Landscape Ecology vol. 7 no. 3 pp 181-194 (1992)
SPB Academic Publishing bv, The Hague

The landscape ecology of large disturbances in the design and management of nature reserves

William L. Baker
Department of Geography and Recreation, University of Wyoming, Laramie, WY 82071, USA

Keywords: disturbances, nature reserves, landscape ecology, management

Large disturbances such as fires and floods are landscape processes that may alter the structure of landscapes in nature reserves. Landscape structure may in turn influence the viability of species and the functioning of ecosystems. Past reserve design and management strategies have been focussed on species and ecosystems rather than on landscape-scale processes, such as disturbance.

An essential feature of a natural disturbance regime is the variation in disturbance attributes (e.g., size, timing, intensity, spatial location). Although some past reserve management policies have included natural disturbances, perpetuating disturbance variation has not been the explicit goal of either reserve design or management.

To design a reserve to perpetuate the natural disturbance process requires consideration of: (1) the size of the reserve in relation to maximum expected disturbance size, (2) the location of the reserve in relation to favored disturbance initiation and export zones and in relation to spatial variation in the disturbance regime, and (3) the feasibility of disturbance control at reserve boundaries, or in reserve buffers.

Disturbance management possibilities are constrained by the design of the reserve and the reserve goals. Where a natural disturbance regime is not feasible, then it is important that the managed disturbance regime mimic historical variation in disturbance sizes and other attributes as well as possible. Manipulating structure on the landscape scale to restore landscapes thought to have been altered by historical disturbance control is premature given our understanding of spatial disturbance processes in landscapes.

1. Introduction

Conservation is moving toward larger scales as environmental problems become more complex, interactive, and global in extent. Global processes, particularly the natural and human processes that affect atmospheric chemistry and ultimately global climate, are increasingly the focus of conservation concern. The contemporary distribution of major biomes may be altered if global warming occurs as a consequence of increases in greenhouse gases (Emanuel *et al.* 1985). Conserving current global form thus requires protection of global processes.

At several levels of biological organization *(i.e.,* populations, species, ecosystems, landscapes), both form and process are conservation concerns. Species conservation efforts have moved beyond simply enclosing a patch of living individuals inside a reserve, to concern with perpetuating other aspects of form, such as genetic variation (Frankel and Soulé 1981) and metapopulation structure, and the processes that perpetuate that form. Ecosystem level conservation similarly requires that the form of the ecosystem, including community composition and diversity, as well as the processes that maintain that form, such as competition, be perpetuated. Recently, conservation attention has turned to the structure of the landscape (Harris 1984; Noss and Harris 1986; Baker 1989a) and the perpetuation of landscape form and process.

Conservation planning is needed that considers all levels of biological organization from population to landscape. Determination of the minimum critical size of a tropical forest reserve from the population viability perspective (Lovejoy *et al.* 1984) is important, but insufficient. It is now apparent that an intact rainforest landscape is essential for maintenance of the tropical forest climate itself (Shukla *et al.* 1990). The success of a reserve designed to protect populations is thus determined in part by the processes in, and structure of the landscape in which the reserve resides.

Landscape-scale process and form have in the past been largely viewed in this way-as externalities which should be considered in planning for conservation goals at other levels of biological organization (e.g., Pickett and Thompson 1978), rather than as conservation concerns in their own right. Yet it is now clear that an interdigitated prairie-forest landscape is a different entity and functions differently from a continuous expanse of either prairie or forest. An emerging tenet of landscape ecology is that the patchy structure of landscapes is important to ecological functioning at a variety of levels of biological organization (Forman and Godron 1986), and is itself worthy of conservation and management attention,

The purpose of this article is to review the process of natural disturbance, particularly large disturbances, and the problems in designing and managing nature reserves in order to perpetuate this process and the structure produced by it. There are many processes, as well as other issues and concerns, some of which may be paramount, that must be addressed in designing and managing nature reserves, but natural disturbance, particularly large disturbance, has received little attention. Past policies have not explicitly considered the variation in the disturbance regime that may be an important source of both spatial and temporal variation in landscapes. I begin with a review of the concept of a disturbance regime, then discuss first the design and second the management of a nature reserve with the disturbance regime in mind.

2. Natural disturbances

Natural disturbances occur in virtually all of the earth's major biomes (White 1979). The role of disturbance in maintaining structure at the species, ecosystem, and landscape scale is being increasingly appreciated. Tropical rainforests, for example, were long thought to be rather stable communities seldom disturbed, but are now recognized to be dynamic communities whose diversity is in part due to disturbances ranging from floods to treefalls to fires (Foster 1980).

