Chapter 15

Lynx Conservation in an Ecosystem Management Context

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Abstract—In an ecosystem management context, management for lynx must occur in the context of the needs of other species, watershed health, and a variety of products, outputs, and uses. This chapter presents a management model based on the restoration of historical patterns and processes. We argue that this model is sustainable in a formal sense, practical, and likely to provide for the needs of a variety of species, including lynx. Because our knowledge of lynx biology and disturbance ecology is limited, implementation of this model will be experimental and must be accompanied by a well-planned monitoring program.

Introduction

Our knowledge of lynx ecology and population dynamics in southern boreal forests is limited and based on information obtained at only a few geographic localities (Chapter 13). Because of these information gaps, the conservation of lynx populations in this region must proceed initially with limited knowledge about their habitat relationships and, consequently, with a limited understanding of how to design forest management strategies that will provide for the persistence of lynx populations in the contiguous United States. However, even if we possessed perfect knowledge, management aimed solely at maintaining or improving habitat conditions for a single species across such a broad geographic area would conflict with many other resource management objectives. Managing habitat for lynx may not provide for the needs of other species of conservation concern, or for biodiversity, watershed health, recreation, grazing, mining, timber, or wilderness uses, yet each has a statutory or regulatory basis and a constituency on public lands.

In addition, the scale at which forest management strategies are defined will be a critical consideration for lynx conservation. Because lynx occupy large home ranges and occur at low densities (about one lynx/50 km^2; Chapter 13), the long-term viability of lynx populations cannot be achieved at the spatial scale of relatively small parcels of public land, or even larger units such as individual National Forests or National Parks. Consequently, we believe that lynx conservation in the contiguous United States can only succeed as part of an ecosystem management strategy that is designed to address the needs of a variety of potentially conflicting resource uses over long periods of time and broad spatial scales. In this chapter, we discuss ways in which lynx conservation could be approached within the context of ecosystem management.

The Concept of Ecosystem Management

The concept of ecosystem management dates back to at least 1988 (Agee and Johnson 1988) and, since that time, a variety of definitions have been published (Grumbine 1994), many of which reflect contrasting areas of emphasis. To various constituencies, ecosystem management represents an emphasis on landscape-scale analysis, on science-based management, on adaptive management, on interagency cooperation, or on ecological integrity (Franklin 1997).

The Ecological Society of America defined ecosystem management as "management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function (Christensen et al. 1996)." The central goal or value of ecosystem management is sustainability. We consider the eight components of ecosystem management presented by Christensen et al. (1996) to represent the most useful framework for discussing this concept:

1. Long-term sustainability as a fundamental objective
2. Clear, operational goals
3. Sound ecological models
4. An understanding of complexity and inter-connectedness
5. Recognition of the dynamic nature of ecosystems
6. Attention to context and scale
7. Acknowledgment of humans as ecosystem components
8. A commitment to adaptability and accountability

Sustainability

Long-term ecological, social, and economic sustainability are the central goals of ecosystem management (Christensen et al. 1996). To achieve these goals, management must ensure that ecological resources and processes will be maintained in perpetuity, so that future options for management are not compromised. Operationally, this means keeping all the "pieces" of ecosystem structure, function, and composition, while addressing productivity of wood, water, livestock, minerals, and other resource outputs expected from public lands (Franklin 1993). For example, a forested landscape will only provide a sustainable output of timber if the rate of timber harvest is low enough that the first stands harvested have re-grown to rotation age before the last stand is harvested. It is this aspect of sustainability that is embodied in the ideals of even-aged silviculture and maximum sustained yield. Yet sustainability includes more than timber; it requires maintaining the productivity of the soil (for timber, forage, and other resources) and, more broadly, sustaining ecological function so that native species are perpetuated. Sustainability was once narrowly defined in terms of resource outputs, but the concept now encompasses both the state of the ecosystem and associated resource outputs. The foundation of sustainability is ecological; only by sustaining ecological function can social and economic sustainability be achieved (Committee of Scientists 1999).

Applying a sustainable management model will not necessarily produce a sustainable system, however. Most sustainability paradigms are based on maintaining a state of equilibrium. This is possible in a deterministic world, but seldom occurs at small spatial scales in a stochastic system prone to disturbance, such as in the boreal or subalpine forests typical of lynx habitat. In high-severity fire regimes (Agee 1993), for example, the majority of the land base is burned by only a few large, catastrophic fires (McKelvey and Busse 1996; Minnich 1983). For a system prone to such disturbances to be sustainable, it must be resilient. Resiliency in this context is created by scale; ecosystems are more resilient and, hence, more stable when considered at very large spatial scales. In other words, maintaining ecosystem stability will be most attainable if the size of the extreme events is small in relation to the total area.

