

Chapter 9 Conclusion: Understanding Forest Disturbance and Spatial Pattern, Information Needs, and New Approaches233

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1 Introduction: Structure, Function, and Change of Forest Landscapes

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INTRODUCTION

Forests are inherently dynamic in space and time. Their composition and distribution can change not only through continuous, subtle, and slow forest development and succession, but also through discontinuous, occasional, and sudden natural disturbances (Botkin, 1990; Oliver and Larson, 1996; Spies, 1997). In addition to natural processes, human activities and disturbances are the source of much contemporary forest change (Houghton, 1994; Meyer and Turner, 1994; Riitters et al., 2002). Such

land cover change is widely considered the primary cause of biodiversity decline and species endangerment (Hansen et al., 2001). Monitoring natural and human-caused land cover and forest changes, disturbance processes, and spatial pattern is relevant for the conservation of forest landscapes and their inhabitants (Balmford et al., 2003). In recent years, international political momentum dedicated to conservation of biodiversity and sustainable development has increased (Table 1.1).

Biodiversity conservation and sustainable forest management require the collection of new kinds of forest and land cover information to complement traditional forest databases, model outputs, and field observations. Remote sensing and geographical information systems (GISs) have emerged as key geospatial tools — together with models of all kinds and descriptions — to satisfy increasing information needs of resource managers (Franklin, 2001). But, these are more than tools — they represent essentially new *approaches* to forest disturbance and spatial pattern mapping and analysis because they enable new ways of viewing disturbances and landscapes, which in turn influence our understanding and management practices. Critical developments in the use of remote sensing and GIS approaches include the ability to map biophysical (e.g., Iverson et al., 1989), biochemical (e.g., Roberts et al., 2003), and disturbance (e.g., Gong and Xu, 2003) characteristics of forest landscapes over a wide range of spatial scales and time intervals (Quattrochi and Pellier, 1991; Turner et al., 2003).

This introductory chapter provides a brief landscape ecological foundation for the importance of detecting and monitoring forest disturbances and changes in forest landscape patterns. We discuss monitoring and scale considerations and then describe basic stand and landscape dynamics of interest to resource managers. We introduce landscape metrics, which are then more completely reviewed by Gergel (Chapter 7, this volume). We emphasize a developing understanding of pattern/process reciprocity in forested landscapes, which is then highlighted by several case studies of different disturbance patterns in widely differing forest environments. Immediately following this introduction is background material on pertinent remote sensing and GIS data selection, methods, and applications issues in support of forest pattern analysis and change detection (Chapter 2). This material leads naturally to the suite of illustrative examples of remote sensing and GIS approaches in forest harvest pattern detection (Chapter 3), forest insect defoliation mapping (Chapter 4), monitoring fire disturbance (Chapter 5), and the role of GIS in forest disturbance and change mapping (Chapter 6). Subsequent chapters in this book present specific aspects of spatial pattern analysis, including remote sensing considerations (Chapter 7) and a detailed remote sensing/GIS/pattern analysis case study (Chapter 8) designed to aid in understanding critical resource management issues. Each of these chapters has been selected as a representative perspective on developing remote sensing and GIS approaches, which are increasingly recognized, in combination with field data and modeling methods, as the only feasible way to monitor landscape change over large areas with sufficient spatial detail to allow comparison of resultant patterns of different management or natural disturbance regimes.

TABLE 1.1
Selected National Programs on the Conservation of Biological Diversity and Sustainable Management of Earth Resources
Developed Since the Rio Earth Summit, the United Nations Conference on Environment and Development (UNCED), in 1992

Program	Initiation Year and Organization	Vision	Web Site Address
Convention on Biological Diversity (CBD)	1992, United Nations Environment Programme	International treaty to pursue the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources, including by appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and to technologies, and by appropriate funding	http://www.biodiv.org/default.shtml
The Montreal Process	1994, Inter-Governmental Organization of Forestry Agencies	A working group for the development of criteria and indicators that provide member countries with a common definition of what characterizes sustainable management of temperate and boreal forests	http://www.mpci.org/home_e.html
Pan-European Biological and Landscape Diversity Strategy	1994, Council of Europe	The principal aim of the strategy is to find a consistent response to the decline of biological and landscape diversity in Europe and to ensure the sustainability of the natural environment	http://www.coe.int/t/e/Cultural_Co-operation/Environment/Nature_and_biological_diversity/Biodiversity/default.asp#TopOfPage
Kyoto Protocol	1997, United Nations Framework Convention on Climate Change	Convention on Climate Change sets an overall framework for intergovernmental efforts to tackle the challenge posed by climate change with the Kyoto Protocol committing parties to individual, legally binding targets to limit or reduce their greenhouse emissions	http://unfccc.int/2860.php

TABLE 1.1 (Continued)
Selected National Programs on the Conservation of Biological Diversity and Sustainable Management of Earth Resources Developed Since the Rio Earth Summit, the United Nations Conference on Environment and Development (UNCED), in 1992

Program	Initiation Year and Organization	Vision	Web Site Address
Earth Observation for Sustainable Development of Forests (EOSD)	1999, Canadian Forest Service and Canadian Space Agency	Research and applications program to develop a forest measuring and monitoring system that responds to key policy drivers related to climate change and to report on sustainable forest development of Canada's forest both nationally and internationally; space-based earth observation technologies are used to create products for forest inventory, forest carbon accounting, monitoring sustainable development, and landscape management	http://eosd.cfs.nrcan.gc.ca/
Earth System Science Partnership (ESSP)	2001, DIVERSITAS, International Geosphere-Biosphere Programme (IGBP), International Human Dimensions Programme on Global Environmental Change (IHDP), World Climate Research Programme (WCRP)	The ESSP brings together researchers from diverse fields and from across the globe to undertake an integrated study of the Earth system: its structure and functioning; the changes occurring to the system; and the implications of those changes for global sustainability	http://www.essp.org/
Millennium Ecosystem Assessment (MA)	2001, United Nations	International work program designed to meet the needs of decision makers and the public for scientific information concerning the consequences of ecosystem change for human well-being and options for responding to those changes	http://www.millenniumassessment.org/en/index.aspx

LANDSCAPE ECOLOGY

The traditional focus of forest ecology, management, and planning has been primarily on separate landscape elements such as homogeneous forest stands or habitat patches. The importance of interactions among different elements in a landscape was noted in the early 1980s (Forman, 1981), coincident with the need for forest management strategies to consider landscape structure as a requirement for long-term conservation of biodiversity (Noss, 1983; Risser et al., 1984). It has since become generally accepted that the structure of the landscape influences the ecological processes and functions that are operating within it (Haines-Young and Chopping, 1996). The discipline of *landscape ecology* is now widely recognized as a distinct perspective in resource management and ecological science.

The central goal of landscape ecology is the investigation of the reciprocal effects and interactions of landscape patterns and ecological processes (Turner, 1989). Fundamental to such investigation is the awareness that landscape observation is scale dependent, spatially and temporally, with different landscape patterns and processes discernible from different points of view and time that are specific to the organism (e.g., trees vs. earthworms) or the abiotic process (e.g., carbon gas fluxes) under study (Perera and Euler, 2000). A brief overview of general scale considerations is included in this introductory section; Coops et al. (Chapter 2, this volume) present concrete spatial data selection issues related to scale.

