example considering five plans developed to meet a set of DFCs for four resources in Table 19.1. Of the five plans, all of which are acceptable, plan D would seem to be the most desirable option from the standpoint of the four resources being considered. We could have any number of resources represented, and we could use different rules on which to base our selection of the preferred alternative (no resource must rank below 3, for instance). These rules will be based on the values assigned by stakeholders in an iterative fashion until those involved can agree that the selected alternative is the one most likely to address public values over multiple ownerships now, and into the future. Indeed, the most difficult (and most informative) part of the planning process can be the social debate necessary to ensure that the selected alternative is most likely to meet social goals.

Comparisons among alternative plans should be done jointly with affected constituents and planners, and scientists should be available to assist with interpretation of assumptions. It is entirely possible, and indeed quite likely, that arguments might ensue among the constituents, planners and scientists, once a set of options has been assessed. The arguments could result in a determination that none of the alternative plans are socially acceptable, and that the planning process should start anew. A similarly likely outcome, especially given the number of assumptions upon which future projections are based, is that the affected parties reach agreement on a preferred alternative and implement it; but based on monitoring data, the observed responses are not likely to be what was expected. Gachechiladze-Bozhesku (2012) found that rarely are stakeholders involved in follow-up assessments of the progress of a plan toward meeting DFCs. Should important societal values not be sustained due to these departures from predictions, there may yet again be a reason to start the planning process anew with a greater understanding of the assumptions that must be changed. Even if the system responds as anticipated, and current social values are sustained, social values, including biodiversity goals, are not static. As societal expectations evolve, so do biodiversity goals. Stuff happens. Evolution of cultural mores, in addition to unanticipated events (9/11, tsunamis, wildfires, disease), can drastically alter the perceived values and importance of biodiversity in our cultures. It should be accepted that assessments will need to be revisited as new social issues emerge, due to the dynamic nature of cultural values.

## PLAN EFFECTIVENESS

Plans abound, but do they achieve the intended goals? Recently, there have been several assessments of the effectiveness of species recovery plans and HCPs in preventing further declines in populations of threatened species. Shilling (1997) noted that the number of threatened and endangered species in the United States is increasing monthly, and critical habitat (as defined in the ESA is habitat required for continued existence of the species) is constantly being destroyed. Shilling (1997) contends that the application of HCPs may be making things worse, not better, for these species. Bingham and Noon (1997) also questioned the logic of allowing incidental “take” by having an approved HCP. They found that mitigation solutions are often arbitrary and lacked a basis in the habitat requirements for the species. They proposed that the concept of “core area” (that portion of an animal’s home range that receives disproportionate use) be used as the basis for mitigating habitat loss within an HCP. Clearly, habitat requirements need to be addressed at an appropriate scale when mitigation measures are involved. Despite concerns raised by Bingham and Noon (1997) and Shilling (1997), Langpap and Kerkvliet (2012) found that single-species HCPs have been effective in contributing to species recovery, but that was not necessarily true for multispecies plans. They found that plans that covered larger areas were more likely to be successful than plans that covered smaller areas (Langpap and Kerkvliet 2012).

Plan effectiveness is always in doubt. All of the uncertainties described in Chapter 18 can rear their ugly heads in the face of the best-made plans. Wilhere (2002) made the case that HCPs entail a compromise between regulatory certainty and scientific uncertainty, and previous authors indicated that many HCPs do not adequately address scientific uncertainty. Monitoring the implementation

and effectiveness of the plan is one way of acquiring and applying information to allow continual improvement of the plan (Wilhere 2002). Adaptive management has been promoted as a means of managing in the face of uncertainty, but few HCPs incorporate genuine adaptive management (see Chapter 22). Developing and implementing HCPs and other landscape management plans are costly, both in monitoring the effectiveness of the plan and in the economic value of resources forgone to meet habitat needs. Wilhere (2002) proposed that economic incentives might encourage implementation of more effective plans because it would enable adaptive management. Incentives might include direct payments or tax deductions for reliable information that benefits a species. Indeed, it is quite possible that incentives are more likely to induce creativity in developing and implementing effective plans than regulations; regulations tend to homogenize systems, while incentives can encourage creativity.

## SUMMARY

Goals for a landscape are usually set by stakeholders representing different social values, ecosystem services, and economic values associated with the landscape. Because the landscapes are often large and complex, with many competing values, landscape management plans are essential to reducing the risk of not achieving goals during management. Some goals are established by law; Habitat Conservation Plans are among the regulations that allow “take” of critical habitat for federally listed species in the United States. The ability of a land manager to meet legal requirements and achieve biodiversity goals is dependent on developing plans, identifying a good acceptable plan, implementing it, and monitoring its effectiveness.

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# 20 Ecoregional Assessments and Conservation Priorities

Time is money. Time, money, and commitment are what make habitat management happen in stands, landscapes, regions, and around the globe, but those resources are limited. Every forest manager has a budget and personnel limitations. Consequently, a manager will need to know where to invest those resources to have the greatest impact on the resources of interest. Getting the “biggest bang for the buck” is the approach that most managers want to take. For instance, consider a forest manager in Alabama with three primary goals: bobwhite quail, white-tailed deer, and timber. Patterns of food patches interfaced with cover are important to deer and quail (albeit at different spatial scales), but making a profit is important as well, so the problem becomes one of optimizing habitat qualities for the two game species, while ensuring profitable timber production. One way to approach this problem is to view habitat for the two species as constraints on the timber production, or alternatively, view timber production as a constraint on habitat for the two species. In either case, the resulting decision is one where one group of resources is given more value than another, and the decision resulting from the analysis can be implemented over space and time to achieve the desired goals (assuming some natural disturbance does not come along and change everything).

Now consider problems likely to occur over much larger areas of space and time. How would you decide where to provide habitat for rare species throughout their geographic range, in order to minimize risk of extinction? Or decide which parcels to buy before they are turned into housing developments? Or decide which nuclei of forests to protect from invasive species before they are overrun? Or decide how to coordinate management actions among landowners over a region to achieve biodiversity goals? Just as landscapes provide the context for stand prescriptions and regions provide the context for landscape management plans, global patterns of biodiversity provide the context for regional conservation strategies (Buchanan et al. 2011). Global patterns of biodiversity will only be conserved if the strategies are implemented among stands over landscapes and among landscapes over regions. Strategies are developed from the top down and implemented from the bottom up. Think globally, act locally. Within this context, it is often difficult to know where to invest the time, money, and commitment to achieve these regional goals (Loyola et al. 2009). Regional assessments can provide the context, and prioritization analyses can provide the guidance for investments.

## ECOREGIONAL ASSESSMENTS

Ecoregions are areas of similar climate, topography, soils, and other factors influencing patterns of vegetation and the animals; the processes that support these vegetation and faunal patterns occur and recur predictably (Table 20.1, from Bailey 1980). Ecoregions are often used as the basis for assessments. Habitat conservation and management strategies are developed at the ecoregional scale, which guide landscape management plans, which guide development of stand prescriptions or local management plans.

Ecoregions are displayed as generalized areas of climatically associated patterns of vegetation (Figure 20.1). This map is one of several attempts that have been made at mapping ecoregions, each with differences as influenced by the goals of the organization funding the work. Some systems of delineating ecoregions have greater detail than others (Omernick 1995). Ecoregions are mapped as discrete entities, but in actuality they represent gradients. One will grade into another,

**TABLE 20.1**  
**US National Hierarchy of Ecological Units**

Planning Scale	Utility
Ecoregion	
Global	Broad applicability for modeling and sampling
Continental	Strategic planning and assessment; international planning
Regional	
Subregion	Strategic, multiforest, statewide, and multiagency analysis and assessment
Landscape	Forest or area-wide planning and watershed analysis
Land unit	Project and management area planning and analysis

*Source:* From Bailey, R.G. 1980. *Descriptions of the ecoregions of the United States*. Washington DC: U.S. Department of Agriculture, Forest Service. Misc. Pub. 1391.

and no two places within any one ecoregion are the same. The devil is in the details. As you zoom in on any ecoregion, there is variability in patterns of soil, topography, climate, and vegetation, which occurs locally; hence, the need for a hierarchy of ecological units (Nesser et al. 1994, Keys et al. 1995, McMahon et al. 2001, Table 20.1). Hierarchical patterns of ecological units are useful because they do not follow political boundaries and can provide a framework for addressing issues that cross administrative and jurisdictional boundaries (Probst and Crow 1991). Using ecoregional units as the basis for assessments and development of coarse-filter conservation strategies is also intuitively appealing, because disturbance forces and recovery patterns are often more similar within an ecoregion than among regions. Although local modifications are often needed during landscape planning, ecoregional patterns provide a broad context for assigning goals and objectives that are related to the ranges of variability in ecosystem indicators (historic or future) seen in the ecoregion and can provide a logical link to population viability modeling efforts (Polasky et al. 2005; see Chapter 21).

Although ecoregions often have climatically or topographically defined boundaries, ecoregions are not a spatial scale per se. Some are large and some are small. In fact, the interaction of various ecological states and processes can all occur over a range of spatial scales. Allen and Hoekstra

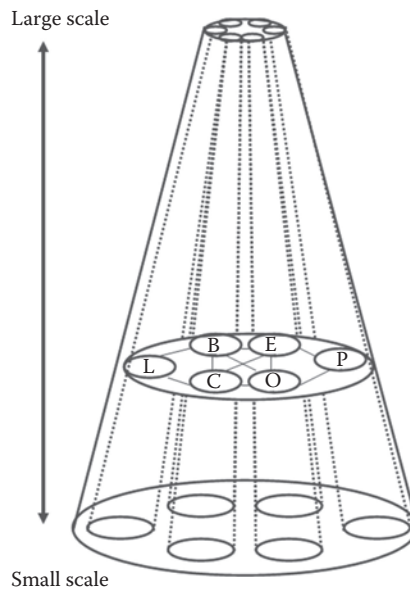


**FIGURE 20.1** Ecoregions of the United States. (Reprinted from Bailey, R.G. 1980. *Descriptions of the ecoregions of the United States*. Washington DC: U.S. Department of Agriculture, Forest Service. Misc. Pub. 1391.)

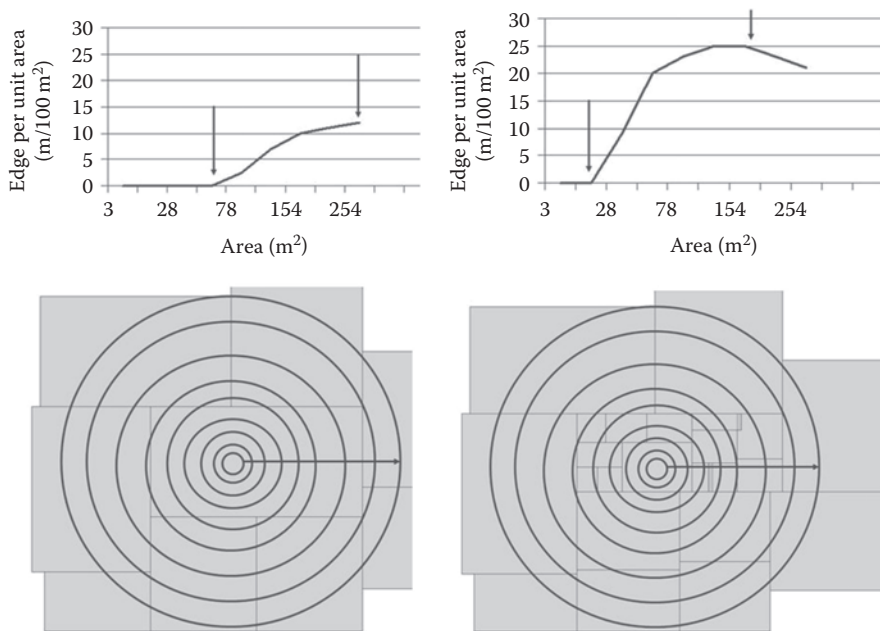


(1990) and Hoekstra et al. (1991) made an excellent case that ecological functional units, such as genotypes, organisms, populations, communities, landscapes, ecoregions, and biomes, all interact over a range of spatial scales, and that there are more potential interactions among these units at small spatial scales than at large spatial scales (because the planet is only so big) (Figure 20.2). So although we can use ecoregional units as a basis for planning, ecoregional units may have, and often do have, genotypes, organisms, populations, communities, and landscapes, that occur within a limited portion of an ecoregion, or which extend beyond ecoregional boundaries. Because ecoregions are developed in a hierarchical manner, conservation strategies for various species often cross ecoregional boundaries and do not always align with those boundaries. Consider a community of large carnivores in the Rocky Mountains containing wolves, lynx, wolverines, and cougars. This community represents a collection of interacting species that clearly transcend multiple ecoregional units in their genotypes, individual organisms, and populations. Using ecoregional units may be useful, but the appropriate level in the classification hierarchy must be interfaced with the spatial domains represented within each of these species, if species conservation is a goal. Nonetheless, ecoregions continue to be used as the basis for prioritizing conservation efforts for a wide range of species (Loyola et al. 2009).

Indeed, it is the spatial scaling properties of the region and the spatial requirements of the species using the region, that interface to provide information about the potential risk of losing species due to changes in patch areas, edges, and other factors that describe the spatial complexity of a region. Wiens (1989) defined “domains of scale” or spatial patterns of patches that emerge as you perceive increasingly large areas of a region. For instance, given two regions, each with different land-use or disturbance histories, patch sizes and configuration might differ (Figure 20.3). Taking a transect from any point on the region and estimating an ecologically important metric over increasingly large areas produce a trend that should increase to an asymptote in a landscape in which all patches were uniformly distributed. But in many regions, patches are not uniformly distributed, and the trend in landscape metrics is not asymptotic (Wheatley 2010). Instead, there



**FIGURE 20.2** Organisms (O), populations (P), communities (C), landscapes (L), ecosystems (E), and biomes (B) all interact over a range of spatial scales, with many levels of interaction possible at small spatial scales than at large scales. (Redrafted and adapted from Allen, T.F.H., and T.W. Hoekstra. 1990. *Journal of Vegetation Science* 1:5–12.)



**FIGURE 20.3** Simplified example of identifying domains of scales in portions of two landscapes. In a landscape where patches are uniform and evenly distributed (left), a threshold in edge density emerges at 50 m<sup>2</sup> and approaches an asymptote at 254 m<sup>2</sup>. In a more heterogeneous landscape (right), two thresholds emerge at 12 m<sup>2</sup> and at 202 m<sup>2</sup> where the patch heterogeneity is expressed in two domains of scale. Depending on the domains of scale associated with a species life history, one landscape may provide higher quality habitat than another.

are thresholds that emerge from these analyses that define domains of scale. These domains can be used as information in regional assessments in two ways. First, the domains identified in current landscapes can be compared to a reference condition (or conditions), such as the conditions that might be seen under the historical range of variability or a desired future condition. An implicit assumption is that, as the domains depart from the reference condition, then there is increasing risk of losing species that are not well adapted to using new domains of scale that fall outside of the historic ranges. In a more explicit analysis, the home range, territory, or metapopulation sizes for a species can be compared to the domains of scale associated with the species' habitat over a region. The greater the disparity between the domains of scale needed by a species and those found in a region, the greater the risk to the species of not having adequate patch sizes, configuration, or connectivity.

In addition, it is often the edges between ecoregional units that can be of considerable interest during conservation planning and management prioritization. On a map, the edges between ecoregions are sharp lines due to the need to classify units, but in reality they are blurred boundaries (Bailey 1980), oftentimes with rich plant and animal communities occurring at climatic or topographic ecotones. Simply viewing the ecoregional unit as the basis for planning, without recognizing the potential importance of the ecoregional hierarchy and ecoregional ecotones, may miss important drivers of species richness. Climate change is likely to change ecoregional boundaries. Beaumont et al. (2011) indicated that of the 238 ecoregions with exceptional biodiversity, 82% are likely to be under significant stress by the year 2100. As ecoregions rearrange in response to climate, it is likely that the domains of scale will also shift, and that species shifts will first be noticed at ecoregion boundaries (Wiens and Blanchet 2010). Cross-region connectivity will be critical to allow species to migrate, if they can, as climates change.

## EXAMPLES OF ECOREGIONAL ASSESSMENTS

Federal, state and NGO groups have been involved in ecoregional assessments across the United States, Canada, Australia, and other countries around the world. The scope of the assessments conducted represents a broad spectrum of spatial scales, processes, and political entities. Some, such as the Forest Ecosystem Management Assessment Team (FEMAT), limited their assessment to federal lands in the Pacific Northwest and focused largely on late successional species (FEMAT 1993). Others, such as Coastal Landscape Analysis and Modeling Systems (CLAMS) project, considered all landowners, long timeframes, and a multitude of processes and species (Spies et al. 2007). In the CLAMS approach, both past and likely future conditions are considered in the face of current and alternative future policies. The Interior Columbia Basin Ecosystem Management Planning (ICBEMP) assessment considered a huge multistate area and resulted in an assessment of forest-related ecosystem process and species over the region (Wisdom et al. 2000, Figure 20.4). States also have conducted much smaller ecoregional analyses, such as the Berkshire Ecoregion Assessment in Massachusetts (Fleming 2006). Oftentimes, when states or federal agencies are involved in assessment, the assessment stops at political boundaries although the ecoregion extends across boundaries. This was the case in the Berkshire assessment. Indeed, the scale of analyses is at times aligned with ecoregional boundaries and at times with political boundaries, but often does not consider the domains of scale of the system being assessed (Wheatley and Johnson 2009). The issue is further confounded when ecoregional analyses include international borders. FEMAT and ICBEMP largely stopped at the Canadian border, although the contributions of resources from Canada were considered as part of the context for the assessment. Nongovernment organizations (NGOs), especially The Nature Conservancy (TNC), also have used ecoregional assessments in their planning



**FIGURE 20.4** U.S. Forest Service and BLM lands covered by the Interior Columbia Basin Ecosystem Management Plan. (Reprinted from the USDA Forest Service Interior Columbia Basin Ecosystem Management Plan. See Wisdom M.J. et al. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia Basin: Broad-scale trends and management implications. USDA Forest Service General Technical Report PNW-GTR-485 for more detail.)



**FIGURE 20.5** Example of a cross-border ecoregional assessment coordinated by The Nature Conservancy. Six ecoregions are included in this assessment. (From Marshall, R. et al. 2004. An ecological analysis of conservation priorities in the Apache Highlands Ecoregion. Prepared by The Nature Conservancy of Arizona, Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, agency and institutional partners. With permission from the Arizona Chapter of The Nature Conservancy.)

and prioritization work. NGOs are not restricted to political boundaries to the degree that states, provinces, and countries might be. For instance, TNC (2001) completed a multistate assessment for the Appalachian forests of Maryland, Virginia, Pennsylvania, and West Virginia and facilitated a cross-border assessment in the southwestern United States and Mexico that included six ecoregions (Marshall et al. 2004, Figure 20.5).

### CONDUCTING AN ECOREGIONAL ANALYSIS

Will you ever be involved in conducting an ecoregional assessment? Well, maybe, but you quite likely will be developing landscape management plans or working within management plans that are tiered to an ecoregional assessment. In order to provide a useful context for landscape plans,

**TABLE 20.2****Primary Steps in an Ecoregional Assessment**

1. Identify the ecoregion and spatial extent to be included in the analysis.
2. Identify the species of conservation concern.
3. Determine the habitat associations of species.
4. Delineate the boundaries of the species range and map distribution within the range.
5. Identify the natural disturbances and human activities.
6. Identify the potential risks to species or its habitat.
7. Map the extent of individual and cumulative risk factors.
8. Identify and develop the management actions.

*Source:* From Wisdom, M.J., M.M. Rowland, and L.H. Suring (eds.). 2005. *Habitat Threats in the Sagebrush Ecosystem: Methods of Regional Assessment and Applications in the Great Basin*. Alliance Communications Group, Allen Press, Lawrence, KS.

ecoregional assessments should be effective in identifying likely risks to species, their habitats and plant communities over large areas. Recent systematic ecoregional assessments and associated conservation planning approaches probably have been more effective at conserving biological diversity than approaches of the past (Margules and Pressey 2000), but expectations of scientists and stakeholders are not always realized (Bottrill et al. 2012). Past approaches often resulted in a biased distribution of lands identified for protection or management specifically for biodiversity goals, with many areas occurring on lands not useful for other purposes such as high elevations and steep slopes (Scott et al. 2001).

Wisdom et al. (2005) outlined the steps for conducting an ecoregional assessment (Table 20.2), and Groves et al. (2002) proposed a process to identify the conservation areas (which may or may not require management) across ecoregions (Table 20.3). Whereas large ecoregional assessments of the past have cost millions of dollars and taken 5 or more years to complete, Groves et al.'s (2002) process has a median cost of \$234,000 per plan (in 2002 U.S. dollars) and an average completion time of just less than 2 years. Cork and Tait (2009) conducted a review of known information, including an identification of high priority conservation areas, and a workshop to develop strategic priorities for biodiversity conservation across Australia. These latter two examples indicate that a regional or national assessment may take many forms and still provide the basis for development of landscape plans and stand prescriptions.

## ASSESSING PATTERNS OF HABITAT AVAILABILITY AND QUALITY

Data used to set goals, develop assessments, and establish priorities often come from remotely sensed sources such as satellite imagery (e.g., LANDSAT), Light Detection and Ranging (LIDAR) data, aerial photography (orthophotos), and the resulting Geographic Information Systems data that are derived from these techniques. Consequently, the data are usually restricted to some minimum grain (perhaps a  $30 \times 30$  m pixel for LANDSAT) and logical extent (simply from the standpoint of managing too many pixels of data). Using satellite data, reflectance values in various spectral bands are usually classified into conditions on the ground that are recognizable to humans as land cover classes. These classes become the basis for further associations as habitat for various species. But each species has its own habitat requirements, so one classification system is unlikely to work well for all species; indeed, Cushman et al. (2010) discourage the use of cover classes at all, recognizing that landscapes are not composed of discrete patches but rather gradients of habitat quality or resource values. Nonetheless, past efforts have classified data into land cover classes that then are related to the likely occurrence of various species and processes across the area of interest. Wildlife Habitat Relationships (WHR) models have traditionally been used to relate classified land types



**TABLE 20.3**  
**A Seven-Step Conservation Planning Framework**

Step 1: Identify the conservation targets:

- Abiotic (physically or environmentally derived targets)
- Communities and ecosystems
- Species: imperiled or endangered, endemic, focal, keystone

Step 2: Collect information and identify the information gaps:

- Use a variety of sources
- Rapid ecological assessments and rapid assessment programs
- Biological inventories
- Workshops with species' experts

Step 3: Establish the conservation goals:

- Address both representation and quality
- Distribute the targets across environmental gradients
- Set a range of realistic goals

Step 4: Assess existing conservation areas:

- Gap analysis

Step 5: Evaluate an ability of conservation targets to persist:

- Use criteria of size, condition, and landscape context
- Use GIS-based "suitability indices" to assess current and future conditions

Step 6: Assemble a portfolio of conservation areas:

- Use site or area selection methods and algorithms as a tool
- Design networks of conservation areas employing biogeographic principles

Step 7: Identify the priority conservation areas:

- Use the criteria of existing protection, conservation value, threat, feasibility, and leverage to prioritize areas

*Source:* From Groves, C.R. et al. 2002. *Bio Science* 52:499–512.

(seral stage and plant communities) to the occurrence of each of the vertebrate species that occur in an area (DeGraaf and Yamasaki 2001, Johnson and O'Neil 2001). For many species the WHR models produce reasonable estimates of the availability of habitat for many species across a region (Block et al. 1994), but not all. Ideally classification would have to be tailored to each species to more accurately reflect the collection of habitat elements important to the species (Betts et al. 2006, Cushman et al. 2010). Further, the grain of the assessment (say  $30 \times 30$  m) may be too large to reliably capture the information needed to assess habitat for some species, both because one classification system does not apply to all species and some species use habitat at scales smaller than the grain size. Consequently, WHR models may be a useful guide to patterns and changes in habitat area and configuration for species over an area, but the reliability of the information varies considerably from species to species. These approaches only provide generalized estimates of habitat availability indicating where the species could occur and not necessarily where it would find habitat of better or worse quality (as it might influence animal fitness). Indeed, knowledge of site-specific sizes, abundance, and distribution of habitat elements would be needed to understand specifically how habitat quality might change from place to place or over time in a region (McComb et al. 2002, Betts et al. 2006, Spies et al. 2007, Cushman et al. 2010). Some of this information could be extracted from air photos and from LIDAR data. These techniques can provide some information on individual trees and even smaller structures that can be seen from the air. Additional information can be derived from generally available GIS layers which includes topography, soils, hydrography, and climate. These classified images and supplementary remotely sensed data can provide the information needed to assess habitat availability for many species. Habitat element information for some



species would better be assessed from the ground. For instance, the amount of cobble in a stream important to torrent salamanders or the presence of hollow trees for swifts will not be reasonably reflected in remotely sensed data. Ground plot information is needed.

Ground plot data are systematically collected from multi-resource inventories such as Forest Inventory and Analysis data on private (and many public) forest lands in the United States and these efforts can be used to infer patterns of habitat availability for species over space and time (Ohmann et al. 1994). In addition, many industrial land managers have continuous forest inventory plots distributed across their properties to monitor tree growth and death, and these plots can be adapted to allow collection of site-specific data on habitat elements as well. Of course these ground inventories are samples and not inventories, and so they have been of limited value when representing habitat availability in situations where both fine-scale habitat elements as well as landscape composition and structure might be important to a species.

Ground plot data have been interfaced with remotely sensed data to allow representation of habitat elements across complex regions (Ohmann and Gregory 2002, Spies et al. 2007, Ohmann et al. 2011). Using this approach, the ground plot data are georeferenced to the physical location, topography, climate, reflectance values, and many other features on GIS layers to create a subset of “informed” pixels (pixels with a ground plot within them). For all pixels that do not have associated ground data (“uninformed pixels”), the same descriptive characteristics are also estimated, but of course there are no corresponding ground plot data associated with the uninformed pixels. To provide a “seamless” representation of ground plot information, characteristics of informed pixels are used to “inform” those pixels without ground plot data that are most similar in these descriptive characteristics. Hence, fine-scaled ground plot data can be imputed to all pixels in the extent of the assessment (Ohmann and Gregory 2002, Ohmann et al. 2011). Once ground plot data have been assigned to the uninformed pixels, then the pixels can be reclassified based on the imputed ground plot data to create species-specific habitat quality maps across the planning area (McComb et al. 2002, Spies et al. 2007, Cushman et al. 2010). These maps can then be used as the basis for assessing net gains and losses of habitat over space and time as well as population viability analysis (see Chapter 21) for species of high risk of being lost from the area in the future.

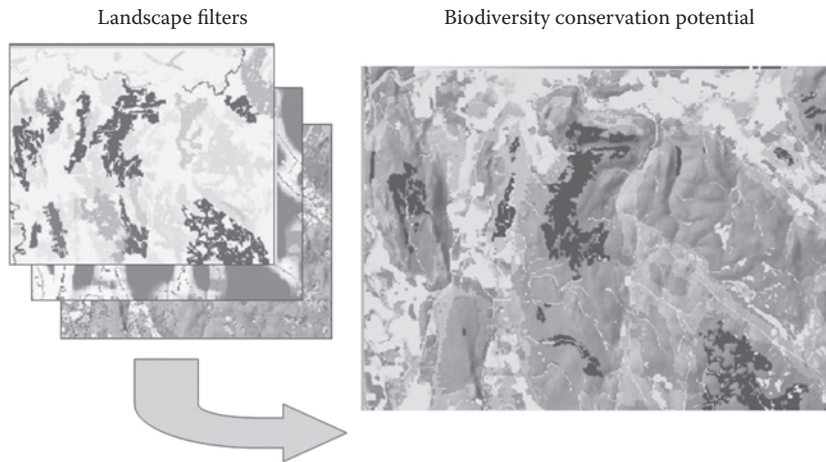
## **PRIORITIZING MANAGEMENT AND ASSESSING POLICIES**

How would you decide which tools to use and approaches to follow to ensure that your biodiversity conservation goals will be effective? The tools available to assist in decision making for biodiversity protection have exploded in number and complexity over the past decade. Gordon et al. (2004) identified over 50 decision support tools that could be used to assist in biodiversity conservation and that number is increasing annually. Choosing which to use, if any, is an overwhelming task and is highly dependent on the specific questions, goals and objectives of the assessment (Johnson et al. 2006). Below are examples of a few commonly used and powerful approaches to assessments and prioritization.

### **Coarse-Filter Approach**

There are numerous examples of how estimates of habitat patterns, availability and quality have been used to provide a means of prioritizing management decisions. These same techniques often can be used to assess alternative management plans or policies across the area of assessment. Since time and money are usually limited when making decisions regarding management to conserve biodiversity, prioritization of the areas to manage or protect becomes paramount.

One such approach is the Conservation Assessment and Prioritization System (CAPS) which uses a coarse filter approach to parcel prioritization (Gordon et al. 2004). The CAPS approach was developed by Dr. Kevin McGarigal and uses potential biodiversity valuation that applies “biodiversity screens” to each patch in the landscape. These screens are applied to a map of predicted natural communities modeled from remotely sensed and GIS data. Biodiversity screens are models

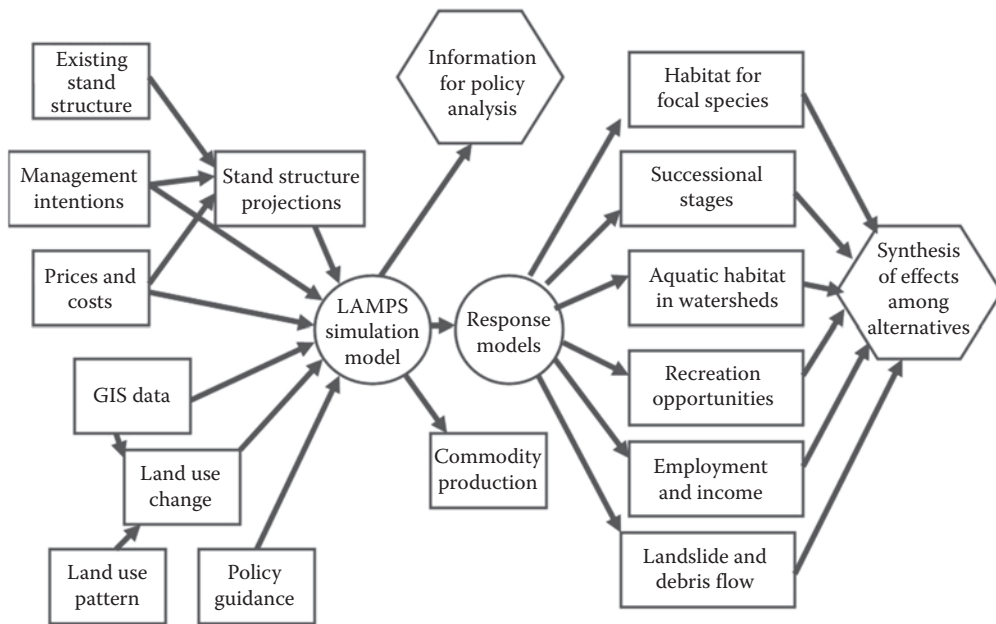


**FIGURE 20.6** Example of the application of biodiversity screens or filters to a landscape resulting in the identification of high priority blocks for management or protection. (Provided by Dr. Kevin McGarigal. With permission.)

that reflect the content, context, spatial character, or condition of a patch to arrive at an index of potential biodiversity value. Stakeholders are involved in deciding how various parameters of the screens guide the identification of high priority patches in the region. Parameters such as the size of a natural community patch, edge contrast, edge density, its proximity to water, the soil type, or road density (among many others) can be identified and weighted to help identify priority patches for management or protection. The result of applying a set of screens is a biodiversity value ranging from 0 (low value for biodiversity conservation) to 1 (high value) for each patch on the landscape that then can be used to highlight those patches of highest value (Figure 20.6). The resulting high priority patches represent areas that may receive special management practices, or could be placed in reserves, conservation easements, or purchased from private landowners to protect species associated with the priority patch characteristics. The species receiving protection include species that are known to be associated with the priority patches as well as those represented in the “hidden diversity,” or those species assumed to be associated with these patch conditions but which have not yet been identified. Further, the approach has been used not only in ecoregional assessments (e.g., the Berkshires), but also in mitigation to replace areas gobbled up by roads and development with patches of appropriate sizes and conditions.

### Integrated Coarse-, Meso-, and Fine-Filter Approaches

Many assessments use a combination of coarse-, meso-, and fine-filter approaches to understand the current conditions across complex ecoregions. Some, such as the Willamette Alternative Futures approach (Hulse et al. 2002) used the likely changes in abundance and distribution of vertebrates across the ecoregion as a primary assessment of current and future effects of alternative future landscapes. They also considered the areal extent and distribution of various plant communities across the planning area, but did not assess the landscape metrics associated with the patches in a patch prioritization manner such as used in CAPS. Nonetheless the results of this effort have been widely used to inform land use planning decisions in the region so that planners can consider the effects on forest land and potential impacts on biodiversity of land use decisions. Another assessment, the Coastal Landscape Analysis and Modeling Systems Project (Spies et al. 2007) was designed to analyze the ecological and socio-economic consequences of various forest policies across multiple ownerships. The process includes a complex set of interacting models that consider the disturbance and regrowth of forests as guided by forest management policies and



**FIGURE 20.7** Complex interacting models produce current and likely future estimates of plant communities, habitat for focal species, and economic goods and services for the Oregon Coast Range under alternative forest policies. (From Spies et al. 2002. *Congruent Management of Multiple Resources: Proceedings from the Wood Compatibility Initiative workshop*. USDA Forest Service Gen. Tech. Rep. PNW-GTR-563.)

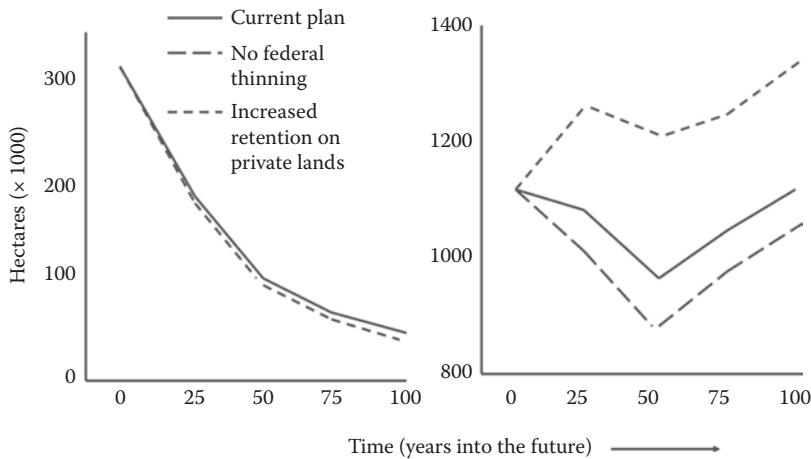
the resulting patterns of plant communities and habitat quality for focal species across the region (Figure 20.7). The results can not only be used to assess alternative forest policies but also identify locations in the region which might be particularly important as core patches or linkages across complex multi-ownership landscapes. The projected changes in plant communities (as a coarse filter index to protecting hidden diversity) as well as changes in habitat quality and distribution for focal species (those selected to represent certain ecological associations) are used to compare policy alternatives (Figure 20.8).

These approaches can be influential with stakeholders because the ability to map plant communities, habitat for various species, other resources, and land ownership can more directly engage stakeholders with scientists around visual portrayals of these resources over space and time (Wright et al. 2009).

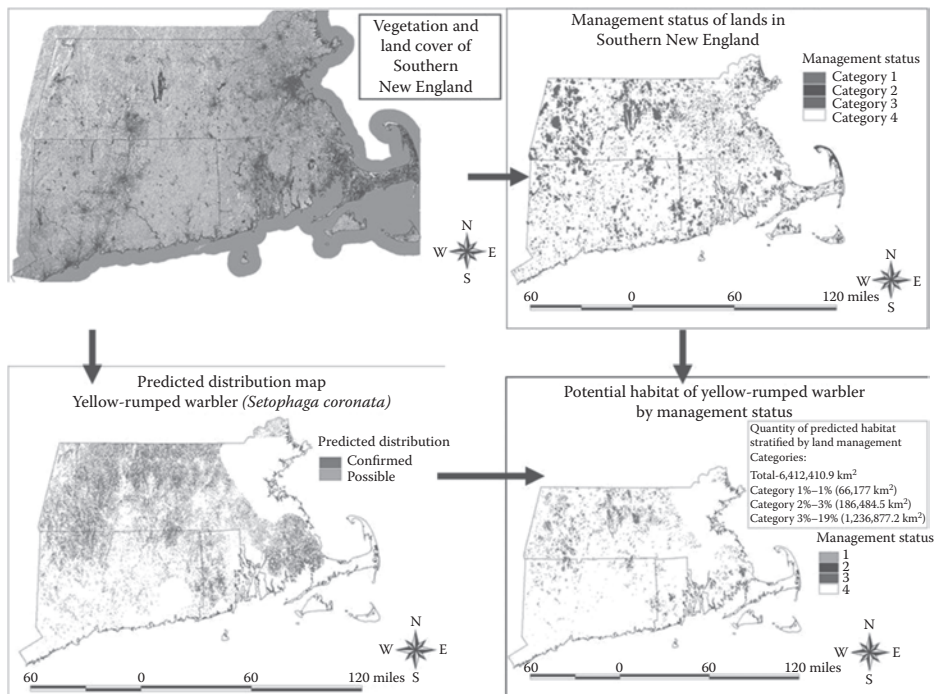
### Fine-Filter Approaches

Other approaches to ecoregion assessments take a species by species approach to identifying areas for particular management or protection. Because most of the species assessments rely on WHR models to develop maps of occurrence of species, the underlying maps can also be used as a coarse filter assessment as well. One such approach is a nationwide effort called Gap Analysis (Scott et al. 1993). The goal of Gap Analysis is to “keep common species common” by identifying those species and plant communities that are not adequately represented on existing conservation lands. By identifying habitat for all vertebrates in a region, Gap Analysis provides information that can be used to make decisions regarding vertebrate species conservation and management.

Gap Analysis consists of three main data layers, a landcover layer, a layer showing the predicted distributions of vertebrate species, and a stewardship layer (Figure 20.9). These layers are used in a Gap Analysis consisting of three primary steps. The first step is to map plant communities to develop a landcover layer. Landcover is mapped using satellite data as well as other supporting information from existing GIS layers, air photos, and ground plot data.



**FIGURE 20.8** Projections of habitat availability for an example focal species (olive-sided flycatchers, right) and a plant community type (hardwoods, left) in the Oregon Coast Range, under three policy alternatives: current policies (solid), no thinning allowed on federal lands (large dashes), and green tree retention on private lands (small dashes). (From Spies, T.A. et al. 2007. *Ecological Applications* 17: 48–65.)



**FIGURE 20.9** Steps in the process of Gap Analysis. Maps of land cover using remotely-sensed and ground-plot data (top left), species distribution based on geographic ranges, and habitat relationships (top right), and management status (bottom left) are overlain to identify gaps in the protection network for each species (bottom right). (From Zuckerberg B., C.R. Griffin, and J.T. Finn. 2004. *A Gap Analysis of southern New England: An analysis of biodiversity for Massachusetts, Connecticut, and Rhode Island*. Final contract report to USGS Biol. Resour. Div., Gap Analysis Program, Moscow, Idaho. With permission.)

The second step is to map predicted distributions of vertebrate species known to breed or use habitat in the region. Known, probable, and possible occurrences are used to define the geographic range of each species. Then a WHR model is developed for each species that relates the land cover data to the likely occurrence of the species across the region. The process does not usually include any assessment of habitat quality or viability.