Ecosystem Context

One of the primary causes for the loss of ecosystem resiliency (i.e., loss of sustainable character) is loss of context resulting from management activities (Allen and Hoekstra 1992). By this, the authors mean a disparity between the spatial and temporal scales of critical ecosystem processes in the natural system, and the area and planning horizon of management activities. For example, cutthroat trout populations were stable in the western United States during historical times, despite the periodic occurrence of large disturbances within the range of the cutthroat, such as the 1910 fire in northern Idaho and western Montana that burned over 1 million ha. Because this fire occurred in only a portion of the species' range, rivers and streams in disturbed areas that lost cutthroat trout habitat eventually recovered and were recolonized by cutthroat trout. Rivers provided connectivity between disturbed and undisturbed areas. Although trout populations in individual streams were prone to local extirpation from such natural disturbances, they were embedded in the larger context of the western United States cutthroat metapopulation. Thus, even catastrophic disturbances had little impact on the long-term sustainability of this system for cutthroat trout.

The situation is similar for lynx, except that the spatial scale that must be considered is much larger. In southern boreal forests, male and female lynx occupy home ranges that average about 150 km^2 and 75 km^2, respectively (Chapter 13), and interbreeding populations occupy an area many times that size. Throughout most of Canada and Alaska, lynx numbers cycle in abundance every 10 years in a pattern of lagged synchrony within the species' range; population peaks in the center of the continent occur approximately two years before they occur in Alaska or Quebec (Chapter 8; see also Ranta et al. 1997). While it is possible that these patterns are independent of one another and driven by exogenous synchronizing factors, such as climate (Elton and Nicholson 1942), the lagged pattern of synchrony in population dynamics suggests some degree of connectivity through dispersal among northern populations (Chapter 2). At least in the 1960s and 1970s, when Canadian population peaks were unusually high, these patterns of lagged synchrony also appeared in trapping records from Minnesota, Montana, and, more weakly, Washington state (Chapter 8). Thus, it is possible that lynx population dynamics in northern tier states of the contiguous United States are linked to those occurring in Canada.

The potential geographic scope of ecosystem context for lynx populations in the contiguous United States is suggested by the historical analysis of lynx distribution presented in Chapter 8. In Washington, lynx populations have persisted only in relatively large blocks of habitat in north-central and northeastern Washington that are adjacent to lynx habitat in southwestern Canada. Similarly, lynx populations in the Clearwater drainage in western Montana occur in the context of the Bob Marshall Wilderness and direct connections to larger habitat areas in Canada. In addition, the documented decline of lynx in New Hampshire is believed to have resulted more from the loss of large-scale habitat connectivity with Quebec than with the loss or alteration of local habitat conditions (Chapter 8; Litvaitis et al. 1991). Thus, for reasons we do not yet fully understand, ecosystem context for lynx appears to occur at least at the scale of ecoprovinces (Bailey 1998; Chapter 8).

Diversity and Complexity

In natural systems, the age distribution of forest stands varies dramatically from that produced under most forest management models. In natural landscapes, stand age distributions produced by large-scale stochastic events, such as forest fires, are more complex than those that result from deterministic, even-aged models of forest management. In most boreal forest systems, the constant-probability stochastic process model (hereafter referred to as the negative exponential model) results in a distribution of stand ages that approximates a negative exponential curve (Fig. 15.1a). For boreal forests with long fire-return intervals, this model approximates natural stand-age distributions, or at least one that is much closer to natural conditions than the truncated distribution produced by even-aged cutting practices (Fig. 15.1c; see also Chapter 3). By maintaining old-growth forests in the landscape, the negative exponential model also retains mature forests, the most temporally stable element on the landscape. Older forests provide a stabilizing influence that is likely to increase ecosystem resiliency. Resulting landscapes have greater levels of complexity and diversity because all age classes are maintained within the landscape.