LANDSCAPE STRUCTURE, FUNCTION, AND CHANGE

When studying the ecology of landscapes, at least three basic elements must be considered and understood: structure, function, and change (Forman, 1995; M. Turner, 1989). Landscape *structure* generally refers to the distribution of energy, material, and species. The spatial relationships of landscape elements are characterized as landscape pattern in two ways (McGarigal and Marks, 1995; Rempel and Csillag, 2003). First, the simple number and amount of different spatial elements within a landscape is generally defined as landscape composition, and this measure is generally considered to be spatially implicit. Second, the arrangement, position, shape, and orientation of spatial elements within a landscape are generally defined as landscape configuration, which is a spatially explicit measure. Within the framework of this book, this meaning of landscape pattern is used to ensure that both the amount and arrangement of spatial elements of interest are included. In contrast, some studies equate landscape pattern strictly with configuration and treat composition as a second landscape characteristic unrelated to pattern (e.g., Martin and McComb, 2002; Miller et al., 2004).

A landscape can be defined as a spatially complex, heterogeneous mosaic in which homogeneous spatial elements or patches are repeated in similar form over an area bounded by the spatial scale at which ecological processes occur (Urban et al., 1987). For example, juvenile dispersal distance has been used to estimate the spatial extent of landscapes in forest birds (Villard et al., 1995); in another example, a third-order watershed could be the appropriate landscape for consideration of water flow and quality (Betts et al., 2002). Mosaic patterns exist at all spatial scales from

submicroscopic to the planet and universe and the type, size, shape, boundary, and arrangement of landscape elements across this mosaic influence a variety of ecological functions.

Landscape *function* generally refers to the flow of energy, materials, and species and the interactions between the mosaic elements (Forman, 1995). Examples range from fundamental abiotic processes, such as cycling of water, carbon, and minerals (Waring and Running, 1998), to biotic processes, including forest succession (Oliver and Larson, 1996), and the dispersal and gene flow of wildlife (e.g., Hansson, 1991). Such biotic and abiotic flows are determined by the landscape structures present, and in turn, landscape structure is created and changed by these flows. The main processes or flows generating landscape structure formation and landscape *change* over time can be considered as natural and anthropogenic disturbances (e.g., wildfire, insect infestation, harvesting); biotic processes (e.g., succession, birth, death, and dispersal); and environmental conditions (e.g., soil quality, terrain, climate) (Levin, 1978). An overview of some of these processes in the forest environment is presented in a subsequent section of this chapter and in later chapters discussing specific disturbance processes.

FOREST MANAGEMENT

The goals of forest management have expanded in recent decades to include values leading to the implementation of different strategies based on concepts of sustained yield, multiple use, and more recently, ecosystem management. Ecosystem management includes the balancing ecological and social (economic and noneconomic) forest values in the context of increasing population growth, resource use, pollution, and the rate and extent of ecosystem alteration (Kimmins, 2004). Concepts of natural disturbance emulation encompass the idea of trying to arrange changes in forests due to human disturbance to more closely approximate those induced by natural processes (Attiwill, 1994; Hunter, 1990). This is an acknowledgment of disturbances as one of the fundamental processes and drivers of landscape structure and functioning at all spatial and temporal scales in the field of landscape ecology (Turner, 1987). Principles of landscape ecology help to make this forest management approach a viable management option by providing a higher-level context for forest management practices (Crow and Perera, 2004).

Emulating natural disturbance aims to guide local forest management by mimicking the natural range of spatial and temporal variation in landscape- and stand-level forest landscape structures created by past natural disturbances in the given location (Bergeron et al., 1999; Hunter, 1999; Kimmins, 2004). The presettlement landscape allowing for natural dynamism is thought to be the ideal condition against which contemporary landscape diversity and composition ought to be evaluated (Noss, 1983; Seymour and Hunter, 1999). The natural disturbance approach builds on the underlying assumption that forest ecosystems, long-term forest stability, and biodiversity will be sustained if the forest structures created by natural disturbances are maintained since they reflect the same conditions under which these ecosystems have evolved (Bunnell, 1995; Engelmark et al., 1993; Hunter, 1990). For example, Hudak et al. (Chapter 8, this volume) provide a case study perspective of forest

harvest and fire disturbance patterns in an area where both disturbances are known to have occurred.

Consideration of the ecological effects of spatial patterns created by forest harvesting is important for the management regime (Franklin and Forman, 1987), and the patterns and processes in landscapes created by natural disturbances generally display greater variation in time and space than traditional silviculture and forest management (Seymour et al., 2002). Disturbance regimes can be described by a variety of characteristics; however, the main components include magnitude, timing, and spatial distribution (Seymour and Hunter, 1999), and each of these will have an impact on the stand- (or patch-) and landscape-level of the forest ecosystem. Magnitude generally describes the intensity or the physical force of the disturbance or the severity of the effect of the disturbance on the landscape element or organism (Seymour and Hunter, 1999; Turner et al., 2001). Timing of a disturbance mainly specifies the frequency, which is often expressed not only as the return interval between disturbances, but also as the duration and seasonality of a disturbance type (Seymour and Hunter, 1999). The spatial distribution of a disturbance refers to the extent, shape, and arrangement of disturbance patches (Seymour and Hunter, 1999).

A review by Seymour et al. (2002) of disturbance regimes in northeastern North America contrasts the differences in aspects of these three main characteristics (magnitude, timing, and spatial distribution) by comparing wildfire with pathogens and insect herbivory. In the investigated cases, wildfires were of stand-replacing magnitude, with a return interval of 806 to 9000 years and a disturbance patch size distribution ranging between 2 and more than 80,000 ha, while pathogens and insect herbivory disturbance was of a magnitude to create smaller canopy gaps, with a return interval and patch size distribution ranging between 50 and 200 years and between 0.0004 and 0.1135 ha, respectively (Seymour et al., 2002; Figure 1.1).

While the natural disturbance approach may be an ecologically sound premise, its constraints and limitations also need to be considered. Some issues to address in the future include (a) society's reluctance to accept this paradigm in ecosystems that experience disturbances that are very large, severe, and frequent; (b) whether past disturbance regime effects will be rendered inapplicable in the future due to long-term climatic variation, invasion of nonnative species, air pollution, human-induced climate change (Kimmins, 2004); and (c) the difficulty in obtaining and interpreting historic disturbance data for adequate conclusions about the natural disturbance characteristics (Appleton and Keeton, 1999).