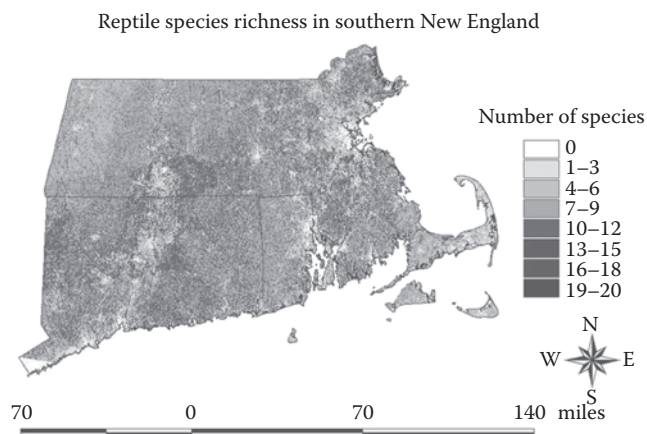
The third step of a gap analysis is to assign a land stewardship rank between one and four to each patch on the assessment area. Status one lands have the highest degree of management for conservation, status four lands have the lowest. Stewardship ranks are based on the long-term intent of the managing entity (owner or steward). Ranks are based on (Scott et al. 1993)

- Permanence of protection from conversion of “natural” land cover to “unnatural” (human-induced barren, arrested succession, cultivated exotic-dominated).
- Amount of the tract protected, with 5% allowance for intensive human use.
- Inclusiveness of the protection, that is, single feature such as wetland versus all biota and habitat
- Type of management program and degree that it is mandated or institutionalized.

The fourth step is to analyze the representation of each species (or plant community) in areas managed for the long-term maintenance of biodiversity. To accomplish this, maps showing animal and plant community distributions are intersected with stewardship maps to identify areas where the species that are not receiving protection based on management status or appropriate management could occur. Identification of high priority areas for protection or management can be based on individual species (Figure 20.9) or on species richness patterns (Figure 20.10). Gap analysis has been completed for every state in the United States and composite assessments across state lines now allow ecoregional analyses. The gap analysis approach has been combined with other biological assessments to allow national mapping of biodiversity indicators (Boykin et al. 2012).

### Utility and Effectiveness of Ecoregional Assessments

Of course the ultimate test of effectiveness is in conserving species and preventing listing of species in the future through hierarchical planning, implementation, and monitoring. Bottrill et al.



**FIGURE 20.10** Example of a composite map of species-richness for reptiles in southern New England. Protection or management of darker areas is more likely to capture more species than lighter areas. (From Zuckerberg B., C.R. Griffin, and J.T. Finn. 2004. A gap analysis of southern New England: An analysis of biodiversity for Massachusetts, Connecticut, and Rhode Island. Final contract report to USGS Biol. Resour. Div., Gap Analysis Program, Moscow, Idaho. With permission.)



(2012) found that the expectations that species would be conserved were not always realized but that additional benefits accrued from ecoregional assessments including improvements in social interactions, attitudes toward conservation, and institutional knowledge. Few regional assessments have monitored effectiveness sufficient to provide evidence for having “saved” species that otherwise would have been regionally or globally eliminated. Measuring effectiveness requires that measurable goals are set to quantify success or failure (Tear et al. 2005). Only through monitoring over long time periods can effectiveness be assessed, and even then how can we know what we did not lose? Monitoring and evaluation provides an opportunity for accountability, as well as an opportunity to learn from past efforts (Bottrill and Pressey 2012). Much of the effectiveness of assessment and planning is based on assumption and faith that to plan objectively and monitor the quantitative objectives is less risky than following unstructured approaches to species conservation.

## SUMMARY

Ecoregional assessments are often used as a context for development and implementation of landscape management plans which provide direction for development and implementation of stand prescriptions. Ecoregions are hierarchically defined classes of vegetation based largely on climatic and other physical features that seem to drive patterns of plant communities. The utility of ecoregional assessments in providing useful contexts for other efforts are highly dependent on matching the appropriate spatial hierarchy with the spatial and temporal scales associated with the species and communities of concern. Assessments can use coarse-, fine-, and meso-filter approaches but most use some combination of these approaches. Usually these approaches are designed to prioritize areas for protection or management. The degree to which these efforts have been effective in protecting species is questionable, but the knowledge gained from the efforts has been significant. The effectiveness of such approaches in conserving species remains largely unknown and will likely only come after years of monitoring over large areas.

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# 21 Viable Populations in Dynamic Forests

I have heard some foresters and biologists say, “We manage habitat, not populations.” The Field of Dreams approach to wildlife management: Build it and they will come. But will they? Can they, given the limited mobility of some species and intervening forest conditions now and 50 years from now? If they do come to the new habitat, how long will they last there? How does their presence contribute to ensuring that the greater population is not going extinct? All of these questions relate to understanding population viability.

Population viability analysis (PVA) is a structured approach to examining population performance—to link population performance with the quantity, quality, and distribution patterns of habitat and to predict the probability of population extinction or persistence. The approach is based on a marriage among the concepts of demography, population dynamics, habitat selection, and landscape dynamics. Many of the concepts covered in this book to this point provide the foundation for an assessment of population viability in forested landscapes. PVA models and analyses originally were developed in response to the concern for the persistence of small or isolated populations, a response to law (e.g., the Endangered Species Act and the National Forest Management Act), and to large-scale regional assessments. More recent applications include predicting the potential success of reintroductions of plants and animals and assessing strategic development of habitat improvement projects (Knight 2012, Bonnot et al. 2013). PVA techniques are both elegant and uncertain. PVA is a modeling exercise and, as Box and Draper (1987) have said, “All models are wrong, but some are useful.” The results from a PVA can be useful in guiding management alternatives if uncertainty is understood and included in the decision-making process.

## EXTINCTION RISKS

It’s a hard world out there. As far as we know, all populations will eventually go extinct (including ours!). Large populations will likely last for many generations; small populations are much more vulnerable to extinction. The crux of the population change issue is encapsulated in this formula:

$$N_{(t+1)} = \lambda N(t)$$

This states that the population size at time + 1 (the future) is equal to a change coefficient ( $\lambda$ ) multiplied times the current population ( $N(t)$ ). Should  $\lambda$  be a negative number over a long time (long being relative to the generation length for the species), then concerns arise regarding the potential for the species to become extinct. At least four factors are likely to influence the risk of extinction, especially in small populations.

First, vital rates of populations vary from year to year, and place to place, simply by chance alone. Birth rates, death rates, and reproductive rates within an age class typically have a mean value over time in a stable population, but there is also a variability associated with that mean value. In some species, that variability is small (species with low but predictable reproduction, such as bears and people), and for others it is quite large (species with irruptive populations, such as voles and quail). These levels of variability in vital rates apply whether the population is large or small, but in small populations the effect can be quite dramatic. A failure to reproduce for a year (or more,

depending on the life span) in a population of 1000 voles may mean a temporary crash, followed by recovery, but a similar event in a population of 10 animals may lead to a population decline from which it cannot recover, due to random shifts in sex ratios and the difficulty in finding mates. This demographic stochasticity is an inherent property of populations, and the variability represented in these vital rates is likely an evolutionary response to unpredictable events of the past. Issues of small populations must be considered relative to the life span of the individuals. Shoemaker et al. (2013) found that long-lived bog turtles have a high probability of persisting even at low population levels.

Second, there are also exogenous events that interact with habitat elements to influence the likelihood that an animal will die, reproduce, or move. Climatic fluctuations from year to year, for instance, can have a significant effect on population change over time (Olson et al. 2004). For instance, an exceptionally rainy year can cause nest failure for some bird species, because nestlings become hypothermic and die. There is some probability that a very wet year will occur, but whether or not it happens is a chance event. The probability of such an event is predictable, but the timing of the actual event is uncertain. These uncertain events, unrelated directly to habitat structure and composition, represent environmental stochasticity. Random events such as storms, droughts, and epizootic diseases that do not affect habitat structure and function, cause vital rates to fluctuate considerably from year to year and place to place. In small populations, these effects can be magnified. An event that increases mortality by 50% in a population of 1000 animals for one year may simply result in population recovery from the remaining 500 individuals over the next few years. But a similar event in a population of 10 individuals can result in an extremely skewed sex ratio by chance alone, causing the population to go extinct.

Third, natural catastrophes, such as hurricanes, fires, and epizootics, which can cause massive changes in vital rates unrelated to habitat structure and composition, are extreme cases of environmental uncertainty. When Hurricane Hugo blasted the Francis Marion National Forest, the majority of nesting cavities for red-cockaded woodpeckers were destroyed (Hooper et al. 1990). These sorts of events cause fluctuations in vital rates that far exceed the expected year-to-year variability seen in most populations and can have devastating consequences for small populations. Dennis et al. (1991) predicted a reasonably high probability of population persistence for Puerto Rican parrots when not accounting for hurricanes, but Hurricane Hugo nearly decimated the population, potentially changing the probability of recovery for this species.

Fourth, in small populations, chance has a huge effect on otherwise subtle changes in genetic variability present in the population. In many instances, the variety of gene expressions, or alleles, represents the potential adaptability of the species to environmental uncertainties. In large populations in dynamic environments, we would expect to see some reasonably high level of heterozygosity (genetic variability) in the population. As populations decline, by chance alone, some alleles may dominate in a population, leading to a preponderance of individuals being homozygous (genetically more uniform) for some traits. This phenomenon is called genetic drift. These changes may make many of the individuals less adaptable to environmental uncertainty associated with those traits. In small populations, this genetic shift can result from increased levels of inbreeding by closely related individuals. In cases where inbreeding leads to reduced fitness in the populations, then inbreeding depression has occurred. In extreme cases, alleles may be entirely eliminated from populations. Regaining that genetic variability would occur either through immigration from surrounding populations (if there are any) or from mutations (most of which are not beneficial). Restrictions in genetic expressions can also be seen as animals disperse to unoccupied patches of habitat and establish a new population. Since a population beginning from just a few individuals is more likely to have a narrow range of alleles in the newly establishing population, the genetic variability in the population can remain low as the population grows, a process termed the *founder effect*. A similar process can occur when a population goes through a genetic bottleneck, or when there is a dramatic decline in the population to low levels, where alleles are lost in the process. During population recovery following the decline, a narrower range of allelic expression may be seen. Founder effects and genetic bottlenecks can lead to populations that are larger

(i.e., recovered) but more vulnerable to environmental uncertainties because the alleles needed to cope with those uncertainties had been lost.

Because these four factors (demographic stochasticity, environmental stochasticity, natural catastrophes, and loss of genetic variability) are exacerbated in small populations, conservation biologists and forest planners spend considerable effort managing habitat to ensure recovery of small populations. Population viability analyses are a set of tools available to planners to address these issues when investigating the potential effects of alternative management strategies.

## GOALS OF PVAs

There are two major goals usually associated with a PVA: (1) to predict short- or long-term rates of change, and (2) to predict the likelihood of extinction (Beissinger and Westphal 1998). At the very least a manager would like to know what the likely trend in a population might be under various management scenarios or forest plans. Being able to estimate if a population is decreasing or increasing can influence the direction of future management efforts if the species is listed as threatened, or if the species is a potential pest. But if the decline is so severe that questions are raised regarding the potential for a population to go extinct, then a new goal emerges for the analysis.

PVA is an approach often used to predict the probability that a population will go extinct in a given number of years—for instance, to estimate a 95% probability of extinction within 1000 years, 100 years, or 10 years. The assumption behind these analyses is that the shorter the potential time to extinction, the greater the risk of losing the species, simply because the actual time to extinction can never be predicted accurately, given all the uncertainties associated with an extinction event.

Most PVA models are demographic in nature, employing age-specific reproduction and survival rates, but genetic implications have been considered in some models as well, especially when managing endangered species. Genetic PVA models incorporate estimates of effective population size, which is the number of breeding individuals effectively contributing to allele frequencies in a population. The effective population size is usually smaller than the absolute population size, unless breeding is random and the chance of inbreeding is negligible—two assumptions likely to be violated in small populations. From the standpoint of potentially losing alleles in a population, especially in small populations, effective population sizes must be considered.

## PVA MODELS

There are five predominant types of demographic models that have been used in analyses of population trends and extinction analyses (Beissinger and Westphal 1998). First, analytical models have been used to examine behavior of a system (or assumptions behind other models), and not usually used to make population predictions (Beissinger and Westphal 1998). Analytical approaches also have been used to simply relate populations to current and likely future conditions using statistical techniques, such as regression, logistic regression, or classification and regression tree analyses. These approaches may be used to examine how a population might react to a change in abundance of habitat, or connectivity, or other factors that do not explicitly take into consideration the demographics of the population, nor its movement capabilities. Analytical approaches such as these may in fact miss changes in populations that could be caused by unrelated changes in birth or death rates. Developing associations does not prove a cause-and-effect relationship, only that several things are related to one another in some way. For instance, using a regression relationship between the number of woodpeckers and the number of snags would be useful for some planning processes, unless some other factor, such as West Nile Virus, caused the population to decline, independent of snag density. Knowledge of birth, death, and survival rates can help to consider both density-dependent and density-independent causes of population change.

A second type of analysis that could be used to assess population trends is a deterministic single-population model. These models are generally based on a Leslie Matrix or a matrix of survival rates



and reproduction rates in each of several age classes, to predict change in populations over years (Leslie 1945). In this approach, estimates of birth and death rates by age class are developed from field data, such as banding returns or radio telemetry data. These estimates are used to calculate survival rates, in conjunction with information on reproduction in each age class to calculate population changes from one time step to another. This approach is among the simplest models requiring the least amount of data but it assumes that demographic rates are constant (Caswell 2001). This kind of model has been used to assess changes in marbled murrelet populations, for example (Beissinger 1995, Peery and Henry 2010).

Stochastic single-population models overcome the assumption of constant demographic rates and include variability in estimated demographic rates in the calculations. Interestingly, this approach was adapted from Leslie models and first used in the development of forest management models (Usher 1969). Estimates of variability are incorporated into birth and death rates (and hence survival probabilities), and reproduction rates, to allow multiple projections of population change over time, reflecting the variability in demographic factors. Because these variances represent the stochastic properties of population change, each projection produces a unique trajectory and ending population size. The results of each projection can then be averaged or summarized to develop confidence intervals around population trends and probabilities associated with extinctions. This is a quite commonly used approach, because it attempts to introduce reality into projections, and has been used for species such as brushtail possums in managed forests of Australia (Lindenmayer 1993). Both deterministic and stochastic population models are typically applied to single populations. When a population is segregated spatially into interacting sub-populations, then a different approach is usually taken.

Metapopulation models are used to assess the interacting dynamics among sub-populations representing a metapopulation structure. The metapopulation dynamics are incorporated into the demographic model by using patch-specific demographic rates and dispersal probabilities between the patches (Beissinger and Westphal 1998). Dispersal rules are developed based on patch size and distances between patches. Patch quality, usually indexed to a carrying capacity, can be assumed to vary among sub-populations as well. This approach has been broadly used with many species that are assumed to have a metapopulation structure. For instance, Beier (1996) used a metapopulation model to assess cougar population trends in western United States. When the condition between the patches becomes important to population trends and the configuration and dynamics of the patch conditions is important, then spatially explicit approaches become more useful in assessing risks to species.

Spatially explicit models are designed to consider population dynamics on complex landscapes with varying matrix conditions between patches. These approaches also lend themselves well to assessing the interacting dynamics of populations and underlying vegetation conditions. The approach incorporates a spatial distribution of resources related to habitat quality, as well as movement rules for dispersing animals, to assess responses to land-use changes or management policies (Wilhere and Schumaker 2001). Because of the spatial detail needed and the dynamics associated with underlying resource layers, these sorts of models typically require enormous amounts of data and, until recently, were most often used with species representing a very high risk of loss and associated very high economic importance. These approaches have been used for northern spotted owls (Lamberson et al. 1994) but also, in a broader capacity, for a suite of species in a large ecoregional assessment (Schumaker et al. 2004). With increased computing capabilities and demographic data for more species, spatially explicit PVAs are now commonly conducted in conjunction with dynamic landscape models.

## CONDUCTING A PVA FOR A FOREST-ASSOCIATED SPECIES

Because forests are inherently dynamic due to disturbance and regrowth, typically, a dynamic, spatially explicit model is used. HEXSIM (<http://www.hexsim.net/>) is an example of the type of model



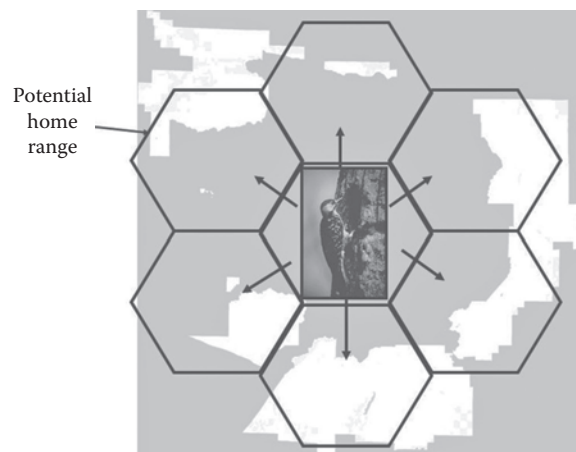
typically used for these sorts of analyses (Marcot et al. 2013, Schumaker 2013). There are eight primary steps to conducting such an analysis (other models require similar, but not identical steps).

First, maps of habitat quality for the species must be developed over the extent of the analysis area. This usually entails application of a wildlife habitat-relationships model (WHR, Johnson and O'Neil 2001), but may entail a more detailed habitat-quality estimate (McComb et al. 2002). The maps must be developed both now and for each time-step into the future, over a period of time deemed adequate to assess population trends. These maps should be based on likely changes in habitat structure and composition, due to management and/or natural disturbances expected to occur over the projection period. If you wish to compare population trends among alternative plans or policies, then you will need a different series of maps for each alternative management strategy being assessed.

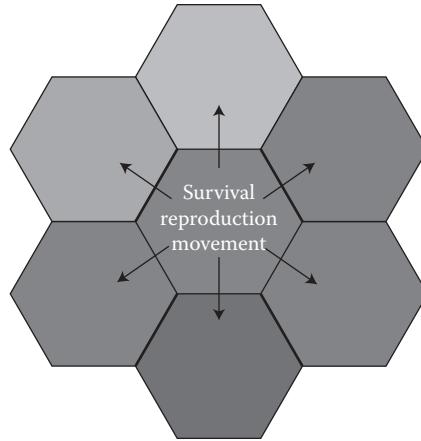
Second, home ranges or territories are assigned to each map. These are usually represented by cells (squares or hexagons) scaled to the home range size or territory for the species being considered, and overlain on the habitat maps (Figure 21.1). The habitat quality is then aggregated through rules representing the amount of habitat of a certain quality that would be needed by a species in its home range to likely achieve a certain level of reproduction or survival. This is done for each in the series of maps, for each management alternative. Where predator-prey relationships are being considered, then this step must be conducted for both predator and prey species being considered in the simulation.

Third, demographic information for the species must be estimated for each age and sex class (although analyses are often restricted to females, and then a sex ratio is estimated to extrapolate to a total population). Age classes may be aggregated into stages if the parameters do not change appreciably from one year-class to another. These parameters include both the estimated average and associated variance for birth rates, reproduction rates, movement rates, and movement direction (if it is not random). These data are typically extracted from published studies, although, for high-priority species, field data collection may be needed to ensure more accurate estimates of these parameters. At the very least, experts on the species are consulted to provide reasonable estimates.

Fourth, these demographic data are then explicitly related to the habitat-quality estimates assigned to the home range cells, such that lower birth and survival rates occur in lower quality patches, and vice versa (Figure 21.2). This must be done for each cell on each map in the time series, for each management alternative. The actual assignment of a value (e.g., survival) to a cell, is typically conducted in a randomized manner. For instance, the survival values assigned to a cell in a given year will reflect the range of values associated with that parameter in that habitat quality class,



**FIGURE 21.1** Simulated home ranges (hexagons) are overlain on maps of habitat availability to estimate the distribution of potentially occupiable home ranges over the extent of the analysis. (Red-cockaded woodpecker photo from Oak Ridge National Laboratory, <http://www.esd.ornl.gov/programs/SERDP/EcoModels/rcw.html>.)



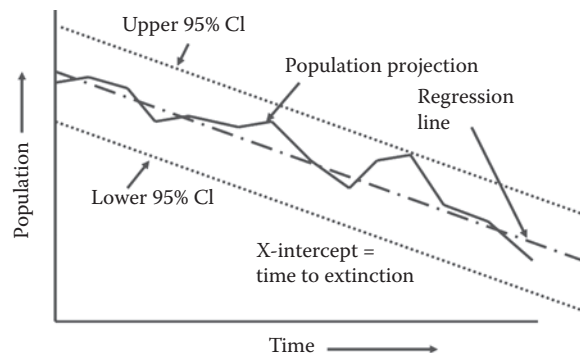
**FIGURE 21.2** Demographic parameters are assigned to each home range (hexagon) based on its likely habitat quality. Should an individual arrive at a home range, then a random draw of parameters, from the range of those associated with the home range quality, is assigned to that individual in that time step.

so that demographic stochasticity is represented. But these assignments are made so that the average value among all cells in a habitat quality class is equal to the mean of the published estimates for that habitat-quality class.

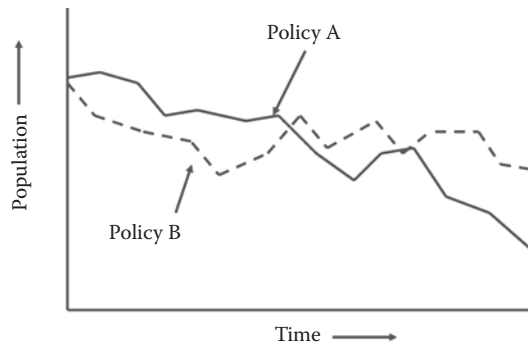
Fifth, at this point, individuals or pairs of animals are assigned to each cell on the landscape at time = 0 (current conditions), such that they represent the known or estimated distribution of organisms over the area of assessment. If the distribution is not known, then a random assignment is made. The landscape is now “seeded” with individuals of various age classes and sexes, in home ranges consisting of varying habitat quality. Sixth, the survival, reproduction, and movement rates associated with each cell are then applied to each individual on the landscape and projected forward one time-step, much the way a Leslie matrix projects a population forward in time.

Seventh, and at the same time, the underlying map of habitat quality is changed to reflect forest disturbance and regrowth. This process is repeated for the number of time-steps in the projection period. But, because assignments of demographic parameters represent a stochastic process, many projections must be made to understand probabilities associated with population trends and extinctions.

Finally, once many projections have been made, then averages, probabilities, and confidence intervals can be assigned at each time step (Figure 21.3).



**FIGURE 21.3** The results from multiple model runs are used to create an average and confidence intervals (CI) in populations over time. With declining populations, calculating the X-intercept can provide an estimate of the time to extinction.



**FIGURE 21.4** Trend lines are compared between alternative management strategies or policies. Those policies, likely to provide the most options for the future (longest time to extinction), are assumed to represent less risk to the species.

The resulting estimate might show a declining trend and an estimated time to extinction of 150 years. Should you believe this estimate? Perhaps, in a relative sense, by comparing this estimate to estimates produced for alternative management plans (Figure 21.4). There are so many assumptions and uncertainties represented in each estimate, that the eighth step is to use an adaptive management approach: compare estimates at each time-step between alternative management approaches (e.g., following NEPA, a no action and a preferred alternative). Indeed, the best use of PVAs is in making projections to compare alternatives, selecting a preferred alternative, implementing it, monitoring the responses, and rerunning the PVA with additional data, allowing continual adaptation and improvements to population estimates. Using this process, we also learn about the habitat and demographic characteristics of the species that seem to most influence population change, and can address those factors during planning and management activities.

## EXAMPLES OF PVA ANALYSES

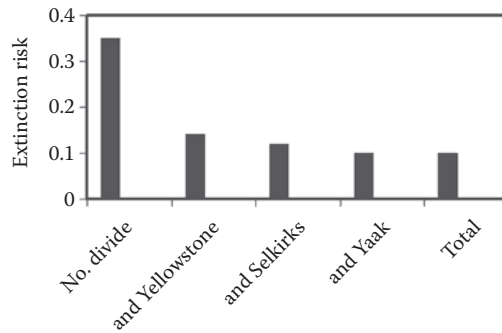
### GRIZZLY BEAR

One of the earliest attempts at conducting a PVA was for grizzly bears in Yellowstone National Park (Shaffer 1983) in the United States. Subsequent approaches proposed by Dennis et al. (1991) and summarized by Morris et al. (1999) provide an estimate of persistence of this large predatory mammal in the lower 48 states of the United States.

The approach taken by Dennis et al. (1991) is somewhat unique, in that the data on which the analysis is based includes exhaustive counts of individuals in the population, including age and sex cohorts. Despite such exhaustive data, the approach is still based on a number of assumptions using multiple stochastic single-population modeling approaches (Morris et al. 1999):

1. The year-to-year variation in the counts reflects the true magnitude of environmentally-driven variation.
2. Inter-annual environmentally-driven variation is not extreme.
3. The population growth rate is not density-dependent.

Dennis et al. (1991) calculated the extinction time for the population of grizzly bears in the Greater Yellowstone ecosystem. They then extrapolated some of the detailed data from the Yellowstone population to other areas, including the Selkirk range in British Columbia, Northern Divide in Washington and British Columbia, and the Yaak River Valley in Montana, to understand the interacting probabilities of population extinction among these known subpopulations of grizzly bears. The analysis assumes that there is no current movement among the subpopulations, so that



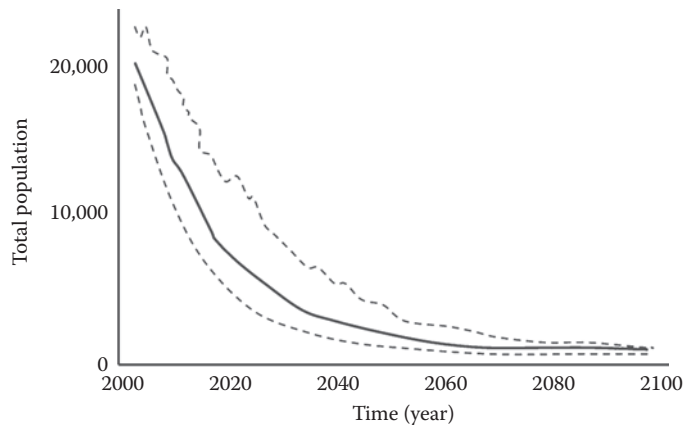
**FIGURE 21.5** Extinction risk for cumulative grizzly bear populations among a collection of protected areas. (Redrafted and adapted from Morris, W. et al. 1999. *A Practical Handbook for Population Viability Analysis*. The Nature Conservancy, Arlington, VA, 47 pp.)

is why multiple single-population analyses were conducted, rather than a metapopulation analysis. They found that for the 500-year projection, there was a 9.6% chance of extinction of all populations (Morris et al. 1999). Considering the populations individually, protecting the Yellowstone population provided the greatest opportunity for reducing extinction risk, but adding the Northern Divide population decreased the probability of extinction from 0.342 (for the Northern Divide alone) to 0.134 (Figure 21.5, Morris et al. 1999).

By adding additional subpopulations, the benefits to reductions in extinction decrease only modestly from that point on (Figure 21.5). So if we wished to be sure that grizzly bears were to remain a component of the ecosystems in the lower 48 states of the United States, then maintaining all of these populations would be a reasonable approach, despite concerns over human safety and property damage. But in this example, the estimate of likely extinction (9.6%) is lower than what we might actually expect, because those populations that are closest to one another are likely to be affected by similar factors influencing their populations (e.g., weather, disease, fire, etc.). But even these simple analyses can be used to understand the potential risks of extinctions if certain local populations are lost.

### MARBLED MURRELET

McShane et al. (2004) described a stochastic metapopulation model used to assess marbled murrelet persistence within six different zones in the Pacific Northwest. They developed a “Zone Model” in which they estimated population projections for each of six zones within the species geographic range, in California, Oregon, and Washington, for 100 years into the future. They assumed that there would be no change in population vital rates over 40 years, and they did not incorporate habitat changes into the model. Similarly, they did not incorporate possible effects of oceanic regime shifts (which could affect foraging efficiency and, hence, survival and reproduction) into the model. The model is a female-only, multi-aged, discrete-time stochastic Leslie Matrix model (Caswell 1989). They found that all zone populations are in decline (over 40 years) with declines of 2.1%–6.2% per decade (Figure 21.6). Further they predicted extinction within 40 years in two zones, and within 100 years in three zones (only one zone population was predicted to extend beyond 100 years), and that the probability of extinction over 100 years is 16%. By modifying their model parameters, they found a reduced rate of decline in two zones if oil spills and gill nets were eliminated as a source of mortality, pointing out the need to consider not just habitat-mediated effects on population changes. More recently, Peery and Henry (2010) used a PVA to predict that control of corvids (crows and jays) near murrelet nests could also lead to an increase in populations.



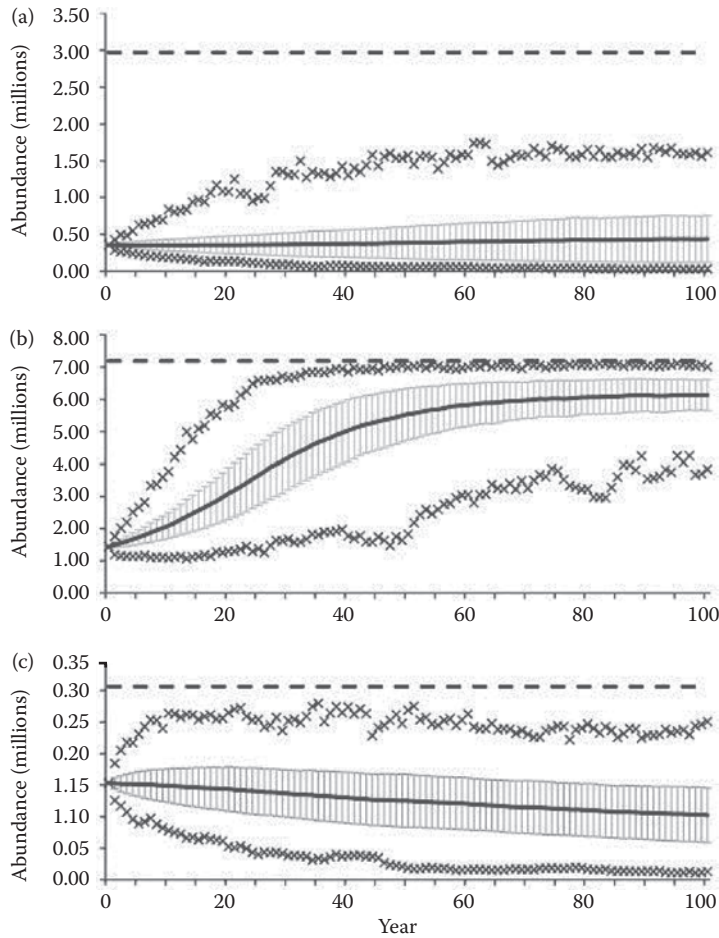
**FIGURE 21.6** Predicted populations of marbled murrelets among six zones in Oregon, Washington and California. Solid line is the mean, dashed lines indicate confidence intervals. (Redrafted based on McShane, C. et al. 2004. Evaluation report for the 5-year status review of the marbled murrelet in Washington, Oregon, and California. Unpublished report. EDAW, Inc. Seattle, Washington. Prepared for the U.S. Fish and Wildlife Service, Region 1. Portland, Oregon. With permission.)

### NEOTROPICAL MIGRANT BIRDS

Bonnot et al. (2011) developed models for three species of breeding birds in the central Hardwoods region of the United States: wood thrush, prairie warbler, and worm-eating warbler. These three species are sensitive to forest fragmentation, and loss of early successional forest, and interior patches of forest. Habitat suitability was defined for each species to develop a base map consisting of 121 patches for the species, ranging in size from 26 ha to 2.6 million ha, and the landscape was held constant (a static landscape) (Bonnot et al. 2011). A number of assumptions were made regarding demographics and dispersal, but the projection of population growth were verified by comparing results to the Breeding Bird Survey (BBS) data from the region in 1966–2007 (Sauer et al. 2008). The predicted annual decline in wood thrush and prairie warblers differed from BBS estimates by less than 1% and 2%, respectively (Figure 21.7). The authors then explored several restoration options for the species: habitat restoration, with both random and strategic locations of restoration efforts; afforestation, to restore forest cover lost historically; and increased survival, by reducing collisions of migrating birds with communication towers (Bonnot et al. 2013). They found that the abundance of prairie warblers and wood thrushes tripled when forests were replanted, or when collisions with communication towers were reduced, and that for all species, strategic placement of restoration efforts within or near existing protected areas was more successful than random placement (Figure 21.8).

### MODEL ERRORS AND UNCERTAINTIES

Models are abstractions of the real world, developed to aid in decision making and to allow managers and scientists to understand complex interacting systems. All model results are incorrect, and we often do not know how incorrect they really are, although the work of Bonnot et al. (2011, 2013), who verified model results based on independent data, indicates that realistic results may be possible for some species. Independent data collected to parameterize the model are not always available, and especially data that shows adverse responses by populations of rare species to management actions that are not only unlikely, but may be illegal. So those using the results from models such as PVAs should verify model projections with independent data when possible, but otherwise



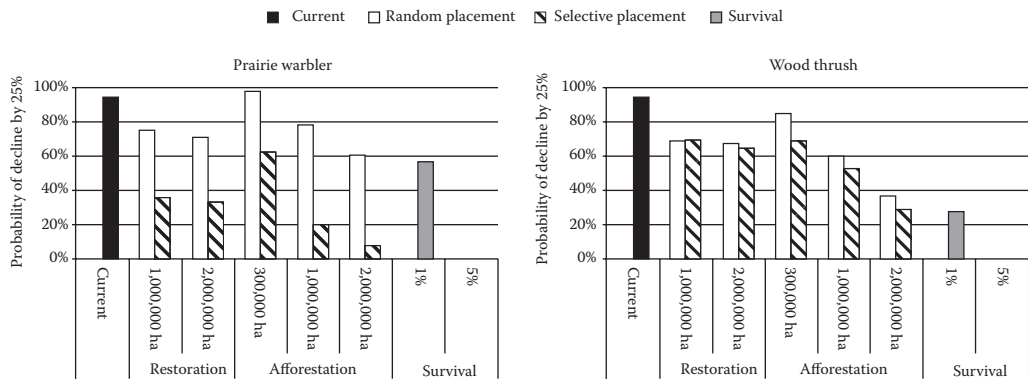
**FIGURE 21.7** Estimated carrying capacities (dotted lines) and projected abundances of wood thrush (a), worm-eating warblers (b), and prairie warblers (c), in the Central Hardwoods Bird Conservation Region. Solid lines are mean abundances  $\pm 1$  SD and are bracketed by minimum and maximum values (x) observed over 1000 simulations. Note: actual growth and trend estimates are based on median abundance rather than mean abundance; therefore, trajectories of graphs may not accurately convey population trends. (Reprinted from *Biological Conservation*, 144, Bonnot, T.W., F.R. Thompson III, and J.J. Millsaugh, Extension of landscape-based population viability models to ecoregional scales for conservation planning, 2041–2053, Copyright 2011, with permission from Elsevier.)

proceed with caution and be fully aware of the potential errors in the models and results. There are four dominant sources of errors—poor data, difficulties in parameter estimation, weak ability to validate or verify models, and effects of alternative model structures—that result in uncertainty in PVA model results (Beissinger and Westphal 1998).

### POOR DATA

PVA models are often described as “data hungry.” They require a broad suite of demographic information on a variety of habitat types ideally collected over many years to adequately represent inherent demographic and environmental stochasticity. Data should be broadly representative, robust, and unbiased. Such data can be very costly, and the data collection requires strict protocols and large sample sizes. The survey of the northern spotted owl is often used as an example of the cost





**FIGURE 21.8** Projected viability of prairie warbler and wood thrush populations resulting from conservation scenarios. Strategic habitat restoration targeted habitat on protected areas, while strategic afforestation created forests in highly forested, productive landscapes. Bars indicate the probability of the regional population declining by 25%. The current scenario assumes the current landscape with no changes. (Reprinted from *Biological Conservation*, 165, Bonnot, T.W. et al., Landscape-based population viability models demonstrate importance of strategic conservation planning for birds, 104–114, Copyright 2013, with permission from Elsevier.)

associated with gaining adequate information over its geographic range to enable credible PVA projections. Millions of dollars have been spent on data collection for this species and millions of dollars are involved in the product values associated with decisions driven, in part, by PVA analyses. Rarely do we have the amount and quality of data for other species that we have for spotted owls. For many species, the vital rates needed to parameterize a PVA are gleaned from the literature, and usually from studies conducted not specifically designed to provide these parameter estimates. For the vast majority of species, PVA projections are better thought of as hypotheses than as estimates. If the estimates are used in decision making, then a set of assumptions and caveats regarding data quality should accompany the population projections.

## DIFFICULTIES IN PARAMETER ESTIMATION

Consider a species' life history, from birth to death. Throughout life, there are events that influence individuals and populations. One critical part of an individual's life that will influence it, if it survives and reproduces, is the period of time immediately after it leaves a nest or den. It is a juvenile, naive to the world, and vulnerable to many factors. Survival during this juvenile stage is a parameter that often emerges as one that greatly influences population dynamics, but it is also one of the most difficult parameters to estimate (Wiens et al. 2006), especially for species that are too small to carry a radio transmitter, or species that are migratory or disperse widely. Dispersal is another parameter that can be difficult to estimate, especially for juvenile individuals in a population (Bonnot et al. 2011), but dispersal estimates are critical to understand the inter-relationships between habitat patch size and proximity (Marcot 2013). In these cases, the technical aspects of PVA modeling may far outweigh our technical capabilities in collecting the data needed to parameterize the models.

## WEAK ABILITY TO VALIDATE OR VERIFY MODELS

Although there are independent datasets that can be used to validate various components of a PVA model (e.g., occupancy rates, survival in some age classes, sex ratios), there are little and often no data available to validate these parameters (confirm accuracy), and rarely are monitoring data available to verify projections (assess consistency with trends), but see Bonnot et al. (2011). Indeed,

due to lag times associated with population responses to management alternatives, some of which may be novel, it will likely take decades or centuries before monitoring data can provide independent assessments of model performance. Fortunately, when monitoring data are collected, then the resulting data can be used to continually improve model performance and accuracy. Oftentimes funding available to develop recovery plans or management plans may not continue over time, as needed, to acquire the necessary monitoring information.

### EFFECTS OF ALTERNATIVE MODEL STRUCTURES

There are several widely used PVA models available now, and there will likely be more in the future, each attempting to include more reality into the model abstractions (Gordon et al. 2004). Each model structure has its own set of strengths and weaknesses with regards to dealing with potential effects of different types of forest management, stochastic events, animal scaling properties and movements, competitors, predators, diseases and parasites, among other factors influencing population dynamics (Beissinger and Westphal 1998). Consequently, projections resulting from one model structure may differ considerably from another. Indeed, projections of future conditions from a range of model structures conducted by different scientists may be one approach to understanding the uncertainties associated with population projections. Consider Bonnot et al.'s (2011) models of bird populations over a region: If they had developed a dynamic landscape model rather than static, would their results have been consistent? If multiple projections using alternative model structures are somewhat consistent in their predicted extinction rates, or population trends, then perhaps more faith can be placed in the results, given the caveats presented in the previous three limitations of all models.

### INTERPRETING RESULTS FROM PVA PROJECTIONS

Given the caveats regarding uncertainty and potential errors described in the previous section, it is important to consider how best to use the results of population viability analyses. First and foremost, do not believe the predictions, at least not with the precision that is often implied by the projections. Predicting the future is easy; predicting it reliably is often impossible. Think of weather predictions, for instance. Predicting the weather a week or more into the future (*The Farmer's Almanac* notwithstanding) is much less reliable than predicting it tomorrow, or in the next hour or next minute. The farther into the future the predictions are made, the less reliable the predictions are likely to be. Long-term predictions may be useful to understand factors such as lag effects and stochastic events but should be viewed with considerable caution. PVA projections are best used over relatively short time periods to make decisions, while using the long-term projections as a context for near-term decisions.

