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# 6. The Natural Disturbance Model

## Ecological Forest Management

The long list of deficiencies associated with the current system of forest management in Alberta indicates that the status quo is no longer tenable. The current system is the product of another era, now out of step with public values and advancing knowledge of forest ecology. Fortunately, Alberta's forest management problems are not unique. Researchers across North America and Europe have spent more than a decade addressing these problems and a substantial body of literature describing alternative approaches now exists. Collectively, these new approaches define a new paradigm of forest management, termed Ecological Forest Management (EFM) (sometimes referred to as Ecosystem Management).

The objectives of EFM are to maintain key ecosystem processes, to conserve native biodiversity, and to provide a stable and sustainable flow of economic benefits from the forest for current and future generations (AFCSSC, 1997: 7). EFM includes the following core elements:

1. The Natural Disturbance Model (NDM). The maintenance of biodiversity in the presence of industrial resource extraction cannot be accomplished through the individual management of species because there are too many species in-

volved and our understanding of their needs is inadequate. The NDM is an alternative approach, based on the assumption that biodiversity can be maintained in the presence of industrial use if industrial practices are made to approximate natural disturbances. In practice, the NDM entails the management of human disturbances to maintain ecological patterns and processes within their typical range.

2. **Protected areas.** Because of limitations with the NDM, and the inherent unpredictability of natural systems, a complete reliance on the NDM to maintain biodiversity would entail substantial risk. The limitations of the NDM do not invalidate its use, but imply that a complementary system of management, specifically designed to maintain biodiversity, must be implemented on a portion of the land base. A system of protected areas capable of maintaining ecological integrity is best suited to this role. Additional roles of protected areas, within the context of EFM, include ecological benchmarks against which the success or failure of the NDM can be assessed, conservation of wilderness, and sites for future research on natural ecological processes
3. **Monitoring.** EFM recognizes that all management scenarios are, in effect, experiments with substantial levels of uncertainty regarding the outcomes. Consequently, monitoring is an integral component of EFM, designed to evaluate whether the system overall is responding as predicted. Using feedback from monitoring, adjustments can be made to assumptions, models, and management practices in an effort to rectify any observed deviations. This process of feedback and adjustment has been termed adaptive management.
4. **Rare and threatened species.** Some species, because they are endangered or highly sensitive to industrial activities, will require extra attention to ensure their viability. Where the range of these species cannot be fully incorporated into protected areas, modifications of the NDM will be required, including specialized restrictions on industrial activities.
5. **Research.** Research is required to support the implementation of the NDM by providing a more complete understanding of ecological processes, including natural disturbance regimes, and determining how human disturbances (e.g., clear-cutting) differ from natural disturbances (e.g., fire). Research is also required as part of the adaptive management process, to determine the causes of any observed deviations from desired management outcomes.
6. **Decision-making.** EFM is not a static set of prescriptions, but a process that evolves in response to changing public values and new scientific information. Rates of forest harvesting and other decisions pertaining to land and resource use are made within the context of the desired future forest, not the growth rate of trees or mill capacity. Furthermore, decision-making is based on a common regional framework that integrates the activities of all users. Public involvement is a key component in identifying and weighing the social, economic, and ecological values to be sustained in the de-

sired future forest. All of the information used in planning and decision-making processes should be available to those who wish to be involved.

This chapter provides a detailed review of the NDM. Other elements of EFM are discussed in subsequent chapters. I begin with a review of fire and forest succession, which are the key ecological processes that the NDM seeks to emulate in the boreal mixedwood forest. I then examine the landscape structures and patterns that arise from these natural processes. In the final section I review the implementation of the NDM.

## Ecological Processes

### *Distribution of Fire Size*

The vast majority of fires are small. Fires up to 2.0 ha in size account for 74% of all fires recorded in the provincial fire database from 1961 to 2000 (ASRD, 2002). A pattern of decreasing frequency with increasing size is also evident in Class E fires (i.e., those over 200 ha in size; Fig. 6.1). Although large fires are rare they are responsible for the majority of the area burned. For example, 98% of the area burned in Alberta from 1961 to 2000 was due to only 5% of the fires. These large fires, some of which have exceeded 100,000 ha in size, play a dominant role in structuring landscape patterns at the largest scales.

### *Patterns in Fire Occurrence over Time*

Large fires are generally associated with so-called "fire years" in which extreme climatic conditions, including extended periods of hot and dry weather, make the forest highly susceptible to burning (Bessie and Johnson, 1995). During fire years multiple extensive burns can occur. For example, in 1981 six fires occurred, each exceeding 100,000 ha in size.

Based on the provincial database of Class E fires an average of 0.65% of the land area in northern Alberta has burned annually since 1980. However, because of the influence of fire years the rate of burning over time has varied tremendously (Fig. 6.2), making it difficult to accurately characterize the mean rate of burning (Armstrong, 1999). Furthermore, studies of charcoal and pollen in lake sediments have demonstrated that the mean rate of burning has fluctuated widely over the centuries, likely in response to long-term climatic changes (Bergeron et al., 1998).

Since the 1950s fire suppression efforts have steadily increased in terms of dollars spent and area controlled (Murphy, 1985). By 1971 a policy of total suppression across the entire province was

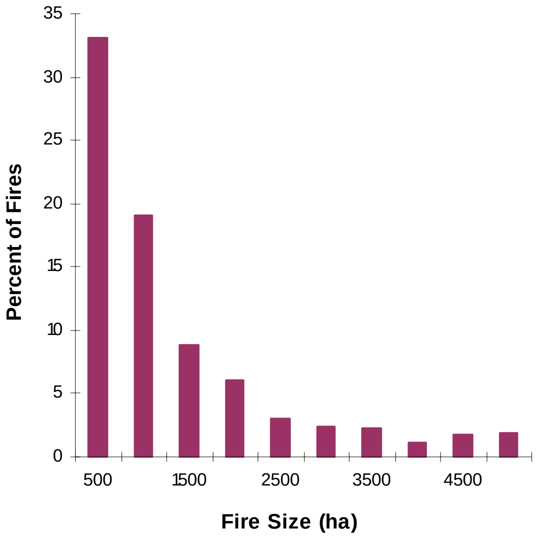

Fig. 6.1 Distribution of fire size for Class E fires in Alberta from 1961 to 2000. Only fires up to 5,000 ha are shown. Source: ASRD, 2002.