SCALE

Every organism is an "observer" of the environment, and every observer looks at the world through a filter, imposing a perceptual bias that influences the recognition of natural systems (Levin, 1992). Science, in general, can be seen as a product of the way the world is seen, constrained by the space and time within which humans inhabit the world (Church, 1996). There is little doubt that ecologists' perceptions have been revolutionized through availability of satellite imagery; for example:

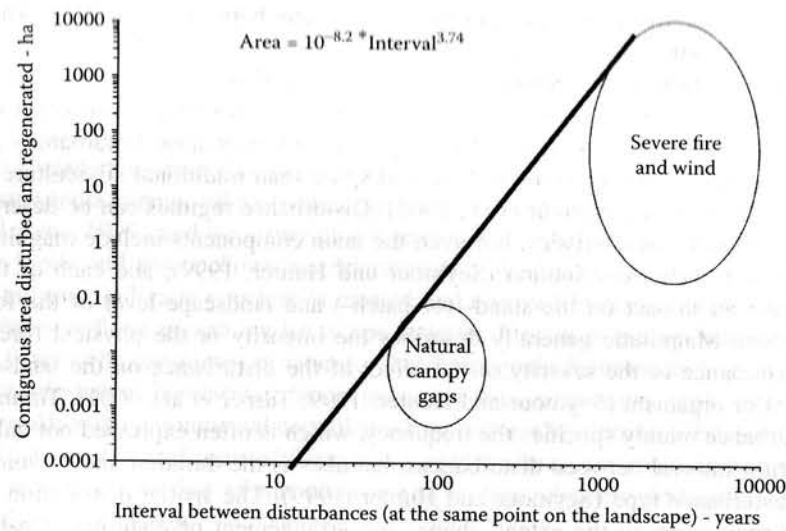


FIGURE 1.1 Boundaries of natural variation in studies of disturbance in northeastern North American forests. The hand-fitted diagonal boundary line defines the upper limits on these disturbance parameters in combination, all of which fall in the lower right of the diagram. Upper limits of the area and return interval of severe fires and windstorms were truncated at 10^4 Ha and 10^4 years, respectively. (Adapted from Seymour et al., 2002.)

- “Images from satellites have revolutionized our perception and approaches to understanding landscapes and regions” (Forman, 1995: p. 35)
- “More than any other factor, it was this perspective provided by satellite imagery that changed the ... manager’s views about the main threats to the panda’s survival” (Mackinnon and de Wulf, 1994, p. 130)

Scale is a strong determinant of viewing, and interpreting the environment and the interest in scale-related research is rapidly increasing (Schneider, 1994). Scale is often understood simply as dimensions of time and space, but has been defined in various more complex ways; for example, Church (1996) considered scale as a relative measure set by the resolution of measurements. Schneider (1994, p.3) defined scale as “the resolution within the range of a measured quantity.” Common to all scientific definitions of scale, however, is a recognition of the temporal and spatial dimensions (Lillesand and Kiefer, 2000; Wiens, 1989).

SPATIAL SCALE

In ecology, spatial scale is usually considered as the product of *grain* and *extent* (Forman, 1995; Wiens, 1989), which, in remote sensing, relate to *resolution* (pixel size) and *area of coverage*, respectively (Lillesand and Kiefer, 2000). A remote sensing scientist will typically define spatial scale as a proportion, a ratio of length on a map to actual length. Small scale, therefore, suggests that a large area is covered; in other words, the difference between actual and mapping size is great (coarse

spatial detail). An ecologist’s typical definition of spatial scale is the level or degree of spatial resolution and spatial extent perceived or considered. Ecologists understand a small-scale study to encompass a small area with fine spatial detail. Overall extent and grain define the upper and lower limits of resolution of a study; they are analogous to the overall size of a sieve and its mesh size (Wiens, 1989). The spatial scale at which measurements or observations are taken influences the recognition of spatial patterns and underlying processes of the environment and of the organisms under study (Wiens, 1989); this has been called *intrinsic* scale, which may determine the type of spatial patterns observed. “The intrinsic scale is a property of the ecological process of interest, for example, tree fall, competition, stomatal control, or microclimate feedbacks, and it is governed in part by the size of the individual organisms (or events) and in part by the range of their interactions with their environment” (Malingreau and Belward, 1992, p. 2291). Others (e.g., Hunsaker et al., 2001) have been keen to understand the *uncertainty* associated with spatial data at different scales.

Remotely sensed imagery is an optimal way to collect spatial data across multiple nested or hierarchical scales; imagery can provide synoptic coverage over large areas, enabling investigations at the landscape scale, or more detailed imagery can be collected representing smaller areas, most practically through some form of sampling framework. As always, limitations exist in the quantities of spatial resolution and area of coverage that can be obtained. Spatial resolution of imagery depends on the sensor spectral sensitivity, and the instantaneous field of view, while the area of coverage depends on the satellite or airborne altitude (swath width) and the instrument total field of view (Lillesand and Kiefer, 2000; Richards and Jia, 1999). Landsat satellites typically cover an area of 185×185 km with a sensor spatial resolution or pixel size of 30×30 m for most of the spectral bands; other satellites carrying Advanced Very High Resolution Radiometer (AVHRR) sensors cover an area of 2394×2394 km with a spatial resolution of approximately 1.1 km. More details on these fundamental concepts are presented in Chapter 2 of this volume.

TEMPORAL SCALE

Temporal scale refers to the frequency with which an observation is made (Lillesand and Kiefer, 2000), but similar to the spatial scale, it is made up of two components, the temporal resolution and the temporal extent. The key to temporal scale is change over time, and this pattern or trend may change with hours, days, months, years, or centuries. Depending on the research question and the object under study, the temporal scale of the investigation can be very different. For each source of imagery, the temporal resolution — a sensor-specific component of scale — must be quantified. Satellites passing frequently over the same area translates into a higher temporal resolution for a given sensor package; for example, the temporal resolution is 24 days for Indian Resource Satellite (IRS)–P2 satellites (Richards and Jia, 1999), but 1 day for satellites carrying the AVHRR (Malingreau and Belward, 1992). In addition, the original start of data collection for different sensor packages determines the maximum possible temporal extent of any earth observation study. Operable satellites launched many years ago translate into a higher temporal extent; for example, the

IRS-P2 satellite was launched in October 1994 (Richards and Jia, 1999), while AVHRR satellites were launched in several National Oceanic and Atmospheric Administration series between June 1979 and May 1991. Clearly, the ability to monitor frequent landscape changes at the temporal scale desired (e.g., daily) may be limited by the temporal resolution and extent of a given satellite platform.

RESEARCH DESIGN AND INTERPRETATION

Understanding the effect of scale on the detection and understanding of patterns and causal mechanisms is one step toward the development of common ecological theories within scales (Wiens, 1989). There is no single proper scale at which all sampling ought to be undertaken (Levin, 1992; Wiens, 1989), and there are no simple rules to select automatically the appropriate scales of attention (Meentemeyer, 1989). Ecological structure, function, and change are dependent on spatial and temporal scale (Turner, 1989). The identification of the appropriate scale to use will depend on the organism or phenomenon under investigation. A species- or phenomenon-centered approach, with recognition of its intrinsic scale to the identification of structure, is most relevant in the research design and analysis of forest landscapes.