The medieval philosopher William of Occam once stated: "One should not increase, beyond what is necessary, the number of entities required to explain anything" (known as Occam's razor). "Keep it simple, stupid" (the KISS principle) is the saying that my high school math teacher espoused. Do not use complex models when simpler, easier-to-comprehend models will provide you with estimates that are of value in decision making. Of course, this is always a tradeoff, because populations and ecosystems are inherently complex. The natural tendency of decision makers is to use the models that best approach reality (most complex) even when they may not be necessary to reach an informed decision.

Model results, if used cautiously, can be used to understand relationships among the various factors influencing population change and to diagnose causes of decline and potential for recovery. These heuristic aspects of modeling (using modeling to teach us something about the system) may represent the most useful approach to PVA modeling, in that it can help generate hypotheses regarding the factors that seem to be most likely to lead a population to extinction (Marcot et al. 2013). These hypotheses can then be tested in the field and controlled settings to allow us to hone in on key factors that can lead to a more efficient population recovery.

Trends are more important than numerical predictions, because numerical predictions will change as parameters are improved. So it is best to use the projection results in a relative—rather than absolute—sense (Beissinger and Westphal 1998). Comparing extinction probabilities, population trends, or times to extinction among management options is more appropriate than saying that the population will go extinct in 40–65 years, for example. These relative comparisons become even more valuable when independent scientists, using a variety of model structures, all produce estimates that support (or contradict) one another. Decisions based on PVAs are usually made based on the credibility of the results, and the credibility is usually much greater in a relative, rather than absolute, sense.

PVA results are often used within a process called risk analysis. Risk analysis is a structured way of analyzing decisions, and the potential effects of those decisions, when the outcomes are uncertain. The process involves identifying options, quantifying or assigning probabilities, and evaluating and selecting management options. This approach is often used when there are multiple interest groups with different and conflicting objectives, the outcomes of management alternatives are uncertain, and any decision may have serious consequences. Risk analysis attempts to structure and quantify management options to help the decision maker to understand the consequences of action or inaction, and choose a decision path. For instance, the Bonnot et al. (2013) PVAs allow managers to understand the potential for management actions to improve habitat and reverse downward trends. The relative effectiveness of different alternatives, in combination with an estimate of costs, can be used to develop an estimate of the return on investment of alternatives, and also assess which alternatives are most socially acceptable.

## SUMMARY

Fine-filter analyses often entail an assessment of population trends, and risks of extinction of species, from all or a significant part of its range. These assessments are integral to the development of recovery plans for threatened species, and have been conducted for a wide range of species during ecoregional assessments. There is a range of model structures used to assess future population trends representing a range of ecosystem complexities that are included in the model structures. Those that consider a dynamic landscape, particularly relative to the generation length of the species under consideration, are often selected for use with forest-associated species. Due to the complexities of the model structures however, the results of these projections are best interpreted in a relative—rather than an absolute—sense, to allow comparisons among management alternatives. Further, trends predicted into the near future are usually more reliable than long-term projections. Issues such as data quality, inadequate validation, and environmental uncertainties, all influence the utility of these projections.

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# 22 Monitoring Habitat Elements and Populations

Implementation of any stand prescription or forest management plan is done with some uncertainty that the actions will achieve the desired results. Nothing in life is certain (except death!). Following implementation, managers should expect to change plans based on measurements taken to see if the implemented plan is meeting their needs. If not, then mid-course corrections will be necessary. Many natural resource management organizations use some form of adaptive management as a way of anticipating changes to plans and continually improving them (Baker and Read 2011; McCaw et al. 2011, Westgate et al. 2013).

## ADAPTIVE MANAGEMENT

Adaptive management is a process to find better ways of meeting natural resource management goals by treating management as a hypothesis (Allan and Stankey 2009). The results of the process also identify gaps in our understanding of ecosystem responses to management activities. Adaptive management incorporates learning into the process, and the data collected during monitoring provides feedback about the effectiveness of preferred or alternative management practices. The information gained from the process can help to reduce the uncertainty associated with ecosystem and human system responses to management, but successful adaptive management projects are not common (Howes et al. 2010). Westgate et al. (2013) reviewed 1336 scientific papers describing adaptive management projects, of which only 13 applied the concepts of adaptive management to the problem. Westgate et al. (2013) suggested that attempts to apply adaptive management could be improved by better collaboration between scientists and stakeholders, better risk analysis associated with not conducting adaptive management, and ensuring that projects are relevant to the management needs of the affected stakeholders.

Adaptive management has been classified as both active and passive (Walters and Holling 1990). Passive adaptive management is a process where the “best” management option and associated actions are identified, implemented, and monitored. The monitoring may or may not include unmanaged reference areas as points of comparison to the managed areas. The changes observed over time in the managed and reference areas are documented, and the information is used to alter future plans. The manager learns by managing and monitoring, but the information that is gained from the process is limited, especially if reference areas are not used. Without reference areas, we do not know if changes over time are due to management or some other exogenous factors.

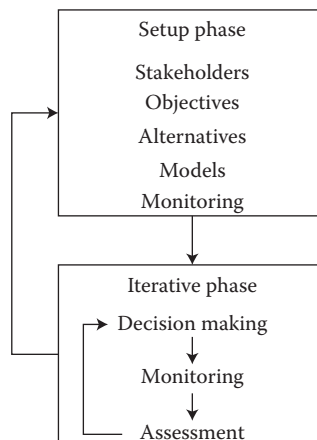
Active adaptive management treats the process of management much more like a scientific experiment than passive adaptive management. Under active adaptive management, management approaches are treated as hypotheses to be tested. The hypotheses are developed specifically to identify knowledge gaps, and management actions are designed to fill those gaps. Typically, the hypotheses are developed following modeling of the system responses (e.g., using forest growth models or landscape dynamics models), to understand how the system might respond, and then use management to see if it responds as intended. Reference areas are used as controls to test responses of ecosystems, and human systems, to management. By collecting monitoring data in a more structured hypothesis-testing framework, system responses can be quantified and used to identify probabilities associated with achieving desired outcomes in the future. Whereas passive

adaptive management is somewhat reactive in approach (reacting to monitoring data), active adaptive management is proactive and follows a formal experimental design.

Traditionally, adaptive management has six steps (Munks et al. 2010), but Williams (2011) described adaptive management as multiple steps having two phases (Figure 22.1). The first phase is a setup phase to engage stakeholders, set objectives, identify alternative approaches, identify models to understand likely futures, and develop a monitoring framework. The second phase is one of decision making, followed by monitoring, followed by assessment of the monitoring data, and then decision making is revisited (Williams 2011). Development of a monitoring plan is at the heart of adaptive management. Monitoring can be conducted to understand if a management plan was implemented correctly, if it was effective, or if the underlying assumptions are valid.

Implementation monitoring is conducted to see if the plan is being implemented on the ground as it is described in the plan (Roccaforte et al. 2010). Are standards and guidelines being followed? Is the appropriate number of snags, logs, and trees being left after harvest? Are harvest boundaries being respected? Are designated skid trails used? Answering these sorts of questions is important, because future stand or landscape conditions are often dependent on correct implementation of the plan. If the plan is not implemented correctly, then there is far greater likelihood that the goals of the plan will not be realized.

Effectiveness monitoring follows implementation monitoring and is designed to determine if habitat elements, populations, or processes are responding as expected and effectively achieving your management goals. Are trees growing as anticipated? Are trees and shrubs producing mast? Are focal species populations persisting and/or growing as anticipated? Answering these questions allows you to know if the management is effective, and, if not, then the results provide evidence for making changes to the plan. Deluca et al. (2010) reported that effectiveness monitoring is rarely conducted on U.S. Forest Service lands, largely due to budget constraints, and so they offered the following three integrated approaches to more efficiently conduct effectiveness monitoring: (1) pursue low-cost multiparty monitoring conducted by a collective of stakeholders, including citizens, conservation groups, timber interests, and agency personnel; (2) conduct highly detailed ecosystem monitoring on a statistically selected number of forest restoration sites by region; and (3) conduct spatial analysis of remotely sensed data as direct or proxy variables to evaluate ecosystem response



**FIGURE 22.1** Traditional adaptive management cycle (above) and two-phase learning in adaptive management (below). In two-phase adaptive management, technical learning involves an iterative sequence of decision making, monitoring, and assessment. Process and institutional learning involves periodic reconsideration of the adaptive management setup. (Reprinted from *Journal of Environmental Management*, 92, Williams, B.K., Passive and active adaptive management: Approaches and an example, 1371–1378, Copyright 2011, with permission from Elsevier.)



to restoration activities. Innovative approaches such as these, along with careful selection of factors to monitor, may make adaptive management and monitoring more widely accepted.

Plans are almost always based on some assumptions. Monitoring of key processes, such as tree and shrub growth rates, population changes, or changes in animal fitness, can produce information that can be used to test assumptions (McComb et al. 2010). Validation monitoring provides the basis for reducing uncertainty associated with assumptions and provides a framework for understanding and interpreting the results of effectiveness monitoring.

With data in hand from implementation, effectiveness, and validation monitoring, the managers can periodically evaluate the responses of the system to the management actions, understand better why they are seeing certain responses, and make adjustments to their management actions in a way that increases the likelihood that they will be more effective in the future. Once these adjustments have been made, it is important to reassess the problem among involved publics to ensure that new concerns and opportunities are identified as a result of making the proposed adjustments. Then the entire process begins anew and, at least theoretically, allows managers to continually improve their ability to meet desired goals.

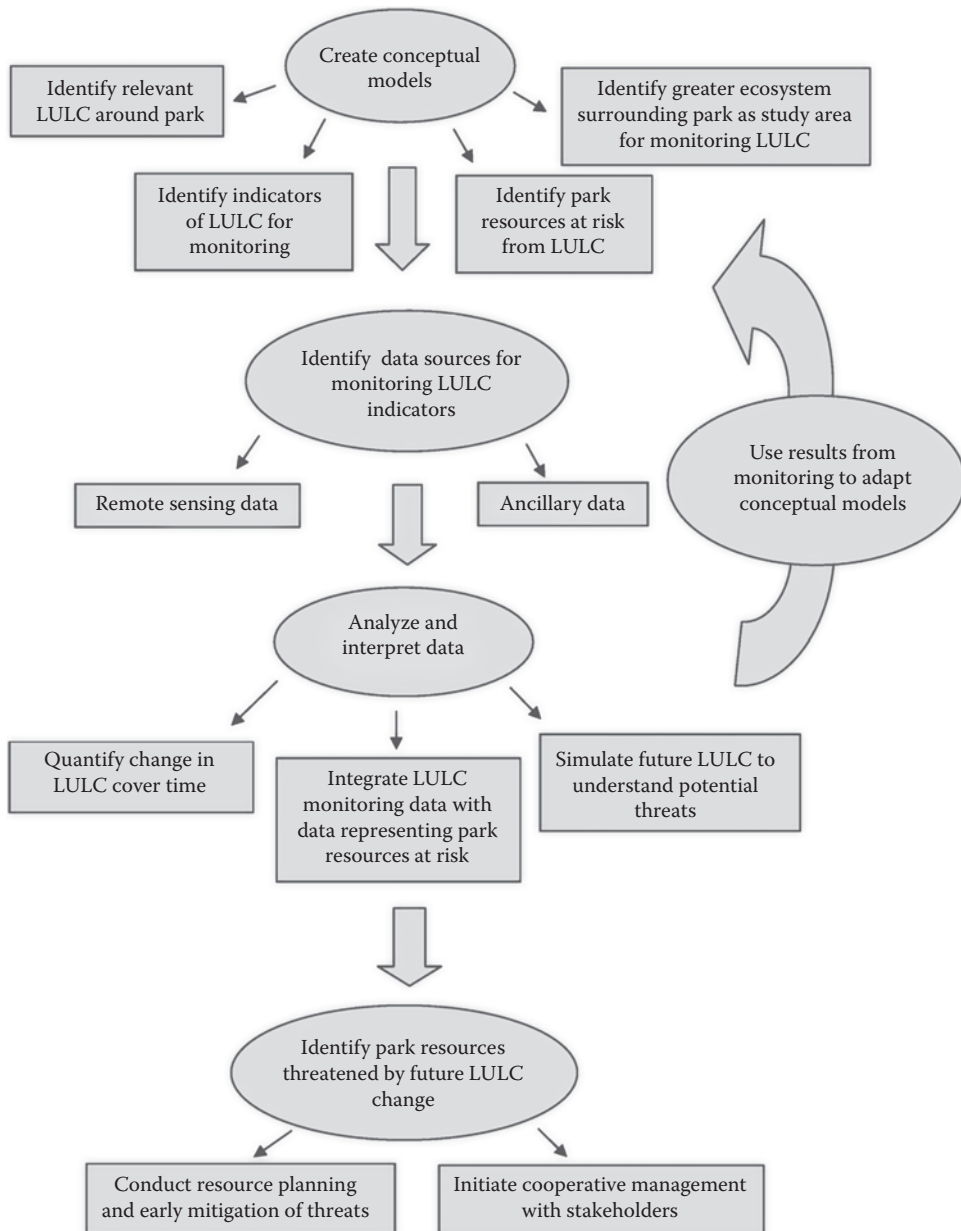
We may monitor habitat types, habitat elements, or populations depending on the goals of the monitoring program. Clearly, it is important to know if the habitat structure, function, and patterns are developing as expected under the plan. Managers may also want to know how populations respond to management. Monitoring populations can be critical in assessing and identifying the potential, or actual impacts of management on the persistence of a species, over some or all of its geographic range. Ideally, the monitoring should allow managers to evaluate (either directly or indirectly) the effects of management actions on factors associated with animal fitness, such as fertility, recruitment rate, survivorship, and mortality. These types of data are very expensive to acquire, so, more often, monitoring of populations is based on occurrence or abundance.

A properly designed monitoring effort allows managers and biologists to understand the long-term dependency of selected species on various habitat components. Resource availability is dynamic for all or most species. Consequently, the challenge, when developing a monitoring plan, is assessing whether changes in occurrence, abundance, or fitness in a population is independent from or related to changes in habitat availability and quality (Cody 1985). Managers will ideally want to identify and incorporate the interactions between habitat and population change in order to make informed management decisions through an adaptive management process (Barrett and Salwasser 1982). The ability to understand why populations are changing in certain directions will depend on the habitat requirements of the species and coincident monitoring of the population and the habitat. The ability to correlate animal and habitat data allows decision makers to better predict the effects of management on populations. However, monitoring plans must address two critical considerations to allow for an accurate and sound comparison of population and habitat data (Jones 1986). First, both animal and habitat data must be collected on the same sites. This allows monitoring programs to document the fluctuations in population density and distribution with respect to changes in the physical and spatial arrangement of habitat elements. Second, the level of detail identified for sampling for both species and their habitats must be determined before associations can be made. This level of sampling will ultimately depend on the objective of the monitoring program. Most monitoring programs deal with the presence/not detected data for many species of rare plants and animals; and, generally, more detailed data is collected on habitat elements in an effort to describe habitat conditions. However, if the objective of the monitoring protocol is assessing habitat quality and its influences on a species' demography, then data must be collected on a species' ability to survive and reproduce (e.g., mortality, survivorship, predation, parasitism) (Cody 1985).

## DESIGNING MONITORING PLANS

Designing a monitoring program is analogous to designing a research project, and a well-designed monitoring program should be every bit as rigorous as a research project; but a monitoring program

is designed to allow managers and stakeholders a chance to learn new information, as well as adapt management actions to achieve their goals. The first step in the process of developing a monitoring plan is to identify, clearly and concisely, the questions to be answered by the monitoring data. Questions can be more easily articulated if the monitoring plan is based on a conceptual framework that describes the states and processes involved in the system being monitored (Jones et al. 2009, Figure 22.2). By understanding the pieces of the system that stakeholders and managers are



**FIGURE 22.2** General steps outlining an approach for monitoring land use and cover change around parks and other protected areas. (Reprinted from *Remote Sensing of Environment*, 113, Jones, D.A. et al., Monitoring land use and cover around parks: A conceptual approach, 1346–1356, Copyright 2009, with permission from Elsevier.)

confident of, they then can focus attention on the critical aspects of the managed system that must be measured, over time, to understand if management is achieving goals. Stakeholders must be involved in this stage of developing a monitoring plan. Once each question and associated goal has been articulated, then the steps indicated in each of the following sections should be taken.

### SELECTION OF INDICATORS

Considering your management plan as a hypothesis, what are the indicators or response variables that you will measure? Indicators are those factors that you hypothesize are responding to the management action. Based on the conceptual framework for the system being monitored and input from stakeholders, provide the rationale for selecting specific indicators or attributes. Why was this indicator selected over others? What are the benefits associated with this indicator? What are the limitations? What are the key habitat elements and population responses that are described in your desired future condition(s) (DFCs)? Desirable characteristics of indicators include (Vesely et al. 2006; McComb et al. 2010) those that

- Have dynamics that are consistent with the element or population of interest
- Are sensitive enough to provide an early warning of change
- Have low natural variability
- Provide continuous assessment over a wide range of environmental conditions
- Have dynamics that can be easily attributed to either natural cycles or anthropogenic stressors
- Are distributed over a wide geographical area and/or are very numerous
- Are harvested, endemic, alien, species of special interest, or have protected status
- Can be accurately and precisely estimated
- Have costs of measurement that are not prohibitive
- Have monitoring results that can be interpreted and explained
- Are low impact to measure
- Have measurable results that are repeatable with different personnel

### DESCRIBE THE SCOPE OF INFERENCE

The scope of inference represents the space and time over which your data can be used to assess changes in the response variable with some known level of certainty. Data must be collected in a manner that provides an unbiased estimate of your response variable from the planning area. Samples should be allocated in a randomized or stratified random manner, with points selected from a pool representing the entire scope of inference. Extrapolating data beyond the scope of inference is done with increasing uncertainty as one departs more and more from the conditions sampled from within the scope of inference. Indeed, broadcasting from the monitoring data (extrapolating to other units of space outside the scope of inference) and forecasting (predicting trends into the future from existing trends) must be done with great care because the confidence limits on the projects increase exponentially beyond the bounds of the data (Kimmins et al. 2010; Munks et al. 2010).

Oftentimes, managers wish to sample large areas so that the results of the monitoring effort can be used more efficiently, but they quickly face a tradeoff. That tradeoff is to monitor over a large spatial extent, so that results are broadly applicable, versus sampling over a small area with less variability to increase the precision of the data (and more likely detect trends). The variability in the indicator will likely increase as the spatial extent of the study increases. As the variance of the indicator increases, the probability of detecting a difference between treatments or of detecting a trend over time will decrease. Funding for the monitoring program often dictates what represents a reasonable level of sampling intensity. Generally, smaller replicated sites from a larger scope of

inference can provide information that is more broadly applicable, but yet would have sufficient statistical power to detect changes.

### DESCRIBE THE EXPERIMENTAL DESIGN

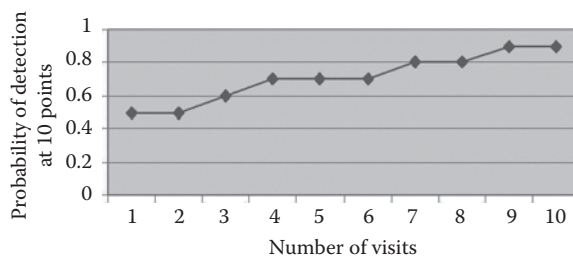
The experimental design will depend on the goals for the monitoring program and may be adapted over time to address new questions that emerge as data is analyzed (Lindenmayer and Likens 2009; Lindenmayer et al. 2011). Consider how the results of data analysis will be used. Will the analysis be used to assess occurrence, trends, patterns, or effects? Estimating occurrence may entail an estimate of the probability of occurrence at a site with an associated estimate of confidence (McComb et al. 2010). Estimating trends often involves a time-series regression, with confidence intervals to understand both the slope of the trend and the uncertainty associated with the trend based on the variability in the data (Hutto and Belote 2013). Estimating patterns may involve use of an analysis of variance (ANOVA), *t*-test, or multivariate analysis (e.g., principal components analysis) to understand if the indicator (response variable) differs between or among areas having different management actions. Estimating ecological effects or management effectiveness on an indicator typically requires a before–after, control-impact (BACI) approach so that we can understand the causes (management action) and effects (relative level of response) associated with our implemented plan (Hutto and Belote 2013).

Because monitoring data are generally collected over time to detect trends or effects, the data are often not independent from one time period to the next. The data collected at one time are related to the conditions when data were collected at a previous time, and this violates a basic assumption when using standard statistical techniques such as ANOVA or regression. This issue can lead to concluding that trends exist when they really do not, simply due to this temporal dependence because of the estimate of variance that is not accurate (too small) leading to a false conclusion (Hurlbert 1984). Repeated measures analyses are often necessary to ensure that estimates of variance between or among treatments reflect this lack of independence (Foster 2001). Similarly, when analyzing spatial data, data collected at one spot can be closely correlated with data collected at nearby spots, so spatial autocorrelation must be addressed (Zhang et al. 2012).

### SAMPLING INTENSITY, FREQUENCY, AND DURATION

“How many samples do I need?” “How long should I monitor?” Those are two of the most commonly asked questions when planning a monitoring program. Use of existing data, or conducting a pilot study, allows planners to conduct a power analysis to decide how many samples are needed to detect a difference or trend, with an acceptable level of confidence (Magurran et al. 2010; Danielsen et al. 2011). Because pilot study data are often collected over a limited timeframe or area, alternative approaches should be considered. Data stabilization approaches are used to assess sample size. For instance, if data on animal density were collected from 20 sites, extending out from some central location, and the variance represented in the data is plotted over number of samples, then the variance should stabilize at some number of samples. Once that asymptote in variance has been reached, then adding additional samples is not likely to influence your estimate of the inherent variability in the response variable, at least not under current conditions. It is also useful to consider how your estimate of variance might change as the plan is implemented, so that your sample intensity in the future is also adequate to address your monitoring question. A similar approach can be taken to identify the number of samples that might be needed to establish the probability of occurrence of a species at a site (Figure 22.3).

“How often should I collect data?” Some habitat elements and populations change very slowly (e.g., snags falling) over time, while others change quickly (browse biomass following a disturbance). The rates of change in the element should dictate the frequency with which monitoring data is collected. Sampling snag-fall every year for 100 years is both inefficient and unnecessary, given



**FIGURE 22.3** Hypothetical change in probability of detecting a species with increasing number of visits or samples. Note that an asymptote is reached at 9 or 10 samples. Sampling beyond this point will not likely improve the precision of your estimate.

the likely changes that would be seen from year to year, either in snags or the species that use them. Sampling every 5 or 10 years would provide useful information at a fraction of the cost.

“How long should I monitor before I can stop?” Sampling duration depends on the time that you think it would take to achieve the DFC and the time that you feel you should monitor the DFC to ensure that it is likely to persist once it is reached. By using an adaptive management approach, with continual improvements in management approaches, then monitoring may continue indefinitely with some response variables being dropped, others added, and some retained as the process continues (Lindenmayer and Likens 2009).

## MONITORING HABITAT ELEMENTS

If the purpose of monitoring includes understanding changes in habitat availability or quality following management, monitoring must measure and document appropriate habitat elements important to the key species, communities, or ecosystems, of concern to the managers and associated publics (see Chapter 4 for a documentation of habitat elements). These data are collected over various scales; however, the relevant scale at which to collect and interpret the data will be defined by the characteristics of the organism(s) of concern. The data should include: the organism’s use of habitat, such as the geographic range; metapopulation structure; home range; resource patch selection; and ultimate resources used by each species identified in the monitoring plan.

Measurement of many habitat elements has been incorporated into existing forest monitoring programs that use ground-plot measurements to document changes in vegetation (e.g., USDA Forest Service Forest Inventory and Analysis) and may represent existing data that can be used in a monitoring program. If these data are collected at permanently marked points, then it is possible to document changes in habitat elements over time. This is especially important since the carrying capacity of habitat for a species is dynamic and depends on a number of resources that can fluctuate over time and space due to natural (e.g., floods, hurricanes) and/or anthropogenic (e.g., harvesting, forest clearing) disturbances.

Remotely sensed data are being used more frequently in association with ground-plot information to assess patterns of change over large areas. Landsat images are commonly used as the basis for assessing change in landscapes over time. The Landsat program placed satellites into orbit around the Earth to collect environmental data about the Earth’s surface (Richards et al. 1999). The reflectance of various wavelengths of light is captured by sensors on the satellite and is stored as discrete numbers assigned to a place on the Earth, typically to a  $30 \times 30$  m square or pixel, but resolution on new satellites is now down to 10 m. The values are specific to a particular place at a particular time and a spectrum of reflectance values (brightness) that are assigned to each pixel (Richards et al. 1999). The area that each pixel covers on the ground represents the resolution or grain of the information. Satellite data must be classified to be of use. Once classified, resulting maps can provide a record of change in classified elements over large areas over time. Despite increasing resolution and

decreasing grain size, Landsat data are less able to provide information on changes in habitat quality, species distribution, and fine-scale disturbances. Instead, very high resolution (VHR) optical data from satellite-based optical sensors are being used more frequently (Nagendra et al. 2013). Light detection and ranging (LiDAR) and synthetic aperture radar (SAR) data, when used in combination with optical data, can even allow users to detect changes in the three-dimensional structure of habitat elements (Nagendra et al. 2013). Further, ground-plot data can be integrated with remotely sensed data to provide more detailed estimates of change in fine-scale habitat elements (Ohmann and Gregory 2002).

The accuracy of interpretation of aerial photographs and classified satellite imagery depends on “ground-truthing” (visiting spots on the ground to see if the class is accurately represented) and subsequent accuracy assessments, but the classification scheme must be designed to assess habitat for the species of interest. Indeed, Cushman et al. (2010) proposed that habitat type classifications are inconsistently applied to landscapes and that a species-specific gradient-based analysis is a more meaningful representation of habitat quality over complex landscapes. Gradient-based landscape analyses are quite likely the approach that will be used in the future, but for the past several decades, landscape analyses have been based on patch-level metrics. Biologists often use a hierarchical approach to define patches based on vegetation types, landform, soil composition, or other factors that are deemed pertinent by the managers and biologists (Kerr 1986). Once a criterion is determined for classification, the landscape can be separated into discrete units so that any additional ground samples can be stratified among vegetation types. The Resource Inventory Committee of British Columbia has outlined a good approach to vegetation stratification (Resource Inventory Branch 1998):

1. Delineate the project area boundary.
2. Conduct a literature review of the habitat requirements of the focal species. If there is enough available and accurate information on the habitat requirements of the species, then it may be possible to identify those vegetation components that relate to habitat quality. However, caution should be used when relying on habitat associations from previous studies since many studies may not be applicable to the region of study or the species of concern.
3. Develop a system of habitat stratification that you expect will coincide with species’ habitat requirements.
4. Use maps, aerial photographs, or satellite imagery to review and select sample units that are reflective of the study area.
5. Evaluate the availability of each habitat strata within the study area.

It is important to keep in mind that one classification scheme will not meet the needs of identifying habitat for all species. If we (humans) classify vegetation as we see structure and composition, then those patterns may or may not relate well to the way that various species respond to patches or gradients of vegetation (Cushman et al. 2010). A classification system that is designed for each species is most likely to reasonably allow an understanding of how habitat is changing for each species over space and time.

## MONITORING FOR SPECIES OCCURRENCE

Assume that you are concerned that management actions will impact a habitat element or a species that could be present in an area. How sure do you want to be that the species occurs in the proposed management area? Do you want to know with 100% confidence if a species occurs in an area proposed for management? Or can you be 95% sure; 90%? The answer to that question will dictate both the sampling design and the level of intensity with which you inventory the site to estimate presence and absence. The more rare or cryptic the species, the more samples that will be needed to assess



presence; and, if the species is rare enough, then the sampling intensity can become logistically prohibitive. In that case, other indicators of occurrence may need to be considered. Consider the following possibilities when identifying the indicators of occurrence in an area (Vesely et al. 2006):

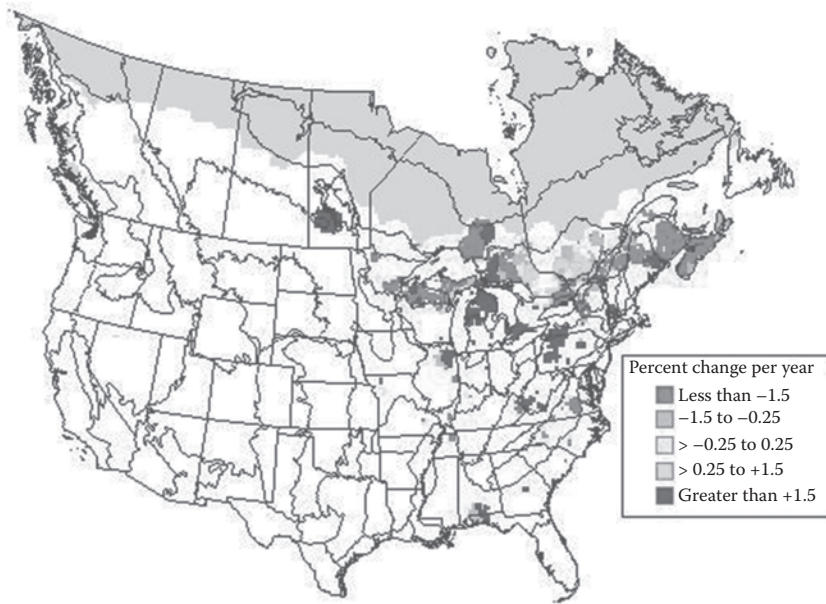
1. Direct observation of a reproducing individual (female with young)
2. Direct observation of an individual, reproductive status unknown
3. Direct observation of an active nest site
4. Observation of an active resting site or other cover
5. Observation of evidence of occurrence, such as tracks, seeds, pollen
6. Identification of habitat characteristics associated with the species

Any of the above indicators could provide evidence of occurrence and, hence, potential vulnerability to management, but the confidence placed in the results will decrease from number 1 to number 6 for most species, based on the likelihood that the fitness of individuals could be affected by the management action.

Opportunistic observations of individuals, nest sites, or habitat elements can be of some value to managers, but are often of not much use in a monitoring framework, except to provide preliminary or additional information. For instance, GPS-tracked locations of a species observed incidentally over a three-year period, could be plotted on a map, and some information could be derived from the map (known locations). The problem with using these data points in a formal monitoring protocol is that they are not collected within an experimental design. Undoubtedly, there are biases associated with where people are or are not likely to spend time, with species detectability among vegetative, hydrologic, or topographic conditions, and with varying detectability among age or sex cohorts. Consequently, this information should be maintained, but rarely would it be used as the basis for a formal monitoring design.

## MONITORING TRENDS

Long-term monitoring of populations to establish trends is often used within monitoring programs. Such monitoring programs provide information on changes in populations or habitat availability, but they do not necessarily indicate why populations are changing. For instance, consider the changes in woodcock populations over a 27-year period (Figure 22.5). Clearly, the number of singing male woodcocks has declined markedly over this time period. This information is very important, in that it indicates that additional study is needed to understand why the changes have occurred. Are singing males simply less detectable in 1995 than they were in 1968? Are populations actually declining? If so, are the declines due to changes in habitat on the nesting grounds? Wintering grounds? Migratory flyways? Is the population being over-hunted? Are there disease, parasitism, or predator effects, which are causing these declines? Are these declines uniform over the range of the species or are there regional patterns of decline? All of these questions indicate the need to consider an adaptive monitoring approach, in which the questions asked and the indicators used can change over time in response to what is learned and what questions are raised (Lindenmayer and Likens 2009). Analysis of regional patterns indicates that the woodcock declines may not be uniform over the region (Figure 22.4). Declines are apparent in the northeastern United States but not uniformly throughout the Lake States. The Breeding Bird Survey data indicates that there are areas where declines have been significant (Sauer et al. 2012), and the work by Bruggink and Kendall (1995) indicates that the magnitude of the declines in some areas is perhaps even greater than might be indicated by the regional averages. So it would seem that causes for declines are probably driven by effects that are regional. Approaches to understanding the potential causes for change in the abundance of a species can be much more informative when considering changes to a management plan than simply examining trends. The causes for the declines would be addressed at the local scale in a manipulative manner that would allow assessment of cause and effect relationship.

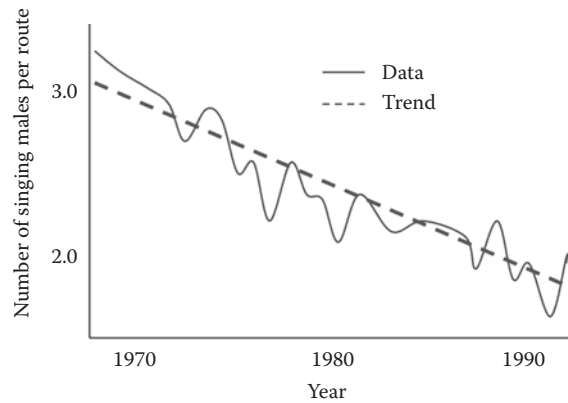


**FIGURE 22.4** Geographic distribution in woodcock population changes. Note that population declines are not uniformly distributed throughout the range of the species. (From Sauer, J.R. et al. 2012. The North American Breeding Bird Survey, Results and Analysis 1966–2011. Version 07.03.2013 USGS Patuxent Wildlife Research Center, Laurel, MD.)

The design of a trend-monitoring program should carefully consider the scope of inference, and if the scope of inference is large (geographic range), monitoring may necessitate coordination over large areas among multiple stakeholders. Site-specific trend analyses will probably be of limited value in many instances, because the fact that species “x” is declining at site “y” is probably not as important as knowing why the species is declining at site “y.” Using trend monitoring to detect increases or declines in abundance is perhaps best applied to high-priority species or to allowing development of associations with regional patterns of habitat availability. These associations then allow the opportunity for a more informed development of hypotheses that can then be tested in manipulative experiments to identify causes for changes.

One aspect of trend monitoring that must be considered carefully is the sampling intensity needed to detect a change in slope over time. Consider Figure 22.5. Annual data were quite variable over the 27-year period, but a trend is still detectable because the slope was so steep. Annual variability caused by population fluctuations and sampling variance can often prevent detection of a statistically significant change in slope. This problem is exacerbated when the slope is not as dramatic as in Figure 22.5. There are several factors that must be considered carefully in the design of a trend-monitoring plan:

1. What is the spatial scale over which you wish to understand if habitat elements or populations are declining or increasing? If it is not the geographic range of a species or subspecies, then what portion will be monitored and how will the information be used?
2. Is the indicator that is selected as unbiased as possible and not likely to vary among time periods, except as would be caused by fluctuating populations?
3. Given the inherent variability in the indicator that is being used, how many samples will be needed in each time period to allow detection of a slope of at least “x” percent per year over time?



**FIGURE 22.5** Trends in woodcock populations over time. Note that the annual data are variable but that the slope is sufficiently steep to allow detection of a trend. (Redrafted and adapted from Bruggink, J.G., and W.L. Kendall. 1995. *American Woodcock Harvest and Breeding Population Status*, 1995. US Fish and Wildlife Service, Laurel, Maryland, 15 pp.)

4. Given the inherent variability in the indicator that is being used, at what point in the trend is action taken to recover the species or reverse the trend? This trigger point should be well before the population reaches an undesirably low level because the manager will first have to understand why the population is declining before action can be taken, and there may be a lag time in population response to any changes in management.
5. Will the data be used to forecast results into the future? If so, recognize that the confidence intervals placed on trend lines diverge dramatically from the line beyond the bounds of the data. Forecasting even a brief period into the future is usually done with little confidence, unless the underlying causes are understood.

Finally, it is important to recognize that auto-correlation among data points is not only likely, but should be expected under traditional designs in which data are analyzed using time-series regression.

## CAUSE AND EFFECT MONITORING DESIGNS

If the monitoring plan being developed is designed to understand the short- or long-term effects of some management action on a population, then the most compelling monitoring design would take advantage of an approach that would assess responses to those actions. Monitoring conducted over large landscapes or multiple sites may use a comparative mensurative approach to assess patterns and infer effects (e.g., McGarigal and McComb 1995; Martin and McComb 2002). This approach allows comparisons between areas that have received management actions and those that have not and is often analyzed using an ANOVA approach. The approach is retrospective and substitutes space for time. Alternatively, the BACI approach allows monitoring to occur on treated and untreated sites, both before and after management has occurred (e.g., Chambers et al. 1999; Bro et al. 2012).

Although the BACI design is usually considered superior to retrospective analyses, BACI designs often are not logistically feasible. On the other hand, retrospective designs—that compare treated sites to untreated sites—raise questions about how representative the untreated sites are of the treated sites prior to treatment. In a retrospective design, the investigator is substituting space (treated vs. untreated sites) for time (pre- vs. posttreatment populations). The assumption behind this approach is that the untreated sites are representative of the treated sites before they were treated. With adequate replication of randomly selected sites, this assumption can be justified, but

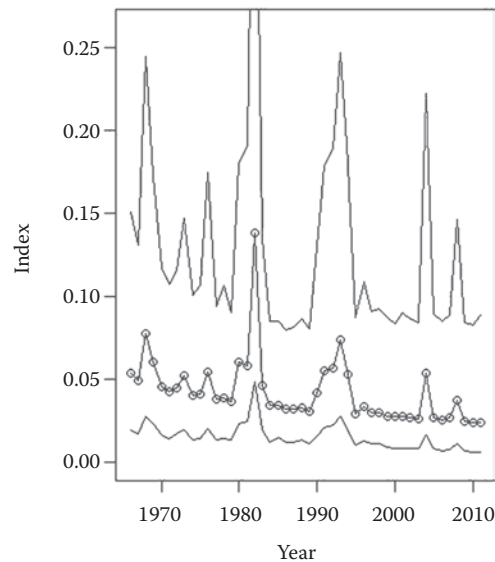
often large-scale monitoring efforts are costly, and logistics may preclude both sufficient replication and random selection of sites. Doubt may persist regarding the actual ability to detect a cause-and-effect relationship using a retrospective approach, especially if the statistical power of the test is low.

BACI designs are more powerful and can establish cause-and-effect relationships, but they can often suffer from nonrandom assignment of treatments to sites, simply due to the logistics involved in harvest planning. Often the location and timing of management actions do not lend themselves to strict experimental protocols. Lack of random selection may limit the scope of inference only to the sites sampled. Nonetheless, with some care in matching control sites to sites that will be treated in the future, there is more that can be learned about the effects of a treatment on a population using this approach than retrospective designs (Rost et al. 2012).