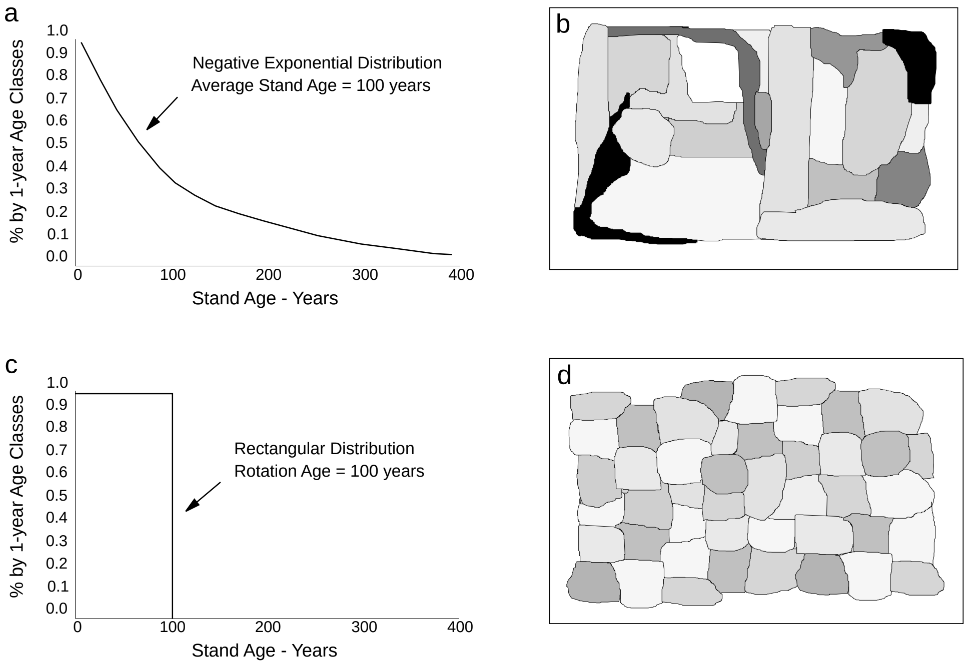

Figure 15.1—A comparison of negative exponential (a, b) and even-aged (c, d) planning paradigms. Graphs on the left depict the stand age distributions, and hypothetical landscape views on the right depict examples of how stands of varying ages, sizes, and shapes could be distributed across the landscape under each planning paradigm. Relative stand age is indicated by increasingly darker shading in the polygons.

Humans as Ecosystem Components

The evaluation of humans as ecosystem components includes both the effects of human perturbations and the needs of society for ecosystem goods and services. Unquestionably, much of the context-based stability that characterized pre-settlement North American ecosystems has been lost. Urbanization and agriculture have increased the isolation and fragmentation of natural areas and reduced the resilience of some parts of those systems to extreme disturbance events; further, this loss of ecosystem context is unlikely to be reversed. We do not believe that a system of reserves maintained by policy or regulation, embedded in a increasingly fragmented and non-natural matrix, can provide for sustainable lynx populations. Inevitable large-scale catastrophes preclude this being a sustainable system from the standpoint of lynx conservation; catastrophes will remove reserves, and there is no mechanism for their replacement or restoration in a reserve/matrix approach. Allen and Hoekstra (1992) have argued that one role of management in sustainable ecosystems is to replace functions that are generally provided in the natural context, but have been lost due to human influences. For this reason, we argue for a system of active management in those land designations where it is permitted. Management of these lands must be cognisant of and limited by the dependency of various species on the condition of these areas given the likelihood of divergent management in adjacent private lands, as well as inevitable natural disturbances.

Simple and rigid prescriptions cannot succeed when the management goals are as complex as maintaining multiple species of wide-ranging vertebrates in natural ecosystems. We simply do not have the knowledge to prescribe procedures that will always achieve these objectives, and we probably never will. Hence, managers cannot be simply technicians; they must be creative thinkers and explorers who use adaptive management to achieve complex and shifting resource management objectives (Walters 1986). We need to build into the system the mechanisms to learn as we go, and the system must be flexible; managers must be able to react quickly to new information to minimize ecosystem damage and maximize conservation opportunities. The necessary regulatory framework must also be in place to allow managers to do so.

Single-Species Habitat Management

Several of the single species-based habitat management models used in the past embodied some of the concepts of ecosystem management because the goal of these approaches was to provide for a broad array of ecological values by ensuring that the habitat needs of key species were met. The National Forest Management Act (NFMA) of 1976, with its attendant planning regulations, included the concept of "indicator" species as a means of maintaining biological diversity on National Forest lands. Most indicator species were chosen on the basis of their presumed sensitivity to ecological changes resulting from management actions. However, the concept was widely criticized as being of little value for monitoring and responding to changes in ecological conditions resulting from management (Patton 1987; Landres et al. 1988), and its use was largely abandoned. "Umbrella" species are conceptually similar to indicators, but their use is based on the assumption that protecting habitat for rare species with large spatial needs will provide for the needs of a variety of more common species with smaller spatial requirements (Berger 1997). Another link between single species with higher levels of biological organization is the concept of "keystone" species (Paine 1969). Keystones are species that make disproportionately large contributions to community or ecosystem function relative to their abundance (Mills et al. 1993).

The Endangered Species Act (ESA) acknowledges the importance of ecosystems, but conservation efforts mandated by the ESA typically have focused on restoring populations of individual species (Christensen et al. 1996). Recently, however, conservation strategies for certain listed species, including the northern spotted owl (USDA and USDI 1994a), bull trout (Rieman and McIntyre 1993), and Pacific salmon stocks (USDA and USDI 1994b), have adopted multi-species habitat conservation approaches. These efforts exemplify the ongoing transformation of endangered species conservation from a focus on individual species, to one that more closely reflects the concepts embodied in ecosystem management. While single-species or "fine-filter" conservation will continue to be mandated by law, we believe that the long-term viability of multiple species will best be addressed with a combination of ecosystem-scale or "coarse-filter" strategies (Haufler et al. 1996; Hunter et al. 1988) and fine-filter strategies for sensitive or rare species whose persistence may require special management actions.

Application of Ecosystem Management to Lynx Conservation

The coarse-filter strategy involves the analysis of pattern and process at large spatial scales. The overall goal is the retention of most biotic diversity and ecological functions through the restoration of ecosystem processes. In this strategy, fine-filter approaches, such as the maintenance of indicator, umbrella, or keystone species, serve as a check on the efficacy of the coarse filter. They also provide insights into modifications that may be necessary to meet the needs of rare or sensitive species not met with a coarse-filter approach. Applying both a coarse and fine filter to the process of ecosystem management increases the probability that key species will be retained, while generating a dynamic and sustainable landscape. For species of concern, such as the lynx, fine-filter measures are mandated by law. The ecosystem management paradigm, when applied to the conservation of lynx, incorporates these species-oriented measures into broad, coarse-filter landscape planning.