Arbitrary scale choices can be avoided by analyzing the variance of measurements across many scales using techniques such as the nearest neighbor method (Davis et al., 2000), semivariance analysis (Meisel and Turner, 1998), and several other univariate (spatial correlograms and spectral analysis) and multivariate methods (Mantel test and Mantel correlogram; Legendre and Fortin, 1989). Statistical approaches are typically based on the observation that variance increases as transitions are approached in hierarchical systems (O'Neill et al., 1986). Peaks of unusually high variance indicate scales at which the between-group differences are especially large, which suggest the representation of the scale of natural aggregation or patchiness of vegetation (Greig-Smith, 1952) or organisms; this is sometimes referred to as the boundary of a scale domain (Wiens, 1989). A method of identifying the appropriate scale of remotely sensed imagery uses a high spatial resolution image characterized statistically and then subsequently collapsed to successively coarser spatial resolutions while calculating local variance (Woodcock and Strahler, 1987). The image resolution at which local variance is highest can be deemed the appropriate remote sensing scale in relation to the structural components of the ground.

PROCESSES GENERATE PATTERNS

Remote sensing of terrestrial ecosystems in support of resource management involves identifying ecosystems and their biological, ecological, and physical characteristics (Franklin, 2001). The definition of an ecosystem and the relevant characteristics vary with the resource managed and the issue under consideration. Therefore, the expectations that ecologists might have of remote sensing will vary; for example, species composition and the physical arrangement of the vegetation can be remotely sensed and used to describe or infer ecosystem attributes using straightforward methods and readily available data. Advances in remote sensing technology continue to expand the capacity to monitor changes of interest in ecosystems and

resource management (Wulder et al., 2004). Forest ecosystems change over time because the trees must grow to survive, due to competition among trees, interactions among trophic levels, and large-scale disturbances. Certain aspects of the current state of ecosystem dynamism can be inferred from individual, remotely sensed images, and other aspects can only be assessed using a time series of images. In this section, we provide ecological background on the remote sensing of ecosystem attributes with special attention to the dynamic nature of these ecosystem attributes, the landscape structure, and composition.

FOREST STAND DYNAMICS

Current understanding of patterns and processes of stand development have been fully described by Oliver and Larson (1996). Their synthesis is useful as a basis for understanding the potential contributions of remote sensing. *Disturbance*, meaning the death of trees that frees growing space, is fundamentally important for stand development. Oliver and Larson (1996) distinguished between autogenic and allogenic forms of disturbance; *autogenic* processes cause death of individual trees for reasons that are particular to the tree and ecosystem, and *allogenic* forms of disturbance arise outside of the affected trees or ecosystem. For ease of explaining the processes involved in stand dynamics and the stand structures that result, Oliver and Larson first described long-term stand development following a major disturbance, including autogenic processes responsible for death of trees, and then incorporated the impacts allogenic forms of disturbance imposed on this underlying pattern of stand development. Oliver and Larson pointed out that stand development has been investigated from two perspectives, one based on describing stand structures and the other based on understanding stand developmental processes. The latter approach has great value to resource management because it leads to greater capacity for predicting changes to stands over time. Individual remotely sensed images may be well suited to the stand structural approach to understanding stand dynamics, while stand development typically requires multitemporal resolution imagery. Ecological knowledge must be used to interpret the remotely sensed images to ensure maximum information extraction occurs from available remotely sensed data (Graetz, 1990).

Forest ecosystems pass through four stages during the course of stand development (Figure 1.2). The period immediately following a major disturbance is the stand initiation stage. During this stage, the important process in stand dynamics is the establishment of a cohort of vegetation. New vegetation becomes established when the preexisting vegetation is killed; the number of species and the number of plants that establish themselves and grow to fill the unoccupied growing space depends on the ecoclimatic zone, site capacity to supply essential materials (nutrients and water), and the relative amount of growing space that is made available and the manner in which it is made available. The period of recruitment ends when the community of trees first comes to fully occupy the available growing space. At this time, the ecosystem enters the stem exclusion stage. Competition among established trees is the dominant process affecting ecosystem development and structure during the stem exclusion stage. Inherent differences among species affect the course of competition and consequently the stand structures that develop. Virtually no growing

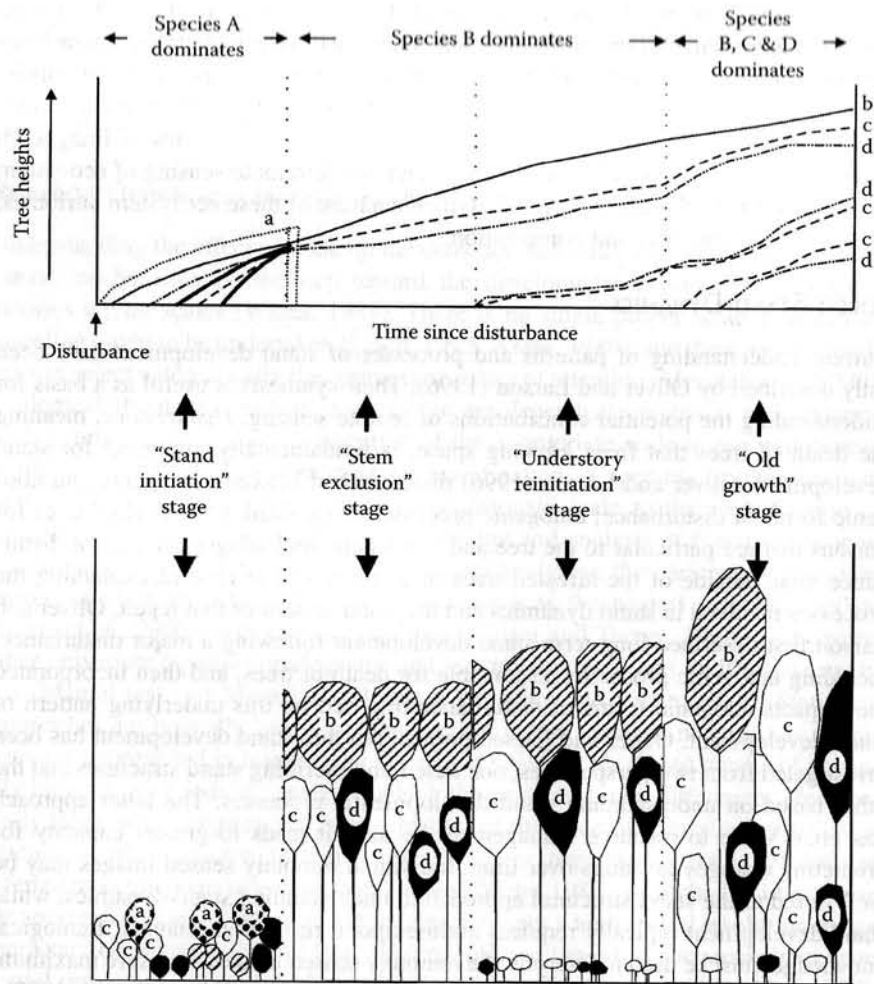


FIGURE 1.2 Schematic stages of stand development following major disturbances. All trees forming the forest start soon after the disturbance; however, the dominant tree type changes as stem number decreases and vertical stratification of species progresses. The height attained and the time lapsed during each stage vary with species, disturbance, and site. (Adapted from Oliver and Larson, 1996.)

space becomes available for the establishment of additional trees as the result of density-dependent mortality (competition). At about the time that the height growth of successful competitors becomes negligible, these trees begin losing their ability to maintain their "grip" on the growing space. This diminished capacity might be abetted by disease or the activities of insects commonly found in the ecosystem and eventually some trees die.