## ARE DATA ALREADY AVAILABLE AND SUFFICIENT?

Before embarking on collection of new data to monitor management implementation or effectiveness, or validation of assumptions, it is always prudent to ask if data already exists to address the monitoring questions. Existing data may not be better than no data at all if the data are of poor quality or have inherent biases. Consider the following questions when evaluating the adequacy of existing data to address a monitoring question:

1. *Are samples independent?* Are observations in the dataset representing units to which a treatment has been applied? Taking ten samples from one harvest unit is not the same as taking one sample from ten harvest units. In the former example, the samples are subsamples of one treatment area; in the latter example, there is one sample in each of ten replicate units (Hurlbert 1984). Further, if the species under consideration has a home-range size smaller than the average harvest-unit size, then sampling the species in harvest units probably represents reasonably independent samples. If the species under consideration has a home range that spans numerous harvest units, then the selection of harvest units to sample should be based on ensuring to the degree possible that one animal is unlikely to use more than one harvest unit.
2. *How were the data collected?* What sources of variability in the data may be caused by the sampling methodology (e.g., observer bias, inconsistencies in methods, etc.)? If sample variability is too high because of sampling error, or if an inherent bias exists (sampling along roads with and without tree cover), then the ability to detect differences or trends will decrease.
3. *Were sites selected randomly?* If not, then there may be (likely is) bias introduced into the data that should raise doubts with regard to the accuracy of the resulting relationships or differences. Although it may be possible to account for biases, interpretation of monitoring results should be conducted with caution regarding inherent bias.
4. *What effect size is reasonable?* An effect size is the difference (or slope) that you could detect given your sample size, sampling error, and the probability of making an error (as indicated by an alpha level), when rejecting a null hypothesis (that there is no difference between treatments or no trend over time). Even a well-designed study may simply not have the sample size adequate to detect a difference or relationship that is real, simply because the study was constrained by resources, rare responses, or other factors that increase the sample variance and decrease the effect size. Again, how this is dealt with depends on the question being asked. Which is more important—to detect a relationship that is real, or to say that there is no relationship when there really is none? In many instances, where monitoring is designed to detect an effect of a management action, the former is more important (especially using the precautionary principle). In that case, the alpha level may be increased (from say 0.05 to 0.10 or more) to make it more likely that an effect will be detected, but in so doing you will be proportionally more likely to say a relationship is real, when it is really not.



**FIGURE 22.6** Changes in detections of yellow-billed cuckoos in New Hampshire 1966–2011. Limited detections, and what appears to be an outlier in the early 1980s, limits our ability to say with certainty that there is a negative trend. But using the precautionary principle, we might conclude that there is a negative trend and change management plans accordingly. (From Sauer, J.R. et al. 2012. The North American Breeding Bird Survey, Results and Analysis 1966–2011. Version 07.03.2013 USGS Patuxent Wildlife Research Center, Laurel, MD.)

5. *What is the scope of inference?* From what area were samples selected? Over what time period? Are the results of the work likely to be applicable to your area? The more different the conditions under which the data was collected from the conditions in your area, the less confidence you should place in the results.

Given the cautions indicated above, it is reasonable and correct to use data that is already available to inform and focus the questions to be asked by a monitoring protocol. For instance, results from the Breeding Bird Survey (Sauer et al. 2012) include a credibility index that flags imprecise, small sample size or otherwise questionable results. For instance, yellow-billed cuckoos have shown a significant decline in southern New England over the past 34 years (Figure 22.6), but the data are deficient when considering regional changes in abundance due to low detection levels (Sauer et al. 2012). Further, an examination of the data would indicate that the one estimate in the early 1980s may be an outlier and may have an overriding effect on the results. In this example, it would be useful to delete that datum and rerun the analyses and then determine if the declining relationship still holds.

Use of existing data and an understanding of data quality can be of value in identifying areas of a management plan that are based on weak data or assumptions. Those factors that are based on assumptions or weak data and which seem quite likely to be influencing the ability to understand management effects should become the focus of questions to be answered by the monitoring plan.

## MAKING DECISIONS WITH DATA

Once you have collected data, then you need to decide what to do with it. Say that your monitoring data of population change over time under current management practices produced a chart similar to that in Figure 22.5. At what point along the *x*-axis do you decide that it is time to change your

management approach—1975, 1985, 1995? Do you wait and collect more data and make a decision in 2025? At what time is a decision to change management soon enough to reduce a declining trend but not too late to make a change that may be moot? Those decisions should be clearly articulated in a management plan. We know that we cannot meet the needs of all species in the same stand or small forest at the same time; there will always be species that are increasing in abundance and others that are declining. When defining your desired future conditions, the expected changes in area of habitat, populations, or frequency of occurrence of all species of concern can be described, and then monitoring can be conducted to see if trends are progressing as expected. Deciding when to make management changes can be based on when the rates of change depart from the expected, to a degree specified in the plan. It is important to consider these decision thresholds (sometimes referred to as “trigger points”) prior to implementing a management plan, rather than waiting until data have been collected and analyzed (Block et al. 2001). Nonetheless, unanticipated results may arise during monitoring, and this should trigger a reinitiation of the adaptive management process.

## EXAMPLES OF APPROACHES TO MONITORING

Species respond to habitat availability and quality at multiple scales, and management occurs over a range of spatial and temporal scales. Consequently, a monitoring plan usually takes either a management-centered approach or an organism-centered approach. Regardless of the approach taken, the scope of inference from the monitoring data will be influenced by the interaction between these two approaches and their inherent scales of space and time. The scale of the management actions relative to the scale associated with populations should help identify a set of questions that can be addressed by different data types. For instance, consider the following examples based on the designs proposed by Vesely et al. (2006):

### MONITORING CLONAL PLANTS

Given our lack of knowledge of the distribution of a clonal plant species, we are concerned that timber management plans could have a direct impact on remaining populations that have not yet been identified in our district. How will we know if a timber sale will impact this species?

In this example, the plant species may have a geographic range extending well beyond the timber sale boundaries and may extend over multiple forest ownerships. The concern is that populations of this species are patchily distributed and poorly known. We may be concerned that population expansion and persistence may be highly dependent on mobility of propagules among population patches, and that additional loss of existing patches may exacerbate loss of the population over a significant portion of its range. Consequently, the primary goal of a monitoring effort should be to identify the probability of occurrence of the species in a timber sale. A survey of all (or a random sample of) impending timber sales will provide the land manager with additional information with regard to the distribution of the species. Although information may be collected that is related to fitness of the clone (size, number of propagules, etc.), the primary information needed is an estimate of the probability of occurrence of the organism, prior to and following management actions. Indeed, this survey approach also can lend itself well to development of a secondary monitoring approach, which utilizes a manipulative experiment. Identification of sites where the species occurs can provide the opportunity for random assignment of manipulations and control areas to understand the effect of management on the persistence of the species.

### MONITORING THE OCCURRENCE OF A SMALL MAMMAL SPECIES

Given the uncertainty in the distribution of a species of small mammal over a forest, will a planned timber harvest have an undue impact on a large proportion of individuals of this species on this forest?



In this example, the species' geographic range extends well beyond the boundaries of the ownership, but the manager needs a context within which to understand the potential for adverse effects on the species. Based on survey information, it is clear that the species occurs in areas that are planned for harvest, but do they occur elsewhere in the ownership? We need to have an unbiased estimate of the abundance of the species over the entire planning area to understand if the proposed management activities indeed represent the potential to impact a significant portion of the population for this species. With an estimate of abundance that extends over the ownership (or forest, or watershed, etc.), one can estimate (with known levels of confidence) if the proposed management activities might affect 1% of the habitat or population for this species, or 80% of the habitat or population.

#### MONITORING TRENDS IN A SALAMANDER SUBPOPULATION

Given the history of land management on a forest and the plans for future management, will these management actions be associated with the abundance and distribution of a subpopulation of a salamander species that we know occurs in our forest?

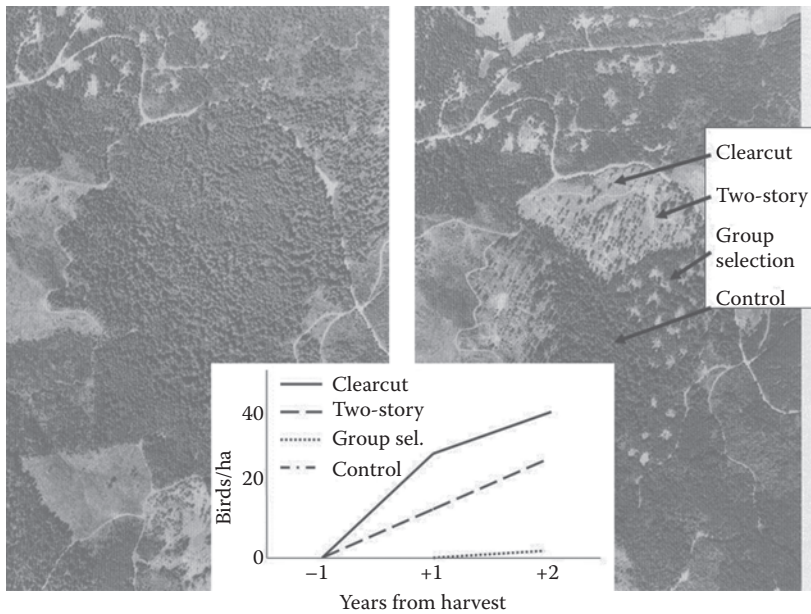
In this example, the species, again, has a geographic range that extends beyond the boundaries of the forest, but there is concern that a subpopulation of a relatively immobile species may occur in our forest. The concern is that the subpopulation may decline in abundance over time, as a result of the past and projected management activities on the forest. The goal is to document trends in abundance over time. Changes in abundance, or even in occurrence, may be difficult to detect at a local scale (timber harvest, road building), because individuals are patchily distributed; but, cumulatively, over space and time, impacts could become apparent. Consequently, this *trends*-monitoring approach should extend over that portion of the forest where the species is known or likely to occur and provide an estimate of abundance of the species at that scale over time.

#### MONITORING RESPONSE OF NEOTROPICAL MIGRANT BIRDS TO FOREST MANAGEMENT

Concern has been expressed for several species of neotropical migrant birds whose geographic range extends across the forest. Is the proposed stand management causing changes in the abundance of these species?

In this example, we are dealing with a species that is probably widely distributed, reasonably long-lived, and spends only a portion of its life in the area affected by proposed management. One could develop a trends-monitoring framework for this species, but the data resulting from that effort would only indicate an association (or not) with time. It would not allow the manager to understand the *cause-and-effect* relationship between populations and management actions. In this case, there are several strata that must be identified relative to the management actions. Can the forest be stratified into portions that will not receive management and others that will receive management? If so, then are the areas in each stratum sufficiently large to monitor abundance of those portions of the populations over time? Monitoring populations in both strata, prior to and following management actions, imposed within one of the strata, would allow the managers to understand if changes occur in abundance or reproductive output. For instance, if populations in both managed and unmanaged areas declined over time, then the managers might conclude that population change is independent of any management effects, and some larger pervasive factor is leading to decline (e.g., changes in habitat on wintering grounds). On the other hand, should the population in the unmanaged stratum change at a rate different from that on the managed stratum, then the difference could be caused by management actions and lead managers to change their plan.

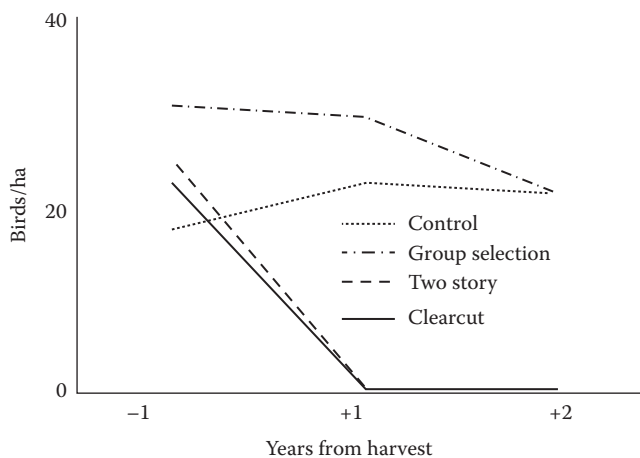
In Figure 22.7, one of three replicate areas is shown prior to and following the management actions that included clearcut with reserves, two-story stands, and group-selection stands. A central control area can also be seen in the post-treatment photo. The results from this effort produced predictable responses, but the responses could clearly be linked to the treatments. White-crowned sparrows were not present on any of the pretreatment sites, but were clearly abundant on the clearcut



**FIGURE 22.7** One of three replicate areas used to assess breeding bird response to treatments using a BACI experimental design and associated changes in white-crowned sparrow detections. (Redrafted from Chambers, C.L., W.C. McComb, and J.C. Tappeiner. 1999. *Ecological Applications* 9:171–185. With permission from the Ecological Society of America.)

and two-story stands, following treatment (Figure 22.7). The treatments caused a response in the abundance of this species.

Conversely, hermit warblers declined in abundance on these two treatments but remained fairly constant on the control- and group-selection treatments (Figure 22.8). Consequently, for this forest, we could predict, with 95% confidence, that future management actions such as these would produce comparable changes in the abundances of these two species. To fully understand why these



**FIGURE 22.8** Changes in abundances of hermit warblers following three management actions. (Redrafted from Chambers, C.L., W.C. McComb, and J.C. Tappeiner. 1999. *Ecological Applications* 9:171–185. With permission from the Ecological Society of America.)

changes might have been observed, ancillary data on habitat elements important to these species also could be collected. If the treatments caused changes in important habitat elements, then the reasons for the effects become clearer (Chambers et al. 1999). These habitat-relationships analyses can be particularly informative when developing predictive models of changes in abundance or fitness of an organism, based on changes in habitat elements caused by management actions or natural disturbances.

## MONITORING HABITAT ELEMENTS

Finally, say that we believe the most likely factor affecting the change in populations of a wide-ranging raptor is change in nest-site availability. Populations are low, and the probability of detecting a change in abundance or fitness of this species over a large forest is very low. Managers may decide to monitor habitat elements that are associated with demographic characteristics of the species, rather than try to monitor the population itself. So we might ask: “How is management affecting the abundance and distribution of potential nest trees for this species?”

Ideally, monitoring of the habitat elements and the associated demographic processes can be conducted to assess cause-and-effect relationships (see above), but with rare or wide-ranging species, this may not be possible. The monitoring data needed to develop *wildlife habitat relationships* includes an unbiased estimate of the availability of key habitat elements that are assumed to be associated with a demographic characteristic of the species and an estimate of the demographic characteristics assumed to be associated with each habitat element. It is important that the monitoring framework for the vegetation component of the habitat relationships must be implemented at spatial and temporal scales consistent with those used by the species of interest.

## SUMMARY

Adaptive management is a formal process of treating management plans as hypotheses, implementing the plans, and monitoring the implementation and effectiveness of the actions and validating key assumptions. The information gained from this process is then used to refine and improve future actions. Design of monitoring plans includes several key steps once goals have been identified: identify the appropriate response variables; identify the scope of inference; establish the experimental design; and estimate the appropriate sampling intensity, frequency, and duration. Monitoring of habitat elements usually includes use of both ground-based and remotely-sensed data. Monitoring of habitat element and population may be conducted to survey the probability of occurrence of species or elements, examine trends, or establish cause-and-effect relationships. Deciding how to use the information to make changes in management direction should be a key part of the description of the desired future conditions in the management plan.

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# 23 Forest Sustainability and Habitat Management

Sustainable. Does it mean leaving resources for the next generation that are equal to or greater than those that we enjoy? Or does it mean not losing any of Leopold's pieces (see Chapter 1)? Or using resources wisely for the greatest good in the long run as Pinchot suggested? Or restoring as much as possible to wilderness to be honored as a place of beauty and spirit as John Muir might have suggested? Clearly, sustainability, literally, is in the eye of the beholder. Equally as clear is that society is demanding more services, *ecosystem services*, from our forests than they ever have before.

I have heard some foresters say that they manage forests sustainably. They cut trees, they plant trees, the trees grow, and they cut them again. Trees are indeed a renewable resource and theoretically production of wood fiber can continue for a very long period of time using currently accepted silvicultural practices. But in the past 20 years, society has raised questions about what exactly is being sustained:  $2 \times 4$  inches, plywood, and pulp? Water? Habitat for various wildlife species? Recreation? Aesthetics? Medicinal plants? Nontimber products? More? During the 1980s and 1990s, large areas of tropical forests were cleared for agriculture, or for planting exotic tree species for timber production, and some forests were simply cut and abandoned.

Philosopher George Santayana once stated: "Those who forget the past are doomed to repeat it." And indeed they have. This is a story that has repeated itself from the northeastern United States, through the Deep South and parts of the west, but also in New Zealand, Australia, and many countries in South America, leading to an increasing number of questions regarding forest sustainability.

In 1992, 172 governments and 2400 representatives of nongovernmental organizations (NGOs) participated in a meeting in Rio de Janeiro, Brazil, in what became known as the Earth Summit. One outcome of that meeting was the endorsement of a set of 17 nonbinding forest principles that provided guidance for management and conservation of global forests. Also during this meeting, an agreement on biodiversity conservation was reached, which stated that:

...the signatories must develop plans for protecting habitat and species; provide funds and technology to help developing countries provide protection; ensure commercial access to biological resources for development and share revenues fairly among source countries and developers; and establish safety regulations and accept liability for risks associated with biotechnology development.

One hundred fifty-three nations endorsed this agreement. The United States did not.

The very next year, in 1993, another meeting was held in Montreal, Canada, that culminated in the Montreal Process. Participants included countries with temperate and boreal forests, including the United States. They developed a framework for measuring the progress of each country toward sustainable forest management. The framework included 7 criteria and 67 indicators, known collectively as the Montreal Process Criteria & Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (Washburn and Block 2001). The first criterion dealt with the conservation of biodiversity. This process and the discussions that led to it represented the social expectations—if not demands—that forests be managed in a way that ensured sustainability of a variety of resources, including, and especially, biodiversity.

## DEFINING THE RESOURCES TO BE SUSTAINED

Managing forests sustainably requires that those people responsible for forest management understand which resources need to be sustained. The Montreal Process provides an internationally accepted list of resources that should be considered during forest management (listed in order of criteria in the document, Washburn and Block 2001):

1. Conservation of biological diversity, including ecosystem diversity (5 indicators), species diversity (2 indicators), and genetic diversity (2 indicators).
2. Maintenance of productive capacity of forest ecosystems (5 indicators).
3. Maintenance of forest ecosystem health and vitality (3 indicators).
4. Conservation and maintenance of soil and water resources (8 indicators).
5. Maintenance of forest contribution to global carbon cycles (3 indicators).
6. Maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of societies, including production and consumption (6 indicators), recreation and tourism (3 indicators), investment in the forest sector (4 indicators), cultural, social, and spiritual needs and values (2 indicators), and employment and community needs (4 indicators).
7. Legal, institutional, and economic framework for forest conservation and sustainable management, including a legal framework (5 indicators), institutional framework (5 indicators), economic framework (2 indicators), a means to measure and monitor changes (3 indicators), and to conduct and apply research and development (5 indicators).

Several of these criteria specifically identify areas of forest management where habitat must be explicitly considered. Clearly, Criterion 1 identifies ecosystems, species, and genes that must be conserved. Managing forests in ways that achieve this goal is the motivation for writing this book. This indicator also ensures representation of forest types across a landscape and addresses levels of fragmentation of forests. Species richness, rare species protection, and population viability must be considered, as must species on the periphery of their geographic range. Other indicators also have implicit habitat management goals: plantations of native tree species, harvest of nontimber forest products (including berry-producing plants), management within the historical range of variability, maintaining ecological processes, providing coarse woody debris (as a carbon store), as well as socioeconomic goals such as employment and social stability. Many of the concepts addressed so far in this book relate directly to sustainable forest management as outlined in the Montreal Process. The devil is in the details, however, when trying to decide how to apply these concepts to forests at local, regional, and global scales.

## SCALES OF SUSTAINABILITY

The Montreal Process framework is useful for assessing the status and trends of forest services and is designed to enhance international communication about sustainable forest management (Washburn and Block 2001). With increased information within and among nations regarding the indicators identified in the Montreal Process, policy makers and stakeholders may be better able to make informed forest management decisions locally, regionally, and globally. How do these indicators scale over space and time? Clearly, there is a temporal limit to sustainability. Societies change and so do their values. Climates change and so does the ability of ecosystems to provide services. Species go extinct, including humans. Stars die and planets grow cold. So the first step in developing a sustainable forest management plan is to define the scope of the plan. How far into the future will we strive to remain sustainable? As long as possible, recognizing that it is a continually moving target? For 100 years? 1000 years? Setting these goals may seem moot, but they do influence the actions that will be taken today to ensure that goals in the future are attainable.

We also need to define the size of the area over which we are striving to be sustainable. The Oregon Department of Forestry has used the Montreal Process criteria and indicators as a basis for developing a sustainable forest management plan for state forests. The Oregon Department of Forestry planners defined the spatial scale as one ownership. The department's ability to achieve its goals will be in part dependent on how its neighbors view forest management and how landowner decisions aggregate to contribute to regional goals, and how regional contributions aggregate to achieve global goals. Think globally, act locally. Although defining the temporal and spatial framework within which sustainable forest management will be conducted, it is equally as important to define the context within which sustainable forest management is being conducted, and how your actions on your lands contribute to some larger aggregate goal. The concept of explicitly considering the context for sustainable forest management is particularly important when addressing Criterion 1. It should be clear by now that no single landowner and no set of reserves will meet the needs for all species and sustain global biodiversity along with other ecosystem services.

## **HUMANS ARE PART OF THE SYSTEM**

The crux of the issue regarding sustainable forest management is: How can we provide the depth and breadth of ecosystem services, that include economic products over space and time demanded by society. Humans set the agenda, humans are part of the problem, and humans are part of the solution. It's all about us. Forest managers are granted a social license to practice forestry in a way that they choose, so long as ecosystem services are provided. How does society grant that license? Unlike wildlife, which, in the United States, is a public resource that occurs on public and private lands, forests are considered private resources. This political dichotomy of ownership is a challenge when managing habitat on private lands to achieve public wildlife goals. Society benefits from forests regardless of whose land they occur on. Policies and laws (see next chapter), the Montreal Process, and certification by a third party are mechanisms used to ensure that those benefits are sustained. Occasionally, incentives, such as selling carbon credits to forest owners are also used. So, although one person or company may own forest land, what they are allowed to do on forest land is influenced by society, both in their own country, as well as in others.

## **ARE WE MAKING PROGRESS?**

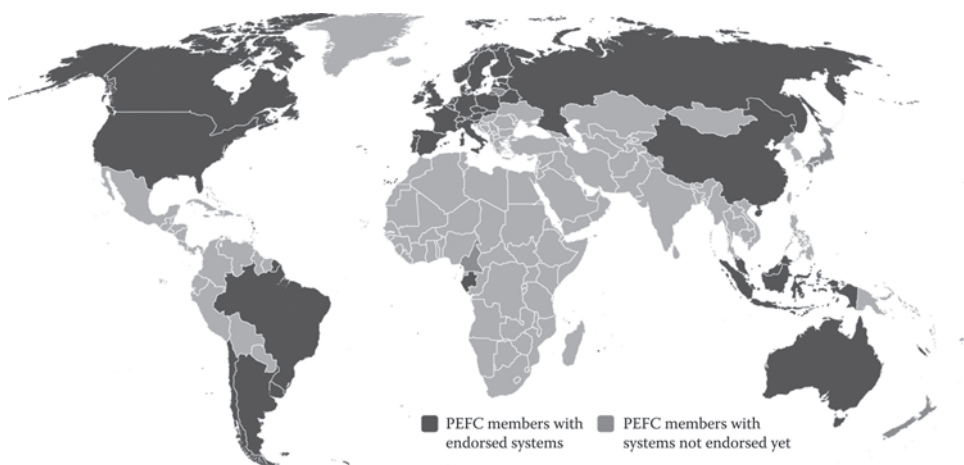
Twenty years after the Rio Summit, how have we done in sustaining Earth's resources for future generations? Rio +20 was a conference held in Rio de Janeiro 2012 to take stock of the progress made over the previous 20 years. A 17-page document, "The Zero Document," framed the discussion by calling for greater coordination around environmental economic policies, support for developing countries, greater use of technology, and a framework for global governance (Martella and Smaczniak 2013). In essence, this document, and the resulting discussion, had at its core the philosophy of not sliding back on any progress already made. A total of 283 statements, describing the desired future condition of the planet, were provided, and signatory countries reaffirmed their commitments to achieving these sustainability goals. The general sense of an assessment of progress was that change has been modest at best. However laudable the goals of the Rio Summit may be, the task of achieving sustainability among all countries is daunting, if not impossible. It is impossible for growth to continue indefinitely, given that resources are finite (Mace 2012). Uncertainty of future climates further confounds our prediction of resource limits, since climate may change resource availability, at least geographically, on the planet. To be successful and make clear progress over the next 20 years, Martella and Smaczniak (2013) identified several obstacles that must be overcome: (1) global environmental governance must be more cohesive and synergistic; (2) organizations need to more effectively coordinate their efforts; (3) resources available to implement strategies need to be more cost-effective; (4) nongovernmental organizations (NGOs) must be given more authority; and (5) NGOs and businesses need to be equal partners in developing

solutions. To date, the Rio Summit has been successful in raising awareness around issues and proposing solutions to improve the fate of future generations. Rio +20 confirmed that awareness. It will be up to participating countries, NGOs, educational institutions, and businesses to begin to work together to bring about true change that will lead to reduced population growth, a reduction in the rate of climate change, a higher quality of life for more individuals, and less disparity between the “haves” and the “have nots.” This is a very tall order for politicians who view successes in 2-, 4-, or 6-year intervals, depending on their position and country’s government. Sustainability can only be achieved by continued pressure and participation from the citizens of the planet, if we hope to achieve sustainability for future generations.

## FOREST CERTIFICATION

Third-party certification is one mechanism that is an outgrowth of the Montreal Process, and it is used around the world to help ensure that forests are managed sustainably and that biodiversity conservation is considered during management. Certification that forests are managed sustainably is based on an audit by an objective organization. In this process, forest management practices are evaluated against a set of standards by an external certification organization. Certification economically benefits landowners because, at least in theory, certified forest products yield a higher return on an investment, while assuring consumers that their purchase comes from a forest whose management meets certain standards (Washburn and Block 2001). Certification also provides an opportunity to shift land management behavior to be more consistent with biodiversity conservation goals, something unlikely to happen without policies or processes such as certification (Primmer and Karppinen 2010). Certification has two interconnected pieces: third-party certification of forest management activities that typically address some or all of the Montreal Process criteria and indicators and certification of chain of custody. Chain of custody is important, because, as wood products pass from stump to mill and mill to wholesaler and wholesaler to retailer, there is assurance to the consumer that the wood purchased did indeed come from a certified forest.

There is a variety of third-party certification organizations in the world, and over 125 million ha (>3% of the world’s forests) are certified by one or more organizations (Rametsteiner and Simula 2003). Over 90% of these certified forests are in North America and Europe (Figure 23.1). The goal of conserving biodiversity in tropical forests, using sustainable forest management principles, has largely failed, but voluntary certification on private lands in temperate forests has been quite



**FIGURE 23.1** Map of PEFC-certified forests in the world. (Map from PEFC. With permission.)

successful. The degree to which certification is effective in conserving biodiversity is unknown, but assumed to be significant in temperate regions, though perhaps dependent on the certifying body (National Wildlife Federation et al. 2001). In the United States, there are two primary certification organizations: The Forest Stewardship Council (FSC) and Sustainable Forestry Initiative (SFI). Canada (e.g., Canadian Standards Association), Europe (e.g., Pan European Forest Certification, PEFC), and other countries and regions (e.g., the Keurhout Foundation in Holland, and National Timber Certification Council in Malaysia) have additional certifying bodies (Rametsteiner and Simula 2003). The FSC originated in Europe and is active in 57 countries around the world, including the United States (Perera and Vlosky 2006). FSC is a nonprofit organization that evolved from the Earth Summit agreements. SFI was established by the American Forest and Paper Association (AF&PA). It is founded in a comprehensive system of principles, objectives, and performance measures developed to integrate both responsible environmental practices and sound business practices (Perera and Vlosky 2006; Sustainable Forestry Initiative 2006). SFI certification includes first- (self) and second- (SFI board) party audits, as well as independent third-party audits of conformance to standards (Perera and Vlosky 2006). Since its establishment, over 124 million acres of forestland in North America have undergone third-party SFI certification (SFI 2006).

Consumers seem willing to support green certified products in North America and Europe, so certification of forest lands in those countries continues to grow (Rametsteiner and Simula 2003). The cost to the landowner (of being audited to receive certification) is high, and cost-prohibitive for owners of small parcels. Since most forest land in eastern United States is in small parcels, most is not certified, except where innovative mechanisms such as woodland cooperatives are used to create an economy of scale (Barten et al. 2001, Figure 23.2).

When a landowner wishes to have her (or his) property certified, she (or he) makes a request to the certification body. The application for certification usually entails an application fee (or membership in an organization, e.g., American Forest and Paper Association). Auditors then visit the property and compare what they see in owner records and on the ground with the standards for certification set by the certifying body. For third-party certification, the auditors are independent of the company or organization being certified to avoid conflict of interest. Certification is given when the auditors' report indicates that there are no noncompliances. If there are instances of noncompliance, then the landowner is given a fixed amount of time to address the issues before certification would then be granted. Periodic or annual audits are required by the certifying body. The landowner must



**FIGURE 23.2** Owners of small forest tracts in Massachusetts have developed an economy of scale through development of the Massachusetts Woodlands Cooperative. The cooperative is certified by the Forest Stewardship Council. (Photo by Susan Campbell. With permission.)



not only pay the application fee but also the costs associated with the audit; hence, for small owners, the cost per certified forest acre can be prohibitively high.

Certification schemes can be broadly categorized into two groups: performance-based and process-based (Layton et al. 2002; Perera and Vlosky 2006). Performance-based standards define specific performance levels for various aspects of forest management; it is the framework used by FSC. Process-based schemes provide a systematic approach to developing, implementing, monitoring, and evaluating environmental policies, but they do not stipulate performance standards; it is the framework used by SFI (Layton et al. 2002; Perera and Vlosky 2006). The two certification processes differ in the ways that they consider the approach to ensure that biodiversity conservation and management of habitat for various wildlife species (Table 23.1). In both approaches, the metrics used to define compliance or noncompliance may be areas of concern for landowners contemplating certification; however, competition between FSC and SFI has led to changes that are bringing the two certification schemes closer together (Overdevest 2010).

Layton et al. (2002) reviewed the metrics that are used and approaches taken to certification, specifically those dealing with wildlife and biodiversity evaluations. They concluded that auditors often indicate that professional judgment is used as the basis for determining if a criterion or indicator is met. Since audit teams vary from owner to owner, the consistency of evaluation is

**TABLE 23.1**

**Comparison of FSC and SFI Certification Approaches Dealing with Protection of Biological Diversity and Management of Habitat for Various Wildlife Species**

Indicator	FSC	SFI
The status of forest-dependent species at risk of not maintaining viable breeding populations, as determined by legislation or scientific assessment.	Safeguards shall exist, which protect rare, threatened, and endangered species, and their habitats. Conservation zones and protection areas shall be established... Ecological functions and values shall be maintained intact, enhanced, or restored, including ... genetic, species, and ecosystem diversity...	Program participants shall apply knowledge gained through research, science, technology, and field experience, to manage wildlife habitat, and contribute to the conservation of biological diversity.
Biological diversity	Landowners are required to maintain, enhance, or restore the long-term integrity of natural habitats, ecological processes, soil, water, and stand development.	Each company develops its own policies, programs, and plans to contribute to the conservation of biological diversity and manage sites of ecological significance.
Stand age–class distribution	Forest owners and managers maintain or restore portions of the forest to the range and distribution of age classes of trees that result from processes that would naturally occur on the site.	Program participants shall have policies, programs and plans to promote habitat diversity at the stand and landscape-level.
Habitat diversity	A diversity of habitats for native species is protected, maintained, and/or enhanced.	Program participants shall apply knowledge gained through research, science, technology, and field experience, to manage wildlife habitat and contribute to the conservation of biological diversity.

*Source:* Based on Washburn, M., and N. Block. 2001. *Comparing Forest Management Certification Systems and the Montreal Process Criteria and Indicators*. Pinchot Institute, Washington, DC; the National Wildlife Federation, Natural Resources Council of Maine, and Environmental Advocates. 2001. *A Comparison of the American Forest & Paper Association's Sustainable Forestry Initiative and the Forest Stewardship Council's Certification System*. National Wildlife Federation, Washington, DC.



brought into question. Certification bodies have struggled with objective measures of performance, especially when evaluating indicators of biodiversity. Lindenmayer et al. (2000) proposed the use of structure-based indicators, such as structural complexity of stands and composition of plant species, connectivity, and heterogeneity in biodiversity conservation, and these may have utility in certification audits as well.

## EFFECTIVENESS OF CERTIFICATION

Internationally, certification is likely to be more successful where governments enforce forestry laws, provide financial incentives for certified forestry, provide land tenure security, and where forestry is profitable (Ebeling and Yasué 2009). Consequently, there are few developing countries where forest certification is likely to achieve widespread success (Ebeling and Yasué 2009). In the United States, Moore et al. (2012) found that forest certification was successful in causing change in forestry, environmental, social, and economic/system practices. Firms certified by FSC made more environmental/forest management changes, and firms certified by SFI made more economic/system changes (Moore et al. 2012).

How effective has certification been in conserving biodiversity? Apparently much depends on how certification is implemented. Elbakidze et al. (2011) compared indicators of biodiversity conservation between FSC-certified lands in Sweden and Russia and found that indicators of conservation spanned multiple spatial scales in Russia but were limited largely to stands in Sweden. So, although the standards for certification were the same between countries, implementation differed—and, hence, the potential for conserving biodiversity. Further, standards in certification, as well as in policies, set thresholds that land managers attempt to meet, thereby homogenizing the landscape, and stand elements, to threshold levels, rather than encouraging variability in representation of habitat elements (Johansson et al. 2013). For instance, if the threshold for snag retention was 10 snags per hectare, then falling below that threshold could risk noncompliance, but going above that threshold by very much may risk economic loss. Allowing variability in habitat elements over large areas may be more representative of the way in which natural systems function.

There are indications that independent audits provide an incentive for improving forest management consistent with the intent of the Montreal Process (Ramesteiner and Simula 2003). Gullison (2003) concluded that landowners that undergo FSC-certification are more likely to improve management with respect to the value of managed forests to biodiversity conservation. But the incentives associated with certification are not sufficient to attract most producers to seek certification, particularly in tropical countries where the costs of improving management to meet FSC guidelines are significantly greater than any market benefits they may receive (Gullison 2003). If FSC certification is to make greater inroads, particularly in tropical countries, significant investments will be needed, both to increase the benefits and reduce the costs of certification (Gullison 2003). Conservation investors will need to carefully consider the biodiversity benefits that will be generated from such investments versus the benefits generated from investing in more traditional approaches to biodiversity conservation (Gullison 2003); but a clear quantification of the impacts of green certification on conservation of biodiversity has not been completed. Nussbaum and Simula (2004) provided a series of case studies indicating potential benefits of certification to biodiversity conservation. They concluded that improved conservation of biodiversity appears to be a consistent benefit of certification, as evidenced by increased protection of representative ecosystems and rare, threatened or endangered species, but few data are available on which to base rigorous assessment. Loehle et al. (2005) evaluated associations between a set of landscape structure metrics that could be used in certification programs and bird species richness. Their work suggested that indicators addressing aspects of landscape structure may be useful in evaluating potential contributions to biodiversity conservation. Nonetheless, the extent to which certification has led to improved ecosystem functions, reduced risk to loss of biodiversity, or greater probability of persistence of endangered species is still unclear (Nussbaum and Simula 2004). Nonetheless,

certification represents a pragmatic international system of private governance that could lead to increased levels of biodiversity conservation. We should anticipate that the processes and standards will evolve and improve over time, and the disparate approaches currently used among certifying organizations will likely coalesce into a common framework. What is needed, however, is a monitoring system that clearly documents the gains and losses of biodiversity and ecosystem processes from certified and uncertified forests.

## **SUSTAINING AND MARKETING ECOSYSTEM SERVICES: HABITAT BANKING**

Habitat banking was defined by Caldecott and Dickie (2011) as a market-based system where credits from habitat creation or set-asides (beneficial actions) can be purchased to offset the loss of habitat from development or other human uses (environmental damage). Credits can be produced, purchased, and used, when habitat loss is imminent, or stored to be used later, as needed. An assumption is that the credits and debits are associated with the gain or loss of biodiversity, which is based on habitat losses or gains (Caldecott and Dickie 2011). For rare habitat types, an overriding concern of management or development is the net loss in the area or capability of the remaining habitat to support desired species. Consequently, wetlands protection acts are in place in many states and countries to ensure that no further net loss of wetlands occurs. This approach means that wetlands in urban areas along roadsides or other highly impacted areas receive protection, even though they may not be particularly functional, depending on the landscape context in which they exist. The same could be true of old-growth stands, savannahs, meadows, or other under-represented habitat types. As humans build houses, create roads, and install similar infrastructure that remove these habitat types, they face a legal dilemma. One solution to minimizing a net loss is to mitigate any loss by setting aside or creating protected habitat elsewhere, potentially in a more functional landscape. Of course, such a measure costs money (usually), so if a developer wishes to fill in a wetland along a roadside to build a mall, he could pay a landowner in a less impacted area to set aside wetlands, or create new wetlands, which would replace the wetlands lost through development. Habitat banking is a procedure used to achieve a no-net-loss policy or objective. Given the correct topography and soils, wetlands could be created and become functional within a few years or decades. Creating old-growth forests, however, could take centuries, so this approach does have limitations (Morris et al. 2006). It is for this reason that habitat creation or habitat offsets have been brought into question as being a viable way of allowing sustainable development with no net loss of rare habitat types (Morris et al. 2006).

Not all habitat types are equal in their ability to support a species of concern (or ecosystem of concern), and not all species are equally common or rare. Consequently, Resource Equivalency Methods (REMs) or Ecological Equivalency methods are used to estimate values associated with the losses and the values associated with the banked habitat, so that comparable habitat area and value is used in the habitat offset (Ozdemiroglu et al. 2009; Quétier and Lavorel 2011). Estimating the size of the compensation area can be done by dividing the present conservation value of the site to be developed, by the predicted future conservation value of a compensation area of the same size (Moilanen et al. 2009). As noted above, it may take decades or centuries for some habitat values to be realized, and there is a risk that the future values may be delayed or not realized at all. Moilanen et al. (2009) developed a method to take into consideration the risks associated with the uncertainty of achieving future values.

There are still questions regarding the effectiveness of habitat banking efforts that monitoring will be able to address over time; the approach is one that has wide appeal because it assigns a value to habitat that can then be traded, much as carbon credits and dollars can be exchanged. In fact, depending on the context of the habitat used in offsets or created, the value of the created habitat could be greater (or less) than the habitat lost to development. Bridging approaches between economics and the ecological sciences, holds promise for a more meaningful dialog between those who promote economic development and those who promote ecological protection.

## SUMMARY

Society demands increasingly more ecosystem services from forest lands, both public and private. International conferences and agreements have developed a set of principles defining sustainable forest management, and these principles have been included in certification protocols. Sustainable forest management certification is private governance of forest management, and provides the social license for managers to continue to manage forests. Enrollment in certification programs has grown dramatically since the 1990s, especially in temperate forests. The incentives to landowners for certification in tropical forests, especially in developing countries, are not apparent, especially given the high costs associated with certification. Although there appear to be benefits of certification to biodiversity conservation, based on changes in biodiversity indicators, long-term monitoring of certified and uncertified lands has yet to be conducted. Nonetheless, the generally accepted assumption is that certified forests should do more to conserve aspects of biodiversity than uncertified forests. Habitat offsets and habitat banking are approaches to help realize a no-net-loss of habitat due to development and other land uses that remove habitat for certain species. Approaches that ensure that the value of the habitat created (credits) is meaningfully exchanged for habitat lost (debits) have been developed, but monitoring is needed to assess assumptions of ecological equivalency, especially when values may take decades or centuries to be realized.

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# 24 Regulatory and Legal Considerations

One way of influencing change in the way forestry is practiced to benefit habitat for animals, or conservation of biodiversity, is through incentive programs such as certification (see previous chapter), tax relief, or compensation for ecosystem services through the form of easements, land purchases, purchases of carbon credits, or habitat banking (Pagiola et al. 2002). More typically, certain practices are prescribed by law. In the United States, laws are policies enacted by legislature and signed into law by an executive branch and enforced through a judicial branch of government. There are certain things that society values strongly enough to prescribe it. Keep species. Manage forests for sustained yield. Do not participate in trade of globally endangered species.