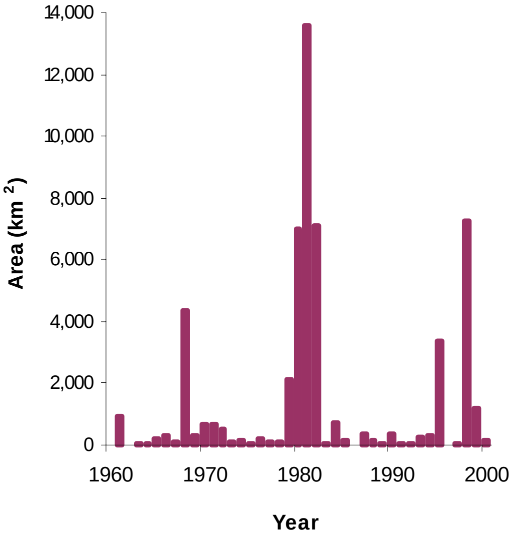

Fig. 6.2. Annual area burned in Alberta from 1961 to 2000. Source: ASRD, 2002.

in place (Murphy, 1985). Annual expenditures on fire suppression and control are currently $127 million (average for 1996-2000; CFS, 2002). But these efforts have not been accompanied by a decreasing trend in the annual area burned (Fig. 6.2). While there is some evidence that fire suppression has reduced the number and size of fires when climatic conditions are not extreme, suppression does not appear to have been effective in stopping large fire events (Larsen, 1997; Johnson et al., 1998; Campbell, 1999). Similar observations have been made elsewhere, indicating that Alberta is not unique in this regard (Agee, 1997; Moritz, 1997). It appears that large fires that occur during fire years, accounting for much of the area burned over time, are virtually unpreventable.

### *Spatial Patterns in Fire Occurrence*

At the provincial scale the impact of large fires is clearly evident (Fig. 6.3). The patchy nature of the burning causes substantial regional differences in the age structure of the forest. Many large regions in the province have not burned at all in the past 50 years. Although there is no clear pattern in the distribution of fires, differences in the rate of burning among Natural Subregions have been detected through statistical analysis (Andison, 1997).

At the landscape scale fire patterns are characterized by marked variability. After an extended period of hot and dry weather most types of forest are susceptible to burning and the patterns produced are primarily a function of wind speed and direction (Bessie and Johnson, 1995). Oblong fires oriented in the direction of the wind are typical (Fig. 6.4), though fires that burn for an extended period will have complex shapes due to changes in wind direction over time. The intensity of burning varies in response to weather variables (e.g., precipitation and wind speed), physical features of the landscape (e.g., slope), and stand type (Foster, 1983). Some patches of forest remain unburned because they were downwind of firebreaks such as lakes, streams, and wetlands (Eberhart and Woodard, 1987).

Under less extreme climatic conditions fires are often smaller and less intense and physical features of the landscape have a greater influence on their behaviour. For example, a fire break that produces an unburned island in an intense, rapidly moving fire might completely block the forward progress of a less intense fire (Eberhart and Woodard, 1987). Furthermore, when climatic conditions are not extreme forest stands will vary in their susceptibility to burning and thereby also influence fire behaviour (Bessie and Johnson, 1995). Recent studies have shown that the probability of a fire starting in aspen stands, and the proportion of available aspen that is burned in

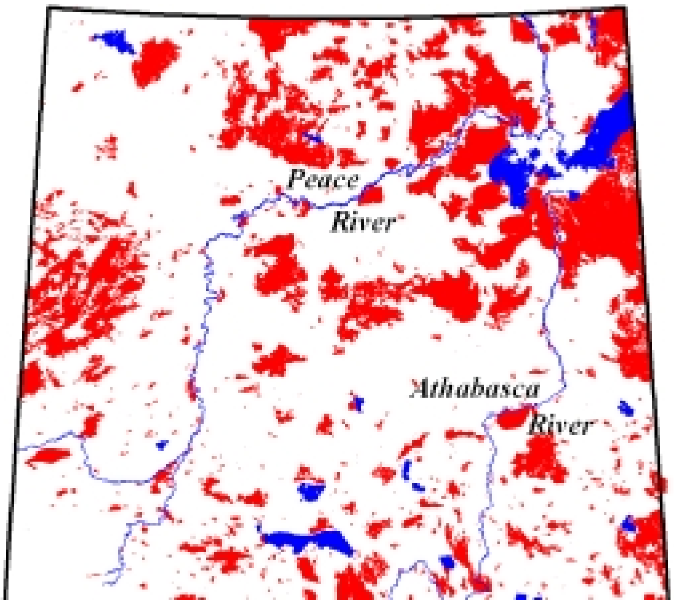

Fig. 6.3. Distribution of Class E fires in northern Alberta from 1950 to 2000. (Map: ASRD, 2002)