Species that have been less successful in competing in previous years may now expand to fill the vacated growing space and consequently come to dominate the overstory. However, if some of the growing space that comes available is captured by

a ground story, particularly a ground story that includes advanced regeneration of some tree species, then the stand enters the understory reinitiation stage. Stand structure becomes increasingly complex with the onset of the understory reinitiation stage; the advanced regeneration does not have sufficient growing space to form a lower strata of the canopy. Consequently, the ecosystem remains dominated by the cohort of trees that were established after the initial disturbance. At a later time in stand development, the autogenic processes release growing space in sufficiently large areas to cause patches to return to the stand initiation stage, and as a result the ecosystem enters the old-growth stage. With the release of growing space in these patches, advanced growth is released, and other regeneration mechanisms operate to cause a new cohort of trees to become established. The establishment of patches of vegetation of new cohorts continues until all of the original cohort has been replaced, and at this time an old-growth stand exists. In nature, this stage of development is seldom achieved because in many parts of the globe large-scale disturbances return the entire ecosystem to the stand initiation stage. Other forests are influenced by gap-replacing disturbance, and there continues to be considerable debate about the historical frequency of gap versus stand-replacing disturbances. One possible valuable application of remote sensing would be to test some of the assumptions about the frequency and extent of gap versus stand-replacing forest disturbances (Wulder et al., 2004).

Oliver and Larson (1996) presented that it is more common for a variety of tree species ranging from pioneers to long-lived, shade-tolerant species to become established during the stand initiation stage (known as *initial floristics*) than for later seral stage species to become established after early seral stage species have occupied the site, modified the environment, and lived a substantial portion of their life cycle (*relay floristics*). This is in contrast to ideas of early ecologists, who imagined that stand development involved a succession of stand cover types. Moreover, Oliver and Larson (1996) show that forest ecosystems commonly develop a stratified mixed stand structure during the stand initiation and stem exclusion stages. In stratified mixed stands, the pioneer species grow most rapidly in the years immediately following a disturbance and dominate the overstory in the years immediately following the disturbance. Species with inherently slow initial height growth but capable of surviving in shade, albeit with even slower growth rates, sort themselves into lower strata during the early years of the stem exclusion stage. Species that initially dominate the upper stratum are usually shorter lived than the more tolerant species in lower strata, and hence eventually the lower strata are freed from suppression and dominate the overstory. The difference between the initial floristic pattern of stand establishment and the relay floristic pattern has practical significance when interpreting the pattern of stand development of stratified mixed stands. In the past, stratified mixed stands have been sometimes misinterpreted to be uneven-aged stands, whereas in reality members of all strata became established in response to the same disturbance. This distinction is particularly important when devising silvicultural interventions to maintain or promote particular stand structures.

Site characteristics such as microclimate and soil conditions vary spatially, affecting the mix of species that becomes established in the various ecosystems that make up a landscape. During the stand initiation stage, site characteristics can be viewed as environmental "sieves" through which species must pass to become

established. For example, species vary in their capacity to tolerate drought, grow on nutrient-poor soils, become established on cold sites, withstand exposure, and survive in shade. Many remotely sensed images only contain information about the uppermost canopy layer and not about lower strata and the ground story, but knowledge of ecological habits of the tree species and of the stand development patterns operating in the region can be used to better interpret current stages in stand development of the observed ecosystems and their future stand structures. Some promising new image data types with three-dimensional capabilities (e.g., light detection and ranging, LIDAR) are described by Coops et al. (Chapter 2, this volume).

EFFECTS OF DISTURBANCE ON STAND DYNAMICS AND LANDSCAPES

Fire, windthrow, insect attack, and harvesting are examples of allogenic disturbances. Each type of disturbance has a different impact on ecosystems and landscapes, thereby having diverse effects on the stand structure created, the species that can become established in the growing space made available by the disturbance, and changes to the soil and site conditions necessary for tree growth. The frequency and spatial extent of major disturbances affect the proportion of a landscape in each stage of stand development at any point in time. Remote sensing provides data for monitoring disturbances and documenting their effects on each ecosystem in a landscape. These data can provide a means to monitor the subsequent stand development for much larger numbers of ecosystems than could be measured by field surveys.

The type and severity of disturbance affects the success that can be achieved by each regeneration strategy. For example, forest fires commonly consume the forest floor, thereby eliminating advanced regeneration and therefore some species such as balsam fir, which rely on advanced growth to become established after disturbance and are prevented from being a future part of the ecosystem after fire. Clark and Bobbe (Chapter 5, this volume) provide background and an example of using remote sensing for portraying fire impacts, and Hudak et al. (Chapter 8, this volume) include fire disturbance in the presented case study. A contrasting example on the role of disturbance in favoring particular regeneration strategies is what happens when the disturbance removes selected species from the overstory but does not eliminate the ground story, such as occurs with outbreaks of defoliating insects (Seymour, 1992; Figure 1.3). In these instances, species regenerating from advanced growth might have an advantage in acquiring the growing space made available, and species that are intolerant of shade might be limited in their capacity to regenerate. Hall et al. (Chapter 4, this volume) develop an approach useful in mapping insect disturbance using remote sensing.

Disturbances can create atypical stand structures in ecosystems that had been in the stem exclusion or understory reinitiation stages at the time of disturbance. These relatively young stands would typically have complete overstories of trees belonging to one cohort if only allogenic processes were in play, but after a major disturbance that does not completely eliminate the original cohort, these stands will have more than one cohort visible from above. Such disturbed stands will exhibit greater spatial variability in stand structure than undisturbed stands. It is possible that these disturbed stands will have structure commonly associated with old-growth stands, and these structures might mislead some into believing that they are stable, old-growth