Some policies are set at local levels, such as counties, towns, and even neighborhoods. Zoning laws, building permits, and noise limitations are all set locally and enforced locally. The layers of policies, laws, regulatory agencies, and responsibilities regarding forest management and wildlife conservation, are at times overwhelming, especially to private forest land managers, and that is where the crux of the habitat management problem often lies. Wildlife are public resources whose habitat is most often controlled by private landowners with private property rights. It is relatively easy to envision an ecosystem management plan for a public property, whereby the outcomes of implementing the plan are a set of ecosystem services valued by society, using public land for the public good (Thomas et al. 2006). Private property owners have property rights, and, in some places, they may also have water and mineral rights, restricting what society can demand from their land (Bliss et al. 1997). So although society may say it wants active habitat management for a rare species on private lands, unless the land is deemed critical habitat for an endangered species, society cannot make the landowner do anything—unless there are laws.

## INTERNATIONAL LAWS AND AGREEMENTS

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) is an international agreement among governments designed to protect species worldwide. Its aim is to ensure that international trade in wild animals and plants does not threaten their survival (Mace and Lande 1991). CITES was drafted as a result of a resolution adopted in 1963 at a meeting of members of The World Conservation Union (IUCN) and signed in Washington DC in 1973 (Hutton and Dickson 2000). There are now 169 signatories. Today, it provides varying degrees of protection to more than 30,000 species of animals and plants, whether they are sold alive or sold as animal or plant parts (e.g., ivory). This resolution has no habitat provisions; however, several legislations in the United States authorize Congress to appropriate funds to aid in international efforts at habitat conservation. The International Environment Protection Act of 1983 and the Neotropical Migratory Bird Conservation Act of 2000 provide funds to other countries to benefit species of animals, particularly those which cross international boundaries (Elliott et al. 2005).

Each country has its own set of laws regarding conservation of wildlife, forests, and habitat. Laws in some countries are extensive and enforced rigorously. In some countries laws exist but are not consistently enforced. In all cases, political pressures can lead to variable interpretation of the laws.

Countries often take an issue-by-issue approach in their contributions to global environmental issues, each contributing to global governances, such as the management of tropical forests, or to address climate change (Long 2010). Private networks (e.g., in certification) are becoming

more important in global governance because they span international boundaries and bring consistent standards and guidelines to international issues (Long 2010). The Reduced Emissions of carbon dioxide caused by Deforestation and Degradation (REDD+) program is one private network approach to reducing greenhouse gas emissions from forest loss or certain types of management, while also promoting public values associated with biodiversity and human well-being (Long 2010). Consequently, we are seeing layers of laws and policies in which political boundaries are spanned by private networks such as forest certification and REDD+ (Phelps et al. 2010; Nguon and Kulakowski 2013). Understanding the laws and private network governance effects on landscape change, and the ability for landscapes to meet desired future conditions, is critical to development of a forest management plan. I use examples of the hierarchical layers of laws in the United States to illustrate the complexities facing habitat and forest managers, but similar layers and problems are found in other countries.

## NATIONAL LAWS

Since the beginning of the conservation debates in the United States between Gifford Pinchot and John Muir, legislation has been proposed that would influence how public and private lands are managed to provide public goods and services (Table 24.1). Many of the earliest legislative acts that affected private landowners had more to do with influencing harvest rates of commercially important species of trees and animals than with governing management on their lands (Bean and Rowland 1997). But as early as 1937, the Federal Aid in Wildlife Restoration Act (also known as the Pittman–Robertson Act) matched federal funds, collected from a tax on firearms, with state funds, to allow management of habitat to benefit many wildlife species (Oehler 2003). Although the original focus of these purchases was for game species, many nongame species benefited indirectly. Use of these funds for habitat management is now viewed more broadly to benefit hunted and nonhunted species in many states (Oehler 2003).

During the 1960s and 1970s, a suite of environmental laws emerged reflecting concern over sustained production of goods and services from public lands. The Multiple Use Sustained Yield Act, Endangered Species Act, National Environmental Policy Act, and National Forest Management Act represent pieces of legislation that continue to shape the way that federal lands are managed and continue to fuel the debates regarding priorities for federal lands (Hibbard and Madsen 2003). One of these, the Endangered Species Act, also has far-reaching influence on private lands. When a species is listed as threatened or endangered by the appropriate federal agency (usually the U.S. Fish and Wildlife Service), “taking” of individuals of protected species by a private citizen constitutes a violation of federal law (Sagoff 1997). The Endangered Species Act (ESA) requires the federal government to designate “critical habitat” for any species it lists under the ESA. Once designated, then any alteration of critical habitat that imperils one or more individuals of any protected species constitutes a violation of the law (Sagoff 1997), and the landowner can be prosecuted unless she (or he) has an incidental “take” permit (Smallwood 2000). The Habitat Conservation Plans (HCPs) are designed to provide no-net-loss of a species, while allowing private landowners the opportunity to continue managing their lands (see Chapter 18). The issue of controlling private property rights through federal law has met with considerable resistance, but because most habitat for many protected species occurs on private, not public, lands, the provision continues to represent a powerful tool for habitat protection and recovery. Unfortunately, there are no provisions for fair compensation of the property owner when ESA restricts harvest of trees for commercial gain (Innes et al. 1998). A federal compensation program, similar to what is used under the Conservation Reserve Program or Wildlife Habitat Incentives Program in the Farm Bill, would probably considerably ease the tension between private forest landowners and federal regulatory agencies.

The Endangered Species Act has been criticized on a variety of fronts. Some say that listing usually occurs too late to effectively recover many species (Neel et al. 2012). Others claim that the rigid framework of no-loss of species is inconsistent with the dynamic nature of ecosystems (Benson



**TABLE 24.1****Examples of US Laws That Influence the Ability of Public and Private Forest Land Managers to Provide Habitat for Animals or to Conserve Biodiversity**

Timber Protection Act of 1922	Protects timber on federal lands from fire, disease, and insects
Migratory Bird Conservation Act of 1929	Established procedures for acquisition, by purchase, rental or gift, of areas for migratory birds
Federal Aid in Wildlife Restoration Act of 1937	Provides federal aid to states for management and restoration of wildlife, including acquisition and improvement of wildlife habitat
Taylor Grazing Act of 1934	Governs grazing on public lands
Transfer of Certain Real Property for Wildlife Conservation Purposes Act of 1948	Real property that is no longer needed by a federal agency can be transferred to the Secretary of the Interior if the land has particular value for migratory birds, or to a state agency for other wildlife conservation purposes
Multiple Use Sustained Yield Act of 1960	Established purposes for the National Forest System, including outdoor recreation, range, timber, watershed, and fish and wildlife
Sikes Act of 1960	Provides for planning, development, and maintenance of fish and wildlife resources on military reservations
McIntire-Stennis Act of 1962	Authorized a formula fund for forest research in all states
Wilderness Act of 1964	Considers all interior roadless lands >5000 acres for inclusion into the National Wilderness Preservation System
National Environmental Policy Act of 1969	Ensures that environmental values are given appropriate consideration, along with economic and technical considerations
Alaska Native Claims Settlement Act of 1971	Authorized Alaska Natives to select and receive title to 44 million acres of public land in Alaska
Endangered Species Act of 1973	Provided for the conservation of ecosystems upon which threatened and endangered species of fish, wildlife, and plants depend, and implemented the CITES agreement
Federal Land Policy and Management Act of 1976	Constitutes the “organic act” for the Bureau of Land Management and governs most uses of the public lands
National Forest Management Act of 1976	Constitutes the “organic act” for the Forest Service
Public Rangeland Improvement Act of 1978	Improved conditions of public rangelands for grazing, wildlife habitat, and other uses
Cooperative Forestry Assistance Act of 1978	Provides for cooperation on forest management issues with non-Federal forest landowners
Renewable Resources Extension Act of 1978	Increased extension emphasis in renewable resources, including fish, wildlife, and water resources, on private forest and range lands
Alaska National Interest Lands Conservation Act of 1980	Provided 79.54 million acres of refuge land in Alaska
Fish and Wildlife Conservation Act 1980	Provides financial and technical assistance to the states for the development, revision, and implementation of conservation plans and programs for nongame fish and wildlife
Tax Deductions for Conservation Easements of 1980	Stipulates that a taxpayer may take a deduction for a “qualified real property interest” contributed to a charitable organization exclusively for conservation purposes protected in perpetuity
International Environment Protection Act of 1983	Assists other countries in wildlife and plant protection efforts in order to preserve biological diversity
Food Security Act of 1985	Contains several provisions that contribute to wetland conservation including the Swampbuster Provision, the Conservation Reserve Program (CRP) and the Wetland Reserve Program (WRP)
Cave Resources Protection Act of 1988	Provides for management and protection of caves and their resources on federal lands

*(Continued)*

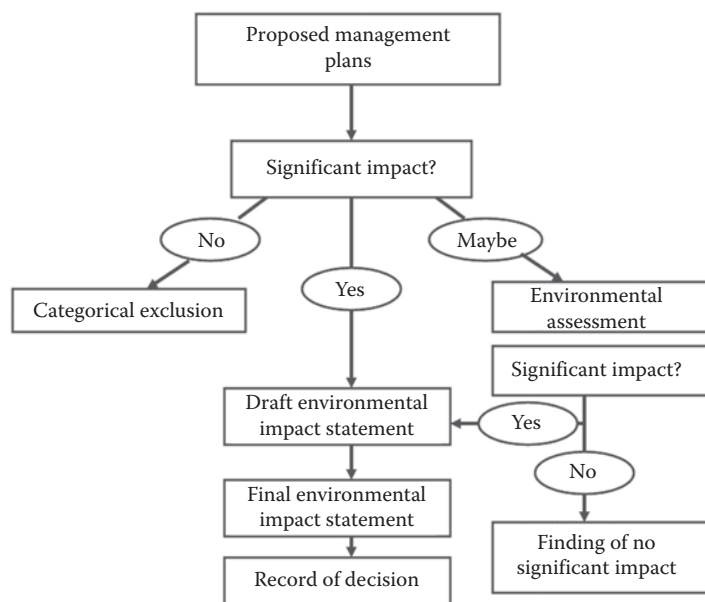
**TABLE 24.1 (Continued)****Examples of US Laws That Influence the Ability of Public and Private Forest Land Managers to Provide Habitat for Animals or to Conserve Biodiversity**

Emergency Wetlands Resources Act of 1986	Authorized the purchase of wetlands with Land and Water Conservation Fund monies
Land Remote Sensing Policy Act of 1992	Directs that Landsat 7 acquire high priority land remote-sensing data in order to meet the needs of the U.S. Global Change Research Program
Hawaii Tropical Forest Recovery Act of 1992	Provides grants, contracts, and cooperative agreements to promote sound management and conservation of tropical forests in the United States
Partnerships for Wildlife Act of 1992	Authorizes grants to the states for programs and projects to conserve nongame species
National Wildlife Refuge Acts	Various acts to establish specific refuges
Neotropical Migratory Bird Conservation Act of 2000	Provides grants to countries in Latin America and the Caribbean, and within the United States, for the conservation of neotropical migratory birds that winter south of the border and summer in North America
State Wildlife Grants program of 2001	Supports cost-effective conservation aimed at preventing wildlife from becoming endangered
Healthy Forests Restoration Act of 2003	Reduces the risk of catastrophic fire to communities, helps save the lives of firefighters and citizens, and protects threatened and endangered species

*Source:* Extracted from the US Fish and Wildlife Service Laws Digest, Washington, DC.

2012). Add, on top of these concerns, the effects of climate change (Povilitis and Suckling 2010), and it is no wonder that ESA continues to be questioned. Nonetheless, it is the strongest law we have currently in the United States to minimize risk of loss of species.

Another powerful federal law, the National Environmental Policy Act (NEPA), is designed to document and prevent unwanted adverse environmental effects of federal actions. Responsibilities of federal agencies, under the National Environmental Policy Act, include scoping of the management action, scientific analysis, social and economic analysis, securing public input, media relations, regulatory analysis, and litigation (Auer et al. 2011). There are instances where this Act can also influence management actions, not only on federal lands, but on nonfederal lands as well (Moser 2000). For instance, if federal funds are used in a habitat improvement project on state or private lands, then NEPA processes would have to be followed (Figure 24.1). It is during this process that environmental impact statements are developed to compare the environmental consequences of proposed and alternative land management actions. The process allows for public involvement and participation, creating a more transparent mechanism for ensuring that public concerns are addressed. The process is also costly, both financially and in timing, often taking a year or more before a record of decision is made. Those involved with NEPA processes can view the process differently. Stern et al. (2010) found that in the U.S. Forest Service, members of interdisciplinary teams conducting NEPA analyses emphasized minimizing negative environmental and social impacts, satisfying stakeholders, and avoiding litigation. Forest Service employees who make decisions based on NEPA processes placed more emphasis on efficiency and less on minimizing impacts (Stern et al. 2010). Advisory employees emphasized transparency in the process (Stern et al. 2010). Consequently, there have been efforts to change or “streamline” NEPA and more rapidly allow management actions to proceed where timing is critical (Smythe and Isber 2003). For instance, the Healthy Forests Restoration Act included provisions to this effect, enabling a more rapid response with regards to salvage logging to reduce risks associated with reburning of forests. Such provisions meet opposition, however, when streamlined procedures are perceived to increase risk of environmental degradation.



**FIGURE 24.1** Flow chart of the NEPA process required of federal land managers and land managers using federal funds in the United States.

Cumulative effects analysis (CEA) is also required as a part of NEPA. Agencies charged with conducting a CEA often encounter challenges in defining the appropriate spatial and temporal scales of analysis, lack of understanding regarding the purpose of a CEA, lack of data, and communicating the results of the analysis (Schultz 2012). Agencies working with other land managers, public and private, could facilitate the effectiveness of CEAs.

A recent federal program has provided new opportunities for biodiversity conservation among the states in the United States. The State Wildlife Grants program of 2001 provides federal dollars to every state and territory to support cost-effective conservation aimed at preventing wildlife from becoming endangered. Funds are allocated to every state according to a formula based on each state's size and population. Each state and territory is responsible for developing a statewide wildlife action plan to identify species, and their habitats, of greatest conservation need, and outline the steps that would be needed to conserve all species and natural areas. This program is supported by a coalition of more than 3000 organizations that are members of Teaming with Wildlife. Such broad-based grassroots support for this initiative has helped to ensure its success. Once in place, a national support program may facilitate cross-boundary conservation needs and enhance collaborative efforts among states (Meretsky et al. 2012).

Federal funding is provided to land management agencies specifically to support habitat acquisition and management. National Wildlife Refuges represent the best example of a federal program of habitat management, but programs within NRCS, such as the Wildlife Habitat Incentives Program (WHIP), provide cost-share habitat improvement support for private landowners as well. Habitat management is also an integral part of land management plans developed in the Forest Service, Bureau of Land Management, National Park Service, and military installations. All tolled, many millions of federal dollars are spent annually on habitat management projects on public and private lands.

Under the 2008 Farm Bill, before state forestry agencies can receive federal funding, an assessment of state and private forest lands must be conducted, including priorities for “enhancing ... biological diversity ... wildlife, wildlife corridors, and wildlife habitat” (Kostyack et al. 2011). Further,

these assessments and strategies are expected to address how forests are to be managed to “adapt to global climate change” (Kostyack et al. 2011).

Federal funding also has been provided to support research and outreach activities to states. The McIntire–Stennis Act and the Renewable Resources Extension Act provide formula funds to land-grant universities, to support forest research and outreach activities, respectively (Thompson and Bullard 2004). These funds are often used to support research and outreach dealing with forest management, wildlife habitat management, and forest biodiversity conservation. This effort supplements efforts within the research programs of the various federal agencies (U.S. Fish and Wildlife Service, U.S.G.S. Biological Resources Division, U.S. EPA Office of Research and Development) and outreach arms of federal agencies (e.g., Forest Service State and Private Forestry program). Private industry also supports research in forest wildlife habitat management, largely through the forest industry-supported organization, National Council on Air and Stream Improvement (NCASI). Information from all of these research efforts is extended to public and private land managers, and to the general public, to enable more effective management of land and water resources.

## STATE LAWS

State laws, generally, have focused on traditional game management approaches, with a lower priority given to protection of nongame species and biodiversity in general, but that trend seems to be changing. There is increasingly more pressure being placed on states to accept responsibility for biodiversity conservation and protection of endangered species. Most states do not have funding mechanisms adequate to address these issues. Nonetheless, many states do have state laws protecting endangered species and policies that influence management of habitat for these and other species (George et al. 1998). I provide a few examples of state wildlife laws that influence habitat management; these examples were largely extracted from Musgrave and Stein’s (1993) overview of state wildlife laws.

Massachusetts identifies significant habitats for rare, threatened, and endangered species under its state endangered species act. Since most land in the state is privately owned, there are significant impacts on private forest-land owners. Before a landowner can alter forest in a “significant habitat” (with some exceptions), the landowner must provide to the regulatory agency (MassWildlife) the following:

- A complete description of the project
- Alternatives to the proposed project
- Impacts of the proposed project on the subject species
- Plans for protection of the subject species and mitigation measures to be taken to offset these impacts
- A description of potential economic effects of the proposed project on the landowner and the community

A permit to proceed with a project would be granted only upon finding that the proposed action will not reduce the viability of the significant habitat to support the population of the subject species. Massachusetts also has developed a landowner incentives program for habitat improvement (Figure 24.2). So, rather than simply regulating private landowner actions, the state encourages landowners to apply to participate in a cost-share program that would benefit certain species. Regulations and incentives represent the carrot and stick approaches to maintaining biodiversity.

Louisiana also has policies regarding habitat improvement, and protection for certain species. The state is authorized to contract with a private landowner for use of lands for at least 25 years, to allow establishment of wildlife management areas, with a tax-free incentive to the landowner. In addition, the Louisiana Department of Wildlife and Fisheries established a “Louisiana Acres for Wildlife” program, which provides landowner assistance in habitat management. Biologists provide planting stock, guidance for habitat management actions, and habitat evaluation surveys. The



**FIGURE 24.2** Grasslands and early successional woodlands managed for birds using Wildlife Habitat Incentives Program grant dollars, Arcadia Wildlife Sanctuary, Easthampton, Massachusetts.

program is voluntary, and participants must have  $>0.4$  ha of land and agree that management continues for at least one year. To date, the focus of this program has been on game species production, but clearly there are benefits for nongame species as well. Many other states, such as Oregon and Kentucky, have similar programs, allowing the state to contract with landowners to establish wildlife refuges or management areas. Most states are also authorized to purchase lands to protect habitat, though, until recently, these purchases have been primarily to allow hunting of game species.

Minnesota's regulations include provisions for improvement or acquisition of "critical natural habitat." Identification of what constitutes a critical natural habitat is based on

- Significance as existing or potential habitat for fish and wildlife, and providing fish- and wildlife-oriented recreation.
- Significance to maintain or enhance native plant, fish, or wildlife species, designated as endangered or threatened.
- Presence of native ecological communities that are now uncommon or diminishing.
- Significance to protect or enhance natural features within, or contiguous to, natural areas, including fish spawning areas, wildlife management areas, scientific and natural areas, riparian habitat, and fish and wildlife management projects.

Purchase of lands for these purposes is restricted by availability of state funds, so areas must be prioritized, based on the above factors. States such as Massachusetts must prioritize purchases, because urbanization is causing purchase prices for important patches of habitat to be excessively high.

The degree to which various states have begun to shift from a game management focus on their lands, to an ecosystem management approach, is high variable. Because most state wildlife agencies are funded primarily from hunting and fishing license sales, expenditures on nongame species can easily be brought into question by their constituents. However, most state biologists have realized that "keeping all the pieces" is a correct course of action that will benefit both game and nongame species (Jacobson et al. 2010). Indeed, reform of the wildlife conservation institution was proposed by Jacobson et al. (2010) who recommended the following changes to maintain relevancy to constituents: "broad-based funding, trustee-based governance, multidisciplinary science as the basis of recommendations from professional staff, and involvement of diverse stakeholders and partners."

Many states also have forest practices acts or policies restricting forest management actions to minimize adverse effects of forest harvest and management on various ecosystem services. Initially, these were largely focused on water quality and habitat for fish (e.g., Oregon and Washington Forest Practices Acts) but have expanded to include headwater and vernal pool amphibians (e.g., Massachusetts) and upland species (snag and green tree requirements in Oregon). Massachusetts, for instance, requires that a cutting plan be submitted to a state service forester, who reviews it for compliance with state laws and also ensures that the plan is reviewed for presence of state-listed rare species by biologists in the Natural Heritage Program of MassWildlife. Such an approach may then lead to additional restrictions on activities, based on the possible presence of a rare species and its habitat needs. The level of restrictions and level of review, both prior to and following a harvest, varies considerably from state to state. Often separate state agencies within the same state are involved in forest practices law enforcement and habitat management enforcement. Such an approach can, at times, pit one set of natural resources professionals against another. Approaches that allow both groups to identify a desired future condition (DFC), and mutually agree on approaches to work toward that DFC, would often be more effective. In addition, increased levels of coordination of habitat management among states can increase effectiveness for many species, because animals usually do not respond to political boundaries. Animals cross state lines during migration and dispersal. As climate change places additional pressures on geographic ranges, cross-boundary movements may become even more important. Interstate coordination of habitat acquisition, leasing, management, and prioritization may improve the effectiveness of approaches among participating states (Meretsky et al. 2012).

### MUNICIPAL POLICIES

Local governments are leaders in the development of policies that influence local production of ecosystem services, because local communities are the direct beneficiaries of these regulations (Hirokawa 2011). Habitat is one of those regulated ecosystem services and is considered explicitly in the policies enacted in some counties and towns. These sorts of policies vary tremendously and are often associated with land-use laws such as zoning laws, urban growth boundary designations, and building permit laws. For instance, Wisconsin County Forests are governed by the County Forest Law requiring that these forests be managed for multiple uses, such as forest products, recreation, wildlife habitat, and watershed protection. Twenty-eight northern Wisconsin counties own and manage nearly one million hectares of county forest lands. Counties are required to develop comprehensive land-use plans for each forest. Eau Claire, Wisconsin, manages a county forest specifically as habitat for wildlife “common to Wisconsin.” This plan recognizes that each species requires different forest conditions ranging from recently disturbed to old growth. The county also pays particular attention to endangered species on the property (Karner Blue Butterfly, a federally listed endangered species) and has developed a habitat conservation plan for the Karner Blue Butterfly.

In the northeastern United States, county governance is not as important as town governance. Towns have policies that influence habitat management and availability, and these vary widely from town to town. Few explicitly consider wildlife habitat. The town of Milan, New York, however, participates in an effort to “... protect the integrity and value of Milan’s natural areas, and protect the town’s watershed and significant biological resources.” The townspeople recognize that the diverse natural resources of the town are particularly vulnerable to adverse impacts associated with development and sprawl. They are conducting habitat assessments to provide baseline information and improve the town planners’ ability to protect significant biological resources in the face of increasing development pressure. The habitat assessment program takes into consideration a number of environmental impacts associated with development, including direct loss of habitat and reduced populations, habitat fragmentation and adverse edge effects, increasing effects of invasive species, degraded water quality, and increased pollution. Such a detailed approach to planning is becoming



increasingly common, especially in communities where land-use change is rapid, and local communities become increasingly concerned about the long-term adverse effects of such changes.

There is also a growing movement toward more community-based conservation approaches, in which local communities benefit from conservation-associated efforts (Kothari et al. 2013). Such approaches can be particularly effective in cultures where locally derived values are achievable in concert with conservation of biodiversity. Community forests in northern New England in the United States were managed based on shared governance, inclusive participation, shared distribution of benefits, and effectiveness in achieving conservation goals (Lyman et al. 2013). Community forests have been a part of Mexican culture for many years. Known as *ejidos*, community forest lands comprise 80% of Mexico's forests. These community lands are used for forest products as well as agricultural production. Community managed forests tend to have lower annual deforestation rates than protected forests (Porter-Bolland et al. 2012). In situations where the local community's social and economic needs are considered, community forests have the capacity to offer protection to many aspects of biodiversity in forests managed to meet human needs, as well as the needs of other species. Similar examples of community forests can be found in Costa Rica and Nepal (Kothari et al. 2013).

## POLICY ANALYSIS

The layers upon layers of policies that apply to forests—across, within, and among countries around the world—can easily overwhelm managers of forest lands. Policies are many, complex, and frequently changing. It is a challenge to develop forest management plans that follow current policies. By having foresters work with biologists to develop plans, it is more likely that pertinent forest practices, rules, and wildlife laws, will be followed. Assuming that all policies are followed (clearly an incorrect assumption!) will they achieve society's goals? Will the application of the federal, state, and local policies to forest parcels, across a mixed-ownership landscape, lead to better biodiversity protection in 10 years? 100 years? Will forest management continue to be economically viable? In many instances, the answer to these questions from policy makers is, "We think so." But in nearly every case, it is unclear how effective any single policy might be in achieving its goal, and even less certain when considered in the presence of a set of additional policies, some with entirely different objectives.

Uncertainty of achieving desired results from policy implementation is related to the complexity or messiness of the policy problem. Lackey (2006) indicated that messy ecological policy problems often share several qualities: (1) complexity—innumerable options and trade-offs; (2) polarization—clashes between competing values; (3) winners and losers—for each policy choice, some will clearly benefit, some will be harmed, and the consequences for others is uncertain; (4) delayed consequences—no immediate "fix" and the benefits, if any, of painful concessions will often not be evident for decades; (5) decision distortion—advocates often appeal to strongly held values and distort or hide the real policy choices and their consequences; (6) national versus regional conflict—national (or international) priorities often differ substantially from those at the local or regional level; and (7) ambiguous role for science—science is often not pivotal in evaluating policy options, but science often ends up serving inappropriately as a surrogate for debates over values and preferences.

Lackey (2006) then proposed nine axioms that are typical of most current ecological policy problems that must be considered during policy development and debate: (1) the policy and political dynamic is a zero-sum game; (2) the distribution of benefits and costs is more important than the ratio of total benefits to total costs; (3) the most politically viable policy choice spreads the benefits to a broad majority, with the costs limited to a narrow minority of the population; (4) potential losers are usually more assertive and vocal than potential winners and are, therefore, disproportionately important in decision making; (5) many advocates will cloak their arguments as science to mask their personal policy preferences; (6) even with complete and accurate scientific information, most policy issues remain divisive; (7) demonizing policy advocates supporting competing policy options

is often more effective than presenting rigorous analytical arguments; (8) if something can be measured accurately and with confidence, it is probably not particularly relevant in decision making; and (9) the meaning of words matters greatly, and arguments over their precise meaning are often surrogates for debates over values.

Although scientific information is just one element of complex political deliberations in a democracy (Lackey 2006), it can—and has—influenced policy direction. One approach to considering the potential effectiveness of policies is to conduct a *policy analysis*, an organized projection of how implementation of the policy over space and time might possibly affect the resources valued by society. It is most useful to think of policies as hypotheses. The ones enacted now are being tested now, especially if we monitor the results. But there are two primary questions that should be asked by policy makers in particular, and society in general: (1) Are current policies achieving our goals? and (2) Would an alternative policy be more effective in achieving our goals?

Regional policy analyses that examine the tradeoffs among policies on economic, social, and ecological values are becoming more common. Analyses have been conducted in Canada (Hauer et al. 2010), Finland (Primmer 2011), and the United States (Spies et al. 2007; Rittenhouse and Rissman 2012), among others. Spies et al. (2007) used the Oregon Coast Range as a case study to examine how forest policies might affect various measures of biodiversity over a multiownership region. The dominant federal policy, the Northwest Forest Plan, is designed to increase habitat for species associated with late successional forests on federal lands (FEMAT 1993). Using projections of forest landscape development under pertinent federal and state policies, the forests of the region are expected to move toward the historical range of variation for most age classes of forest (Spies et al. 2007) and improve habitat for associated late successional species. But habitat recovery for some species may take >100 years; a very long-term policy would need to be in place! In addition, the current set of policies is expected to result in declines in diverse early successional forests and in hardwood forests in the region, with predicted declines in habitat availability for several species associated with these types of forests (McComb et al. 2007). These unanticipated changes in forest composition and structure may cause policy makers to develop alternative policies to ensure that such changes do not lead to a landscape in which additional species are placed at risk. Projections such as these represent “thought-experiments” that can provide policy makers insights into the possible outcomes of forest management policies (Oreskes 1997).

## HOW DECISIONS IN THE UNITED STATES INFLUENCE HABITAT IN THE WORLD

Many nations of the world have complex forest-land management policies that are enforced to ensure that ecosystem services demanded by society are provided into the foreseeable future. But the degree to which various countries have international policies addressing biodiversity conservation (the United States does), reducing the effect of climate change (the United States does not), or trade in endangered species varies tremendously. And policies enacted in the United States that conserve biodiversity can have significant impacts on biodiversity conservation in other countries.

The world has become a smaller place with global transportation and economies. All countries aspire to a high standard of living. They want a clean environment, a safe place to live, adequate healthcare, and, to the degree possible, stability and security in the lives of its citizens. Some countries can afford such a set of conditions (the “haves”), while others struggle to provide the basic services needed for human survival (the “have-nots”), and, of course, these extremes represent endpoints of a spectrum. The “haves” not only can afford to meet the expectations of most of its citizens most of the time, they also often have large appetites for resources such as energy, water, and timber. If the timber does not come from the “haves” then it will come from the “have-nots,” the countries least able to afford enforcement of environmental laws. Where the “have-not” countries also represent areas of high biodiversity (e.g., some tropical countries), the “have” countries conserve their biodiversity while threatening it in other countries and worldwide. How do we solve this problem? Altruism, a human quality of placing others before yourself, is a human behavior

that could be espoused by both citizens and the politicians who represent them. Realizing that for the greater good of both humanity and biodiversity, some societies must have less so that others can have more, would begin to address these problems of inequity. Use less energy, less water, less wood, make fewer babies, accept stable economies, and do not expect them to continue to grow, grow the food you eat, grow the wood that you use. These things have not happened yet. It is not clear what it might take for altruism to be more broadly expressed in the world.

## **CASE STUDY: THE ENDANGERED SPECIES ACT AND CLIMATE CHANGE**

Nearly every climate scientist in the world agrees that the Earth's climate is changing at a rate that is unprecedented, at least as presented in the geologic record. The most noticeable changes thus far have been at the poles and in high alpine areas. The image of the polar bear on an ice flow that is shrinking, with nothing but ocean surrounding it, true or not, has captured the attention of many people. But what happens to those species found at the poles, on mountain tops, along shorelines (with sea-level rise), and in other ecosystems, where the changing climate will lead to novel conditions that these organisms have never faced? To call them threatened seems an easy case to make, unless they can move to new sites, adapt to new conditions, or be moved to sites habitable by them (Kostyack et al. 2011).

In the United States, these ecosystems at risk of loss or significant change due to climate change could be considered critical habitat under the Endangered Species Act. Indeed, by 2008, 60% of the ESA Recovery Plans referenced climate change (Polvilitis and Suckling 2010). Considering climate change in Habitat Conservation Plans has also gotten some attention. Bernazzani et al. (2012) provided nine recommendations for integrating climate change into the HCP process: (1) identify species at risk from climate change; (2) explore new strategies for reserve design; (3) increase emphasis on corridors, linkages, and connectivity; (4) develop anticipatory adaptation measures; (5) manage for diversity; (6) consider assisted migration; (7) include climate change in scenarios of water management; (8) develop future-oriented management actions; and (9) increase linkages between the conservation strategy and adaptive management/monitoring programs. Other scientists have offered suggestions for taking proactive measures to mitigate the potential adverse effects of climate change in forests.

Spies et al. (2010) suggested that forest-stand and landscape structure could be managed to increase ecological resistance and resilience; species and genotypes could be actively moved to new locations; and multiownership planning could be implemented over regions, to make adaptive actions more effective. But remarkably little progress has been made in addressing the effects of climate change on endangered and threatened species. Taking actions to address climate change effects is more often discussed than implemented, and institutional barriers within agencies can limit what can be accomplished (Jantarasami et al. 2010). Archie et al. (2012) cited lack of information at relevant scales, budget constraints, lack of specific agency direction, and lack of useful information, to be the most common barriers to adaption planning. Budget constraints and perceived lack of importance to the public were obstacles to implementation of adaptation projects (Archie et al. 2012).

So what can we do to minimize the risk of losing species when little management has been implemented, and the rate of change in climates is rapid? Scientists need to be careful to cite the risks to species from climate change, without becoming viewed as policy advocates, if we are likely to see a change in policies that will significantly reduce the rate of accumulation of greenhouse gases. Without objective policy analyses that demonstrate risks associated with different courses of action, we may stay in policy debates, while options decrease due to the rapidity of change around the globe.

## **SUMMARY**

The myriad interacting policies at the federal, state, and municipal levels influence the amount and distribution of habitat for various species, from forest stands to global forests. Policies have

begun to evolve at all levels from a strong focus on habitat for game species to one of biodiversity conservation. Increased levels of coordination in policy formulation across political lines (international, interstate, cross-county) may be necessary to ensure biodiversity conservation in forested landscapes. Analyses of the future implications of forest policies can allow policy makers to better understand the time needed to see the results of their policies, if there are unanticipated consequences from their policies, and if alternative policies might better achieve their goals.

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# 25 Should I Manage the Forest?

Most of this book has addressed active management: Taking actions to achieve habitat goals for a species, a community or contributing to biodiversity conservation, and doing that while also considering the potential for providing wood and nontimber products for people. Many people do not feel compelled to manage their forests. The millions of small private landowners in the United States and Canada may own their forests for reasons other than timber, woodcock, or deer. They just like to have a forest. To walk through it, see it, sit in it, and listen to the birds in it (regardless of species). Except when a disturbance happens, forests change slowly. They provide a place that evokes stability, security, and spirituality. For people who view forests in this way, management is not only unnecessary, it is disruptive and evokes instability, insecurity, and flies in the face of personal spirituality. They may extend those feelings to all forests, regardless of who owns them, because, after all, we are merely temporary tenants on the Earth, regardless of what we pay for the pieces we use. So, for many people doing nothing is a perfectly acceptable management decision (Kittredge and Kittredge 1998). Doing nothing is indeed a management decision.

Other landowners wish to have certain animal species to hunt, wood to sell or burn, leaf colors to enjoy, as well as clean water to drink. They choose a more active management decision. Sometimes, this involves regeneration methods that other landowners and neighbors do not find acceptable. NIMBY—not in my back yard—is a phrase that captures the essence of the disagreement between the two philosophies (Shindler et al. 2002). Some people want to restore a forest to a previous, “better” condition, with “better” being something valued by the landowner, land manager, neighbor, or society. Restoration represents an example of a philosophy associated with active management, just as does “doing nothing” and “commodity production.” These philosophies are all points on a spectrum of values and behaviors. Restoration often represents the middle ground—the production of a desired future condition that is neither utilitarian nor “let nature take its course.” I use restoration as an example of a management philosophy that we can examine more closely.

## WHAT DOES RESTORATION MEAN?

Restoration is a noble goal—but restored to what? What is a reasonable target? What is our reference condition? Is it an ecological condition? Cultural? Both? Is it even possible to now “restore” a system that has been changed markedly due to recent intense human activities, climate change, invasive species, or toxic compounds? Higgs (2005) distinguished between the terms “ecological restoration” and “restoration ecology.” *Restoration ecology* is the suite of scientific practices that constitute an emergent subdiscipline of ecology (Higgs 2005). *Ecological restoration* uses practices that represent restoration ecology as well as human and natural sciences, politics, technologies, economic factors, and cultural dimensions (Higgs 2005). Maintaining a broader approach to restoration requires respect for knowledge in addition to science, and especially the recognition of a morality that is beyond the scope of science alone. Some have made an analogy between ecological restoration and human health (Schaefer 2006). Schaefer (2006) suggested that portraying the human body as a metaphor of a natural ecosystem can be useful in identifying the breadth of strategies used to restore the “natural” environment. The use of science in restoration is analogous to the use of technology to address a disease through drugs or surgery, while the cultural and ecological interactions are analogous to restoring health to the human body through a broader holistic/preventative approach to cultivate the mind–body connection (Schaefer 2006). The integration of the tools, concepts, and examples described in this book, in conjunction with cultural values and individual

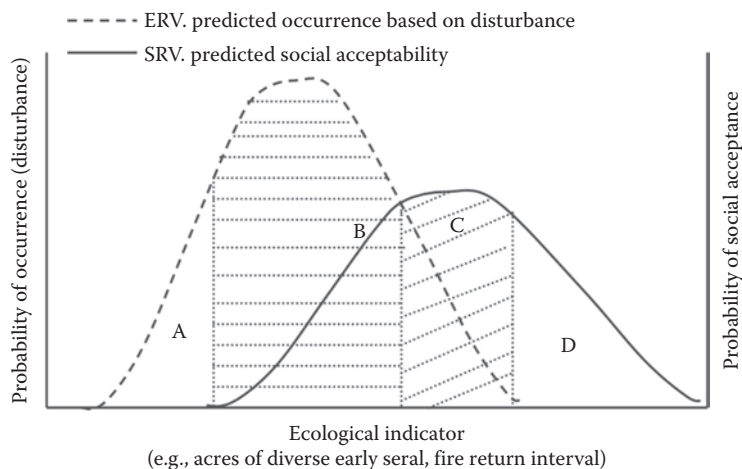
beliefs, can be used to set restoration goals and objectives. It is not just about science, but about the people who use science and other tools to restore a system to meet their needs. "Restoration" to one person may be quite different from what another person pictures in his or her mind. Coming to agreement on what "restoration" means, to both the individuals and the community, is central to setting restoration goals where many people are affected by a management decision. Agreement is difficult, especially where individual property rights enter into the discussions. Some proportion, at times a majority, of the citizens in an area (e.g., the watershed councils that have formed in states across the United States) can agree to share a core set of goals and move the area toward that goal, recognizing that not all landowners will share those goals.

## HUMAN REQUIREMENTS AS CONSTRAINTS ON GOALS

The historical range of variability (HRV) has often been proposed as a concept that can be used by forest land managers to guide conservation of ecosystem functions and biodiversity (Morgan et al. 1994; Landres et al. 1999; Swetnam et al. 1999; Keane et al. 2009). The rationale for use of the historical range of variability in certain ecosystem properties is that biodiversity was assumed to persist, albeit with fluctuations in populations, over thousands of years of disturbance and recovery. Further, the concept assumes that, as contemporary conditions depart from historical processes and states, the risk of losing species, both known and unknown, increases.

The following ideas reflect a conceptual framework—developed by Dr. Sally Duncan, Dr. Norm Johnson and me—for understanding the range of variability in a way that considers not only HRV as traditionally defined, but also likely future ranges of variability, given ecological processes and conditions that society finds acceptable (Duncan et al. 2010). Authors have often used a probability distribution to reflect the probability of occurrence of certain ecological states or indicators, over some reasonably long period of time (e.g., Wimberly and Ohmann 2004), in order to represent an HRV. The probabilities are derived from the outcome of physical disturbances that have occurred over this timeframe, and typically included Native American disturbances, but not usually contemporary disturbances produced since European colonization. Clearly humans have been, are, and will be, forces of disturbance and recovery in forested landscapes. The range of variability expressed at any time reflects human and biophysical drivers of landscape change, while recognizing both temporal- and spatial-scale dependency. A key component of our understanding of the expression of variability in an ecosystem is the effect of a social range of variability (SRV). SRV reflects the cultural mores that collectively influence the range of conditions that society finds acceptable. Integration of the SRVs of the past, present, and future, with disturbance and recovery during the past, present, and future, interact to define both historical and future ranges of variability in ecosystem indicators: an ecological range of variability (ERV) and a social range of variability (SRV) (Figure 25.1). The ecological range of variability is a product of ecosystem disturbance and recovery from any cause: physical, biological, or human. The patterns of disturbance and recovery, over space and through time, presumably produced a set of conditions that supported biodiversity historically and will influence biodiversity into the future. Societies have been more or less effective in influencing disturbance frequencies, intensities, sizes, and durations over time, with classic examples of recent short-term influences being flood-control dams, fire control, and harvest of old-growth trees. But historical influences of humans on ecological conditions also are apparent. The Kalapuya Americans burned the Willamette Valley regularly, maintaining a savannah system (Boag 1992). Native Americans also contributed to the fire frequencies of the longleaf pine savannahs of the Deep South (Denevan 1992).