Broad-Scale Patterns of Lynx Occurrence

Our analysis of the current and historic distribution of lynx populations in the United States indicates that large, contiguous areas of suitable habitat are necessary for population persistence (Chapter 8). All lynx populations that have long histories of commercial trapping in the contiguous Unites States are those whose geographic range is contiguous with or adjacent to larger areas of suitable habitat north of the Canadian border, including populations in Washington, Montana, and Minnesota. Because individual lynx may disperse >500 km in straight-line distance (Chapter 9), lynx populations in southern boreal forests may be augmented at various times by immigrations from the Canadian taiga. However, the degree to which the persistence of southern populations depends on the rescue effect of immigrations from Canada is unknown (Chapter 8).

We cannot assume that lynx populations in the contiguous United States will be maintained by dispersal of lynx from Canada, nor that connectivity with larger habitat areas in Canada will be maintained in perpetuity. Although cooperative conservation efforts with Canadian land management agencies should be explored in all areas of adjacent lynx habitat, we believe that lynx conservation efforts in the contiguous United States should be addressed at geographic scales that will provide for the persistence of resident populations of lynx, regardless of periodic augmentations that may occur from other areas. Clearly, ecoprovince-wide planning is necessary to provide the broad-scale information necessary for effective conservation of lynx.

The following calculation (included here for illustrative purposes only) provides a sense of the smallest scale at which conservation planning will need to be addressed. The estimated density of a local population of 25 resident lynx in north-central Washington studied with radiotelemetry from 1980 to 1987, was about one lynx per 50 km^2 (Chapter 13; Brittell et al. 1989, unpublished; Koehler 1990). Thus, conservation of this small group of animals, which is probably not viable as a geographically closed population (see Chapter 2), would require a planning area at least 1,250 km^2 in size (25 x 50 km^2). However, this estimate represents a minimum area requirement over a relatively short period of time; to provide sufficient habitat for the population size to fluctuate around a long-term mean of 25 animals, the actual conservation area would need to be considerably larger.

Broad-Scale Habitat Management

Unlike organisms that are tightly linked to a particular forest condition, lynx use a variety of forest age and structural classes (Chapter 13). Preferred prey for lynx are also associated with a variety of forest types. Red squirrels are closely tied to mature, cone-producing forests, whereas snowshoe hares generally reach highest abundance in younger seral stages. Dense horizontal structure appears to be important for snowshoe hares, but such conditions develop at different stages in different forest types (Chapters 6 and 7). These observations suggest that forests managed for lynx should contain a mixture of age classes and structural conditions. How such habitats should be interspersed to benefit lynx, and how to maintain this interspersion in space and time, therefore become central issues in habitat management.

Existing lynx habitat management plans in the contiguous United States are generally focused at relatively small spatial scales, and emphasize the production of young forest to provide foraging habitat, and the maintenance of a few, small patches of older forest to provide denning habitat (e.g., Washington State Department of Natural Resources 1996). These plans are based on the premise that if a few small areas of old-growth forest are provided within the matrix for denning habitat, landscapes managed primarily for a spatial and temporal mosaic of high-quality snowshoe hare habitat will provide for the long-term persistence of lynx populations (Koehler and Brittell 1990). However, this approach is based on untested hypotheses and does not address the divergent needs of other species that inhabit these landscapes. When faced with uncertainty concerning the biological requirements of an organism or of a group of organisms, land management plans must be conservative in their retention of habitat components. Because we know little of the details of lynx habitat relationships at the stand scale or the necessary juxtaposition of stand types, landscape-scale habitat management strategies should provide for a continuum of stand ages in a variety of spatial configurations.

The maintenance of all age classes in commercial forests has been viewed as problematic because even-aged forest management removes all stands older than the rotation age, eliminating both habitat and future management options for organisms associated with late-successional forests. On the other hand, removing all disturbance from a site by suppressing fire and refraining from cutting will, in most areas, preclude stand replacement and impact organisms associated with younger forest conditions. Our challenge, from the perspective of maintaining lynx and their prey in the context of ecosystem management, is to design management strategies that result in dynamic, sustainable landscapes that approximate the composition of natural systems.

Natural Disturbance as a Management Template

Natural disturbance in boreal forests, due to its stochastic nature, tends to generate forests composed of widely varying ages and conditions (Chapter 3). These conditions emerge as a direct consequence of disturbance stochasticity; one drainage may burn frequently, whereas an adjacent drainage may avoid fire for centuries. For example, within a burned drainage variation in wind patterns may leave unburned patches, and diurnal variation in moisture and temperature during a fire may result in some areas having high tree survival, whereas others do not. This stochastic process leads to a landscape with a mosaic of conditions at various spatial scales. A very large, stand-replacing fire has little mosaic within its boundaries (Eberhart and Woodard 1987), but viewed at the spatial scale of the region and the temporal scale of 100 years, it represents a patch in a fire mosaic on the landscape. Such an event produces a large, contiguous area of young forest but, in the larger context, there will be areas of very old forest as well.