Disturbances create a mosaic of patches in landscapes. The structure of this patch mosaic (e.g., patch density, distance between patches, patch size distribution) is important to many species (Forman and Godron 1986). This structure, in landscapes subject to large disturbances, is controlled by the characteristics of the disturbance regime as well as by the landscape itself.

A disturbance can be defined as ". . . any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (White and Pickett 1985 p. 7). What constitutes disturbance, as opposed to normal fluctuation, is dependent on the scale of observation as well as the level of biological organization being considered (Rykiel 1985; Pickett *et al.* 1989). Here I am most concerned with the landscape scale of observation (hundreds of meters to kilometers).

Spatial and temporal sets of disturbance patches are the components of a disturbance regime. Each patch created by a disturbance has a set of attributes, and a disturbance regime can be characterized as the set of frequency or probability density distributions for each patch attribute (Table 1, Figure 1). The form of each distribution can be illustrated by a histogram and quantitatively characterized by values of parameters (e.g., mean, variance, skewness) of standard statistical distributions (e.g., normal, Weibull) that have been fit to empirical data (e.g., Johnson and Van Wagner 1985).

Most of the attribute distributions, including type, shape, intensity, severity, edge, and orientation have never been determined empirically. For example, it has been recognized that multiple disturbance types (e.g., wind, fire, avalanches) affect

Table 1. Some attributes of individual disturbance patches, and the sets of disturbance patches that comprise a disturbance regime.

Each Disturbance Patch	
Type - type of disturbance	
Size - land area disturbed	
Shape - measure by a shape index (cf. Austin 1984)	
Intensity - physical force of a disturbance	
Severity - damage caused by a disturbance	
Timing - temporal setting of a disturbance	
Spatial location - spatial setting of a disturbance	
Edge - total length of the perimeter of a patch	
Orientation - compass direction of central axis of a patch	
Spatial or Temporal Sets of Disturbance Patches	
Type distribution	
Size distribution	
Shape distribution	
Intensity distribution	
Severity distribution	
Timing distribution	
Spatial distribution	
Edge distribution	
Orientation distribution	

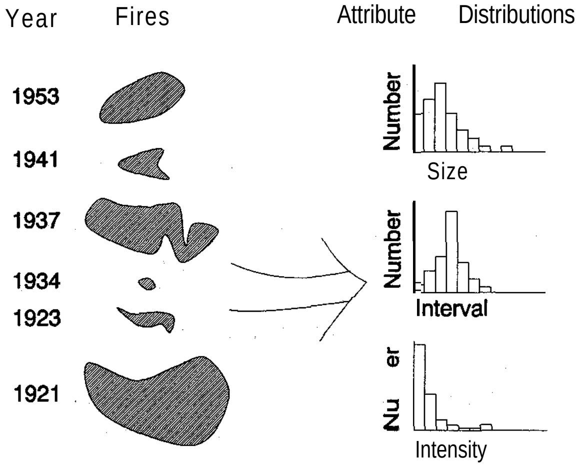

Fig. 1. Some examples of attribute distributions for a disturbance regime.

many landscapes (e.g., Reiners and Lang 1979), yet no one has quantified the type distribution. As a result, disturbance regimes have typically been characterized by a limited set of properties of one or two attributes (White 1979; White and Pickett 1985; Christensen 1988). In the remainder of this section I will discuss what is known about the size, timing, and spatial distributions, as these are better known and perhaps more important.

2. 1. Size distributions

Patch size distributions have not been determined for many disturbance regimes. Size distributions are perhaps best known for treefall gaps, where they have been adequately fit by a negative exponential (Foster and Reiners 1986), a power function (Hubbell and Foster 1986), or a lognormal distribution (Runkle 1982; Arriaga 1988). Size distributions for wave-generated gaps in intertidal landscapes are also lognormally distributed (Paine and Levin 1981). Size distributions for fires have not been statistically fit, but appear to have a negative exponential (van Wagtendonk 1986; Baker 1989a) or power function distribution (Minnich 1983).

2.2. Timing distributions

There are two aspects of timing that are commonly used to characterize disturbance regimes. These are (1) parameters describing the distribution of intervas between disturbances and (2) the frequency of disturbances. Often only the mean of a disturbance interval distribution is reported, with other parameters of the distribution ignored. Disturbance frequency, or the number of disturbances during a particular time interval within some specified land area, is simply the inverse of the mean disturbance interval.