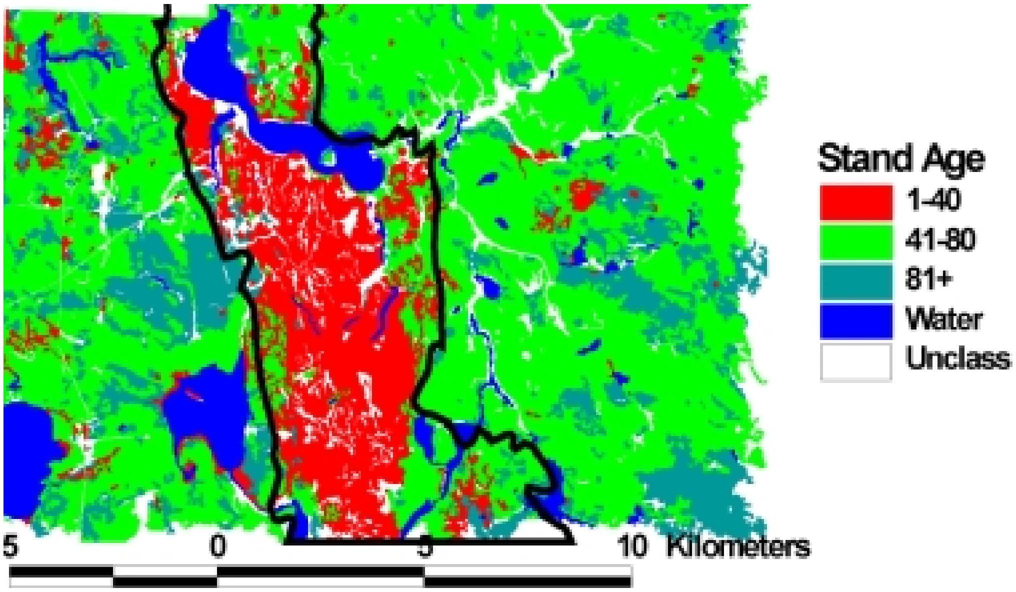

Fig. 6.4. Stand age structure following a large fire that occurred in northeast Alberta in 1968. The fire (outlined in black) was 283 km$^2$ in size, of which 91 km$^2$ is shown. Note the arrangement of large patches of young forest produced by the fire and the patches of older forest within the fire boundary, representing fire skips. (Map: Forest Watch Alberta)

large fires, are both lower than in other forest types (Cumming et al., 1998; Cumming, 2001). This implies that the threshold for burning aspen stands is relatively high, though further research is required to confirm these findings.

### *Forest Succession*

The following synthesis of forest succession in the boreal mixedwood was derived from papers by Kabzems et al., 1986: 62; Frelich and Reich, 1995; Cumming et al., 1996; Lieffers et al., 1996; Bergeron et al., 1998; Kneeshaw and Bergeron, 1998; Bridge and Johnson, 1999; Greene and Charron, 1999; and Cumming et al., 2000.

Although the intensity of burning within and among fires is variable, the large fires responsible for most burning generally kill most aboveground vegetation. The regeneration of the forest is influenced by a variety of regional and local site characteristics (e.g., moisture regime, soil type) and by availability of seeds. The result is that the large post-fire patches fairly rapidly differentiate into smaller forest stands that differ in vegetation composition and successional pathway.

Moisture regime has the greatest influence on forest succession. In the boreal mixedwood of Alberta moist sites are characterized by stands of black spruce and larch, medium sites by aspen and white spruce, and dry sites by pine. However, substantial variation exists within these coarse associations.

Succession on moist and dry sites is relatively straight-forward in that the original black spruce or pine stand is generally replaced with the same stand type after fire, though often with some component of aspen. Succession on medium sites is more complex. Aspen regenerates aggressively after fire through root suckering and is virtually always present in regenerating stands (even on sites previously dominated by white spruce, since few sites are completely devoid of aspen). The regeneration of white spruce is more variable. Regions within a burn that are more than about 100 m from the edge of the burn or from an unburned island will not be fully regenerated. Furthermore, seed production is variable from year to year and optimal production may not coincide with the immediate post-fire period which is best for seedling survival. Also, fire intensity influences the suitability of the seed bed. Hot fires that expose bare soil are required for optimal seedling germination and survival.

Because of the variability in white spruce regeneration, several outcomes are possible on medium sites. Wherever white spruce seed is available and the seedbed is suitable, an even-aged mixed stand of white spruce and aspen can be expected. Because aspen cannot regenerate in the shade of a conifer canopy the stand converts to pure spruce in approximately 100 years. If white spruce seed is available but the seedbed is unsuitable then the stand initially regenerates to aspen, but spruce enters incrementally (often germinating on old logs). The result is an uneven-age mixed stand, which also eventually becomes pure spruce, though over a longer period of time. Finally, if there is an inadequate source of white spruce seed, the stand regenerates to aspen, with or without scattered white spruce at low density. Instead of transforming to a white spruce stand the stand is more likely to become a self-perpetuating uneven-aged aspen stand. This occurs through incremental aspen regeneration in small forest gaps produced by wind damage or mortality associated with self-thinning. If the stand is not burned for a long interval (i.e., 200-300 years) the density of spruce may incrementally increase to the point of dominance.

The successional pathways for medium sites described above represent points along a continuum of possible trajectories. The relative frequency of occurrence of these trajectories at large spatial scales has not been determined, limiting our ability to interpret and predict landscape patterns.

Forest succession is influenced by other site characteristics in addition to moisture regime. Surface features and soil type are two of the most important factors, operating at both local and landscape scales. Because of unique combinations of moisture, soil, and surface features along riparian areas the composition of vegetation and age structure of stands here is usually distinct from the rest of the forest.

## Forest Structure and Pattern

### *Stand Structure*

Forest stands can be characterized by their structural attributes, including measures of the dominant tree species (e.g., density, percent canopy closure), understory, snags, and downed logs. These attributes undergo a predictable pattern of change as stands age and together they define a set of structural stages common to all stands types (Fig. 6.5). The duration of each structural stage differs among stand types because of differences in the rate of tree maturation. For example, aspen stands begin to acquire old-growth characteristics by 100 years, whereas stands of white spruce are still in the mature stage at this time (Timoney, 2001).