Johnson et al. (1995) and Agee (Chapter 3) argue that naturally occurring age distributions in forested landscapes, especially landscapes composed of boreal and subalpine forests (Johnson et al. 1995), often fit negative exponential functions, indicating a constant-probability stochastic process. While this model may not always produce the best fit (Chapter 3), all stochastic processes produce a characteristic signature in the resulting age distribution, including asymptotic "tails" in the older age classes (Fig. 15.1a; Van Wagner 1978; McKelvey and Lamberson 1994). In contrast, cyclic processes, such as even-aged rotation forestry, truncate the age distribution (Fig. 15.1c). In a cyclic system, there is no formal mechanism to generate stands or elements older than rotation age. However, in a forested landscape with a negative exponential stand-age distribution, about 36% of the landscape will be older than the average stand age, about 13% older than twice the average stand age, and about 5% older than three times the average stand age (Table 15.1). Although there is no maximum stand age in this system (Finney 1994), old stands that have not burned will be subject to other sources of mortality, such as insects and disease.

Table 15.1—Equilibrium time since disturbance for forests with a negative exponential stand age distribution. Where time since disturbance exceeds the longevity of individual trees, the stand ages will be younger than are indicated in this table, and the forests will be characterized by "gap" processes due to overstory mortality.

Fire return interval (average stand age)	Percent of area (by age class) <50	50-100	100-200	200-300	>300
100 years	40	24	23	9	4
150 years	29	20	25	13	13
200 years	23	17	24	14	22
300 years	16	13	20	15	36

The retention of older stands essentially mimics the process of ecological "escape" in systems characterized by large-scale disturbances. In a catastrophic fire, variation in fire intensities and the vagaries of weather allow forest elements at many scales to survive, including individual trees, small groups of trees, stands, or entire watersheds. As multiple fires overlap over many years, there will be small areas embedded in larger areas of younger forest that have, by chance or location, survived multiple fires. This escape of older elements is not effectively mimicked by embedding fixed reserves in a landscape of cyclic management. In a stochastic system, with each new disturbance event, older forests may be destroyed or altered, or they may be left intact; thus, late-successional stands are not spatially assigned and maintained, but emerge dynamically within the disturbance process. A variety of other differences, some of them dramatic, also distinguish fixed-rotation from stochastic systems. For example, if a natural fire-return interval is 100 years, and a 100-year clearcut rotation is implemented to mimic natural rates of disturbance, the average stand age in the managed forest will be 50 years. In a stochastic system with a negative exponential stand-age distribution, however, the same landscape would have an average stand age equal to the fire-return interval, or 100 years.

Assuming that the scale of disturbance corresponds with the movement capabilities of individual lynx, landscapes generated through these stochastic processes would provide an amenable environment for lynx by producing areas of young, dense forest without removing the older elements. If the natural fire regime can be estimated for an area that is to be managed for lynx, then it is logical to use this rate to design management strategies. The overall rate of disturbance would be an aggregate of the cumulative effects of timber harvest, fire, and other natural disturbances (catastrophic windthrow or insect epidemics) on age structure. Therefore, in areas subjected to large wildland fires, or where inholdings are managed for short-rotation timber yield using even-aged management, timber harvest (including salvage) may be inappropriate for decades. In some regions, significant amounts of wilderness or National Park land will be part of the planning area, and management of natural fires must be incorporated into planning. We assume here that fire and timber harvest are similar processes in their capacity to initiate secondary succession. In addition, we would argue for a planned rate of disturbance somewhat less than the historically derived estimate. It is easy to increase the disturbance rate if desirable, but much harder to recover old-growth forests removed through overly aggressive management.

Spatial Patterns Resulting from Natural Disturbance Processes

Applying the negative exponential model has been suggested as an ecosystem management approach (Johnson et al. 1995), and does conform to many of Christensen et al.'s (1996) required properties: it produces sustainable patterns, maintains ecosystem complexity, and is primarily derived from an analysis of natural processes, specifically fire. It does not, however, directly address the larger scale issues of landscape pattern and connectivity. In these regards the model is neutral; it can be applied in a variety of configurations.

In even-aged timber management systems employing clearcutting, harvest units are generally similar in size (Fig. 15.1d). For example, a maximum clearcut size of 40 ac (16 ha) has been mandated on Forest Service lands for many years. Fires come in many sizes, however. In most forested landscapes, the majority of fires are small, whereas the majority of acreage burned has resulted from a few, very large fires (Chapter 3). Although this pattern is probably not feasible or even desirable as a management goal outside of wilderness, there are many advantages associated with producing fewer, larger cutting units. A disturbance pattern characterized by a few large blocks (Fig. 15.1b) is desirable if large areas of interior forest habitat are a management goal, or if processes associated with high edge densities, such as predation, competition, or nest parasitism, are potentially serious problems. It also vastly decreases the number of roads that need to be built. Lower road densities mean less sediment and mass failure into streams, reduced interactions between people and native biota, and slower invasion by exotic weed species. Costs of road construction and maintenance are decreased and, if a site is to be broadcast burned, the cost per acre declines as the area to be burned increases.