The widespread use of mean disturbance intervals and disturbance frequencies is based in part upon the assumption that disturbance intervals are symmetrically distributed about the mean. The mean fire interval (Romme 1980), for example, is the basis of most of our understanding of fire regimes. Mean disturbance intervals or disturbance frequencies have been calculated or estimated for fires (e.g., Martin 1982), floods (Baker 1990), windstorms (Canham and Loucks 1984), and a variety of other disturbances (White 1979).

However, empirical studies of disturbance interval distributions, such as fire interval distributions, do not support the assumption that disturbance intervals are symmetrically distributed about the mean. Distributions derived from empirical data have a variety of shapes (Johnson 1979; Baker 1989b). When disturbance interval distributions are skewed, then parameters other than the mean may be more meaningful (Romme 1980). A more appropriate approach for these distributions may be to fit a flexible statistical distribution to the empirical disturbance interval data, and use the parameters of the fitted distribution to compare disturbance regimes. For example, the Weibull model has been found to adequately fit empirical distributions of fire intervals, some of which are highly skewed (Johnson 1979; Johnson and Van Wagner 1985; Baker 1989b). This distribution has parameters for shape, scale, and location which can be used to compare disturbance interval distributions between sites.

Variation in disturbance intervals alone, ignoring variation in disturbance size, may result in temporally varying landscape structure. There may be long times without disturbance that allow landscape patches to become comparatively old, followed by times of frequent disturbance that produce young landscape patches. For species sensitive to the age structure of the patch mosaic, variation in disturbance intervals may produce important fluctuations in the availability of habitat (Romme and Knight 1982) that should not be ignored in landscape management.

2.3. Spatial distributions

Spatial variation in the location of disturbances can be substantial even on a local scale. Variation in disturbance initiation and spread may be influenced by topographic setting, elevation, aspect, and slope, as well as by the condition of the vegetation (White 1979; Fowler and Asleson 1984; van Wagtendonk 1986; Foster 1988; Turner and Romme, in press). The result is spatial variation in the density and size, and other attributes of the patch mosaic. Although some generalizations have been made about the factors influencing the spatial location of disturbances (e.g., Swanson *et al.* 1990), there have been only a few studies that have specifically examined quantitative aspects of the spatial variation in the patch mosaic (e.g., Chou *et al.* 1990). Nonetheless, this spatial variation can be expected to be important to species that are sensitive to the structure of the patch mosaic.

Spatial variation in the timing attribute has been studied, however, at both local and continental scales. On a local scale, it has been found that within a 400,000 ha area of relatively uniform topography, fire interval distributions varied from right-skewed to left-skewed, and the Weibull scale parameter (related to expected recurrence interval) varied from about 30 to about 120 (Baker 1989b). On the continental scale, disturbance intervals vary as a result of spatial variation in climatic conditions, vegetation type, and physical setting. Reported mean fire intervals in the United States, for example, vary from less than 10 years in dry pine forests to about 800 years in the forests of the northeastern United States (Turner and Romme, in press). Past disturbance management policies have not explicitly recognized the local spatial variation in the disturbance regime.

3. Design principles

A broad reserve design goal for perpetuating a natural disturbance regime is to ensure that the essential attributes of the disturbance regime (Table 1) are all perpetuated as well as possible within a reserve. This has not in the past been an explicit design goal. As a result, the essential spatial and temporal variation in landscape structure that is a consequence of a fully active disturbance regime is missing from many reserves. When the disturbance regime has been considered, the size attribute of disturbances in relation to the size of reserves has commanded the most attention, although the location of the reserve, the placement of its boundaries, and the possibility of using buffers as part of a biosphere reserve model also warrant consideration.

3.1. Reserve size

It has been suggested that the primary design consideration for disturbances is that a reserve contains a 'minimum dynamic area' (Pickett and Thompson 1978). This area is defined by these authors as '. . . the smallest area with a natural disturbance regime, which maintains internal recolonization sources, and hence minimizes extinction' (p. 34). This definition is thus solely in relation to species viability, which does not necessarily equate with perpetuation of disturbance-induced landscape structure or with ecosystem preservation.

For perpetuating the disturbance regime itself, it may be desirable to find a minimum land area within which a population of disturbance patches has a temporally stable structure (Baker 1989a). The goal would be to ensure that reserve size is sufficient so that disturbances are 'incorporated' (sensu Urban *et al.* 1987) within the reserve. What this means is that the location of patches may shift spatially, but the composite structure of the overall mosaic of patches will remain temporally stable, producing what can be called a 'shifting-mosaic steady state' (Bormann and Likens 1979). A simulation study (Shugart and West 1981) suggested that disturbances could be incorporated within land areas about 50 times the size of disturbances. Empirical studies, however, have not found shifting-mosaic steady states in temperate zone forests subject to stand-destroying fires (Romme 1982), even in a reserve of 400,000 ha (Baker 1989a), which is more than 50 times the mean patch size.