The probability distribution of an ecological indicator arising from past disturbances (Figure 25.1) is directly influenced by the social range of variability, which can be described as a probability distribution of ecosystem conditions, which are socially acceptable. For example, rivers channelized or dammed for irrigation and flood control (socially acceptable, at varying levels through time) no longer nourish wetlands during and after flooding (a natural disturbance) (Shafroth et al.



**FIGURE 25.1** Conceptual diagram reflecting the interaction between the ecological range of variability resulting from disturbances and recovery in ecosystems and the social range of variability reflecting social acceptability of ecological conditions. It is the tension between these two forces that produce the range of conditions that we see expressed across forested landscape over time.

2002). Suppressed wildfires (socially desirable) can no longer regulate levels of insect and disease populations (natural disturbance) (McCullough et al. 1998). Thus, the ERV and SRV are inextricably linked. Even such disturbances as hurricanes and ice storms, today, may have their frequency affected by “desired” human activity across the planet, which is influencing our climate (Knutson and Tuleya 1999).

Historically, the full range of the probability distribution—in an ecological indicator, including the part that may be socially unacceptable—may have been observed simply because human societies had insufficient influence on biophysical disturbance regimes. Earlier societies could not limit the occurrence of some unacceptable conditions driven largely by biophysical forces (e.g., stand replacement wild fire). However, given the size of the current human populations, and their wealth and technological power, human societies today are capable of exerting significant pressure on the disturbance-based probability distribution, thereby altering possible future outcomes/trajectories. As the SRV departs from the ERV, social pressures can change drivers of ecosystem disturbance, and limit the expression of the ERV (bolded line in Figure 25.1). The more disparate the ERV and SRV, the greater is the potential for society to influence the expression of the ERV. This disparity is often reflected in the various management options considered for an area and the policies guiding the direction of land management. That disparity, or tension between ERV and SRV, will wax and wane depending on changes in social values, as well as changes in nonhuman disturbance dynamics.

Given that human systems and ecosystems are inextricably linked, how can we be sure that decisions made today will not have unanticipated negative consequences in our future or in our children’s futures? Science can provide the facts, but people also hold beliefs. Facts can be proven, beliefs cannot. It is through the dynamic combination of knowledge with evolving beliefs that change in societal goals occurs. Many events and experiences, both biophysical and social, affect beliefs and knowledge—hence the ever-changing nature of societal goals.

## DEVELOPING A PERSONAL MANAGEMENT PHILOSOPHY

As you go through your professional career you will use a variety of tools to make decisions: scientific data, belief, intuition, and stakeholder opinions, among others. All of these tools pass through

the filter of your personal philosophy. Your core values and your beliefs give rise to your behaviors, using all of these tools. Two people using the same tools to make a decision, but having different management philosophies, will often arrive at different decisions. And these differences are not always apparent to those making the decisions. The differences are often questioned by those holding different philosophies, and oftentimes in public settings. It is wise to be prepared to explain your personal philosophy to others; not to impose it on them, but merely to explain it clearly and concisely. Although those with different philosophies may never agree with you, they might understand you, thereby making dialogue possible. It is also important to keep in mind that our personal philosophies usually evolve throughout our lives. Checking in on your personal philosophy periodically through introspection is also prudent when faced with making daily management decisions. In addition, as you move into supervisory positions, you might consider encouraging management teams of all kinds, and at all levels, to similarly reassess their philosophies and share them with each other.

Below I provide a glimpse into my personal philosophy, not to impose it on you, not to try to convince you that it is correct, but to make you think. Some things that I have written you may not agree with, and some may make you angry. If you do get angry, that is fine if it makes you think about why you have that reaction, and how your philosophy differs from mine. Use my philosophy as a springboard to your own.

## OUR PLACE ON EARTH

Humans are a species, and as such we have habitat needs to allow us to persist on this planet. We are simply one of millions of species on this Earth, sharing space, energy, and time together. In my view, resources were not placed on this Earth for us to use by some omniscient, omnipotent, or omni-everything being. We just are. Our life arose from the resources on this Earth around our parents; the material bits of us will return to the resources of the Earth; and our actions during our brief lives form the legacy that we leave to other humans and other species. Each of us is a blip in time and space—or rather seven billion blips, each trying to survive, each using resources that could be used by other people, or other species, on this planet. Each blip arising from the Earth, each returning to the Earth. The Earth is our source, and as far as we know, it is the only source of any life for many light years in any direction. It is prudent to treat it as we would our own home and share it not only with those other individuals and species that are with us now, but also with those that will come after us. In my view, we have a moral obligation to consider the effects of our seven billion lives on the other species with which we share this planet; to not pollute it; to not overuse it; to not poop in our own petri dish. To recognize that we are a part of it, not separate, and how we make collective decisions affects the lives of other people, and other species.

## LIVING SIMPLY AND SUSTAINABLY

Energy can be changed from one form to another, but it cannot be created or destroyed. The total amount of energy in the universe is constant, merely changing from one form to another. The laws of thermodynamics apply to humans and all other life forms. From a practical standpoint, there is a solar constant so energy input is a zero-sum game. Allocating more energy to human needs and desires leaves less energy for other organisms. How do we redistribute energy on Earth to allow coexistence with other organisms? Reducing population growth is an obvious first step. The carrying capacity for humans on Earth is likely around 12 billion (Cohen 1995), but what is the carrying capacity of the Earth for humans while ensuring the coexistence with other forms of life? Have we surpassed it already? Quite likely. Granted that technology and scientific advancement can provide us with a marginal gain in our ability to support humans and other species, but we are faced with the fact that there is only so much energy to be allocated to all species and organisms. Technology is simply fussing with the edges, fine-tuning the allocations. We will reach a point of diminishing returns on technological advancements to increasing human carrying capacity (Czech 2003).

If we were to seriously start now in reducing human populations, there will be lag time to population reduction. Consequently, reducing energy consumption is another step. Our lives are based on growth economies. Consumption is good; it spurs economic growth. For how long? If money is indeed a surrogate for energy, and there is a solar constant, then how can energy consumption continually increase? It cannot. Accepting stable economies rather than continually growing economies is something that contemporary societies must begin to grapple with or the differences in the quality of life between the “haves” and the “have nots” will continue to increase, leading to greater and greater political instability (Briggs and Weissbecker 2011).

Even if we reduce populations, control consumption, live more simply, and reduce carbon emissions and other pollutants on our Earth, will it lead to continued coexistence with other species? Much depends on not only how much of the Earth’s renewable resources we consume, but where and when. Even doing nothing comes at a cost. The location, frequency, and the intensity of resource-use matter to our own long-term well-being, and the well-being of other species. There are limits to the resistance and resilience of the ecosystems in which we all live. As we use the Earth to meet our own needs, it is important to remember that it must be given the time to recover from those uses, to allow ecosystems to express resilience to both human and natural disturbances.

### LEAVING THE WORLD A BETTER PLACE

We all have our own ideas of “better.” Much of how various people define “better” has to do with their timeframe. Better today, this week, next year, or the next generation? Taking the long view, and assuming that most people, most of the time, will want to leave the world a better place for future generations, then it becomes clearer that we cannot have it all now and in the future too. We could live in a way that allows us to be happy and prosper and also allows those who come after us to live lives as good or better as ours. We can provide that next generation with the management options they would need to develop and implement their own definitions for “sustainable.” Leaving such a suitable legacy for our children takes commitment and discipline and will mean that societies with the most will have to be more satisfied with not having even more (Figure 25.2). The insatiable appetites for resources from rich countries pass the burden of production and loss of sustainability on to other societies. If we enter the global arena of natural resources management as an equal, sharing the benefits and the costs with other societies, then I suspect that the image of the “haves” would improve significantly among the “have nots.” This would not only provide us



**FIGURE 25.2** Management philosophies often involve insurance that there are always intergenerational opportunities for use of wildlife and forest resources.



with an opportunity to live in greater harmony than we do now, it would also provide all children of the future the options and opportunities that many do not enjoy now, including the opportunity to live on a diverse living planet.

## SUMMARY

Management of the Earth's resources to meet human needs and provide habitat for various species is dependant on the values and beliefs of people. We all approach management with the tools of science, beliefs, opinions, and culture, and pass those tools through the filter of our own philosophy. Defining a philosophy of forest and habitat management is important as you interact with others holding, oftentimes, different views. Developing and articulating your philosophy of management is something that should be done at the outset of your career, and revisited periodically, as your personal philosophy evolves throughout your life.

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# Appendix 1: Common and Scientific Names of Species Mentioned in the Text

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## Mammals

<i>Alces alces</i>	Moose
<i>Aplodontia rufa</i>	Mountain beaver
<i>Arborimus albipes</i>	White-footed vole
<i>Arborimus longicaudus</i>	Red tree vole
<i>Bettongia penicillata</i>	Woylie
<i>Bison bison</i>	Bison
<i>Blarina brevicauda</i>	Short-tailed shrew
<i>Clethrionomys gapperi</i>	Gapper's red-backed vole
<i>Canis latrans</i>	Coyote
<i>Canis lupus</i>	Wolf
<i>Castor canadensis</i>	Beaver
<i>Cervus elaphus</i>	Elk
<i>Clethrionomys californicus</i>	Western (California) red-backed vole
<i>Didelphis virginiana</i>	Virginia opossum
<i>Elephas maximus</i>	Asian elephant
<i>Erithizon dorsatum</i>	Porcupine
<i>Felis concolor</i>	Florida panther, cougar
<i>Felis lynx</i>	Lynx
<i>Glaucomys sabrinus</i>	Northern flying squirrel
<i>Glaucomys volans</i>	Southern flying squirrel
<i>Gulo gulo</i>	Wolverine
<i>Lepus americanus</i>	Snowshoe hare
<i>Lepus</i> spp.	Hares
<i>Lontra canadensis</i>	River otter
<i>Marmota monax</i>	Woodchuck
<i>Martes americana</i>	American marten
<i>Martes pennanti</i>	Fisher
<i>Mephitis mephitis</i>	Striped skunk
<i>Microtus oregoni</i>	Creeping vole
<i>Microtus</i> spp.	Voles
<i>Mustela vison</i>	Mink
<i>Myotis</i> spp.	Myotis bats
<i>Myotis volans</i>	Long-legged bat
<i>Neotoma</i> spp.	Woodrats
<i>Odocoileus hemionus</i>	Mule deer
<i>Odocoileus hemionus sitkensis</i>	Sitka black-tailed deer
<i>Odocoileus</i> spp.	Deer
<i>Odocoileus virginianus</i>	White-tailed deer
<i>Ondatra zibethicus</i>	Muskrat

<i>Peromyscus leucopus</i>	White-footed mice
<i>Peromyscus maniculatus</i>	Deer mouse
<i>Procyon lotor</i>	Raccoon
<i>Rangifer tarandus</i>	Caribou
<i>Rattus norvegicus</i>	Norway rat
<i>S. trowbridgii</i>	Trowbridge's shrew
<i>Sciurus carolinensis</i>	Gray squirrel
<i>Sciurus niger</i>	Fox squirrel
<i>Sorex pacificus</i>	Pacific shrew
<i>Sorex palustris</i>	Water shrew
<i>Sorex vagrans</i>	Vagrant shrew
<i>Sus scrofa</i>	Domestic pig
<i>Sylvilagus floridanus</i>	Cottontail rabbit
<i>Tamias striatus</i>	Eastern chipmunk
<i>Tamiasciurus douglasii</i>	Douglas squirrel
<i>Thomomys</i> spp.	Pocket gophers
<i>Trichosurus vulpecula</i>	Brush-tail possum
<i>Ursus americanus</i>	Black bear
<i>Ursus arctos horribilis</i>	Grizzly bear
<i>Ursus</i> spp.	Bears
<i>Vulpes vulpes</i>	Red fox
<i>Zapus</i> spp.	Jumping mice
<i>Zapus trinotatus</i>	Pacific jumping mouse

#### Birds

<i>Accipiter gentilis</i>	Northern goshawk
<i>Actitis macularia</i>	Spotted sandpiper
<i>Aimophila aestivalis</i>	Bachman's sparrow
<i>Aix sponsa</i>	Wood duck
<i>Amazona vittata</i>	Puerto Rican parrot
<i>Anas rubripes</i>	American black duck
<i>Ardea herodias</i>	Great blue heron
<i>Athene cunicularia</i>	Burrowing owl
<i>Bonasa umbellus</i>	Ruffed grouse
<i>Brachyramphus marmoratus</i>	Marbled murrelet
<i>Bubo virginianus</i>	Great horned owl
<i>Bucephala clangula</i>	Common goldeneye
<i>Buteo jamaicensis</i>	Red-tailed hawk
<i>Buteo swainsoni</i>	Swainson's hawk
<i>Campephilus principalis</i>	Ivory-billed woodpecker
<i>Catharus bicknelli</i>	Bicknell's thrush
<i>Catharus guttatus</i>	Hermit thrush
<i>Catharus ustulatus</i>	Swainson's thrush
<i>Certhia americana</i>	Brown creeper
<i>Ceryle alcyon</i>	Belted kingfisher
<i>Chaetura</i> spp.	Swifts
<i>Charadrius vociferous</i>	Killdeer
<i>Chaetura pelagica</i>	Chimney swift
<i>Chaetura vauxi</i>	Vaux's swift
<i>Ciconia ciconia</i>	White stork
<i>Cinclus mexicanus</i>	American dipper
<i>Coccyzus americanus</i>	Yellow-billed cuckoo
<i>Colaptes auratus</i>	Common flicker

*Colinus virginianus*  
*Columba fasciata*  
*Columba livia*  
*Contopus cooperi*  
*Corvus corax*  
*Dolichonyx oryzivorus*  
*Dryocopus pileatus*  
*Ectopistes migratorius*  
*Empidonax hammondi*  
*Falci pennis canadensis*  
*Falco peregrinus*  
*Gavia immer*  
*Haliaeetus leucocephalus*  
*Histrionicus histrionicus*  
*Hylocichla mustelina*  
*Ixoreus naevius*  
*Junco hyemalis*  
*Melanerpes formicivorus*  
*Mniotilta varia*  
*Myiopsitta monachus*  
*Parus bicolor*  
*Passer domesticus*  
*Phasianus colchicus*  
*Picoides borealis*  
*Poecile atricapilla*  
*Scolopax minor*  
*Seiurus aurocapillus*  
*Setophaga discolor*  
*Setophaga pensylvanica*  
*Setophaga petechia*  
*Setophaga pinus*  
*Setophaga virens*  
*Sialia sialis*  
*Sitta pusilla*  
*Spinus tristis*  
*Strix occidentalis caurina*  
*Strix varia*  
*Sturnella magna*  
*Sturnus vulgaris*  
*Thryothorus ludovicianus*  
*Troglodytes aedon*  
*Troglodytes troglodytes*  
*Turdus migratorius*  
*Vermivora celata*  
*Vireo olivaceus*  
*Wilsonia canadensis*

Bobwhite quail  
 Band-tailed pigeon  
 Rock dove  
 Olive-sided flycatcher  
 Common raven  
 Bobolink  
 Pileated woodpecker  
 Passenger pigeon  
 Hammond's flycatcher  
 Spruce grouse  
 Peregrine falcon  
 Common loon  
 Bald eagle  
 Harlequin duck  
 Wood thrush  
 Varied thrush  
 Dark-eyed junco  
 Acorn woodpecker  
 Black-and-white warbler  
 Monk parakeet  
 Tufted titmouse  
 House sparrow  
 Ring-necked pheasant  
 Red-cockaded woodpecker  
 Black-capped chickadee  
 Woodcock  
 Ovenbird  
 Prairie warbler  
 Chestnut-sided warbler  
 Yellow warbler  
 Pine warbler  
 Black-throated green warbler  
 Eastern bluebird  
 Brown-headed nuthatch  
 American goldfinch  
 Northern spotted owl  
 Barred owl  
 Eastern meadowlark  
 European starling  
 Carolina wren  
 House wren  
 Winter wren  
 American robin  
 Orange-crowned warbler  
 Red-eyed vireo  
 Canada warbler

### Amphibians

*Ambystoma maculatum*  
*Ambystoma opacum*  
*Aneides ferreus*  
*Ascaphus truei*  
*Dicamptodon tenebrosus*

Spotted salamander  
 Marbled salamander  
 Clouded salamander  
 Tailed frog  
 Pacific giant salamander

*Ensatina eschscholtzii*  
*Gyrinophilus porphyriticus*  
*Hyla* sp.  
*Ochlerotatus triseriatus*  
*Plethodon cinereus*  
*Plethodon dunni*  
*Plethodon* spp.  
*Plethodon stormi*  
*Plethodon welleri*  
*Rana aurora draytonii*  
*Rana cascadae*  
*Rana catesbeiana*  
*Rana septentrionalis*  
*Rana sylvatica*  
*Rhyacotriton variegatus*  
*Taricha granulosa*  
*Zenaida macroura*

*Agkistrodon contortrix*  
*Chelydra serpentina*  
*Clemmys marmorata*  
*Crotalus* spp.  
*Crotalus viridis*  
*Elaphe obsoleta*  
*Gopherus polyphemus*  
*Opheodrys aestivus*  
*Sceloporus occidentalis*  
*Terrapene carolina*

*Adelges tsugae*  
*Blattella* spp.  
*Camponotus* spp.  
*Chrysomela confluenta*  
*Dendroctonus* spp.  
*Dendroctonus frontalis*  
*Lycæides melissa samuelis*  
*Lymantria dispar*

*Carassi carassius*  
*Oncorhynchus kisutch*  
*Oncorhynchus clarki*  
*Salmo* spp.  
*Salvelinus fontinalis*

*Abies balsamea*  
*Abies grandis*  
*Acer macrophyllum*  
*Acer pensylvanicum*  
*Acer platanoides*  
*Acer rubrum*

Ensatina salamander  
 Spring salamander  
 Tree frog  
 Tree hole mosquito  
 Red-back salamander  
 Dunn's salamander  
 Slimy salamander  
 Siskiyou Mountains salamander  
 Weller's salamander  
 California red-legged frog  
 Cascades frog  
 Bullfrog  
 Mink frog  
 Wood frog  
 Southern torrent salamander  
 Rough-skinned newts  
 Mourning dove

### Reptiles

Northern copperhead  
 Snapping turtle  
 Western pond turtle  
 Rattlesnake  
 Prairie rattlesnake  
 Rat snake  
 Gopher tortoise  
 Rough green snake  
 Western fence lizard  
 Box turtle

### Insects

Hemlock wooly adelgid  
 Cockroaches  
 Carpenter ant  
 Leaf beetles  
 Bark beetles  
 Southern pine beetle  
 Karner blue butterfly  
 Gypsy moth

### Fish

Carp  
 Coho salmon  
 Cutthroat trout  
 Trout  
 Brook trout

### Plants

Balsam fir  
 Grand fir  
 Bigleaf maple  
 Striped maple  
 Norway maple  
 Red maple

<i>Acer saccharum</i>	Sugar maple
<i>Acer saccharinum</i>	Silver maple
<i>Acer spicatum</i>	Mountain maple
<i>Acer</i> spp.	Maples
<i>Ailanthus altissima</i>	Tree of Heaven
<i>Alnus rubra</i>	Red alder
<i>Amelanchier</i> spp.	Serviceberries
<i>Arbutus menziesii</i>	Pacific madrone
<i>Berberis</i> spp.	Barberries
<i>Betula alleghaniensis</i>	Yellow birch
<i>Betula lenta</i>	Black birch
<i>Betula papyrifera</i>	White birch
<i>Betula populifolia</i>	Gray birch
<i>Betula</i> spp.	Birches
<i>Brachypodium sylvaticum</i>	False brome
<i>Carya ovata</i>	Shagbark hickory
<i>Carya</i> spp.	Hickories
<i>Castanea dentata</i>	American chestnut
<i>Cirsium</i> spp.	Thistles
<i>Chamaecyparis lawsoniana</i>	Port-Orford cedar
<i>Cornus florida</i>	Flowering dogwood
<i>Cornus</i> spp.	Dogwoods
<i>Corylus cornuta</i>	Hazlenut
<i>Crataegus</i> spp.	Hawthorns
<i>Cytisus scoparius</i>	Scotch broom
<i>Elaeagnus umbellata</i>	Autumn olive
<i>Endothia parasitica</i>	Chestnut blight fungus
<i>Fagus grandifolia</i>	American Beech
<i>Fomes pini</i>	Red heart disease
<i>Galium</i> spp.	Bedstraws
<i>Ilex</i> spp.	Hollies
<i>Ilex verticillata</i>	Winterberry
<i>Ilex vomitoria</i>	Yaupon
<i>Kalmia latifolia</i>	Mountain laurel
<i>Juglans</i> spp.	Walnuts
<i>Liquidambar styraciflua</i>	Sweetgum
<i>Liriodendron tulipifera</i>	Yellow-poplar
<i>Magnolia fraseri</i>	Fraser magnolia
<i>Melaleuca quinquenervia</i>	Australian paperbark tree
<i>Nyssa aquatica</i>	Water tupelo
<i>Nyssa sylvatica</i>	Blackgum
<i>Phaeocryptopus gaeumannii</i>	Swiss needle cast
<i>Phytophthora lateralis</i>	Root rot
<i>Picea glauca</i>	White spruce
<i>Picea mariana</i>	Black spruce
<i>Picea rubens</i>	Red spruce
<i>Picea sitchensis</i>	Sitka spruce
<i>Picea</i> spp.	Spruces
<i>Pinus banksiana</i>	Jack pine
<i>Pinus contorta</i>	Lodgepole pine
<i>Pinus echinata</i>	Shortleaf pine
<i>Pinus elliotii</i>	Slash pine

<i>Pinus jeffreyi</i>	Jeffrey pine
<i>Pinus palustris</i>	Longleaf pine
<i>Pinus ponderosa</i>	Ponderosa pine
<i>Pinus resinosa</i>	Red pine
<i>Pinus</i> spp.	Pines
<i>Pinus taeda</i>	Loblolly pine
<i>Populus deltoides</i>	Eastern cottonwood
<i>Populus trichocarpa</i>	Black cottonwood
<i>Populus</i> spp.	Cottonwoods
<i>Populus</i> spp.	Aspens
<i>Populus tremuloides</i>	Quaking aspen
<i>Prunus pensylvanica</i>	Pin cherry
<i>Prunus</i> spp.	Cherries
<i>Pseudotsuga menziesii</i>	Douglas-fir
<i>Quercus alba</i>	White oak
<i>Quercus garryana</i>	Oregon white oak
<i>Quercus nigra</i>	Water oak
<i>Quercus palustris</i>	Pin oak
<i>Quercus</i> spp.	Oaks
<i>Quercus velutina</i>	Black oaks
<i>Quercus alba</i>	White oak
<i>Quercus bicolor</i>	Swamp white oak
<i>Quercus coccinea</i>	Scarlet oak
<i>Quercus prinus</i>	Chestnut oak
<i>Quercus rubra</i>	Northern red oak
<i>Rosa multiflora</i>	Multiflora rose
<i>Rubus spectabilis</i>	Salmonberry
<i>Rubus</i> spp.	Brambles, raspberries, blackberries
<i>Salix</i> spp.	Willows
<i>Taxodium distichum</i>	Baldcypress
<i>Thuja plicata</i>	Western redcedar
<i>Tsuga canadensis</i>	Eastern hemlock
<i>Tsuga heterophylla</i>	Western hemlock
<i>Viburnum acerifolium</i>	Maple-leaf viburnum
<i>Viburnum alnifolium</i>	Hobblebush
<i>Viburnum</i> spp.	Viburnums
<i>Viburnum trilobum</i>	High-bush cranberry

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## Appendix 2: Glossary

**Active adaptive management:** Management is treated as a hypothesis to be tested using monitoring data

**Adaptive management:** A process of continual improvement in management using monitoring data to refine plans

**Advance regeneration:** Seedlings and saplings present in the stand prior to a disturbance that releases them

**Aerial photographs:** A capture of the reflectance of items on the Earth's surface on a photographic film

**Allele:** Expression of a gene

**Allochthonous material:** Leaves, needles, and plant parts that fall into a water body

**Alluvial:** Downstream movement of soils

**Artificial regeneration:** Planting seedlings or seeds usually at a particular spacing to establish a new stand

**Barrier:** An intervening patch type with a low probability of survival

**Basal area:** Cross-sectional area of all trees on a hectare or acre at 1.4 m above ground

**Broadcasting:** Extrapolating data to other units of space outside of the scope of inference

**Brood:** A cohort of young birds

**Brood parasite:** Birds that reproduce by laying their eggs in the nests of other birds

**Browse:** Herbivore consumption of woody plants

**Carrying capacity:** A point in population growth where births equal deaths and further population growth is limited

**Chain of custody:** A process that assures the consumer that wood products came from a certified forest

**Clearcut:** A regeneration method in which all or most trees are removed to allow establishment of a new cohort of trees

**Codominant:** Trees in an even-aged stand receiving full sunlight from above and comprising the main canopy layer

**Community:** An assemblages of populations over space and through time

**Composition:** The types or classes of features in an areas, such as species of plants and types of soils

**Connectivity:** The degree to which the landscape facilitates or impedes movement among habitat patches

**Context:** An area beyond the extent that we are not managing but it affects the function of our landscape

**Core:** The interior of a patch

**Corridor:** An intervening patch type with a high probability of survival

**Critical habitat:** Specific areas and habitat elements essential to the conservation of species listed under the U.S. Endangered Species Act

**Crown classes:** Differentiation of trees into classes in response to growth rates and competition in an even-aged stand

**Cutting cycle:** Period of time between harvests when some trees of all tree diameters in an uneven-aged stand are cut

**Decomposition pathway:** An energy web passing through decomposers

**Deferred rotation:** Also known as a clearcut with reserves; retains some trees through two rotations

**Demographic stochasticity:** The variability represented in vital rates owing to fluctuations in survival and reproduction

**Desired future condition:** A description of the structure and composition of a stand or landscape that you wish to achieve

**Diameter-limit cutting:** Cutting of all the trees above some minimum diameter during each cutting cycle in an uneven-aged stand

**Digestible energy:** That portion of food that can be used by an animal for energy and nutrients

**Dominant crown class:** Uppermost trees in an even-aged stand receiving sunlight from above and from the sides

**Dynamic carrying capacity:** Changing carrying capacity due to fluctuations in resource availability

**Dynamic corridor:** A corridor that “floats” across the landscape over time to provide connectivity at all times

**Ecological restoration:** Uses of practices of restoration ecology as well as human and natural sciences, politics, technologies, economic factors, and cultural dimensions

**Ecosystem management:** A management approach designed to increase the likelihood that it will be socially sustainable

**Ecosystem services:** Services provided by ecosystems to meet society’s needs, including but not restricted to commodities

**Ectotherm:** A species that receives most of its body heat from the surrounding environment

**Edge associates:** Species that find the best quality habitat where there is access to required resources in two or more vegetation patch types

**Edge density:** Edge length per unit area

**Edge specialist:** A species likely to only occur where edges between two or more vegetative patch types exist

**Effect size:** The difference (or slope) that you could detect given your sample size, sampling error, and the probability of making an error when rejecting a null hypothesis

**Effectiveness monitoring:** Monitoring designed to determine whether habitat elements, populations, or processes are responding as expected and effectively achieving management goals

**Endotherm:** A species that generates its own body heat

**Environmental stochasticity:** Uncertain environmental events that influence population vital rates

**Establishment cut:** Second step in a shelterwood regeneration method to release trees to produce seeds and to provide growing space for regeneration

**Eutrophic system:** Nutrient-rich aquatic system

**Extent:** The outer bounds of the landscape over which we are managing resources

**Extinction vortex:** Accelerated population declines irreversibly leading to extinction

**Fecundity:** Number of young produced per female over a given time period

**Filter approach:** An approach to biodiversity conservation that employs coarse-, meso-, and fine-filter management strategies

**First-order selection:** Selection of a geographic range by a species

**Fledgling:** A bird that successfully leaves the nest

**Forecasting:** Predicting trends into the future, based on past trends

**Forest interior species:** A species that avoids edges and uses the core of a patch

**Forest structure:** The physical architecture of a forest in three dimensions

**Forest type:** Forest community dominated by representative tree species

**Founder effect:** Low genetic variation often seen in a newly established population

**Fourth-order selection:** Selection of specific food and cover resources acquired from the patches used by the individual within its home range

**Genetic bottleneck:** Marked decline in a population resulting in loss of alleles

**Genetic drift:** Some alleles may dominate in small populations by chance alone

**Grain:** The smallest unit of space in a landscape that we identify and use in an assessment or management plan

**Grazing:** Herbivore consumption of herbaceous plants

- Group selection:** Creation of small openings in a stand to establish patches of regeneration an uneven-aged stand
- Guild:** A group of species that share common nesting or feeding resources
- Habitat:** The set of resources necessary to support a population over space and through time
- Habitat conservation plan:** A plan designed to offset any harmful effects of a proposed activity on endangered or threatened species allowing issuance of an incidental take permit
- Habitat element:** Piece of a forest important to many species, such as vertical structure, dead wood, tree size, plant species, and forage
- Habitat fragmentation:** A process whereby a habitat for a species is progressively subdivided into smaller, geometrically more complex, and more isolated fragments
- Habitat generalist:** A species that can use a broad suite of food and cover resources
- Habitat selection:** A set of complex behaviors that each species has evolved to ensure fitness in a population
- Habitat specialist:** Species that use a narrow set of resources
- Habitat types:** Vegetation type or other discrete class of the environment that is associated with some species
- Hard mast:** Hard fruits such as nuts and acorns
- Harvesting systems:** The means of removing the trees from the site and to a landing during forest management
- Heuristic:** Use of models to teach us something about the system
- Home range:** Area that an individual (or pair of individuals) uses to acquire the resources that it needs to survive and reproduce
- Human commensal:** A species that typically is associated with humans
- Hyporheic zone:** Subsurface saturated sediments along the stream bottom
- Ideal despotic distribution:** A distribution of individuals reflecting high individual fitness in the highest quality patches at lower than expected densities caused by territoriality
- Ideal free distribution:** A distribution of individuals reflecting the freedom of each individual to choose the patch that will provide the greatest energy or other required resources
- Implementation monitoring:** Measurements that document compliance with a stand prescription or management plan
- Incidental take permit:** A permit issued by the USFWS to allow activities that might incidentally harm (or “take”) species listed as endangered or threatened under the Endangered Species Act
- Indicator species:** Species that are assumed to be surrogates for other species having similar resource needs
- Individual tree selection:** Removal of one or a few trees from a location in the stand to create a canopy gap to allow tree regeneration to occur
- Induced edge:** Edge between two patch types of different successional condition
- Inherent edge:** Edge formed by differences in the floristic composition of two patches
- Intermediate crown class:** Trees in an even-aged stand receiving partial sunlight from above
- Intra-riparian gradients:** Continuum of conditions from the headwaters to the confluence with larger water bodies
- Intrinsic rate of natural increase:** Each species’ potential for population increase
- Lambda:** The population parameter used to estimate population change
- Landscape:** A complex mosaic of interacting patches
- Logistic growth:** As resources become limiting, population growth becomes asymptotic
- Longevity:** The age at death of the average animal in a population
- Marsh:** Wetlands dominated by nonwoody vegetation
- Matrix:** The landscape patch type within which focal patches are embedded
- Matrix management:** Managing the matrix condition to be made more permeable to dispersing organisms

- Metapopulation:** A population distributed among smaller, interacting sub-populations that contribute to overall population persistence
- Mortality rate:** The number of animals that die per unit of time (usually 1 year) divided by the number of animals alive at the beginning of the time period
- Natality:** The number of young individuals born or hatched per unit of time
- Natural catastrophe:** Extreme case of environmental uncertainty such as hurricanes, fires, and epizootics that can cause massive changes in vital rates
- Natural cavity:** Tree hole resulting from fungal decay
- Natural regeneration:** Stand regeneration from seedling establishment or sprouting following the disturbance
- Neotropical migratory bird:** Birds that nest in the northern hemisphere but migrate to the tropics during the winter
- Oligotrophic:** Nutrient-poor aquatic systems
- Orographic effects:** As air is moved over mountains, it increases in elevation, cools and moisture precipitates
- Orthophoto maps:** Aerial photos corrected for distortion and usually with topographic information superimposed
- Overwood removal:** Final step in a shelterwood regeneration method to release newly established regeneration
- Passive adaptive management:** The “best” management option is identified, implemented, and monitored
- Phreatophytic vegetation:** Vegetation associated with high soil moisture or free water
- Policy analysis:** An organized projection of how implementation of the policy over space and time might affect the resources valued by society
- Population:** Self-sustaining assemblages of individuals of a species over space and through time
- Population viability analysis:** A structured approach to examining population performance based on demographic characteristics and habitat quantity and quality
- Preparatory harvest:** First step in a shelterwood regeneration method to encourage seed production
- Prescriptions:** Silvicultural management plans for stands
- Primary cavity nester:** A species that excavates a cavity in living or dead wood
- Proximate cue:** An element of structure and/or composition that an individual uses to predict resource availability
- Q-factor:** The factor by which the number of trees in one diameter class is multiplied to get the number in the next smallest diameter class in an uneven-aged stand
- Refereed journal:** Scientific literature in which papers are reviewed and can be accepted or rejected based on review by peers
- Response variable:** Specific indicator or metric used to test a hypothesis
- Restoration ecology:** The suite of scientific practices that constitute an emergent subdiscipline of ecology designed to return functions to systems where they have been eliminated
- Riparian area:** The interface between the water and the land
- Riparian associate:** A species that tends to be found more commonly near water but does not require free water directly
- Riparian obligate:** A species that requires free water
- Risk analysis:** A structured way of analyzing the potential effects of decisions when outcomes are uncertain
- Rotation:** A complete growing cycle in an even-aged silvicultural system
- Rotation age:** The stand age when the stand is harvested and a new even-aged stand is regenerated
- Satellite imagery:** Reflectance values collected by satellites for discrete places on the Earth
- Scope of inference:** The space and time over which data can be used to assess changes in a response variable
- Secondary cavity user:** Species that use natural cavities or those created by primary cavity nesters

**Second-order selection:** Establishment of a home range

**Seedbank:** Seeds stored in the soil

**Seedbed:** Growing site for seedlings and sprouts

**Seed-tree regeneration method:** Natural regeneration is established by leaving some trees after harvest to provide a seed source

**Serpentine soil:** A soil enriched in toxic metals, including nickel, magnesium, barium, and chromium, and lacking in calcium

**Shade intolerant:** Plant species that do not survive under low light conditions, and grow well only under full sunlight

**Shade tolerant:** Plant species that can survive under low light conditions

**Shelterwood regeneration:** Natural regeneration needs protection from sun or frost so a light canopy cover is maintained after harvest

**Shifting gap phase:** Forests maintained by frequent small-scale gap disturbances

**Silviculture:** The art and practice of managing forest stands to achieve specific objectives

**Sink habitat:** Habitat patches in which populations are declining or are maintained by immigration

**Site fidelity:** A behavior in which an individual returns annually to the same location despite drastic changes in the habitat

**Site index:** Height of the dominant trees in an even-aged stand at a specified age

**Soft mast:** Soft fruits such as berries and drupes

**Source habitat:** Patches in which individuals are fit enough to support a stable or growing population

**Source patch:** During dispersal, the patch that a disperser is leaving from

**Stand initiation:** Early stage of stand development following a stand-replacement disturbance

**Stand:** Unit of homogeneous forest vegetation used as the basis for management

**Static corridors:** Maintaining connectivity in a fixed location

**Stepping stone:** Small patches of habitat close to one another to enhance connectivity between high-quality patches

**Stocking:** The degree to which a site is occupied by trees of various sizes

**Structure:** Physical features of the environment such as vegetation, soils, and topography

**Suppressed tree:** Trees in an even-aged stand occurring below the main canopy in the stand

**Survival:** The number of animals that live through a time period and is the converse of mortality

**Survivorship functions:** Types 1, 2, and 3 refer to high, medium, and low survival rates of juveniles, respectively

**Swamp:** A wetland dominated by woody vegetation

**Target patch:** During dispersal, the patch that a disperser is going to

**Target tree size:** The diameter class representing the largest harvestable trees in an uneven-aged stand

**Territory:** The space, usually around a nest, that an individual or pair defends from other individuals

**Thermal neutral zone:** The range of ambient temperatures where an animal has to expend the least amount of energy to maintain a constant body temperature

**Third-order selection:** Use of patches within a home range where resources are available to meet an individual's needs

**Trans-riparian gradients:** Changes in conditions as you move from the edge of the stream into upslope forests

**Trophic level:** The feeding position in a food web

**Ultimate resources:** Food, cover, and other resources needed for survival

**Validation monitoring:** Measurements that provide the basis for testing assumptions

**Vernal pools:** Isolated ponds and wetlands that hold water for only a part of the year

**Wolf trees:** Large and often deformed legacy trees from the previous stand





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# Appendix 3: Measuring and Interpreting Habitat Elements

Basic to understanding current conditions and desired future conditions in stands and landscapes is measurement and interpretation of habitat elements. This field exercise introduces you to a few simple techniques for measuring the availability of key habitat elements. More comprehensive information on field sampling of habitat elements can be found in Bookhout (1994), James and Shugart (1970), Hays et al. (1981), and Noon (1981).

## METHODS

Some habitat elements are particularly important to many species depending on their size, distribution, and abundance. These include percent cover, height, density, and biomass of trees, shrubs, grasses, forbs, and dead wood. Other habitat elements are associated with only a few species, such as stream gradients (e.g., beaver; Allen 1983) and forest basal area (e.g., downy woodpecker; Schroeder 1982). Visit two areas with very different management histories such as a recent clearcut and an unmanaged forest. Then compare habitat elements between the two stand types and assess the relative habitat quality for a species between them using life history information, a habitat suitability index model, and a geographic information system.

## RANDOM SAMPLING

Probably, the most important part of sampling habitat is to sample randomly within the area of interest (stand, watershed, stream system, etc.). Systematic or subjective sampling can introduce bias into your estimates and lead to erroneous conclusions. In this example you will be sampling two stands. Within your stand you should collect a random sample of data describing the habitat elements. For the purposes of this exercise, you will collect data from three or more randomly located points in each stand.

1. Using a random numbers table (nearly all statistics books have these) first select a three-digit number that is a bearing (in degrees) that will lead you into the stand. If the number that you select does not lead you into your stand, then select another number until you have a bearing that will work.
2. Select another three-digit number that is a distance in meters. Using your compass to establish the bearing and either a 30 m tape measure or pacing, measure along the assigned bearing the randomly selected distance and establish a sample point. You will collect habitat data at this point. Once you have completed collecting data at this point, you repeat the process of random number selection three or more times in this stand and then three or more times in another stand.

## MEASURING DENSITY

One of the most common habitat elements that you will measure is density of items, usually trees, snags, logs, shrubs, or other plants. Density is simply a count of the elements over a specified area. When estimating the density of trees, you usually will count all the trees in a circular plot, usually 0.04 ha (0.1 acre) in size. Saplings and tall shrubs are usually measured in a 0.004 ha (0.01 acre) plot. Small shrubs and tree seedlings are usually measured in a 0.0004 ha (0.001 acre) plot.