Juxtaposition of Habitat

The application of a fixed rotation age, in addition to the negative consequences of truncating the forest age distribution, also produces fixed landscape geometries (Fig. 15.1d). Once an area is cut, it is not scheduled to be re-entered (except for thinning) until it is again at rotation age. Because the adjacent areas will be younger than the rotation age, the next cut will conform to the original boundaries. Even if protected reserves are embedded within an even-aged rotation system, only a relatively small number of age juxtapositions and geometries are possible within each landscape.

However, the juxtaposition of stand ages in a variety of configurations and the edge habitats that result may provide essential ecological complexity at the landscape scale. A predator that needs thermal cover and nest sites in large snags or down logs typically found in old-growth forests, may forage in herbivore-rich early seral stands if they are nearby. For example, northern spotted owls use edge areas between old and young forest preferentially in those areas where young forests contain high concentrations of woodrats (Zabel et al. 1995). Additionally, most organisms have habitat requirements that change seasonally or stochastically; what is optimal in the summer may not be optimal in the winter, and what is optimal in a mild winter may not be useful during extreme conditions. By developing forests with a mixture of young and old stands in a variety of spatial combinations, we are likely to provide habitat requirements for a broad array of organisms.

Stand selection by fire in the taiga is not truly random, but it is more so than in the southern boreal forest types of the contiguous United States. In more southern areas, natural barriers and topographic features such as slope and aspect may provide locations where older forest is more likely to be found. These "refugia" (e.g., Camp et al. 1997) are not permanently fixed on the landscape, but tend to be located in areas that are infrequently disturbed, such as moist sites with topographic protection from wind. Northern aspects and areas near stream confluences are likely to become refuges from fire, whereas steep south- and west-facing slopes are likely to be burned more often. Locally-defined habitat conditions, such as these, should be used to develop spatially explicit landscape configurations.

Heterogeneity at the Local Scale

Although individual lynx may respond to landscape conditions at the scale of 50-100 km^2, their primary prey species respond to much smaller scale patterns, on the order of 1-10 ha. A landscape-scale management strategy that emphasizes stochastic models does not prescribe the size or location of treatments on the landscape, nor does it foreclose options concerning the types of treatments involved; treatments can include either timber harvest or prescribed fire. Additionally, activities on private lands, and naturally occurring events such as lightning fires, wind-throw, insect-induced mortality, etc., also need to be factored into landscape planning. Some of these events and activities may be predictable, while others will be surprises. Accordingly, management strategies must be flexible at several spatial scales.

Snowshoe hares generally occur in areas of dense forest cover, including shrubs and "doghair" thickets of small trees (Chapters 6 and 7). These structures are common in naturally regenerating areas after fire, but do not result from standard, even-aged forestry practices (Daniel et al. 1979). Thus, we believe that natural regeneration and stand development will likely benefit hares and, ultimately, lynx. Creative silviculture is required when management goals extend beyond wood production. This might translate to heavier reliance on natural regeneration after wildland fires. Where harvest mandates artificial regeneration in a short time frame, the planting might be clustered such that the time of early seral dominance will be spatially variable across a unit. If pre-commercial thinning is considered, it should be recognized that the usual objective of increasing the diameter growth of residual trees may be inappropriate for lynx. Leaving some doghair stands may be good for lynx; thinning others so heavily that additional conifer regeneration occurs in the unit may also be appropriate. Because fires leave large amounts of woody material standing and down, management should also leave substantial amounts of woody material in representative size classes, regardless of treatment.

The planning area also needs to encompass transitional areas between what is considered suitable habitat for lynx. Although the role of such areas in maintaining lynx populations is poorly understood, the condition of these areas will likely affect movement and survival during dispersal. For example, in lower elevation zones with mixed-severity fire regimes and fire-resistant trees, such as western larch, it may be appropriate to leave green trees on site. Large units may contain a variety of silvicultural treatments (e.g., clearcuts, as well as areas of green-tree retention at various stem densities) that grade into one another. In these transitional areas, some proportion should be treated with overstory removal significant enough that dense conifer regeneration will result, unless such areas already exist.

Minimum Area and Age-Class Requirements for Applying the Negative Exponential Model

We have argued that effective management of wide-ranging species, such as the lynx, requires planning at large spatial scales. In this context, it is important to note that stochastic disturbance processes only approach equilibrium at large spatial scales. With a negative exponential age distribution, very old forests, which make up a relatively small proportion of the landscape, will be lost by chance alone if planning is done at too small a scale. Johnson et al. (1995) provide equations for estimating the appropriate scale based on fire-return interval, projected age of oldest patch, average patch size, and size of the management unit. Because historic fire-return intervals vary substantially among southern boreal forests (Chapter 3), spatial planning should be based on sub-regional fire-return intervals.