Much of the structure of young stands is due

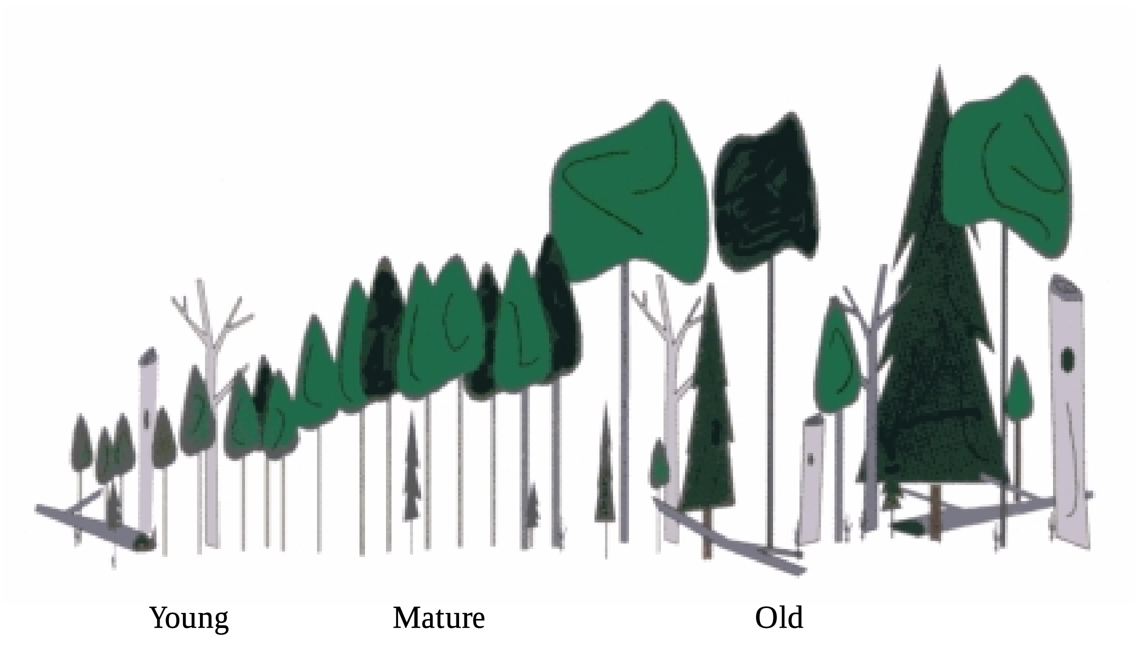

Fig. 6.5. Schematic representation of stand structure as a function of stand age (Graphic: I. Adams).

to the structural legacy left by the pre-fire stand. In all but the most intense fires, large trees are killed but not consumed and in time these dead trees become snags and then downed logs on the forest floor, providing structural diversity to the stand for several decades (Lee et al., 1997). Diversity in young stands is also enhanced by the openness of the canopy which permits light and warmth to reach the forest floor, stimulating understory growth. As a consequence of these multiple influences, structural diversity in young stands is intermediate between that of mature and old stands (Stelfox, 1995, *v*).

The transition of stands to the mature stage is marked by closure of the canopy. Mature stands are typified by a dense growth of relatively even-aged trees and reduced understory development (Stelfox, 1995: *v*). Self-thinning of the trees begins at this stage, but gap formation is not yet a prominent feature. The legacy of pre-fire aspen trees has diminished (Lee and Crites, 1999), though large-diameter conifer logs may persist. Mature stands have the lowest levels of structural diversity.

The transition from mature to old stands is gradual. The key changes include canopy breakup and release of understory plants, emergence of secondary canopy species, and accumulation of snags and downed logs (Stelfox, 1995: *v*). Relative to younger stages, old stands have trees of many ages and have more large canopy trees, large snags and large downed logs. Overall, structural diversity is highest in old stands and this is reflected in the highest species richness in both plants and animals (Stelfox, 1995: *vi*).

Stand structure is also influenced by stand type. The differences between coniferous and deciduous stands are of particular significance. Mixedwood stands, because they combine the features of both coniferous and deciduous stands, have a unique structural composition that is of importance to many species (Stelfox, 1995: *viii*).

### *Age Structure*

If forests burned at a constant rate in a spatially random pattern then the age class distribution of stands would follow a curve similar to that shown in Fig. 6.6. The extended tail of this curve reflects the fact that through chance some stands escape burning for very long periods. The slope of the curve depends on the mean annual rate of burning. On actual landscapes this theoretical distribution is rarely observed, as illustrated by stands in Forest Management Unit L1 (2900 km$^2$; Fig. 6.6). The main reason for the discrepancy is that burning does not occur at a constant rate, but instead occurs in pulses associated with fire years (Fig. 6.2). Furthermore, most burning is clumped

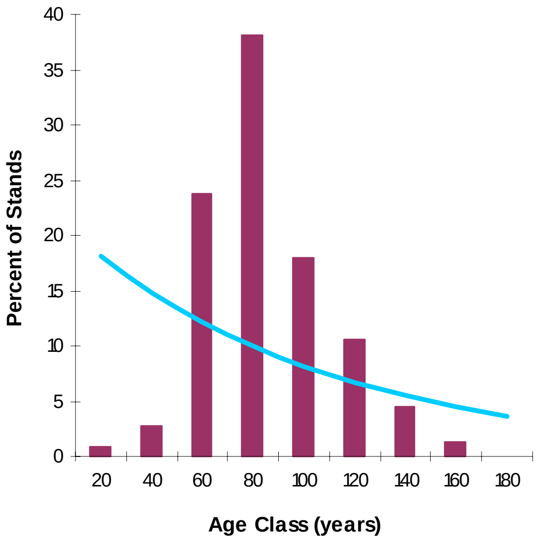

Fig. 6.6. Age class distribution of forest stands in FMU L1 (columns) and theoretical age distribution (line)assuming a constant rate of burn of 1.0% . Source: Al-Pac inventory.

as a consequence of large fires (Fig. 6.3). Because of these processes the age structure of the forest in any given area is more a function of the variability in fire occurrence, expressed locally, than it is of the mean rate of burning (Armstrong, 1999). This effect is most pronounced at small scales, but even at the provincial scale, equilibrium conditions are not observed due to long-term shifts in climatic conditions (Bergeron et al., 1998).

### *Stand Size*

The distribution of stand size reflects an interplay between fire, site conditions, and forest succession. Large fires produce large uniform patches, albeit with many small unburned islands, and succession differentiates these patches into smaller units in response to differences in site characteristics and seed availability (Lieffers et al., 1996). The net result is that stand size is distributed in a pattern similar to forest fire size (Fig. 6.7), but on average, stands are substantially smaller than fires.