1. From plot center, measure out in each cardinal direction (N, E, S, W) 11.3 m (37.2 ft) (the radius of a 0.04 ha [0.1 acre] plot). Mark these places with flagging.
2. Using a diameter tape or a Biltmore stick, measure the diameter at 1.3 m (4.5 ft) above ground of all live trees in the plot that are >15 cm (6 in) dbh (diameter at breast height) and record the species of each tree. Repeat this procedure for all dead trees >15 cm dbh. Expand this sample to 1 ha (or acre) estimate by multiplying the estimates by 25 to convert to a per hectare estimate (or multiply by 10 to get a per acre estimate). This procedure can be repeated for smaller plot sizes to estimate seedling numbers, and so on.

## ESTIMATING PERCENT COVER

Using your four 11.3 m (37.2 ft) radii as transects, walk along each stopping at five equidistant points along each transect. At each of these points, you will estimate canopy cover. There are a number of techniques available to estimate canopy cover, including moosehorns (Garrison 1949) and densimeters (Lemmon 1957). A simple approach to estimating cover is to estimate the presence or absence of vegetation using a sighting tube (a piece of PVC pipe with crosshairs) (James and Shugart 1970). At each of the 20 points on your transects, look directly up and see if the crosshairs intersect vegetation (if so record a "1") or sky (if so, record a "0"). Repeat this at each of the five points on each of the four transects.

1. Tally the number of "1"s recorded from these points.
2. Divide by 20 to estimate percent cover.
3. How would you use this technique to measure understory herbaceous cover?

## ESTIMATING HEIGHT

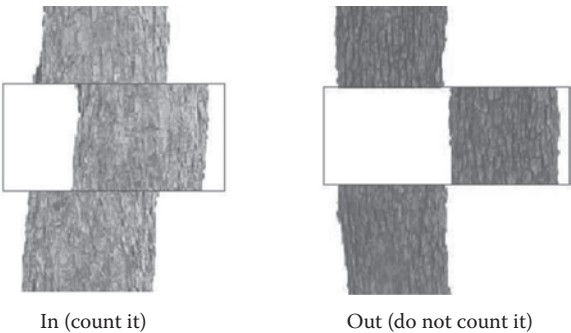
Use a clinometer with a percent scale (look through the view finder and you should see two scales, with units given on them if you look straight up or straight down).

1. Measure 30 m (100 ft) from the base of the tree or other object that you wish to measure.
2. Looking through the view finder, align the horizontal line in the view finder with the top of the tree. Record the number on the percent scale (top).
3. Looking through the view finder, align the horizontal line in the view finder with the base of the tree. Record the number on the percent scale (bottom).
4. If the top number is positive and the bottom number is negative (<0) then add the absolute values of these two numbers together to estimate height in feet.
5. If the top number is positive and the bottom number is also positive (>0), then subtract the absolute value of the bottom number from the top number to estimate height in feet.

## ESTIMATING BASAL AREA

Basal area is the cross-sectional area of all woody stems at 1.3 m (4.5 ft) above ground. It is a measure of dominance of a site by trees. The higher the basal area, the greater the dominance by trees. There are two ways to estimate basal area. First, using your estimates of dbh from your sample of trees (see the section on Measuring density, given earlier), you can calculate the area of each stem ( $A = 3.1416 * r^2$ , where  $r = dbh/2$ ). By summing the areas on a 0.04 ha (0.1 acre) plot and then multiplying the total by 25, you can get an estimate of basal area per hectare (multiply by 10 to estimate basal area per acre).

Alternatively, you can use a wedge prism (Figure A3.1). Holding the prism over plot center, look at a tree through the prism. If the image that you see through the prism is connected to the image of the tree outside the prism, then tally the tree and record its species. If the image that you see through the prism is disconnected from the image outside the prism, then do not record the tree. Moving in a circle around the prism that you continue to hold over plot center record all trees that have the prism



**FIGURE A3.1** When using a wedge prism you have two images to compare—the one you see through the prism and the one above or below the prism. If they overlap you count the tree as an “in” tree. If the images do not overlap then the tree is not counted. (Image from Jesse Caputo. With permission.)

image connected to the image outside of the prism regardless of whether they fall in the 0.1 acre plot or not. Tally up the number of trees that were recorded. Generally you will use a 10-factor prism, that is, each tallied trees represents 10 other trees per acre. Multiply the number of trees tallied by 10 and this estimates the basal area in square feet per acre for this site.

**ESTIMATING BIOMASS**

Biomass of vegetation is usually estimated to provide information on food available for herbivores, typically in the winter when browse resources are essential to supporting herbivores (deer, moose, or hares). Herbivores usually will only eat woody growth resulting from the most recent growing season, and during winter, which includes the twigs and buds, but not leaves (which will have fallen off).

Within a 1.1 m (3.7 ft) radius plot, using clippers, clip all of the twigs within the plot that have resulted from the most recent growing season. Remove and discard the leaves and place the twigs in a bag. Return to the lab and weigh the bag with the twigs. Remove the twigs and weigh the empty bag. Subtract the bag weight from the bag + twigs weight to estimate biomass per 0.0004 ha (0.001 acre) plot. Multiply this number by 2500 (or 1000 in acres) to estimate biomass (kilogram) per hectare.

**USING ESTIMATES OF HABITAT ELEMENTS TO ASSESS HABITAT PRESENCE**

If you refer to Table A3.1 as an example (you will have your own numbers from your field samples), consider how you would interpret these data for a species of your choice, in this case downy

**TABLE A3.1**  
**Comparison of Average and Range of Habitat Elements between Clearcut (with a Legacy of Living and Dead Trees) and Uncut Forests, Cadwell Forest, Pelham, MA**

	Clearcut	Mature Forest
Trees >15 cm/ha	3 (0–6)	308 (234–412)
Snags >15 cm/ha	1 (0–2)	22 (4–43)
Basal area/ha	2.4 (0–3)	16 (12–18)
Canopy cover (%)	4 (0–7)	95 (90–100)
Canopy height (m)	23 (18–34)	27 (23–33)
Browse (kg/ha)	1234 (554–2600)	387(122–788)

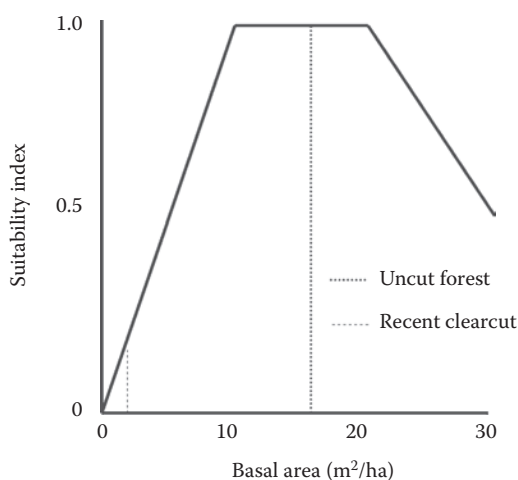
woodpeckers. DeGraaf and Yamasaki (2001, p. 161) describe habitat for downy woodpeckers as: "... woodlands with living and dead trees from 25–60 cm dbh; some dead or living trees must be greater than 15 cm dbh for nesting." Although both sites contain trees and snags of sufficient size, the canopy cover data in Table A3.1 would suggest that the clearcut is not functioning as a woodland and so we would probably not consider it a suitable habitat for downy woodpeckers though they certainly do use snags in openings at times.

## USING ESTIMATES OF HABITAT ELEMENTS TO ASSESS HABITAT SUITABILITY

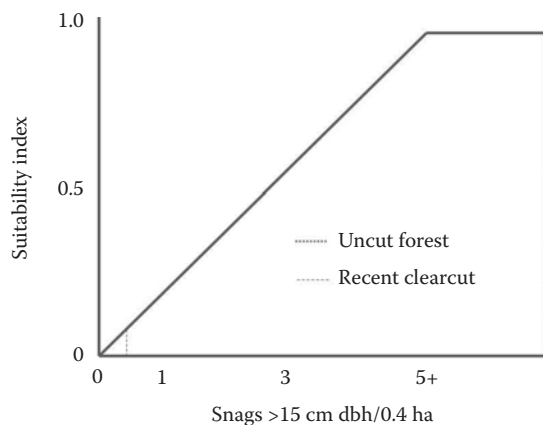
In addition to using your data to understand if a site might be used by a species, habitat suitability index models have been developed to understand whether some sites might provide more suitable habitat than others (e.g., Schroeder 1982). Very few of these models have been validated especially not using fitness as a response variable. Nonetheless they do represent hypotheses based on the assumption that there is a positive relationship between the index and habitat carrying capacity. If we take the example of the downy woodpecker then its habitat suitability is based on two indices: tree basal area (Figure A3.2) and density of snags >15 cm dbh (Figure A3.3). Considering first the uncut stand, note that there is an average of 16 m<sup>2</sup>/ha of basal area and 22 snags/ha (8.8/0.4 ha). The corresponding suitability index score for each variable is 1.0 and the overall habitat suitability is calculated (in this case) as the minimum of the two values. Hence, this should be a very good habitat for downy woodpeckers. In the recent clearcut, however, the suitability index for snags is approximately 0.1 and for basal area is approximately 0.2. Hence, the overall suitability in the recent clearcut for this species is 0.1; not very good and certainly less than in the uncut stand. And in this case, snag density is the factor most limiting habitat quality for downy woodpeckers in the recent clearcut. The best way to use these sorts of models is in a relative sense, to compare one site to another. If we were to use this technique for snowshoe hares habitat assessment, then we might find the recent clearcut to be much better habitat.

## ASSESSING THE DISTRIBUTION OF HABITAT ACROSS A LANDSCAPE

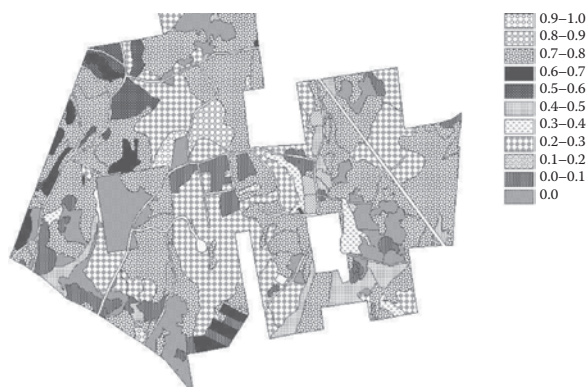
It is often as important to know whether stands are a suitable habitat for a species and how they are arranged on a landscape. In Figure A3.4, a 490 ha forest has been broken into habitat types based



**FIGURE A3.2** Habitat suitability relationship for downy woodpeckers for one of two suitability indices: basal area. (Redrafted from Schroeder, R.L. 1982. *Habitat Suitability Index Models: Downy Woodpecker*. US Fish and Wildl. Serv. FWS/OBS-82/10.38.)



**FIGURE A3.3** Habitat suitability relationship for downy woodpeckers for one of two suitability indices: snag density. (Redrafted from Schroeder, R.L. 1982. *Habitat Suitability Index Models: Downy Woodpecker*. US Fish and Wildl. Serv. FWS/OBS-82/10.38.)



**FIGURE A3.4** An example of a mosaic of habitat patches of varying suitability based on extrapolation of ground inventory data to digitized patches in Cadwell Memorial Forest, Pelham, Massachusetts.

on overstory cover and stand structure. Field samples were taken at 117 points distributed across the forest and habitat elements were sampled at each point. Habitat suitability index values are then calculated at each point and extrapolated to the habitat types as portrayed in this figure to illustrate how habitat availability for a species can be displayed over a landscape. A different pattern would emerge for other species using this same approach, and these would have to be overlain on stands used as the basis for management. In addition, these types of maps can guide harvest planning in order to achieve habitat patterns leading to a desired future condition for the landscape.

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# Appendix 4: Wildlife–Habitat Relationships Models

Since the 1970s, scientists and managers have developed tools that allow them to relate the possible occurrence of a species to a habitat type as the basis for assessing the potential of a unit of land to support populations or communities. A species–habitat-type matrix has long been at the heart of wildlife–habitat relationships (WHR) models such as these, which have been developed for New England, the Blue Mountains, Colorado, the southwestern United States, California, and the Pacific Northwest. These models provide a quick and easy, though not always entirely accurate, ability to relate a species to a habitat type, given knowledge about the structural stage of the habitat type and its location. Each of these models has greater or lesser levels of detail when developing lists of species that could be found in a habitat type or habitat types that a species could be found in. I use a simple hypothetical example of a WHR to illustrate how they are structured and can be used.

## THE CENTRAL HARDWOODS EXAMPLE

I use a subset of habitat types and a subset of species to illustrate how a WHR might be developed and used. In this simplified example, consider three habitat types:

- Grasslands—areas dominated by herbaceous vegetation, including grasses, sedges, and forbs
- Mixed mesophytic hardwoods—upland hardwoods often with 20 or more species represented per acre
- Upland coniferous forest—forests dominated by pines and hemlock

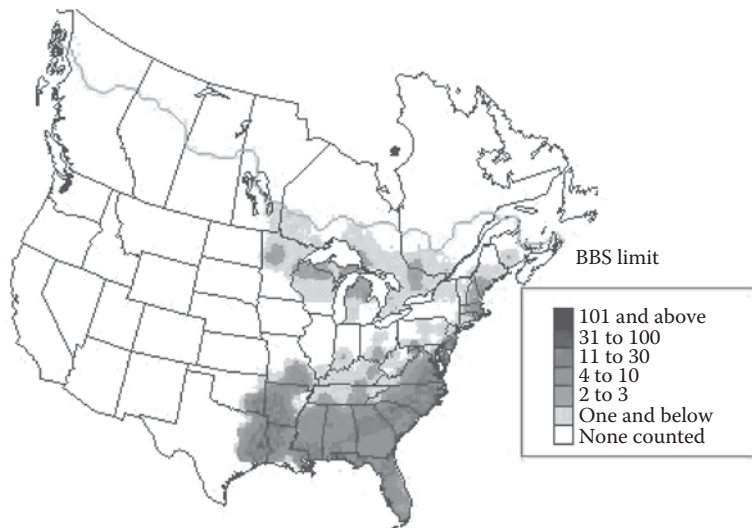
Within the two forested habitat types, we can define four structural states of stand development:

- Seedling shrub—woody vegetation <2 m tall
- Sapling/pole—woody vegetation >2 m tall but <20 cm in dbh (diameter at breast height)
- Sawtimber—woody vegetation 20–50 cm dbh
- Old-growth—woody vegetation representing a range of tree sizes with some trees per hectare >50 cm dbh

I then created a matrix of these conditions and whether or not each of the following species was likely to be found in that habitat-type–structural-condition combination: American goldfinch, Chestnut-sided warbler, Pileated woodpecker, Pine warbler, eastern meadowlark, wood thrush, and black bear. Within this simplified system, we can see that we would expect three of these species to occur in grasslands and four to occur in old-growth conifer forests, and that they would be a different set of species (Table A4.1). So if we applied this model to a forest in southern Indiana, then we would be outside the geographic breeding range of Chestnut-sided warblers and pine warblers might be uncommon (Figures A4.1 and A4.2). So although the model can generate a list of species, the user must assess whether the site being assessed is within the geographic range for the species. However, we can also add value to this simplified assessment of potential occurrence of a species by asking how each habitat type might be used by a species. For instance, a black bear might use a seedling stage of forest for feeding, but an old-growth stage for denning. By designating use of each type-condition combination with an “F” or a “D” rather than an X we know more about how the species could use the habitat type. Some models have further refined this attribution

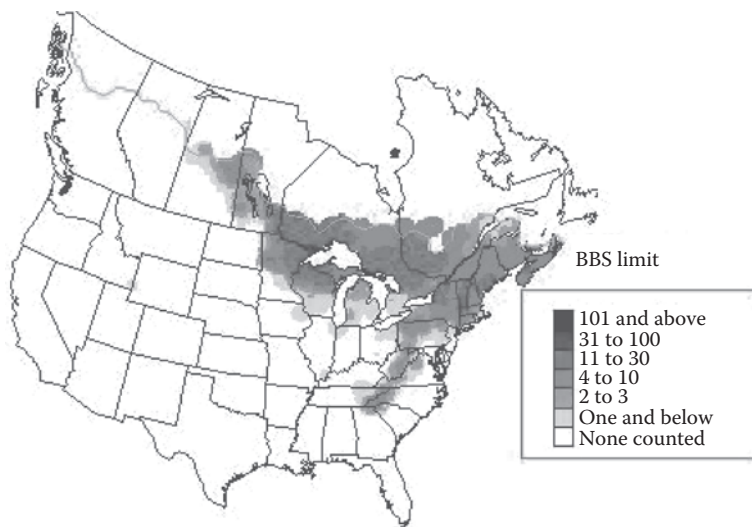
**TABLE A4.1**  
**Simplified WHR Model for a Few Habitat Types and a Few Species from the Central Hardwoods Region**

<i>Habitat Type</i>	Grassland		Hardwood		Hardwood		Hardwood		Hardwood		Hardwood		Conifer		Conifer	
<i>Structural Stage</i>	Grassland	Grassland	Seedling	Sapling	Sawtimber	Old-growth	Seedling	Sapling	Sawtimber	Old-growth	Seedling	Sapling	Sawtimber	Old-growth	Seedling	Sapling
<i>Species</i>																
Rough green snake	x		x				x									
American goldfinch	x		x				x									
Chestnut-sided warbler				x								x				
Pileated woodpecker					x								x			
Pine warbler													x			
Eastern meadowlark	x												x			
Wood thrush																
Black bear			x	x	x	x	x	x	x	x	x	x	x	x	x	x
Total species	3		3	2	2	2	3	2	4	4	3	2	4	4	4	4



**FIGURE A4.1** Geographic distribution of pine warblers in the summer. (From Sauer, J.R. et al. 2012. *The North American Breeding Bird Survey, Results and Analysis 1966–2011*. Version 07.03.2013 USGS Patuxent Wildlife Research Center, Laurel, MD.)

to identify primary and secondary habitat types, where grasslands may be a primary type for goldfinches and seedling stage forests may be secondary, or less commonly used. It might also be important to know whether certain habitat elements are present, such as snags. If we knew that the conifer-sawtimber condition did not have snags >40 cm dbh, then we might conclude that although the habitat type is adequate for feeding pileated woodpeckers, they would be unlikely to nest there. Some WHR models add considerable detail with regard to habitat elements (e.g., Johnson and O’Neil 2001), while others recognize the importance of habitat elements but do not include some of them explicitly within the model (DeGraaf and Yamasaki 2001).



**FIGURE A4.2** Summer geographic range for chestnut-sided warblers. (From Sauer, J.R. et al. 2012. *The North American Breeding Bird Survey, Results and Analysis 1966–2011*. Version 07.03.2013 USGS Patuxent Wildlife Research Center, Laurel, MD.)

## VALUES AND CAUTIONS OF USING WHR MODELS

Given the fact that each species has its own niche and habitat requirements, how well can an approach like this represent habitat for a suite of species? As a first-level assessment of the potential for a site to support species x, y, or z, it may be useful to identify focal species that may need greater attention in a fine filter analysis. Further, if you know that your management actions are likely to shift a stand from one structural stage to another, then the model can be used to provide an estimate of the potential impacts on species found in that area. For instance, in our simple example, if we clearcut a hardwood sawtimber stand, we likely would lose pileated woodpeckers, gain goldfinches and rough green snakes, and black bears would continue to use the site (although for different reasons). What this does not tell us is that goldfinches would be most likely to use the clearcut if thistles (a source of food) were present in the clearcut and that black bears may use the site if it is not too close to people. A WHR model is not a substitute for an approach that identifies the habitat elements important to a species and describes habitat based on the collection of habitat elements needed to support the species. But WHR models can be a first step toward developing more species-specific habitat models.

Because WHR models do not necessarily include all of the habitat elements important to each species that tests of WHR models document errors of omission (species predicted to occur on a site but were not found there) and commission (species were not predicted to occur on a site but were found there). Edwards et al. (1996) reported error rates of 0%–33% among eight national parks in Utah, while Block et al. (1994) reported error rates from 6% to 42% in California. Errors of commission are usually unknown unless independent field verification such as the study conducted by Block et al. (1994). Although WHR models may be useful for large-scale conservation planning, their use for site-specific planning is limited due to these high error rates in some situations and because geographic ranges and detailed habitat elements are not included in some WHR models (the California WHR models now interface with GIS for more accurate representation of species geographic ranges however). Hence, WHR models should be used with caution and the species lists that are derived from them should be assessed carefully to minimize errors of omission.

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# Appendix 5: Projecting Habitat Elements through Time

Once a manager has defined a desired future condition (DFC), then it is important to know whether it is possible to achieve that condition given the mix of plant species on the site and their capacity for growth. If achievement of the DFC is possible, then the manager will need to know what actions will likely be needed and when to achieve the goal and what costs and incomes might be accrued along the way. The U.S. Forest Service developed the forest vegetation simulator (FVS) as a decision support tool for forest managers to use as part of stand and forest plan development (Crookston and Dixon 2005). Because the model has tree growth and mortality functions for most common tree species and because growth varies regionally, FVS has variants designed to simulate the growth of forests in regions across the United States. FVS is a single tree growth and mortality model, meaning that the growth simulations are based on field measurements of a sample of individual trees in a stand.

The model is widely used by public agencies and NGOs and some industries because it is adaptable to a variety of conditions and its ability to use some of the output of the model to understand economics, habitat elements, fuels, and carbon sequestration, among other values associated with forests. Snag dynamics have also been incorporated into some variants of FVS. Further, recent advances have allowed the simulated stand characteristics to be visualized as idealized cartoons of stand structure and composition so that stakeholders can envision what the future conditions might look like, offering a valuable tool for stakeholder input during forest planning.

Although FVS is commonly used to simulate growth of a stand, using FVS to simulate growth of multiple stands simultaneously across a landscape is also possible. The landscape management system (LMS) incorporates FVS (as well as several other growth models) into an overall forest simulation decision support system (Oliver et al. 2012). When landscape visualization is overlain on a topographic map displayed in three dimensions using a digital elevation model, then stakeholders can not only view stands but also landscape change through time from any point in three-dimensional space (Oliver et al. 2012). In this overview of forest stand projection, I first introduce you to simulating stand changes over time and then discuss how these are integrated over landscapes.

## INPUTS

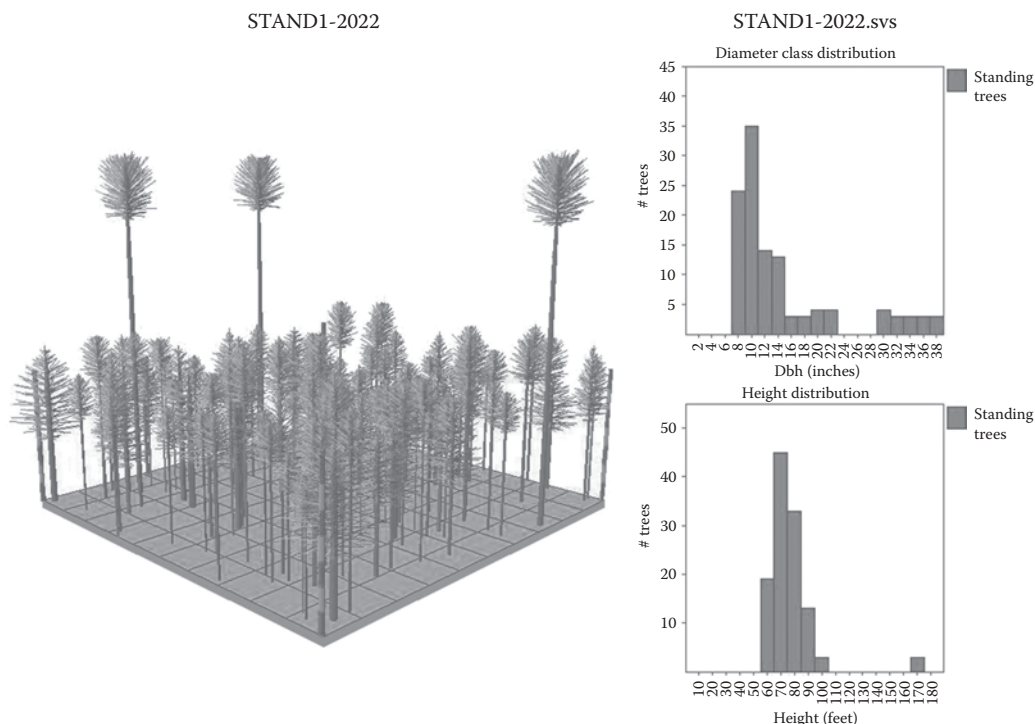
In order to simulate growth of a stand, we need information about the stand and about the trees in the stand. Characteristics of the stand include its location, site index (height of the dominant trees at 50 years of age), stand age (for even-aged stands), slope, aspect, elevation and size (ha), as well as the year that the data were collected to represent current conditions in the stand. In addition, a random sample of trees from the stand must be measured to represent as much as possible all other trees in the stand. See Appendix 3 in this book for examples of how these data can be collected or refer to a text book of forest measurements (e.g., West 2009). For each tree, you record the species, dbh (diameter at breast height), height, and crown ratio (proportion of the tree with living branches), as well as the expansion factor or the number of trees per hectare that each sampled tree represents (e.g., samples from a 0.1 ha plot would have an expansion factor of 10). These data allow the model to represent the current condition of the stand as the basis for all simulations of future conditions.

## PROJECTIONS

Usually the first simulation is simply to grow the stand without any management and assess the changes in conditions over time. I use an example from a stand on the Oregon State University Forests as an example (Figure A5.1) that is two-story stand with a large 20-year-old cohort and scattered 100-year-old trees. Note that the model allows the user to visualize the stand as well as the diameter and height distributions. In addition to the visualization, tables are available to understand basal area, carbon sequestration, vertical structure, fire risk, habitat suitability for selected high interest species, tree species composition, volume tables, and wind hazard assessment for live trees and for dead trees (e.g., snags per acre by size class). Simulating growth of the stand for 50 years results in a different stand structure (Figure A5.2) and the resulting tables tell us that the basal area has increased from 140 to 313 ft<sup>2</sup>/acre during that time, and the average tree has increased diameter from 12 to 22 in dbh. The number of trees per acre has dropped from 119 to 90, with 29 snags per acre produced during that time. Note that the model does not simulate growth of shrubs or herbs. At this point in the simulation the planner/manager should be asking, “Does this simulation seem realistic?” If so, then additional simulations with management actions can be attempted to understand achievement of a DFC. If not, then the underlying parameters in the model may need to be adjusted to more accurately represent conditions on your site.

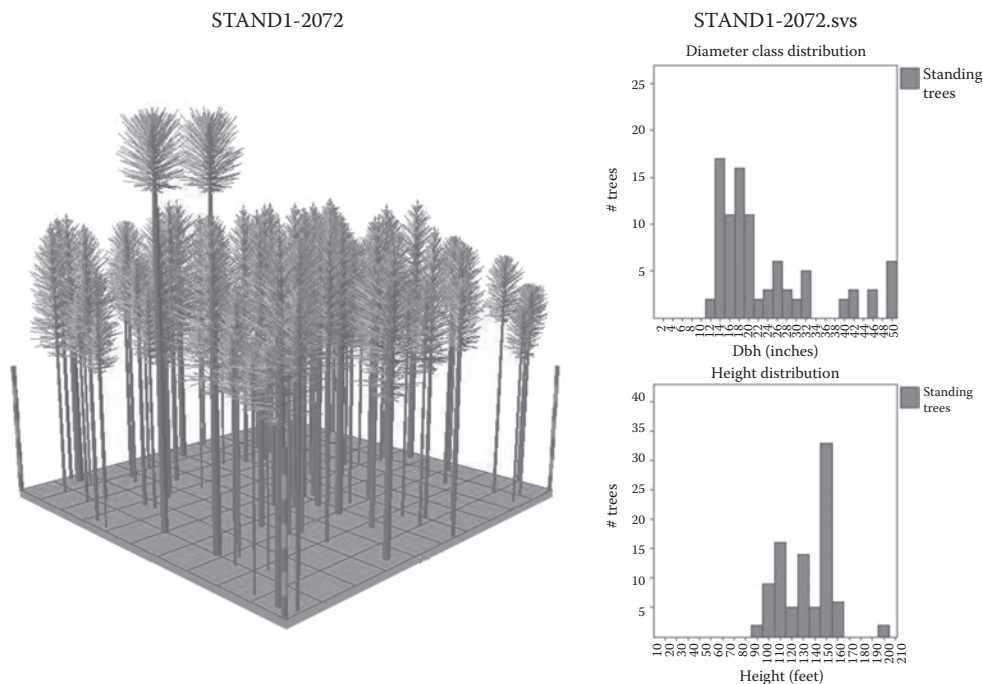
## TREATMENTS TO ACHIEVE A DFC

Let us assume that we define a DFC as a stand with three age cohorts of trees, the oldest of which is 200 years of age, and with both hardwoods and conifers represented in the stand, and with five snags >20 in dbh. One first treatment would be to thin the 20-year-old trees to a level to which



**FIGURE A5.1** Example of a two-story stand 20 years after establishment on McDonald Forest, Corvallis, Oregon, and projected using stand inventory data. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)



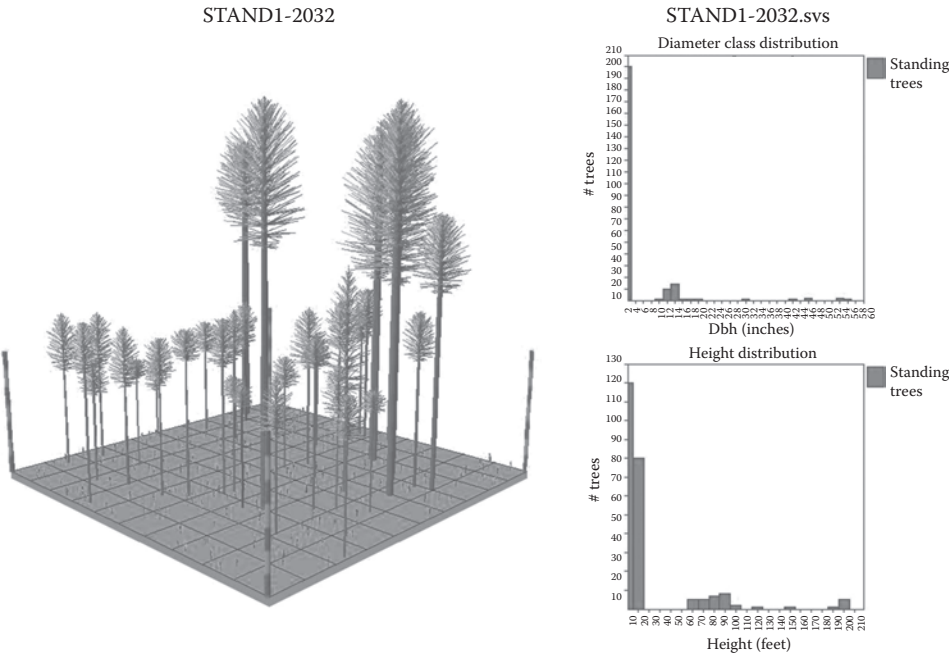


**FIGURE A5.2** Example of the stand illustrated in Figure A5.1 projected 50 years into the future. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

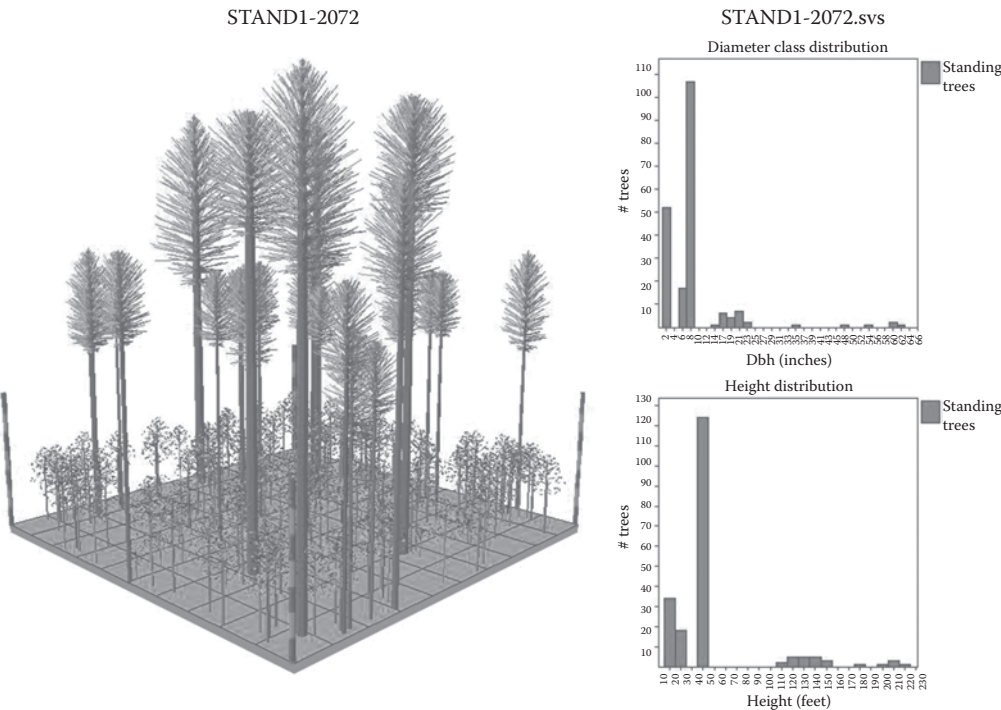
regeneration would become established and receive enough sunlight to grow. I simulated a thin to 30 trees/acre and then planted or relied on natural regeneration of red alder and bigleaf maple (Figure A5.3), which leaves the stand looking very sparse, but allowed removal of 93 thousand board feet of Douglas-fir, which, if sold for \$500/ thousand board feet, would generate \$46,500 based on the harvested volume tables provided by the model. Projecting the stand 50 years into the future, we see a three-storied stand with the lower story consisting largely of hardwoods (Figure A5.4). If there were insufficient snags >20 in dbh, then there are 13 live trees per acre >20 in dbh available to create snags if needed. In fact, the DFC would have been met within 30 years following the harvest. The question then is, how long would this condition persist? Projecting stand growth another 50 years suggests that the stand complexity would persist and potentially increase over the 100 years following the initial harvest, and at that time, there would be 88 thousand board feet per acre, 160 trees/acre, with Douglas-firs as large as 70 in dbh and red alders as large as 18 in dbh.

## COMPARISONS

Of course the approach that I took to achieve the DFC is not the only way to get there and may not even be acceptable to some stakeholders. Repeated lighter thinning may be preferable to one heavy thin. Or a different species mix of regeneration may be desired. Or more income may be required. By simulating different types of treatments at different times, comparisons can be made among multiple approaches to achieve the DFC and then the approach acceptable to stakeholders can be used as the basis for developing a stand management plan.



**FIGURE A5.3** Example of the stand illustrated in Figure A5.1 thinned and planted to establish a third cohort of trees. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)



**FIGURE A5.4** Example of the stand illustrated in Figure A5.3 thinned projected 50 years into the future. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

## INTERPRETATION OF HABITAT ELEMENTS

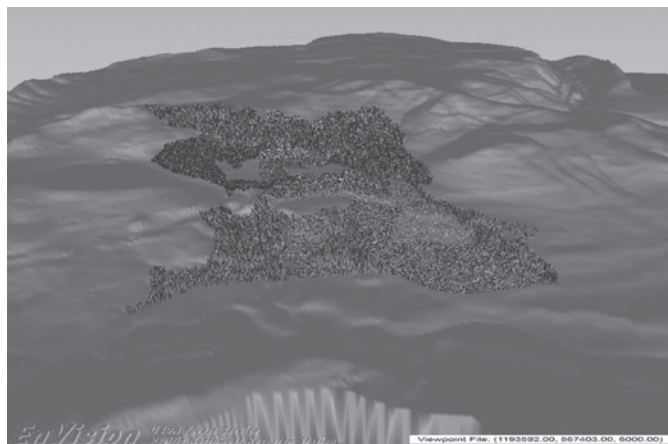
Although some habitat elements such as tree species, tree size, snag abundance and size, and fallen log accumulation can be estimated directly from the tabulated output from the model, other elements must be inferred. For instance, canopy cover can be estimated from the model and used as a surrogate for the potential of the stand to support shrubs and herbs. As the canopy closes, it is likely that shrubs and herbs will decline. As more large and old trees persist in the stand, there is a greater likelihood that some will contain rot and be more likely to form tree cavities as den sites. Hence the ability to fully understand that a DFC might develop is limited using these models, but knowledge of stand dynamics is useful in making inferences.

## STOCHASTIC PROCESSES AND UNCERTAINTIES

Most growth models such as FVS do not explicitly include effects of stochastic processes such as wind, fire, defoliating insects, droughts, floods, and similar events as processes affecting stand development. But clearly these events do occur; so the simulations of stand development are merely representation in the absence of coarse scale disturbances that would cause tree mortality significantly greater than might be found from inter-tree competition mortality. Consequently, it is important to remember that the farther into the future you simulate stand development the more likely one of these stochastic events is to occur, which could significantly affect your ability to achieve a DFC. Simulations such as these are useful to develop plans and modify plans as stands develop but managers realize that unexpected disturbances may cause them to have to plan again following a disturbance.

## PROJECTING LANDSCAPES

Projecting landscape change using LMS is largely a function of simulating the dynamics of many stands simultaneously, something that LMS is designed to do as a part of a planning process. Additional information is needed to describe the location, shape, and position of each stand on the landscape and that information is imported from a Geographic Information System such as ArcGIS. By exporting the digital elevation model and the shape files for each stand, the visualization for each stand is overlain on the landscape (Figure A5.5). By adjusting your position in Envision (the



**FIGURE A5.5** Example of multiple stands simulated across a landscape. (Based on simulations from the Landscape Management System; McCarter, J.M. et al. 1998. *Journal of Forestry* 96(6):17–23.)

visioning tool created by the U.S. Forest Service), you can look at the landscape from different points around it and above it to understand what stakeholders might see. Further tabular data allow a comprehensive landscape level understanding of the availability of habitat elements over the planning area over time.