Knowledge of historic patch sizes will enable some rough ideas of minimum area. If stand ages are distributed according to the negative exponential model, the largest patch should be a small proportion of the total landscape. Assuming 10% as the maximum size for a "small" proportion, and applying this to the north-central Washington landscape discussed earlier in this chapter where fires of 20,000 ha have occurred (e.g., the 1988 White Mountain fires), the minimum planning area might be 200,000 ha (2,000 km^2). If we were less conservative and chose 20%, the minimum area would be 1,000 km^2. This range of 1,000 to 2,000 km^2 compares favorably with the minimum estimate of 1,250 km^2 derived from lynx studies conducted in that area. The size of planning areas need to be based on local conditions and must be linked at broader scales, but these estimates provide a general sense of the scale at which planning would need to take place.

The management approach we have suggested for the conservation of lynx is to imitate the stochastic process of fire by managing for a stand age distribution that approximates the negative exponential model (Fig. 15.1a and 15.1b). Because this model results in an equilibrium distribution of forest age classes, standard methods can be used to estimate sustainable value and volume (McKelvey and Lamberson 1994). This management model is not as difficult to implement as it may seem. The stochastic process being modeled does not need to be randomized in the formal sense; deterministic processes can achieve the same patterns. In fact, within the overall planning area, there is a great deal of freedom concerning the placement and size of management units and the timing of management activities. Any stand can be disturbed at any time as long as the proportions associated with that stand type within the planning area are not exceeded. Accordingly, this strategy is much more spatially flexible than a fixed-rotation system, but it requires larger spatial and temporal frames than have been used historically.

Compatibility of Lynx Conservation with Other Resource Management Objectives

Both the regulatory requirements of state and federal resource management agencies and the concepts embodied in ecosystem management preclude the implementation of a lynx conservation strategy across a broad geographic area that fails to adequately address other resource values, especially the habitat needs of threatened or sensitive species. We propose that other resource management objectives, even those that may conflict in various ways with the needs of lynx, can be accomplished within the context of a habitat management strategy that mimics large-scale stochastic disturbance regimes. A key component of the negative exponential model is that it represents an adaptable and flexible management system, not one with rigid constraints on management options. On the contrary, the dynamic nature of the negative exponential age distribution in both time and space, provides a variety of opportunities for adjusting management actions to meet alternative objectives.

In southern boreal forests, other forest carnivores, including the marten, fisher, and wolverine, are also of conservation concern, but have habitat requirements that differ in many ways from those of the lynx (Ruggiero et al. 1994). Although providing for the habitat needs of these species may conflict with management of lynx habitat at the stand scale, the flexibility inherent in the negative exponential model should enable the attainment of multiple management objectives. In the remainder of this section, we use these three species to present examples of how applying this model to lynx conservation can be compatible with management for species having divergent habitat requirements.

American martens occupy mesic coniferous forests throughout boreal regions of North America, but generally attain their highest densities in late-successional stands with high levels of structural heterogeneity (Buskirk and Powell 1994). In addition, recent work by Hargis et al. (1999) in Utah suggests that interior forest conditions may provide essential breeding habitat for martens. Both martens and lynx occupy habitats characterized by snowy winters, and their ranges overlap extensively in northern portions of the Rocky Mountains, Cascade Range, and northeastern United States. Thus, management of habitat for lynx, especially the creation of early successional stands for snowshoe hare habitat, will impact martens in these regions. Under a stochastic disturbance regime, the resulting successional landscape would provide islands of old-growth habitat nested within an area managed for lynx. However, when applying the negative exponential model, the areal distribution of stand ages is flexible; within the conservation area, a given forest age class could be dispersed, clumped, or a combination of both. Thus, if a primary management objective was to provide habitat for a resident population of martens, or other species closely associated with late-seral forest conditions, older age classes could be clumped so that one or several areas contained high proportions of old growth and interior forest habitat.

Habitats occupied by fishers are similar to martens, but fishers are not as strongly associated with late-successional forests or deep snowpacks (Powell and Zielinski 1994). Fishers occupy a variety of stand ages but prefer forests with high canopy closure and complex structures near the ground. Unlike martens, fishers are not well adapted for traveling or hunting in snow and generally occupy low to mid-elevation forests where deep, soft snow does not accumulate. Although fishers are sympatric with lynx in much of Canada and in some portions of the northeastern United States, historical and current records indicate little overlap in habitats used by these species in the western mountains. Thus, it is unlikely that management of high-elevation boreal forest habitats for lynx will have adverse effects on fisher populations. Because lynx may use lower elevation habitats to move between patches of boreal forest, however, lower elevation zones may contain fisher habitat. Thus, if a management objective was to provide habitat for fishers, a manager might restrict clearcutting or other timber harvest activities designed to create snowshoe hare habitat to upper elevations, where lynx are most likely to occur, but maintain stands at lower elevations in older age classes having high levels of canopy closure. In addition, silvicultural prescriptions designed to provide a diversity of large forest structures in areas of potential fisher habitat would provide den and rest sites and increase prey availability for both martens and fishers.