Stand size is quite sensitive to the system used to classify the landscape. An increase in the number of categories or resolution of interpretation results in a decrease in the average size of stands. In fact, highly detailed inventories, such as the Alberta Vegetation Inventory (AVI), virtually preclude the existence of stands greater than 100 ha (AEP, 1994). This issue must be considered when comparing landscapes and when developing size targets for NDM management.

### *Spatial Arrangement*

The spatial arrangement of forest stands reflects the legacy of fire and local and regional differences in site conditions (Cumming et al., 1996). As a consequence of infrequent large fires, stands of the same age are typically aggregated together (Fig. 6.4). Within the matrix produced by these large fires lie patches of older forest, representing fire skips, and patches of newer forest arising from more recent small fires (Johnson et al., 1998).

Aggregation is also apparent from the perspective of vegetation type (Fig. 6.8). This is largely a consequence of regional patterns in site conditions, especially moisture regime (Bridge and Johnson, 1999). Although most of the boreal region is relatively flat, there are nevertheless significant differences in moisture regime expressed at a variety of scales up to multiple kilometres. These differences in moisture regime are in turn linked to different assemblages of vegetation (Cumming et al., 1996). Fire also plays a role in aggregating stands in that it promotes the establishment of aspen and mixedwood stands at the expense of pure conifer stands.

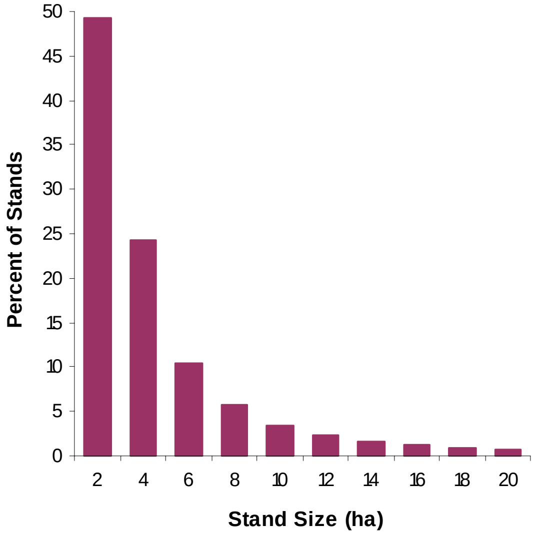

Fig. 6.7. Size distribution of forest stands in FMU L1. Stands > 20 ha are not shown. Source: Al-Pac inventory.

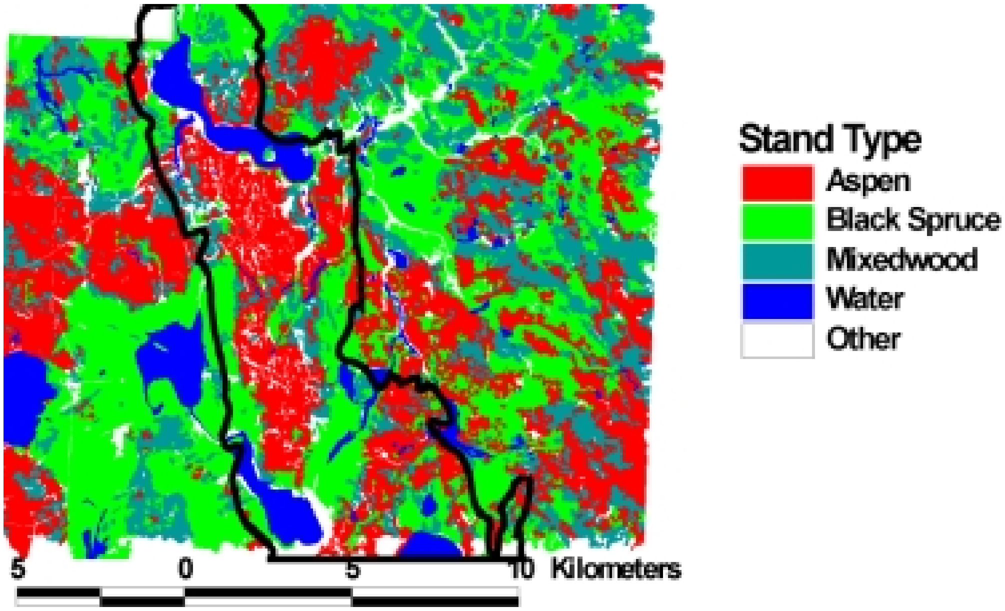

Fig. 6.8. Stand type for the same region illustrated in Fig. 6.4. Note the aggregation of similar stand types at a scale of multiple kilometres. Area labelled Mixedwood includes both mixedwood stands and pure white spruce stands.

The existence of large-scale aggregation notwithstanding, the spatial arrangement of stands at the local level is often highly complex (Fig. 6.9). This complexity reflects local variations in site conditions, seed availability, successional stage, and the irregular boundaries of past fires (Lieffers et al., 1996).

Aquatic features such as rivers, lakes, and wetlands also have an important influence on landscape patterns (Bergeron, 1991). Because they often act as fire breaks there is a greater probability of finding older forest stands in the vicinity of these features than in the remaining landscape (Timoney, 2001). Furthermore, the unique moisture regime, soils, and even microclimate in the vicinity of aquatic features lead to distinct assemblages of vegetation in these areas (Naiman et al., 1993).