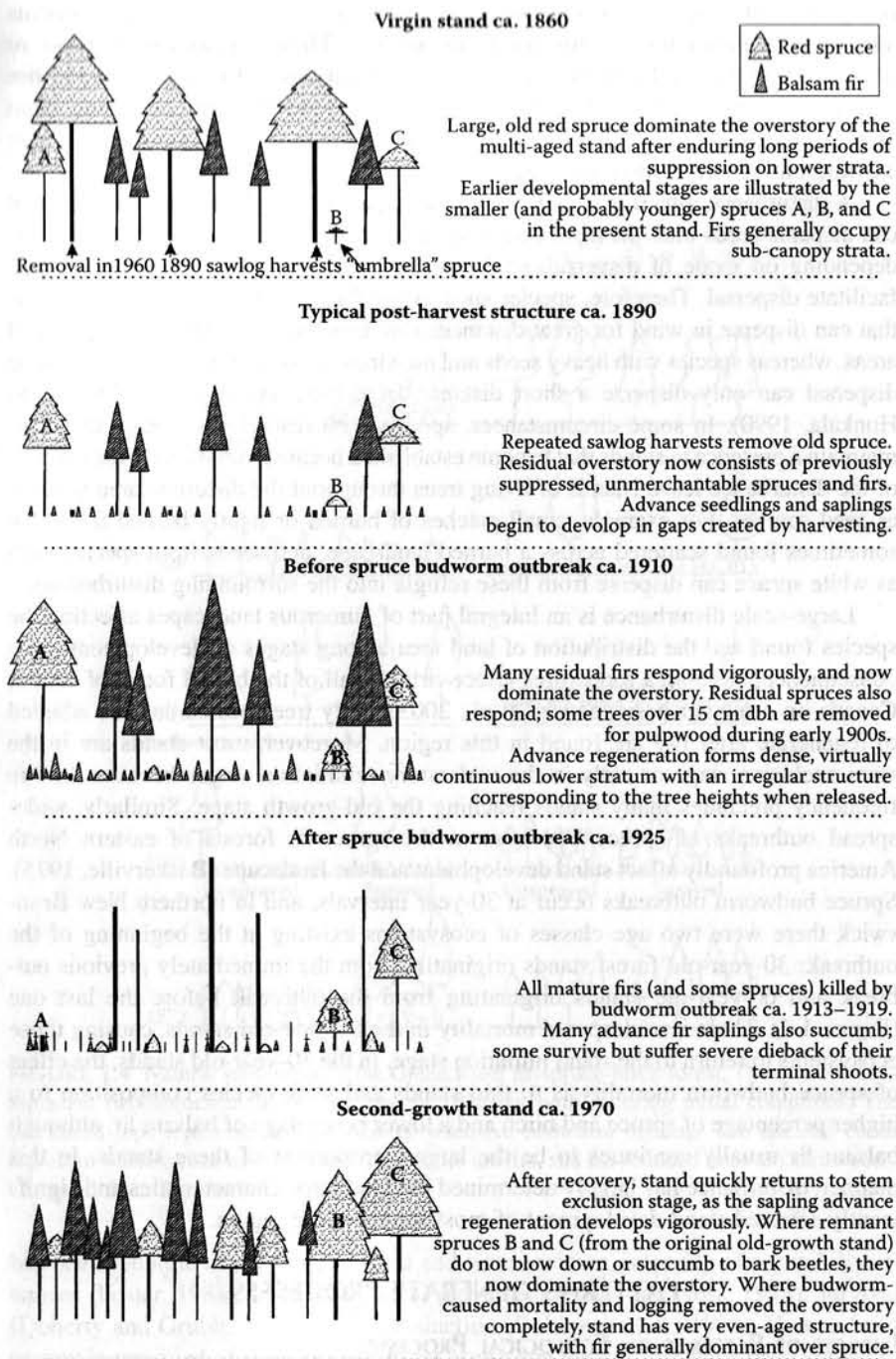


FIGURE 1.3 Development of typical spruce-fir stand after logging and budworm attack circa 1860-1970. (Adapted from Seymour 1992.)

ecosystems. Disturbance can also change the capacity of a site to supply the nutrients and water required for establishment and growth. These effects can increase or decrease growth, and the impact on growth might vary over time. For example, fire can release nutrients held in recalcitrant organic matter, thereby increasing plant growth immediately after the fire, but fire also decreases the total stock of nitrogen existing on the site, which may decrease long-term potential productivity.

A disturbance affecting a large area can reduce the number of tree species that can disperse seeds onto the disturbed areas. Dispersal distance varies among species depending on mode of dispersal, seed size, and special appendages on seeds that facilitate dispersal. Therefore, species such as trembling aspen with very light seeds that can disperse in wind for great distances can disperse seeds onto large disturbed areas, whereas species with heavy seeds and no wings or other appendages to facilitate dispersal can only disperse a short distance from their site of origin (Burns and Honkala, 1990). In some circumstances, species with restricted dispersal distances maintain a presence in stands that become established because variations in the severity of the disturbance leave islands of living trees throughout the disturbed area to serve as seed sources. For example, small patches of burned or lightly burned forest are sometimes found scattered across a burned landscape, and seeds from species such as white spruce can disperse from these refugia into the surrounding disturbed area.

Large-scale disturbance is an integral part of numerous landscapes affecting the species found and the distribution of land area among stages of development. As a consequence, large-area forest fires affect virtually all of the boreal forest of central Canada on a regular basis (Stocks et al., 2003). Only tree species that are adapted to regenerate after fire are found in this region. Moreover, most stands are in the stem exclusion stage or early in the understory reinitiation stages because the fire frequency precludes many stands reaching the old-growth stage. Similarly, widespread outbreaks of spruce budworm in the spruce-fir forests of eastern North America profoundly affect stand development and the landscape (Baskerville, 1975). Spruce budworm outbreaks occur at 30-year intervals, and in northern New Brunswick there were two age classes of ecosystems existing at the beginning of the outbreak: 30-year-old forest stands originating from the immediately previous outbreak and 60-year-old stands originating from the outbreak before the last one (Figure 1.4). There is widespread mortality in the 60-year-old stands, causing those ecosystems to return to the stand initiation stage. In the 30-year-old stands, the effect of spruce budworm mortality is to thin stands and shift species composition to a higher percentage of spruce and birch and a lower percentage of balsam fir, although balsam fir usually continues to be the largest proportion of these stands. In this manner, disturbance has largely determined the landscape characteristics and significantly affected stand development of most stands in the region.

PATTERNS GENERATE PROCESSES

IMPACTS OF PATTERNS ON ECOLOGICAL PROCESSES

Just as physical and ecological processes generate landscape structure, landscape structure influences physical and ecological processes. Specifically, landscape pattern