The reason for the difference between simulation and empirical studies is that Shugart and West's simulation used disturbances that were all the same size, while disturbances in nature vary in size and in timing (e.g., Baker 1989b). Variation in disturbance size alone means that landscape structure must fluctuate temporally (Baker 1989a). If, for example, disturbances are chosen randomly and at equal time intervals from a negative exponential size distribution, then the percentage of the landscape that is newly disturbed will fluctuate temporally. If, in addition, unequal-sized disturbances also occur after unequal time intervals, then more marked fluctuation in landscape structure is likely. Fluctuation in landscape structure is thus a direct consequence of the variance in, and the non-normal shapes of the size and timing distributions that comprise disturbance regimes. Fluctuation in landscape structure will occur in any reserve, but will be less if a reserve is large in relation to maximum disturbance size. Reserve size determines how similar the pattern of fluctuation in the reserve will be to the pattern of fluctuation in the larger landscape (Baker 1989a).

When the goal is the perpetuation of a natural disturbance regime and its associated fluctuation in landscape structure, the best strategy is to make reserves large relative to maximum disturbance size. This strategy will maximize the chance that the disturbance regime and the associated pattern of landscape change in the reserve will be similar to the disturbance regime and pattern of change that would have occurred in the larger landscape. Also, if the reserve is large relative to maximum disturbance size, then management problems will be minimized (White 1987). This is primarily because: (1) disturbances will not threaten to destroy the entire reserve at once, leaving it vulnerable to recolonization by weedy or exotic species from the surrounding human-altered landscape; (2) disturbances will less often spread out of the reserve onto surrounding human-occupied lands where they may have adverse economic effects; and (3) the size distribution of disturbances will not be truncated as a result of the suppression of disturbances when they reach the reserve margin.

Choosing to establish several small, rather than a single large reserve is a poor choice in most cases, because each of the small reserves may truncate the patch size distribution attribute of the disturbance regime (Baker 1989a). On the other hand, when each of several reserves is large relative to maximum disturbance size, then several reserves may be placed in a manner that will maximize the protection of greater variation in the environment than might be possible with a single reserve (Baker 1989a).

Maximum disturbance sizes can be obtained from historical records (e.g., Yancik and Roussopoulos 1982), which typically cover only the last several decades, or from reconstruction in the field (e.g., Heinselman 1973; Reiners and Lang 1979). Field reconstruction is difficult, but historical data on disturbance sizes over the last few hundred years are also important, as discussed below, where prescribed disturbances are to be used.

3.2. Reserve location

If the location of the reserve is not constrained by other demands, then there are two locational concerns from the standpoint of disturbance regimes. First, it is important to place the reserve in such a way that both the disturbance initiation zones and disturbance export zones are contained within the reserve. Without control of favored initiation zones, it will be difficult to manage disturbances in the reserve. Without protection of favored directions of disturbance export, it will be difficult politically to maintain a natural disturbance regime within the reserve.

A few examples will illustrate this concern. In mountainous terrain containing snow avalanche paths it is critical to protect the avalanche source area, the entire track, and the run-out zone (Butler 1980). In riparian landscapes, it is important to have some control of activities in the upstream watershed where floods originate as well as protection of flood-prone areas downstream to which floods will be exported. Favored topographic settings for lightning strikes are often on ridgetops at higher elevations in mountainous settings, although the actual number of initiated fires may be highest at intermediate elevations perpendicular to dominant storm tracks (Fowler and Asleson 1984). In cases where fires typically initiate at intermediate elevations and spread upslope, protection of the export zone at higher elevations is desirable. A local analysis of initiation and export patterns is required for each disturbance type and environmental setting.

It has been proposed that a system of long-rotation forest islands be located along riparian strips as a means to minimize the possibility of destruction by fire and to maximize the connectivity between islands (Harris 1984). This location strategy is one of disturbance avoidance rather than disturbance incorporation. For very small reserves or highly valued fragments this may be a reasonable approach, but it must be recognized that this approach means that the system of islands will include only a small part of the spectrum of natural disturbances that has shaped the coniferous forest biome. As such, the system becomes more a system of habitats for species-level conservation than an ecosystem-level or landscape-level reserve system.