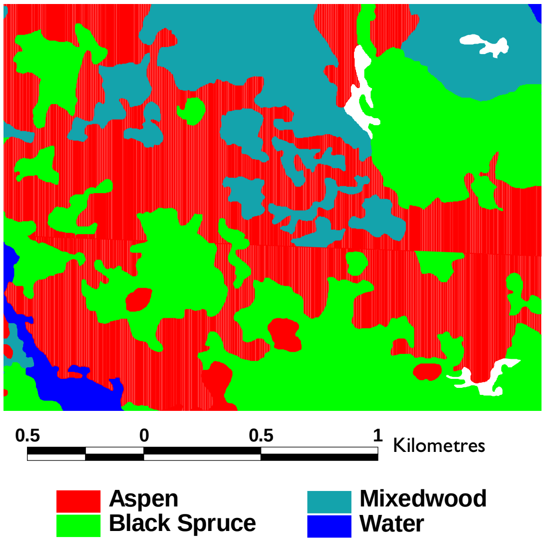

Fig. 6.9. Spatial arrangement of forest stands viewed at small scale. (Map: Forest Watch Alberta)

## Relationship to Biodiversity

Forest species vary widely in their habitat requirements, reflecting diverse strategies for obtaining nutrition, avoiding predation, and meeting other requisites of life (Kirk et al., 1996). Many have specialized requirements reflecting physical and behavioural adaptations designed to minimize competition with other species. For example, the unique beak of the red crossbill, designed to efficiently pry open cones and extract the seeds therein, links this species to patches of coniferous forest with a high density of cones (Godfrey, 1986: 566). Because of such habitat specialization the overall diversity of forest species is dependent upon the diversity of habitat features, which is in turn a consequence of the combined actions of fire and succession (Bunnell, 1995; AFMSC, 1997: 3). Old-growth stands, because of their high structural diversity, generally have the highest levels of plant and animal species diversity (Stelfox, 1995: *vi*). In contrast, in Europe, where long-term traditional forestry has resulted in a simplification of forest structure and pattern, there has been a documented decline in species diversity (Siitonen and Martikainen, 1994; Mikusinski and Angelstam, 1998).

In addition to requirements for specialized habitat types, species also have requirements relating to patterns of habitat distribution. For some species stand size is of particular importance. For example, large stands are required for the optimal nesting success of some songbirds (Paton, 1994; Flather and Sauer, 1996). Another important aspect of habitat distribution is the spatial arrangement (interspersion) of stand types (Keitt et al., 1997). Frequently, a species will use one habitat type for cover and another for foraging, therefore both need to be in proximity (e.g., Romito et al., 1996). At large scales, the aggregation of similar habitat types noted previously may be important for the population viability of species that live in herds or those with limited ability for dispersal (e.g., Schaefer and Pruitt, 1991).

Landscape structure and pattern can also influence animal movement. Riparian zones are used as movement corridors by many species (e.g., Machtans et al., 1996). Also, the profound changes in landscape structure caused by disturbance events can reduce the general permeability of the landscape to animal movement (Donovan et al., 1995). In particular, many birds and mammals are reluctant to cross stands that have been intensely burned or clearcut because they lack sufficient cover (e.g., Steventon and Major, 1982; St. Clair et al., 1998).

It should be noted that natural disturbance and succession do not provide the habitat requirements for all species in all locations. The landscape is in constant flux, and the changes that occur, although temporary, often mean that species must move to survive (particularly after large disturbances). At any fixed location the abundance of many species will rise and fall (often suddenly) as the habitat features of the site change over time.

## Forest Harvesting: The Natural Disturbance Model

### *Overview*

Through natural selection the species inhabiting the boreal forest have developed adaptations for maintaining viability in the face of catastrophic disturbances such as fire (AFMSC, 1997: 3). In fact, periodic disturbance is necessary for the existence of species that depend on the earlier stages of forest succession. Based on these observations it has been suggested that biodiversity can be maintained in the presence of forest harvesting if harvest practices are made to approximate natural disturbances (Hunter, 1993). This forms the basis of the NDM approach to harvesting. Because fire is the dominant stand-replacing disturbance in the boreal mixedwood it has received the greatest attention within the context of the NDM in Alberta. Other disturbances such as insect damage and wind throw have been considered only superficially.

In practice, it is not the actual process of fire that the NDM seeks to approximate, but the structure and pattern resulting from fire and subsequent forest succession (Bergeron et al., 1999). In part, this reflects a recognition that fire and harvesting are fundamentally different processes (Niemela, 1999). Cutting and removing trees is not the same as burning them and leaving them in place, and even more importantly, infrequent large fires account for most of the area burned, whereas sustainable mill operation requires a relatively constant flow of timber from year to year. The assumption (and hope) is that the key to maintaining biodiversity is not necessarily the strict emulation of fire but the maintenance of habitat diversity, however that may be achieved (Bergeron et al., 1999).

Our ability to replicate the structures and patterns produced by natural disturbance is dependent on how well these structures and patterns can be characterized. The first consideration is the selection of a reference landscape. Because landscape structure and pattern change over time and space, due to the impact of large fires and long-term fluctuations in climate, there is no single "correct" landscape or time period to use (Johnson et al., 1998). However, for the simple practical reason that historical landscapes cannot be adequately characterized, using the current landscape for primary guidance is the only realistic option available. The reference landscape should be large (e.g., FMA or Natural Subregion) to avoid undue influence from large historic fires. Also, areas that have already been significantly impacted by industrial use should be excluded from the analysis, or at least be subject to appropriate corrections.

Once a reference landscape has been defined the next step in the implementation of the NDM is the development of operational targets for the forest attributes that characterize forest structure and pattern. The key attributes amenable to measurement include those discussed throughout this chapter: within-stand structure, the distribution of stand type, age, and size, and the spatial arrangement of stands. The attributes are characterized using distributions instead of average values because of the NDM emphasis on maintaining the natural range of variability. Although I have to this point been using forest stands as the primary unit of measure, in practice it is more appropriate to use forest patches. Patches are simply aggegrates of stands with similar features. The idea is to limit the number of different patch types to a quantity that is ecologically meaningful and tractable to work with. For example, a 50-year old aspen stand adjacent to a 60-year old aspen stand would be considered a single patch of "mature" aspen.

### *Within-patch Structure*

Given the importance of the structural legacy left by the pre-disturbance stand (Stelfox, 1995: *vii*), post-harvest targets should be defined for the quantity and distribution of residual live trees, quantity and distribution of standing dead trees, and the quantity and distribution of downed woody material. Targets relating to soil nutrient levels and the disturbance and compaction of the forest floor after harvest should also be defined as these attributes may be affected differently by fire and harvesting (Xu et al., 1999).