Wolverines occupy a broader range of vegetation zones than other forest carnivores, including boreal forest, alpine, and tundra habitats. It has been argued that good wolverine habitat is best defined in terms of an adequate food supply in areas of low human density; however, the habitat ecology of wolverines in western montane boreal forests has been studied at only a few localities (Banci 1994; Copeland 1996, unpublished). Because the spatial requirements of wolverine exceed those of lynx (Ruggiero et al. 1994), areas managed for lynx would be smaller than those required for the conservation of wolverine populations. However, because these areas will likely contain components of subalpine and alpine habitats, relatively extensive areas of potential wolverine habitat would be included within areas managed for lynx. Wolverines generally scavenge for ungulates along valley bottoms and forage and den in remote, high-elevation areas (Hornocker and Hash 1981; Magoun and Copeland 1998). Thus, if managers wished to provide habitat for wolverines, they could pay particular attention in the planning process to ungulate winter range and other aspects of habitat quality for ungulates to provide a consistent supply of carcasses for wolverine to scavenge. In addition, wolverines generally avoid areas of human activity. To limit the threat of human-caused disturbance or mortality, managers could also restrict access to portions of the landscape where wolverines are most likely to occur.

Monitoring

The management model we have presented in this chapter represents a conservative approach, but is largely untested. Consequently, the implementation of these ideas should be considered experimental in nature. Because we are relatively ignorant of the ecological and social consequences of this type of approach, a comprehensive monitoring plan must also be established to assess the current status of vegetation and target species within the planning area and to monitor changes to the landscape and the biota that result from the implementation of this management strategy. Unfortunately, most environmental monitoring programs on public lands have not been carried out as planned, nor have they contributed to decision-making by resource managers or the aversion of biological crises (Noon et al. 1999). The primary reasons given by Noon et al. (1999) for the widespread failure of monitoring programs include: a minimal foundation in ecological theory or knowledge, little logical basis for the selection of target species, no required understanding of cause-and-effect, no determination of the magnitude of change that would result in a management response, and no connection between the results of monitoring and decision-making. Lynx conservation in the context of ecosystem management cannot succeed without a scientifically based monitoring strategy that overcomes the shortcomings of previous monitoring efforts.

An important aspect of our suggested approach to lynx conservation in the context of ecosystem management is the resultant flexibility in planning landscape features (i.e., the kinds, amounts, and arrangements of landscape elements) based on outputs of the negative exponential model. As discussed throughout this chapter, these outputs include information about patch sizes and the vegetative composition of modeled landscapes. In many, if not most, instances the appropriate spatial scale for modeling will subsume more than one administrative unit. Thus, model outputs will have to be disaggregated across administrative boundaries. To accomplish this, highly innovative and integrated planning processes will be needed. Innovative approaches to monitoring the consequences of planning decisions will also be required. Indeed, the success of this approach will depend on monitoring the proportions, patch sizes, and arrangements of the vegetation types and successional stages in each planning area. Given the fundamental importance of pattern as an ecological feature, spatially explicit monitoring to ensure conformity to desired patterns will be essential.

Within this framework, more traditional monitoring activities will also need to be performed. Much has been written on this subject (e.g., Goldsmith 1990; Mulder et al. 1999), but the details are beyond the scope of this chapter. Because a primary objective of management actions will be to enhance habitat conditions for both lynx and snowshoe hares, monitoring of hare populations with pellet transects (e.g., Krebs et al. 1987) and lynx populations with snowtracking or other techniques (e.g., Thompson et al. 1981), will likely be critical components of monitoring strategies. Lastly, we emphasize that effective monitoring programs must include pre-defined criteria for determining how to interpret monitoring results, and what actions to take when monitoring indicates the need to modify management direction.

Conclusions

We believe this approach to conserving lynx populations is compatible with the need to provide for other sensitive species and with the coarse-filter approach suggested by ecosystem management. We do not present a detailed template for the implementation of these ideas, because the specific components of management planning will depend on the natural disturbance regime, current conditions within the landscape, the plant and animal species that inhabit it, and other management objectives. In all of the applications of lynx management, however, we recommend the following:

1. Use natural disturbance patterns, in terms of size, frequency, intensity, and stochasticity for guidance concerning the design and management of landscapes.
2. Engage in spatially explicit landscape planning within very large management areas. Lynx metapopulation dynamics operate at regional scales.
3. Manage for landscapes that contain a continuum of age classes, including both very young and very old forest. Not only is this reflected in patterns that result from fire and other natural disturbances, but it represents a prudent conservation policy in the face of uncertainty. In this regard, be especially cautious with older forests and carefully consider the replacement time for each forest type and forest element. A large-diameter log may take hundreds of years to decay; a small clearing can be created in a few weeks.
4. Provide for a variety of regeneration conditions; some areas should be dense and some more open. Treatments that depend on natural regeneration will be more likely to produce these patterns than will planting. Leave individual trees and variable-sized groups of residual elements if natural disturbance processes suggest these structures are appropriate.
5. Practice adaptive management and consider every action on the landscape to be an experiment. Monitor the efficacy of management actions and adapt strategies if desired conditions are not being achieved.

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