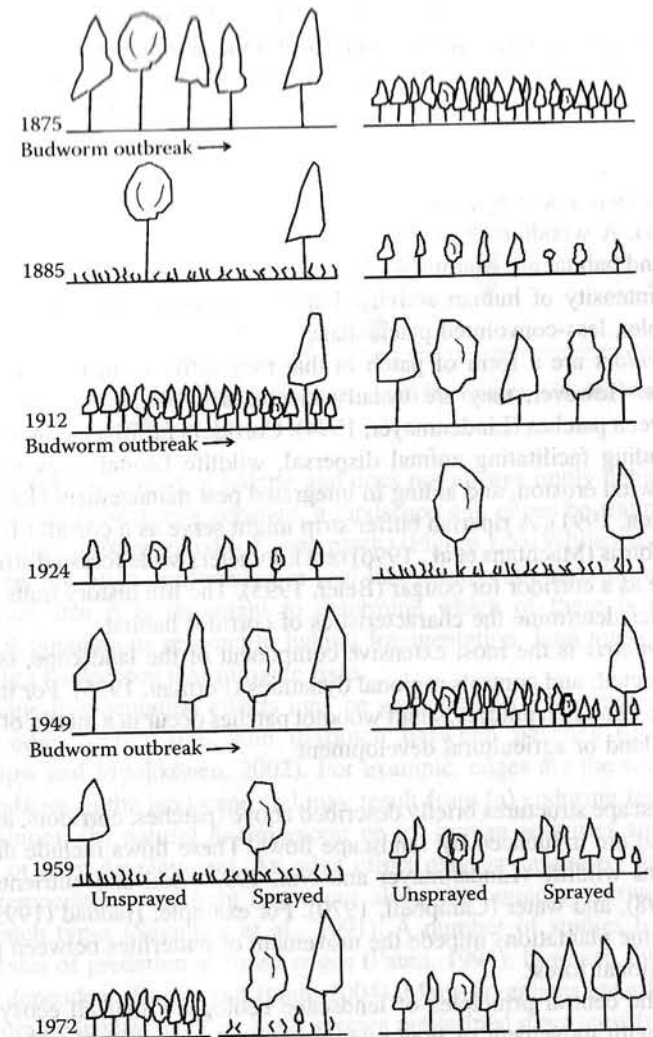


FIGURE 1.4 Natural succession in the Green River fir-spruce-birch forest. The two columns represent two sequences both beginning in 1875 but with differing initial conditions. The blackened tops represent disfiguration by intensive budworm feeding. The hatched cones represent white spruce trees, the unshaded cones are firs, and the rounded trees are hardwoods. (Adapted from Baskerville, 1975.)

has been found to affect rates of wind and water erosion, intensity of natural disturbances (Foster, 1988), plant and animal movement (Beier and Noss, 1998), survival (Doherty and Grubb, 2002), and reproduction (Robinson et al., 1995). Here, a brief review is provided of the components of landscape pattern that have been shown to exert a strong influence on ecological processes. Such patterns are considered priorities for measurement in remote sensing (Gergel, Chapter 7, this volume).

All landscapes are characterized by degrees of heterogeneity (patchiness) at different scales; differing substrates (soils, bedrock), natural disturbances (fire, insect outbreaks), and human activity (forestry, road building) all create patchiness across a landscape. The “patch-corridor-matrix” model (Forman, 1995) has become a central component of landscape ecology in theory and in practice:

1. A *patch* is a homogenous area that differs from its surroundings (Forman, 1995). A woodlot surrounded by farmland and a wetland immersed in upland habitat are examples of patches. Patch shape often correlates with the intensity of human activity. Intense human activity often results in simpler, less-convoluted patch shape
2. *Corridors* are a form of patch in that they differ from the surrounding areas. However, they are usually identified as strips that aid in flows between patches (Lindenmayer, 1994). Corridors fulfill a number of roles, including facilitating animal dispersal, wildlife habitat, preventing soil and wind erosion, and aiding in integrated pest management (Barrett and Bohlen, 1991). A riparian buffer strip might serve as a corridor for forest songbirds (Machtans et al., 1996) or a kilometers-wide forested strip could serve as a corridor for cougar (Beier, 1995). The life history traits of each species determine the characteristics of corridor habitat
3. The *matrix* is the most extensive component of the landscape, is highly connected, and controls regional dynamics (Forman, 1995). For instance, in the Canadian prairies, small woodlot patches occur in a matrix of natural grassland or agricultural development

The landscape structures briefly described above (patches, corridors, and matrix) influence, and are influenced by, landscape flows. These flows include diverse elements such as wildlife (Lindenmayer and Nix, 1993), soil and nutrients (Stanley and Arp, 1998), and water (Campbell, 1970). For example, Haddad (1999) demonstrated that pine plantations impede the movement of butterflies between patches of early successional forest.

One of the central principles of landscape ecology is that all ecosystems are interrelated, with movement or flow rate dropping sharply with distance but more gradually between ecosystems of the same type (Forman, 1995). Thus, a very heterogeneous landscape (with many patch types) is marked by a relatively low degree of movement (flow) and a large amount of resistance.

FRAGMENTATION, CONNECTIVITY, AND ISOLATION

Fragmentation is the “breaking apart” of habitat. This can occur as a result of natural processes such as forest fires or anthropogenic disturbances such as road building or timber harvesting. Different views exist about definitions regarding “habitat loss” and “fragmentation.” Wilcove et al. (1986) suggested that fragmentation is a combination of habitat loss and isolation; however, recently the emerging consensus is that habitat loss and fragmentation should be described separately (Andrén, 1994; Fahrig, 1998, 2002; Mazerolle and Villard, 1999). Fragmentation is often defined

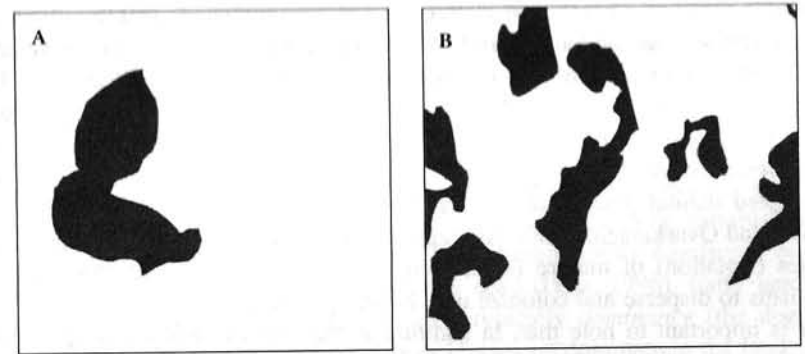


FIGURE 1.5 Unfragmented (A) versus fragmented (B) landscapes with the same amount of habitat present in each landscape.

purely as the breaking apart of habitat and does not always imply habitat loss. For instance, holding habitat area constant, a landscape can either be fragmented (i.e., many patches) or unfragmented (i.e., one patch) (Figure 1.5). While habitat loss and fragmentation are often confounded in real landscapes (i.e., they occur together), we emphasize that it is important to determine which of these is ecologically important; if populations respond to habitat fragmentation, land managers may be able to design landscapes that mitigate risks.

Landscape fragmentation effects may be grouped into a few major categories, including edge, patch size, and distance between patches (connectivity) (Schmiegelow and Mönkkönen, 2002). For example, edges are the result of vegetation boundaries in the landscape and may result from (a) enduring features (soils, drainage, slope); (b) natural disturbances; or (c) human activities such as forest harvesting or farm development. An edge effect may be caused by differences in moisture, temperature, and light that occur along the boundary between different adjacent patch types (Saunders et al., 1991). A number of studies have reported increased rates of predation at forest edges (Paten, 1994); however, this appears to be context dependent (Batáry and Báldi, 2004). Many organisms are affected by the size of favorable habitat patches. Such species are termed *area sensitive* (Freemark and Collins, 1992). Robbins et al. (1989) found that “area” was one of the most significant habitat features for many neotropical migrant bird species. Area sensitivity has also been observed for amphibians (Hager, 1998). Although some debate exists about the area sensitivity of plants, a number of published studies reported lower genetic diversity and higher rates of extinction in smaller populations (Bell et al., 1991; Damman and Cain, 1998).