Because harvesting in Alberta generally involves the removal of most large trees from a site, but fire does not, it would be impossible to achieve structural targets based on the strict emulation of fire (under current rates of harvest) (Niemela, 1999). Consequently, there has been an effort to define targets based on experimental studies that seek to quantify the minimum levels of post-harvest structure required for the maintenance of biodiversity. Recent research by the Alberta Research Council recommends that up to 30% of merchantable trees should be retained in within-stand residuals (Fig. 6.10), unharvested stands, and riparian buffers (Schieck and Song, 2002). Additional research will be required to determine how residual trees within cutblocks should be distributed for maximum benefit (e.g., size of clumps, spatial layout).

Given that fire has minimal influence on the structure of mature and old stands (Lee and Crites, 1999), the aforementioned targets for stand structure relate primarily to young stands. The unique characteristics of old-growth stands, particularly large live trees, large snags, and large downed logs, and the unique assemblages of plants and animals that are associated with them (Stelfox, 1995: *vii*), cannot be created through harvesting. Consequently, old-growth structure must be maintained by retaining older-aged stands on the landscape (Burton et al., 1999; Niemela, 1999).

### *Distribution of Patch Type*

I use patch type to denote a specific combination of age and dominant tree species (e.g., old-growth aspen). The best available guide for the characterization of the target distribution of patch types is the current forest inventory. However, the extreme variability in annual area burned, reflecting the impact of fire years, distorts the distribution of patch age classes relative to the long-term mean, even at the scale of the largest FMAs (Cumming et al., 1996). Furthermore, forest inventories underestimate the proportion of older forest patches on the landscape (Cumming et al., 2000). Unfortunately, there are few options available for improving the estimated target distribution other than expanding the reference landscape to the maximum possible size (e.g., Natural Subregion) and removing biases in the inventory. Modelling techniques can offer qualitative guid-

Fig. 6.10. Recent cutblock demonstrating residual patches of trees left after harvest (Photo: Al-Pac).

ance, but because of deficiencies in our understanding of successional processes and rate of burning among patch types it is not currently possible to generate target patch distributions on the basis of theory.

A comparison between hypothetical targets for the NDM and sustained-yield management illustrates the major differences between these two approaches (Figs. 6.11 and 6.12). Whereas the NDM seeks to maintain the long-term mean distribution of patch types, sustained-yield management seeks only to maintain a continuous supply of timber. Consequently, under sustained-yield management older patch types and mixedwood patch types (and the habitat diversity they represent) are slated for elimination from the landscape (Bergeron et al., 1999).

The rate of harvest and the selection of harvest blocks both influence the distribution of patch types through their effect on forest age structure (Bergeron et al., 1999). Harvest rates that exceed the rate of regeneration of old stands, or the preferential selection of older stands, will result in the loss of older age-class patches from the landscape.

The maintenance of targeted distributions over a typical 200-year planning horizon will require an iterative harvest planning approach involving computer modelling techniques. It should be noted that the distribution targets for patches are only intended to be met at the large spatial scales at which they were defined. At smaller scales (e.g., landscape) substantial variation can be expected, and should in fact be maintained; however, deviations should not exceed the range that is typical for a given scale.

The retention of mixedwood patches on the landscape requires changes to land management

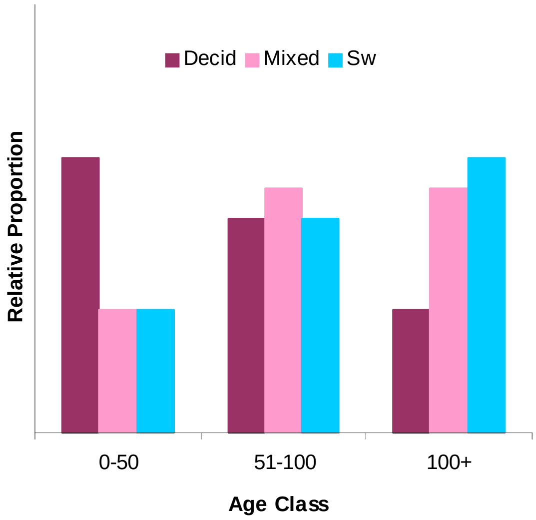

Fig. 6.11. Hypothetical NDM target distribution for patch types that are likely to be influenced by harvesting (Sw = white spruce).

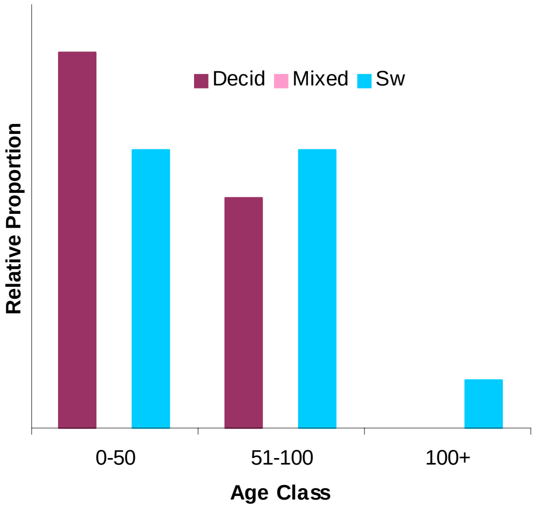

Fig. 6.12. Target distribution for patch types assuming sustained-yield management with an 80-year rotation for deciduous and 120-year rotation for conifer.

policies and practices. Currently in Alberta mixedwood stands are designated as either coniferous or deciduous, based on the volume of conifers at the time of inventory, and managed separately (generally by different operators) (Lieffers and Beck, 1994). Because of this separation of land bases, immature understory trees are generally not retained at harvest and regeneration efforts are (by regulation) intended to produce relatively pure stands, either through plantation techniques on coniferous sites or natural regeneration on deciduous sites.