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# Index

## A

Adaptive management, 264, 297; *see also* Fire management  
active, 297–298  
implementation monitoring, 298  
monitoring, 303  
passive, 297  
process, 8  
strategy, 259  
traditional adaptive management cycle, 298  
validation monitoring, 299  
Adenosine-tri-phosphate (ATP), 5  
Advance regeneration, 83  
balsam fir, 93  
in northern hardwood and boreal forests, 99  
Aerial photographs, 273, 304  
AF&PA, *see* American Forest and Paper Association (AF&PA)  
Agricultural environments, 227  
certification of agricultural lands for wildlife, 234  
edge effects in agricultural settings, 228–229  
loss of forests in industrial agricultural settings, 229–230  
multifunctional landscapes, 233–234  
organic vs. traditional agriculture, 232–233  
Agroforestry, 228, 232  
third party certification, 234  
Alleles, 284, 285  
Allochthonous material, 151  
Alluvial sediments, 148  
American Forest and Paper Association (AF&PA), 321  
American Marten habitat selection, 32–33  
Amphibians, 4, 29, 47, 145, 210, 230–231  
aquatic, 154  
chytridiomycosis in, 72  
common and scientific names, 349–350  
habitat quality, 54  
pond-breeding, 147, 155  
survivorship functions, 29, 30  
Analysis of variance (ANOVA), 302, 307  
Analytical models, 285  
Animal Unit Months (AUMs), 231  
Annual crop management, 230–231  
*Anolis* lizards, 56  
ANOVA, *see* Analysis of variance (ANOVA)  
Artificial regeneration, 83, 97, 99, 121  
Atchafalaya Basin, 149  
ATP, *see* Adenosine-tri-phosphate (ATP)  
AUMs, *see* Animal Unit Months (AUMs)

## B

Bachman's sparrows, 108, 134–138, 140–142  
BACI approach, *see* Before–after, control-impact approach (BACI approach)

Barrier, 18, 48, 145, 216, 221  
maze of potential, 190  
role of, 23  
Basal area, 84–85, 103, 122, 134, 137, 139  
dependency, 105  
estimation, 118, 360–361  
stands increase in, 104  
thinning effect, 108  
BBS, *see* Breeding Bird Survey (BBS)  
BC, *see* British Columbia (BC)  
Beavers, 147, 148, 155–156  
burrow systems, 47  
cottonwoods felled by, 48  
Before–after, control-impact approach (BACI approach), 302, 307–308, 312  
Biodiversity, 1, 237  
higher levels of, 233  
indicators, 13  
loss, 72, 73  
screens, 275, 276  
sustaining, 10  
threat to, 67  
Biodiversity conservation, 14, 59, 75, 79  
advantages of agroforestry, 232  
challenges to managing, 245  
climate change effects, 68  
coarse-filter approach, 240–243  
conceptualization, 239  
fine-filter approaches, 243–245  
forest fragmentation, 206  
forest reserves, 8  
funding, 14  
meso-filter approach, 243  
scientific concept, 238  
setting goals, 239  
spatial scale, 245–246  
time, 247–248  
uncertainty, 248–249  
Biofuels, 68–69, 230  
Biomass, 62, 165  
dead wood, 166, 167, 170–171  
estimation, 361  
human, 5  
pattern of changes, 166  
theoretical changes in, 88  
Birds  
body temperature, 3  
common and scientific names, 348–349  
dispersal pattern, 214  
using flight paths, 213  
forest-interior, 205  
inhabiting woodlots disperse seeds, 229  
migratory, 32  
movement of heavy-seeded tree species, 62  
neotropical migratory, 201, 291, 311–313  
nestling, 4, 45, 76

Birds (*Continued*)

- overwintering, 40
- using retention stands, 98
- survivorship functions, 29, 30
- uneven-aged stands, 122
- West Nile virus in, 72
- Bird species diversity (BSD), 39, 40, 45
- BLM, *see* Bureau of Land Management (BLM)
- "Boom and bust" population pattern, 28
- Breeding Bird Survey (BBS), 291, 305
- British Columbia (BC), 9, 102, 165
  - forests of, 9–11
  - timber harvest in watershed, 10
- Broadcasting, 301
- Brood parasitism, 201, 228
- Brown-headed cowbirds, 200, 201
- Brown-headed nuthatches, 134–137, 142
- Browse, 40, 100, 122, 127
  - availability, 102
  - for deer, 46
  - forage and, 123
  - quantity and quality, 53
  - resources, 361
- BSD, *see* Bird species diversity (BSD)
- Buffer strips, 47, 153
- Bullfrogs, 72, 145
- Bureau of Land Management (BLM), 156
  - lands, 223, 271

## C

- CAPS, *see* Conservation Assessment and Prioritization System (CAPS)
- Cause-and-effect relationship, 311, 313
- Cavity-nesting birds (CNB), 161, 163, 174
- CC stands, *see* Clearcut stands (CC stands)
- CEA, *see* Cumulative effects analysis (CEA)
- Cellulose-managed system
  - dead and damaged trees, 44–45
  - even-aged stands, 39
  - FHD, 39
  - food and cover in, 37
  - forest floor, 47
  - fruit production, 43–44
  - Gopher tortoises, 47
  - herbivores, 48
  - horizontal patchiness, 40
  - niches for species, 38
  - proximity to water, 47
  - shade-tolerant species, 38
  - stages of decay of trees and logs, 44
  - tree species and invertebrate associations, 45–46
  - vertical complexity, 37–40
- CFI plots, *see* Continuous forest inventory plots (CFI plots)
- Chain of custody, 320
- CITES, *see* Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
- CLAMS, *see* Coastal Landscape Analysis and Modeling Systems (CLAMS)
- Clayoquot Sound (CSP), 10
- Clearcut stands (CC stands), 99, 121
- Clearcutting, 96

- Clearcut with reserves, *see* Deferred rotation method
- Climate, 56
  - geographic range of northern copperheads, 58
  - gradient of oak species, 59
  - isotherms for date of first frosts, 58
  - map of moisture stress in Oregon Coast Range, 58
  - moisture, 57
  - patterns of tree pollen from sediments in Wisconsin Lake, 57
  - precipitation, 59
  - predictions of species ranges, 59, 60
- Climate change, 69, 318, 337
  - burning of fossil fuels, 69
  - carbon dioxide concentrations, 71
  - contributions to atmospheric carbon, 70
  - current range of northern bobolink, 71
  - impacts of increase greenhouse gases on Earth's climate, 70
- Clonal plants, monitoring, 310
- CNB, *see* Cavity-nesting birds (CNB)
- Coarse-filter approach, 240, 275–276
  - ecosystem indicators, 243
  - goals and objectives, 260
  - natural-and human-induced disturbances, 242
  - open and early-successional forest conditions, 241
- Coarse-filter approach, 240, 275–276
  - ecosystem indicators, 243
  - goals and objectives, 260
  - natural-and human-induced disturbances, 242
  - open and early-successional forest conditions, 241
- Coastal Landscape Analysis and Modeling Systems (CLAMS), 271
- Codominant trees, 85, 86, 105, 106, 165, 173
- Columbia River, 23
- Commercial thinning, 103, 107
  - Eastern white pine stocking chart, 104
  - site index curves for northern red oak in Lake States, 106
  - stand density index diagram, 105
  - stands increase in basal area over time, 104
- Community, 2
- Composition, 37
  - effects on species composition, 183
  - forest, 40, 93, 94
  - hardwood, 46
  - plant community, 88
  - plant species, 102, 123, 132
  - species, 41, 54
- Connectivity, 218
  - landscape fragmentation, 219
  - landscape resistance map for American black bear, 221
  - landscape structure, 218–219
  - management app roaches to, 222
  - physical and genetic connectivity, 220–222
- Conservation Assessment and Prioritization System (CAPS), 275–276
- Conservation strategies, 267; *see also* Ecoregional assessments
- Context, 199
- Continuous forest inventory plots (CFI plots), 141
- Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), 327
- Core, 202, 277



Corridors, 222  
 Cover, 3, 4, 28, 164  
   canopy, 97, 122, 373  
   conifer, 52  
   estimating percent cover, 360  
   foliage, 39  
   overstory, 137  
   resources, 26, 32, 37, 44  
   shrub, 31, 197  
   thermal, 55  
   types, 2  
   understory, 141  
 Critical habitat, 31, 328  
   natural habitat, 333  
   removal, 253  
 Crown classes, 85, 86  
 CSP, *see* Clayoquot Sound (CSP)  
 Culmination of MAI, 107  
 Cultural effects on habitat patterns, 67  
   climate change, 69–71  
   factors leading to extinction of passenger pigeon, 76  
   invasive species, 71–74  
   land use, 67–69  
   passenger pigeons, humans, and forests, 75–77  
   synergistic effects, 75  
 Cumulative effects analysis (CEA), 331  
 Current forest condition, 259  
 Cutting cycle, 117–119, 122, 124, 127

## D

Data hungry, 292–293  
 Dead trees, 44–45, 161, 191, 370  
 Dead wood, 191  
   biomass, 166, 167, 170–171  
   creation and loss, 182  
   retention, 172  
 Dead wood management, 161  
   changes in dead wood over time, 166–167  
   comparison of decay constants, 167  
   fate of trees, snags, and logs in forests, 162  
   following disturbance, 165–168  
   generalized pattern of changes in dead  
     wood biomass, 166  
   log users, 164–165  
   natural disturbance, 161, 162  
   in Oregon forests, 174–175  
   primary cavity excavators, 161–163  
   ruffed grouse use logs, 164  
   secondary cavity users, 163–164  
   during stand development, 167–168  
 DECAID, 169, 174  
 Decomposition, 165  
   dead wood changes, 166  
   of organic matter, 59  
   organisms effect, 161  
   pathway, 6  
 Deferred rotation method, 97  
 Degradation, 13  
 Demographic stochasticity, 284, 285, 288  
 Density-dependent habitat selection, 26; *see also*  
   Vertebrate habitat selection  
   fixed amount of resources in patch, 27  
   ideal free distribution, 27

Density management diagram, *see* Stocking chart  
 Density measurement, 359–360  
 Desired future condition (DFC), 125, 131, 135–136,  
   255–256, 260, 310, 334, 369; *see also* Red-  
   cockaded woodpecker habitat  
   assessing risk of losing species, 242  
   habitat elements and population responses, 301  
   management actions, 132, 260  
   pathways to, 256  
   spatial scale and, 245  
   stand prescription development, 131–133  
 Diameter-limit cutting, 121–122  
 Digestible energy, 3, 5, 41  
 Direct effects, 101, 102  
   of climate change on biodiversity conservation, 68  
   on habitat for some species, 54  
   of invasive species, 73  
   of rain, 59  
   of reduced pH, 59  
   soils, 56  
 Dispersal, 213  
   distances, 214  
   natal dispersal, 213  
   understanding successful dispersal probability,  
     214–218  
 Disturbance(s), 79  
   bison, elk, and other herbivores, 80  
   frequency, 82  
   frequency, size, severity relationships, 83  
   histogram of fire-size distribution for 2898 lightning  
     fire, 80  
   legacy trees, snags, and logs, 81  
   severity, 81–82  
   size and pattern, 79–81  
   zones of hurricane frequency in New England, 82  
 “Doing nothing” management, 122, 341, 345  
 “Domains of scale”, 209, 269–270, 271  
 Dominant crown class, 38  
 Dominant trees, 85, 96  
   even-aged stand, 38  
   site tree, 157  
 Douglas-fir forests, 54, 87  
 Douglas-fir plantation, 108  
   on forest industry land in western Oregon, 109  
   green-tree retention stand in western Oregon, 111  
   plantation establishment and management in, 109, 110  
   seed tree regeneration method, 110  
 Douglas-fir stand, 83  
   coastal, 105  
   understory, 74  
 Dynamic carrying capacity, 28  
 Dynamic corridors, 222

## E

Earth Summit, 317, 321  
 Eastern white pine stocking chart, 104  
 Ecological  
   psychology, 17  
   restoration, 13–15, 193, 341  
   rotation age, 108  
 Ecological range of variability (ERV), 342, 343  
 Economic rotation age, 107, 108  
 Ecoregional analysis, 272–273

- Ecoregional assessments, 267
  - climate change, 270
  - ecoregional analysis, 272–273
  - examples, 271
  - habitat availability and quality patterns
    - assessment, 273–280
  - spatial scaling properties, 269–270
  - US national hierarchy of ecological units, 268
  - utility and effectiveness, 279–280
- Ecosystem management, 7
- Ecosystem services, 13
  - analysis, 14–15
  - examples, 14
  - sustaining and marketing, 324
- Ectotherms, 4
- Edge associates, 202
- Edge density, 204
- Edge geometry, 203
  - decreasing patch size, 203
  - edge density, 204
  - staggered setting approach, 205
- Edge specialists, 202
- Effectiveness monitoring, 140–141
- Effect size, 308
- Endangered Species Act (ESA), 253, 328, 330, 337
- Endotherms, 4, 55, 57
- Energy, 3, 344
  - pathways through forest, 6
  - production, 68–69
  - savings of cavity-roosting species, 164
- Environmental ethics, 15
  - “actors” and “do-ers”, 16, 17
  - forest activist protects tree from being cut, 16
- Environmental stochasticity, 284, 285, 292
- ERV, *see* Ecological range of variability (ERV)
- ESA, *see* Endangered Species Act (ESA)
- Establishment cut, 96, 97
- Eutrophic systems, 146
- Even-aged stands, 94; *see also* Uneven-aged stands
  - capabilities of site, 95–96
  - change in MAI with stand age, 108
  - commercial thinning, 103–107
  - fertilization, 107
  - identifying legacy elements to retain, 98
  - live tree diameters distribution, 95
  - natural regeneration and planting options, 99–100
  - precommercial thinning, 103
  - regeneration method, 96–98
  - rotation length, 107–108
  - site preparation effects on habitat elements, 98–99
  - vegetation management effects on habitat elements, 100–103
- Extent, 199
- Extinction vortex, 76
- F**
- Fallen logs, 45, 124, 191
- Fecundity, 29
- Federal funding, 331–332
- FEMAT, *see* Forest Ecosystem Management Assessment Team (FEMAT)
- Fencerows, 42, 229, 232
- Fertilization, 107
- FHD, *see* Foliage height diversity (FHD)
- Field and farm management, 230
  - annual crop management, 230–231
  - grazing management, 231
  - influencing woodlot function, 230
  - land sparing, 231–232
- Filter approach, 239–240
  - coarse-filter approaches, 240–243
  - meso-filter approaches, 243
- Fine-filter approach, 243, 276–277
  - in coarse-filter approach, 244
  - gap analysis, 277–279
  - goals and objective, 260
  - indicator species, 245
- Fire frequency, 182, 183
- Fire management, 181
  - changing fire risk through management, 183
  - fire in managed forests, 185
  - fuels treatments on habitat elements, 184–185
  - on habitat elements and succession, 181–183
  - salvage logging, 183–184, 185–187
- First-order selection, 22
- Fish
  - cold headwater streams, 21
  - common and scientific names, 350
  - type III survivorship curves, 30
- Fledgling, 4, 32, 201, 231
- Foliage height diversity (FHD), 39, 40
- Food Alliance, 234
- Forage, 40; *see also* Cellulose-managed system
  - availability and quality, 40
  - black bears, 43
  - change in taxa abundance or community diversity, 41
  - herbivores, 41
  - mycophagists, 42
  - plants with lower lignin and cutin, 42
- Forecasting, 301
- Forest
  - active forest management, 8, 9
  - digestible energy, 5
  - energy pathways, 6
  - floor, 47
  - fragmentation, 206–207
  - gapiness, 86
  - as habitat, 5
  - historical approaches to managing, 6–7
  - interior species, 202, 203, 205, 234
  - management, 7–8
  - society views natural resources, 7
  - type, 21, 31, 40, 87, 126, 128, 173, 207
  - wildlife habitat, 1
- Forest-associated species, PVA for, 286
  - parameters to home range, 288
  - simulated home ranges, 287
  - time-step between alternative management approaches, 289
  - WHR model, 287
- Forest certification, 320
  - certification effectiveness, 323–324
  - certification schemes, 322
- FSC, 321
- FSC vs. SFI certification approaches, 322
- PEFC-certified forests in world, 320
- third-party certification, 320

Forest clearing for agriculture, 68  
 Forest Ecosystem Management Assessment Team (FEMAT), 156, 271  
 Forest management; *see also* Fire management  
   “doing nothing”, 341  
   neotropical migrant birds to, 311–313  
   personal management philosophy development, 343–346  
   restoration, 341–343  
 Forest Practices Code (FPC), 10  
 Forest Stewardship Council (FSC), 321, 322  
   SFI certification approach vs., 322  
 Forest structure, 40, 41, 86  
   characteristics, 117  
   elements, 87  
   rate of change in, 209  
   variability in, 89  
 Forest sustainability, 317  
   forest certification, 320–324  
   humans as part of system, 319–320  
   resources, 318  
   scales of sustainability, 318–319  
 Forest vegetation simulator (FVS), 132, 173, 369  
 Founder effect, 284–285  
 Fourth-order selection, 26, 30  
 FPC, *see* Forest Practices Code (FPC)  
 FRAGSTATS program, 210  
 Fruit production, 43–44  
 FSC, *see* Forest Stewardship Council (FSC)  
 Fuels, 182, 184  
   burning of fossil fuels, 69  
   treatments on habitat elements, 184–185  
 FVS, *see* Forest vegetation simulator (FVS)

## G

*Game Management*, 6, 202  
 Gap-crossing ability, 218–219  
 Gap analysis, 277–279  
 Genetically modified tree seedlings  
   (GM tree seedlings), 100  
 Genetic bottleneck, 284  
 Genetic drift, 284  
 Geographic information systems (GISs), 194, 255  
 Geographic range, 22, 70  
   American marten, 32  
   ESA and, 253  
   marsupials, 203  
   of northern copperheads, 58  
   plant species, 310  
   red-cockaded, 133  
   species, 24–25  
 Geology, 53  
   effects, 54  
   karst geology locations in United States, 54  
   microclimatic characteristics, 55  
   vegetation structure and composition, 53  
 GISs, *see* Geographic information systems (GISs)  
 Global positioning system (GPS), 213  
 GM tree seedlings, *see* Genetically modified tree seedlings (GM tree seedlings)  
 Gopher tortoises, 47  
 GPS, *see* Global positioning system (GPS)  
 Grain, 198, 199, 211, 248, 274

Grazing management, 231  
 Green-tree retention stands (GTR stands), 111, 121  
 Grizzly bear, 29, 289–290  
 “Ground-truthing”, 304  
 Group selection stands (GS stands), 120, 121  
 GTR stands, *see* Green-tree retention stands (GTR stands)  
 Guild, 6, 245

## H

Habitat, 1, 2  
   cover, 4  
   energy flow, 2  
   environmental activism and effects on, 18  
   forests as, 5–8  
   forests of British Columbia, 9–11  
   function, 2–5  
   for humans, 5  
   metabolic rate and ambient temperature relationship, 3  
   patch of, 4–5  
   specialists, 21  
   timber harvest in watershed, 10  
   types, 2  
 Habitat availability and quality patterns assessment, 273  
   coarse-filter approach, 275–276  
   fine-filter approach, 276–277, 277–279  
   ground plot data, 275  
   integrated coarse-filter approach, 276–277  
   meso-filter approach, 276–277  
   prioritizing management and assessing policies, 275  
   seven-step conservation planning framework, 274  
   WHR models, 274  
 Habitat banking, 324  
 Habitat Conservation Plan (HCP), 253, 258–259, 328  
 Habitat distribution assessment, 362–363  
 Habitat elements, 37, 87, 297, 303–304, 313; *see also*  
   Vegetation management effects  
   adaptive management, 297–299  
   basal area estimation, 360–361  
   biomass estimation, 361  
   cause and effect monitoring designs, 307–308  
   clonal plants, 310  
   comparisons, 371–372  
   data availability, 308–309  
   decisions with data, 309–310  
   density measurement, 359–360  
   designing monitoring plans, 299  
   Douglas-fir forests, 87  
   estimation to habitat presence assessment, 361–362  
   estimation to habitat suitability assessment, 362  
   examples, 310  
   experimental design, 302  
   fuels treatments on, 184–185  
   geographic distribution in woodcock population changes, 306  
   habitat distribution assessment, 362–363  
   height estimation, 360  
   hypothetical range of conditions, 90  
   inputs, 369  
   interactions among forest disturbance, climate change, and management, 89  
   interpretation, 373  
   land-use, monitoring, 300

Habitat elements (*Continued*)

- limiting species in urban and suburban settings, 191–192
  - management implications from disturbances, 89–90
  - measuring and interpreting, 359
  - monitoring, 305–307, 313
  - neotropical migrant birds, 311–313
  - percent cover estimation, 360
  - projections, 369, 370, 373–374
  - random sampling, 359
  - salamander subpopulation, 311
  - sampling intensity, frequency, and duration 302–303
  - scope of inference, 301–302
  - selection of indicators, 301
  - site preparation effects on, 98–99
  - small mammal species occurrence,
    - monitoring, 310–311
    - for species occurrence, 304–305
  - stochastic processes, 373
  - successional pathways, 88–89
  - theoretical changes in forest states, 88
  - treatments to achieving DFC, 370–371
  - uncertainties, 373
  - in uneven-aged stands, 122–124
  - in woodlots, 228–229
- Habitat fragmentation, 205
- habitat loss, 206
  - species–area relationships, 206–209
- Habitat generalists, 21
- Habitat quality, 29, 287
- at landscape scale, 200
  - proximate and ultimate cues to, 31–32
- Habitat selection, 21
- American Marten habitat selection, 32–33
  - body mass and home range size relationship, 25
  - deer mice, 22
  - distribution of complex of slimy salamander species, 23
  - first-order selection, 22
  - fourth-order selection, 26
  - habitat quality and demographics relationship, 28–30
  - hierarchical selection, 21–26
  - measurement, 30–31
  - metapopulation distribution, 25
  - population fitness, 29–30
  - population growth over time, 28
  - proximate and ultimate cues to habitat quality, 31–32
  - range maps for geographically restricted species, 23
  - second-order selection, 25
  - social cues in, 32
  - survivorship curves, 30
  - third-order selection, 26
  - trophic level, 26
  - woylie, 24
- Hard mast, 43
- Harvesting system, 96, 172
- HCP, *see* Habitat Conservation Plan (HCP)
- Herbicides, 101–102
- Herbivores, 41, 42, 48
- Heuristic aspects of modeling, 294
- Historical range of variability (HRV), 168, 240, 249, 342
- Home range, 25
- Horizontal patchiness, 40
- HRV, *see* Historical range of variability (HRV)
- Human commensal, 22

- Hydrology, 56, 59, 60
- Hyporheic zone, 155

## I

- ICBEMP assessment, *see* Interior Columbia Basin Ecosystem Management Planning assessment (ICBEMP assessment)
- Ideal despotic distribution, 27
- Ideal free distribution, 27
- IMF, *see* International Monetary Fund (IMF)
- Implementation monitoring, 140, 298
- Incidental take permit, 253–254
- Indicator species, 6, 245
- Indirect effects, 53, 59, 102–103
- Individual tree selection, 120
- Induced edges, 200–201, 205
- “Informed” pixels, 275
- Inherent edges, 200, 201
- Insects, 30, 86
  - bark-dwelling, 32, 46
  - common and scientific names, 350
  - foliage-dwelling, 32
- Integrated coarse-filter approach, 276–277
- Interior Columbia Basin Ecosystem Management Planning assessment (ICBEMP assessment), 271
- Intermediate crown class, 85
- Intermediate trees, 85
- International laws and agreements, 327
  - municipal policies, 334–335
  - national laws, 328–332
  - state laws, 332–334
- International Monetary Fund (IMF), 8
- Intra-riparian gradients, 147, 148
- Intrinsic rate of natural increase, 28
- Invasive species, 71
  - Bullfrogs, 72
  - changes in native plant species, 74
  - direct effects, 73
  - numbers of nonnative plant species, 73
  - understory of Douglas-fir stand, 74
  - whitenose syndrome, 72

## K

- KISS principle, 294

## L

- Lambda, 200
- Land ownership pattern, 153, 157, 245
- Landscape connections, 213
  - connectivity, 218–222
  - dispersal, 213–218
  - gap-crossing ability, 218–222
  - matrix management for wide-ranging species, 222–224
- Landscape management plans, *see* Landscape management system (LMS)
- Landscape management system (LMS), 246, 256, 369
  - considering alternative plans, 260–261
  - current conditions, 255
  - development, 256

DFC, 255–256  
 finding solutions to problems, 261–263  
 goals establishment, 253–255  
 HCP, 258–259  
 one-time step in forest landscape pattern, 257  
 pathways to DFC, 256  
 plan effectiveness, 263–264  
 structure, 259–260  
 Landscapes, 197  
   forest, 199  
   gradient-based analysis, 198  
   habitat area, 209–211  
   habitat quality at landscape scale, 200  
   living on edge, 200–205  
   Swainson's thrushes, 198  
 Land sparing, 231–232  
 Land use, 67  
   energy production and biofuels, 68–69  
   forest clearing for agriculture, 68  
   urbanization, 67–68  
 Late-successional reserves (LSRs), 222  
 Leslie Matrix, 285–286  
 Light detection and ranging (LIDAR), 273, 304  
 Live cavity-tree management, 171  
   dead limbs on live hardwoods and cavities, 171  
   leaving snags and logs, 172  
 Living on edge, 200  
   brown-headed cowbirds, 201  
   edge associates, 202  
   edge geometry, 203–205  
   feature of landscapes, 200  
   induced edges, 200–201  
   species richness, 202  
 LMS, *see* Landscape management system (LMS)  
 Log(s); *see also* Snags  
   creation for wildlife, 173–174  
   monitoring cavity, 174  
   users, 164–165  
 Logistic growth, 28  
 Longevity, 29, 247  
 "Louisiana Acres for Wildlife" program, 332–333  
 LSRs, *see* Late-successional reserves (LSRs)

## M

MAI, *see* Mean annual increment (MAI)  
 Mammals  
   body temperature, 3  
   common and scientific names, 347–348  
   dispersal pattern, 214  
   inhabiting woodlots disperse seeds, 229  
   movement of heavy-seeded tree species, 62  
 Marbled murrelet, 290  
 Marshes, 148  
 Matrix, 205, 210, 213, 222  
 Matrix management, 222  
   BLM lands, 223  
   land allocation pattern, 223, 224  
   for wide-ranging species, 222  
 McIntire–Stennis Act, 332  
 Mean annual increment (MAI), 107  
 Meso-filter approach, 243, 276–277  
   goals and objectives, 260  
 Metapopulation, 215

distribution, 25  
 models, 286  
   stochastic metapopulation model, 290  
 Millennium Ecosystem Assessment Report, 14  
 mitochondrial DNA (mtDNA), 220  
 Moisture, 57  
   soil, 59  
   stress, 58, 59  
   stress gradient, 59  
   tolerance, 120  
 Montreal Process, 317–320, 323  
 Mortality rate, 29  
 mtDNA, *see* mitochondrial DNA (mtDNA)  
 Multifunctional landscapes, 233–234; *see also* Landscape management system (LMS)  
 Municipal policies, 334–335  
 Mycophagists, 42

## N

Nam Choan Dam, 18  
 Nataliity, 29  
 National Council on Air and Stream Improvement (NCASI), 332  
 National Environmental Policy Act (NEPA), 258, 259, 330, 331  
 National laws, 328  
   CEA, 331  
   environmental laws, 328  
   federal funding, 331, 332  
   NEPA, 330, 331  
   US laws, 329–330  
 Natural catastrophe, 284, 285  
 Natural cavity, 171–172  
 Natural regeneration  
   even-aged stands characteristics, 99–100  
   uneven-aged stands characteristics, 120–121  
 Nature Deficit Disorder, 17  
 NCASI, *see* National Council on Air and Stream Improvement (NCASI)  
 Neotropical migrant birds, 201, 291, 311–313  
 NEPA, *see* National Environmental Policy Act (NEPA)  
 Nest boxes, 171  
 Nesting cover, 4  
 NGOs, *see* Nongovernmental organizations (NGOs)  
 Nongovernmental organizations (NGOs), 246, 253, 254, 271, 317, 319  
 Nonregulatory goals, 254  
 Nontraditional management approaches, 125–126  
 Northwest Forest Plan (NWFP), 156, 222, 260–261

## O

Occam's razor, 294  
 Occupational Health and Safety Administration (OSHA), 172  
 OFPA, *see* Oregon Forest Practices Act (OFPA)  
 Old-growth, 86–87  
 Oligotrophic streams, 146, 151  
 Oregon Forest Practices Act (OFPA), 157  
 Organic agriculture, traditional agriculture vs., 232–233  
 Orographic precipitation, 55  
 Orthophotos, 273

OSHA, *see* Occupational Health and Safety Administration (OSHA)  
Overwood removal, 97

## P

Pan European Forest Certification (PEFC), 321  
Passenger pigeons, 75–77, 241  
Passive adaptive management, 297  
PCR, *see* Polymerase chain reaction (PCR)  
PEFC, *see* Pan European Forest Certification (PEFC)  
Percent cover estimation, 360  
Performance-based standards, 322  
Personal management philosophy development, 343  
    leaving world, 345–346  
    living simply and sustainably, 344–345  
    place on Earth, 344  
Pesticides, 101–102  
Phreatophytic vegetation, 155  
Physical environment, 53; *see also* Habitat  
    climate, 56–60  
    geology, 53–55  
    hydrology, 60  
    soils, 56  
    topography, 55–56  
    vegetation patterns, 60–62  
Pittman–Robertson Act, *see* Wildlife Restoration Act  
Plan effectiveness, 263–264  
Planting options  
    even-aged stands characteristics, 99–100  
    uneven-aged stands characteristics, 120–121  
Plants  
    chemical and physical defenses, 42  
    clonal, 310  
    common and scientific names, 350–352  
    GM, 100  
    herbaceous, 40, 97  
    with lower lignin and cutin, 42  
    in partial shade, 44  
    producing chemicals, 48  
    producing phenols, 41  
Policy analysis, 335–336  
Polymerase chain reaction (PCR), 221  
Population, 2, 24, 28, 134  
Population fitness, 2, 29–30  
Population viability analysis (PVA), 283  
    extinction risks, 283–285  
    for forest-associated species, 286–289  
    goals 285  
    grizzly bear, 289–290  
    interpreting results from PVA projections, 294–295  
    marbled murrelet, 290  
    model errors and uncertainties, 291–294  
    models, 285–286  
    neotropical migrant birds, 291  
Precipitation, 56, 59, 183  
    acid, 59  
    orographic, 55  
Precommercial thinning, 103  
Preparatory harvest, 97  
Prescriptions, 131, 133, 141  
Primary cavity excavators, 161  
    primary CNB, 163  
    species, 162

Primary cavity nesters, 44, 45  
Process-based schemes, 322  
Projecting landscapes, 373–374  
Proximate cues, 31–32  
Public resources, 7, 199, 319  
    on private lands, 17  
PVA, *see* Population viability analysis (PVA)

## Q

Q factor, 118, 122–123

## R

Random sampling, 359  
Red-cockaded woodpecker habitat, 133; *see also* Desired future condition (DFC)  
    budget, 141  
    current stand condition, 134–135  
    income and expenses, 142  
    management actions, 136–140  
    monitoring plans, 140–141  
    species background and management options, 133–134  
    stand condition, 137, 138, 139, 140  
    statistics for 7-ha pine stand, 135, 136, 137, 139, 140  
Reduced Emissions of carbon dioxide caused by Deforestation and Degradation program (REDD+ program), 328  
Refereed journal, 260  
Regeneration method, 96  
    clearcutting, 96  
    seed-tree, 96  
    shelterwood, 97  
Regulatory and legal considerations  
    climate change, 337  
    decisions in United States, 336–337  
    Endangered Species Act, 337  
    habitat for animals, 327  
    international laws and agreements, 327–335  
    policy analysis, 335–336  
Regulatory goals, 253–254  
REMs, *see* Resource Equivalency Methods (REMs)  
Reptiles, 4  
    common and scientific names, 350  
    composite map of species-richness, 279  
    distribution, 55–56  
    survivorship curve, 29  
Resource Equivalency Methods (REMs), 324  
Response variable, 30, 301, 302  
Restoration, 341  
    ecology, 341  
    human requirements as constraints, 342–343  
Riparian area management, 145, 146  
    animal associations with, 146–147  
    eutrophic systems, 146  
    in patchwork ownership, 156–157  
Riparian associates, 147  
Riparian buffers, 152  
    beavers, 155–156  
    land ownership pattern, 153  
    principles, 155  
    streamside buffer in managed Oregon forest, 153  
    streamside management areas, 154–155



two species of aquatic amphibians, 154  
width of riparian management area, 153

Riparian functions, 150  
allochthonous material, 151  
hypothesized functional relationships, 151  
tree canopies, 152

Riparian obligates, 147

Riparian zones, gradients within, 147  
alluvial sediments, 148  
dead wood in stream, 150  
hierarchical system of stream orders, 148  
intra-riparian gradients, 147  
stream morphology, 149  
trans-riparian gradients, 149–150

Risk analysis, 295

Rotation, 97

Rotation age, 107

Rotation length, 107–108

Rough-barked trees, 46

**S**

Salamander subpopulation, 311

Salvage logging, 183–184  
boundaries and pattern of Biscuit Fire, 186  
scientific debate with social solution, 185–187

SAR, *see* Synthetic aperture radar (SAR)

Satellite imagery, 273, 304

Scope of inference, 301–302

Second-order selection, 25

Secondary cavity users, 44, 163–164

Seed-tree regeneration method, 96

Seedbank, 86

Seedbed, 79, 101, 119, 121

Seed cut, *see* Establishment cut

Seed tree regeneration method, 110

Serpentine soil, 53

SFI, *see* Sustainable Forestry Initiative (SFI)

Shade-intolerant species, 38, 88, 98, 121, 126

Shade-tolerant species, 38, 86, 88, 120, 121

Shelterbelts, 229

Shelterwood regeneration method, 97

Shifting gap phase, 86

Silviculture, 38, 93  
as forest disturbance, 93–94  
range of management decisions, 93, 94

Single-tree stands (STS), 121

Sink habitat, 27

Site fidelity, 31

Site index, 105–106, 120, 369

Site preparation effects, 98–99

Small mammal species occurrence, monitoring, 310–311

Small privately owned forest management, 126–127

Snags, 44, 45, 81, 98, 135, 161; *see also* Log(s)  
clumping, 172  
creation for wildlife, 173–174  
cumulative species curves, 169  
elevated, 45  
leaving, 172  
monitoring cavity, 174

Social cues, 32

Social facilitation, 75, 76

Social range of variability (SRV), 342, 343

Social values, 15  
ecological psychology, 17  
environmental ethics, 15–17  
TEK, 15

Soft mast, 43, 44, 87, 124

Soils, 31, 53, 56–57

Source habitat, 27, 202

Source patch, 214–215

Spatially explicit models, 286

Spatial scale, 245–246

Specialty crops, 232

Species common and scientific names, 347  
amphibians, 349–350  
birds, 348–349  
fish, 350  
insects, 350  
mammals, 347–348  
plants, 350–352  
reptiles, 350

Species composition, effects on, 183

SRV, *see* Social range of variability (SRV)

“Staggered setting” approach, 205

Stand dynamics, 83  
crown class differentiation, 85  
forest gapiness, 86  
old-growth, 86–87  
stand initiation, 83–85  
stem exclusion, 85  
understory reinitiation, 86

Stand initiation, 83  
advance regeneration, 83  
basal area, 84  
conceptual timeline portraying developmental stages, 84  
stocking, 85

Stand prescription development, 131  
budget, 132  
current stand condition, 132  
DFC, 132  
management actions, 132  
monitoring plans, 132  
references, 132–133  
schedule, 132  
species background, 132

State laws, 332  
critical natural habitat, 333  
forest practices acts, 334  
“Louisiana Acres for Wildlife” program, 332–333  
regulatory agency, 332

State Wildlife Grants program (2001), 331

Static corridors, 222

Stem exclusion, 83, 95

Stepping stone approach, 190, 222–223

Stochastic processes, 373

Stochastic single-population models, 286

Stocking, 85, 124, 135  
chart, 104, 106

Streamside management areas, 152, 154–155, 157

Structure, 37  
forest, 32, 40, 54, 87  
of landscape management plan, 259  
soil, 56  
stand, 86, 103  
vegetation, 53  
vertical, 38, 82, 85, 122

STS, *see* Single-tree stands (STS)  
 Suppressed trees, 85  
 Survival, 5, 29, 215–216, 258  
 Survivorship functions, 29, 30  
 Sustainability, 13, 247, 317; *see also* Forest sustainability  
 Sustainable Forestry Initiative (SFI), 321  
   FSC certification approach vs., 322  
 Swamps, 148  
 Synergistic effects, 75  
 Synthetic aperture radar (SAR), 304

## T

Target patch, 214–215  
 Target tree size, 117–118, 120, 124  
 TEK, *see* Traditional ecological knowledge (TEK)  
 Termites, 166  
 Territory, 25, 27, 133, 153, 172, 246, 270  
 The Nature Conservancy (TNC), 271, 272  
 Thermal neutral zone, 3  
 Third-order selection, 26  
 Third-party certification, 320  
 TNC, *see* The Nature Conservancy (TNC)  
 Topography, 53, 55–56  
 Traditional agriculture, organic agriculture vs., 232–233  
 Traditional ecological knowledge (TEK), 15  
 Trans-riparian gradient, 148–150  
 Tree canopies, 152  
 Tree cavities and dead wood management, 168; *see also*  
   Dead wood management  
     cumulative species curves of snags, 169, 170  
     dead wood retention and harvest system  
       considerations, 172  
     expected range of variability of old-growth forests, 168  
     likely ranges, 169  
     live cavity-tree management in managed stands, 171–172  
     monitoring cavity trees, snags, and logs, 174  
     snags and logs creation for wildlife, 173–174  
 Trees, 234  
   dead and damaged, 44–45  
   monitoring cavity, 174  
   rough-barked tree, 46  
   size and density, 46  
   species and invertebrate associations, 45–46  
   in urban, 194–195  
 Trigger points, 310  
 Trophic level, 26  
 Truffles, 42

## U

Ultimate cues, 31–32  
 Ultimate resources, 32, 37  
 Uncertainty, 89, 200, 248–249, 335  
   of future climates, 319  
   in predictions of species ranges, 60  
 Understory reinitiation, 86  
 Uneven-aged stands, 94, 117; *see also* Even-aged stands  
   challenges to using, 124–125  
   cutting cycles in mixed hardwood pine stand, 127  
   dead and dying trees, 124  
   development, 121–122  
   forage and browse, 123  
   habitat elements in, 122–124

horizontal diversity, 123  
 idealized distribution of tree sizes, 119  
 mast, 124  
 natural regeneration and planting options, 120–121  
 nontraditional management approaches, 125–126  
 Q factor, 118  
 silvicultural approaches, 125  
 site potential, 120  
 small privately owned forest management, 126–127  
 tree densities, 118  
 trees harvested by diameter class, 119  
 uneven-aged regeneration methods, 120  
 vertical complexity arising from regeneration, 126  
 vertical structure, 122–123  
 Uninformed pixels, 275  
 Urban forests, 189–190  
   habitat elements limiting species, 191–192  
   managing trees, parks, and forests in urban, 194–195  
   spaces for habitat management, 191  
   species interpreting “built environment”, 190  
   urban expansion, 193–194  
   urban–rural continuum, 189–190  
   urban streams and wetlands, 192–193  
   wetlands, and mitigation, 193–194  
 Urbanization, 67–68, 333  
 Urban–rural continuum, 189–190  
 Urban streams and wetlands, 192–193  
 Use–availability studies, 31–32  
 U.S. Endangered Species Act, 31

## V

Validation monitoring, 299  
 Vegetation management effects, 100; *see also* Habitat  
   elements; Site preparation effects  
   direct effects, 102  
   herbicides, 101–102  
   indirect effects, 102–103  
   pesticides, 101–102  
 Vegetation patterns, 60  
   mosaic of historic habitat types, 62  
   potential vegetation of North America, 61  
   wildlife habitat types, 61  
 Vernal pools, 60, 147  
 Vertebrate habitat selection, *see* Habitat selection  
 Vertical complexity, 37–40  
   even-aged stands, 39  
   FHD, 39  
   niches for species, 38  
   shade-tolerant species, 38  
 Very high resolution (VHR), 304  
 Vital rates, 29, 200, 283–284, 293

## W

Watershed, 60, 147, 193  
   mid-watershed, 148  
   timber harvest in, 10  
 Wetlands, 193–194  
 WHIP, *see* Wildlife Habitat Incentives Program  
   (WHIP)  
 Whitenose syndrome, 72  
 WHR models, *see* Wildlife–habitat relationships models  
   (WHR models)

- Wildlife habitat, 2
  - Wildlife Habitat Incentives Program (WHIP), 331
  - Wildlife–habitat relationships models (WHR models),
    - 273–274, 287, 313, 365
    - central hardwoods example, 365–367
    - for habitat types and species, 366
    - values and cautions, 368
  - Wildlife Restoration Act, 328
  - Willamette Alternative Futures 255, 276
  - Wolf trees, 85
  - Wood-decaying fungi, 174
  - Woodlands management, 227; *see also* Agricultural environments; Fire management
    - agroforestry, 232
    - field and farm management, 230–232
    - forest fragmentation, 227
    - specialty crops, 232
    - woodlots values to landowners, 228
  - Woodlots
    - farm management, 230–232
    - field management, 230–232
    - habitat elements in, 228–229
    - values to landowners, 228
  - Woylie, 24, 42
- Z**
- Zone Model, 290