In some cases, fragmented landscapes have been shown to exhibit the same characteristics as those observed in island archipelagos by MacArthur and Wilson (1967). Isolation of habitat seems to compound the effect of small patch size on the ability of some species to persist and recolonize. These findings can be understood better if placed in the context of the concept of *metapopulations*. The metapopulation concept requires that population dynamics be studied beyond the scale of local populations. “Equilibrium,” rather than occurring in a single, local population, might

occur as a result of a number of interconnected subpopulations that are distributed across a region (Husband and Barrett, 1996). Population dynamics are the result of a series of local extinctions and recolonizations in habitat patches (Levins, 1970). If the subpopulation of one patch becomes extinct, then it may eventually be recolonized by dispersers from a subpopulation that exists in a neighboring patch. This is the "rescue effect"; for a species to spread or persist, individuals must colonize unoccupied habitat patches at least as frequently as populations become extinct (Hanski and Ovaskainen, 2000). As fragmentation progresses, the distance between patches (isolation) of mature forest increases. This distance limits the ability of organisms to disperse and colonize new habitat patches.

It is important to note that, in addition to the studies briefly described above indicating a significant influence of landscape pattern on species distributions, there are many studies that reveal only weak or nonexistent landscape pattern effects (Delin and Andrén, 1999; Game and Peterken, 1984; McGarigal and McComb, 1995; Schmiegelow et al., 1997; Simberloff and Gotelli, 1984). Indeed, the majority of evidence indicates that it is habitat loss rather than fragmentation per se that is the most important influence on species occurrence, reproduction, and survival (Fahrig, 2003). This appears particularly to be the case in forest mosaics (for reviews, see Bender et al., 1998; Mönkkönen and Reunanen, 1999). This idea reinforces the notion that it is important for remote sensing to provide accurate classifications of landscape composition in addition to input data to analyses of landscape configuration.

Andrén (1994) proposed that landscape configuration is only important below a threshold in the proportion of suitable habitat at the landscape scale. Only at low levels of habitat are patches small and isolated enough to result in patch size effects or restrictions in movement (Gardner et al., 1991). This results in multiplicative impacts of fragmentation on habitat loss. A number of theoretical studies supported this "fragmentation threshold" hypothesis (Fahrig, 1998; Hill and Caswell, 1999; Wiegand et al., 2005; With and King, 1999), but it has rarely been demonstrated in nature (Trzcinski et al., 1999). However, this may be because "suitable habitat" has rarely been defined according to the requirements of individual species.

PREDICTIVE MODELING OF SPECIES OCCURRENCE USING GEOSPATIAL DATA

To determine rates of change in the amount and pattern of habitat at any spatial scale, it is clearly necessary to have accurate definitions of habitat for different species. Remotely sensed data have been used extensively to develop habitat models; these are inexpensive to develop in comparison to models based on detailed vegetation data collected in the field (Osborne et al., 2001; Vernier et al., 2002) and provide an opportunity to generate predictions about species distributions over large spatial extents at relatively fine resolutions (Betts et al., 2006; Gibson et al., 2004; Linke et al., 2005). Such models are usually probabilistic in nature, but a wide range of modeling techniques are available, including classification trees, neural networks, generalized linear models, generalized additive models, and spatial interpolators (Segurado and Araujo, 2004). Models have been developed to cover aspects as diverse as biogeography, conservation biology, climate change research, habitat or species management (Guisan and Zimmermann, 2000), and vegetation mapping (J.

F. Franklin, 1995). As the resolution of remotely sensed data improves, the range of potential applications is likely to increase (Coops et al., Chapter 2, this volume; Wulder et al., 2004).

LANDSCAPE METRICS

To study the effects of landscape structure on ecological processes, it is necessary to develop methods to quantify spatial patterns into measurable variables before links to ecological processes can be determined (Frohn, 1998). *Landscape metrics*, or indices, have been developed to meet this need (Diaz, 1996). Early landscape metric studies presented only a few metrics, typically *dominance* (the degree to which certain kinds of landscape patches or classes predominate in the landscape), *contagion* (the extent to which similar patches are clumped together), and *shape* (the form of an area or a patch as determined by the variation of its border) (Forman, 1995; O'Neill et al., 1988). Today, an extensive array or suite of landscape metrics and indices exists (Elkie et al., 1999; McGarigal and Marks, 1995). The suite of available landscape metrics can be considered to include specific measures of area, edge, shape, core (or interior) area, nearest neighbor/diversity/richness/evenness, interspersion/juxtaposition, contagion/configuration, and connectivity/circuitry (Gergel, Chapter 7, this volume; McGarigal et al., 2000).

The large number of metrics that have been developed to describe and quantify spatial structure often appears to be overwhelming, and the question of metric redundancy has frequently arisen. Initially, use of metrics that have known ecological relevance and application should be considered. However, some standard approaches have been employed to deal with the issues of redundancy and number of metrics for a given application. For example, Riitters et al. (1995) used a factor analysis to reduce to a few components more than 50 specific landscape metrics applied to 85 maps of land use and cover in the United States. Recent studies have concluded that it is possible to identify a parsimonious suite of metrics using principal components analysis to characterize much of the spatial patterns existing in a boreal forest landscape subject to many common disturbance processes (S. Cushman, personal communication, April and October, 2002; Linke and Franklin, in press).

In addition to the issue of appropriate metric selection, there are several other factors known potentially to influence the interpretation and use of landscape metrics (Haines-Young and Chopping, 1996), including, for example, metric uniqueness, sensitivity, abrupt versus continuous edges, statistical quantification, study area extent, and scale or resolution. Another important characteristic of landscape metrics to consider is their actual behavior over a wide range of landscape structures; the instance of nonlinear landscape metric behavior over scale is briefly mentioned here:

1. Hargis et al. (1998) investigated the relationships between six landscape metrics and the proportion of two landcover types across simulated landscapes, also controlling for the size and shape of patches. Most metrics were linearly associated at the lower landcover proportion range but had nonlinear associations at higher proportions, which limits their direct comparability across different regions

- Such nonlinear metric behavior was also found in simulated landscapes in a study of dispersal success on fractal landscapes (With and King, 1999) and in a study specifically designed to detect metric behavior under controlled conditions (Neel et al., 2004)

An awareness of all of these interpretational effects and metric behavioral limitations must be embedded in any landscape quantification attempt. A detailed discussion of spatial pattern analysis using landscape metrics is presented by Gergel in Chapter 7.

CONCLUSION

Understanding forest disturbance and spatial pattern is increasingly recognized as essential to effective and sustainable forest management in many forest environments around the world. We hope that this introduction has provided some insight into the challenges that are further elaborated in later chapters; the developing appetite in landscape ecology and conservation biology for spatial data and models that work with complex phenomena; the relationships between pattern and process, process and pattern; the specific details of remotely sensed and GIS data selection; the importance of scale; the myriad issues in fire and insect, forest harvesting, and other disturbance monitoring; and the emerging role of landscape metrics and modeling landscapes. The literature and practice of forest disturbance and spatial pattern using remote sensing and GIS approaches are diverse and increasing at an astonishing rate as new perspectives and insights take hold. We expect this presentation will be useful to those involved in this interesting and exciting endeavor, in the implementation and continued development of remote sensing and GIS approaches, and in their application to forest ecosystems and processes. We anticipate progress in these areas will help shape future directions in the important work of forest resource mapping, monitoring, and management.

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Methodological Implications

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