In order to maintain mixedwood patches the dual land base system must be abandoned and mixedwood management techniques need to be employed (AFMSC, 1997: 4). Mixedwood management involves the promotion of a more natural process of regeneration in place of plantation techniques. So, for example, in harvesting a mature deciduous stand care would be taken to protect any existing coniferous understory. In time these retained coniferous trees would be available for harvest in a second entry to the site. Additional mixedwood techniques are reviewed by Lieffers et al. (1996). The abandonment of plantation management also lowers the likelihood of environmental disruption from techniques such as ground scarification, introduction of genetically selected trees, fertilization, and the use of herbicides (Lieffers and Beck, 1994; Easton and Martin, 1998). In addition to the ecological benefits, it has been suggested that the overall productivity of the land base is increased through mixedwood management and that increased overall economic returns can be anticipated (Lieffers et al., 1996). Mixedwood management may also be necessary for the viability of the sawlog industry as a shortage of large logs is anticipated under the current short-rotation system (Lieffers and Beck, 1994).

### *Distribution of Patch Size*

The distribution of patch size is much less variable than the distribution of stand age and can readily be derived from inventory data. Operationally, the easiest way of maintaining the size distribution of stands is to define harvest blocks on the basis of existing patch boundaries (see Fig. 6.9). However, for operational reasons it is difficult to fully emulate the natural distribution. Specifically, it is uneconomical to cut very small patches (e.g., < 2 ha) and there is public sentiment against the cutting of very large blocks (e.g., > 400 ha). The continued occurrence of small fires and the adoption of structured harvest techniques in place of conventional clearcutting will to some extent mitigate against the absence of very small harvest blocks. The maintenance of large patches can be achieved through careful planning of the spatial pattern of harvest.

Another issue that must be resolved is a tradeoff in the level of detail used to categorize patch types. If too many categories are used, large patches will be artificially fragmented and therefore be underrepresented in the target distribution. However, decreasing the number of habitat categories increases the likelihood that unique habitat types will be missed. Therefore, a balance must be sought.

The main difference between the NDM target for patch size and the sustained-yield management target is again variability (Fig. 6.13). The NDM seeks to maintain a natural range of patch sizes, whereas the targeted patch size under sustained-yield management is typically fixed for each stand type.

### *Spatial Arrangement of Patches*

Although computer programs such as FragStats can quantify many aspects of landscape pattern, the full measure of complexity cannot be sufficiently characterized to establish a meaningful target. The alternative is to use the actual landscape as a guide.

In contrast to the NDM, the selection of harvest blocks under sustained-yield management is based on commercial value and accessibility. There are no landscape pattern objectives.

As illustrated in Figs. 6.4 and 6.8, the aggregation of similar patch types at large spatial scales is a prominent feature of forest landscapes. However, the maintenance of these large-scale patterns is one of the most difficult aspects of the NDM to implement. The problem is that large-scale patterns are in large part the result of intermittent fires that burn large areas irrespective of patch type, whereas forest harvesting seeks to remove selected patch types at a constant rate from the entire management area. Furthermore, the extensive road network that is established to gain access to stands at the time of harvest results in continued disturbance by other forest users (Hunter, 1993). Little research has been conducted on this issue to date; however, it seems obvious that a shift away from dispersed harvesting will be necessary. Large patches of old-growth are likely at greatest risk; therefore, long-term regional planning that ensures the continual existence of such patches on the landscape (e.g., old-growth reserves) should be a priority (Niemela, 1999). Special attention must also be given to riparian zones and wetlands as these regions generally contain unique vegetation patterns that must be maintained (Naiman et al., 1993).

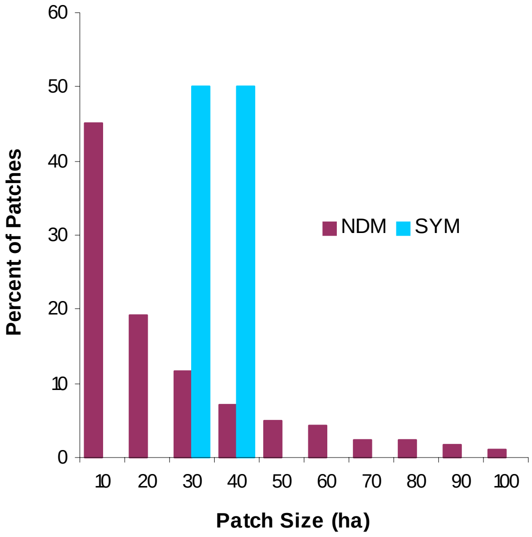

Fig. 6.13. Hypothetical target distribution of patch size for the NDM and sustained-yield management (SYM).

At the local scale, natural shape and local arrangement of patches can be maintained by defining harvest blocks on the basis of existing patch boundaries (see Fig. 6.9). In contrast to the NDM, sustained-yield management makes no attempt to maintain natural landscape patterns; harvest blocks are generally uniform in size and shape and are laid out in a simple grid pattern (Fig. 3.6).

### *Habitats Types not Maintained by the NDM*

Although it should be possible to maintain the majority of habitat types using the NDM, certain habitat types may require special attention. For example, the dead trees remaining after fire constitute critical foraging habitat for black-backed woodpeckers as well as other species (Murphy and Lehnhausen, 1998; Schieck and Song, 2002). Habitat types such as these, that cannot be replicated through harvesting, must be maintained on the landscape through alternative methods in sufficient amounts to ensure the viability of species that depend on them. In the case of post-burn stands this may mean that a proportion of naturally burned stands must be left unsalvaged (Schieck and Song, 2002). In addition to the maintenance of unique habitat types consideration must also be given to the habitat requirements of species that are rare or at risk of extinction.

### *Misuse of the NDM*

Operating within the range of natural variability does not mean that a forestry company can transform the entire forest under its management to the conditions existing at some arbitrarily chosen place and time. If the NDM is properly applied, there should be no significant shifts in the current state of the forest at the regional scale. Variability, which is indeed an important component of the NDM, is primarily expressed at the local scale.

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