The second locational concern is to try to incorporate some of the spatial variation in the disturbance regime. For example, in landscapes subject to fire, areas beyond natural fire breaks may serve as refugia that allow recolonization of disturbed areas. Topographic and elevational variation may similarly provide opportunities for sufficient spatial variation in disturbance regimes so that species with a variety of life history strategies, and ecosystems with a variety of environmental requirements can persist.

3.3. Reserve boundaries

Previous analyses of nature reserve boundaries (Schonewald-Cox and Bayless 1986, Schonewald-Cox 1988) have emphasized that administrative boundaries may not confer needed protection for species and communities due to segmentation of the boundary and adverse effects that extend across the boundary. It is similarly true that administrative boundaries alone may be insufficient to insure that the natural disturbance regime in a reserve is protected. It may be possible, however, to locate the administrative boundary in a manner that minimizes adverse effects on the disturbance regime.

Natural disturbance breaks, for example, make excellent boundaries for reserves as these are locations where disturbances would normally stop spreading. For snow avalanches the valley wall opposite the avalanche path is the disturbance break. The boundary between windward and leeward slopes is an obvious wind disturbance break. Natural fire breaks include lakes and streams, extremely rocky areas such as talus slopes and landslide paths, or areas with little fuel or moist fuels such as

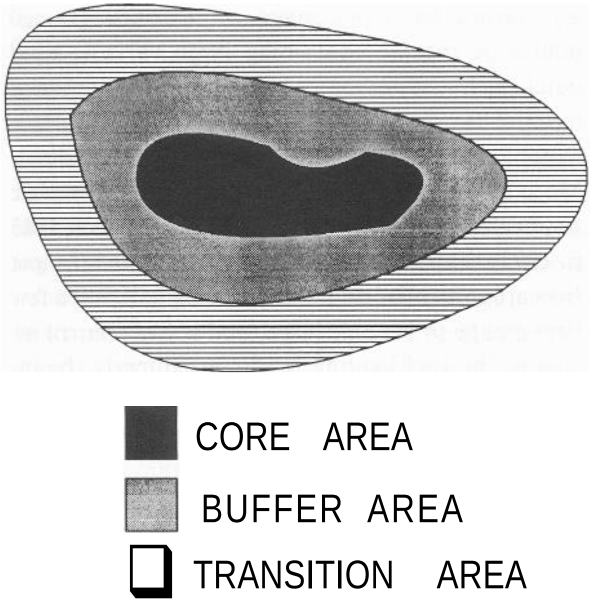

Fig. 2. Major parts of a biosphere reserve

avalanche paths (Malanson and Butler 1984). Where natural disturbance breaks do not exist, roads or other human-constructed breaks can be utilized, but boundaries need to be planned with these in mind.

3.4. The biosphere reserve model

The biosphere reserve model (Gregg *et al.* 1989) and its cousin, the multiple-use-module or MUM (Harris 1984; Noss and Harris 1986), may be good designs for reserves in landscapes subject to natural disturbances. A typical biosphere reserve (Fig. 2) contains a secure central core area designed strictly for nature preservation, a surrounding buffer area in which relatively non-destructive multiple uses may occur, and an outermost transition area in which cooperative economic and research activities that are in harmony with the reserve occur (Gregg *et al.* 1989). A multiple-use-module is like a biosphere reserve, but its central core could contain any area of unusual conservation value, not just a representative ecosystem (Noss and Harris 1986). Increasing attention is now being given to landscape ecology in the design of biosphere reserves (Dyer and Holland 1991), but little attention has been given to design for the natural disturbance regime.

The major advantage of the biosphere reserve model, from the standpoint of natural disturbances, over that of a typical national park or other nature reserve is the presence of buffer zones. If nature reserves become islands in a sea of intensive human land use, then the buffer can absorb some of the effects of external activities. Janzen (1983), for example, argued that it may be better to have cropland than secondary forest around a pristine forest reserve, as fewer species will invade the reserve from cropland. Harris (1984) suggested maintaining managed forests around an old-growth reserve as a way to increase the viability of the reserve and to mimic the patch-mosaic structure of forests.

However, the best buffer for a disturbance regime may differ from the best buffer for maintaining species diversity in a reserve. For the former the best buffer is clearly one which contains a disturbance regime that most closely mimics that in the original ecosystem. For example, secondary forest may be a better buffer than cropland for tropical forest reserves subject to disturbance by cyclones or other high winds, as forest edges next to cleared areas are especially susceptible to blowdown (Foster 1980; Franklin and Forman 1987). Maintaining a secondary forest around a temperate zone old-growth core, in contrast, may increase the export of fires into the core, because of an increase in accidental ignitions or the intentional burning of logging slash in managed forests (Franklin and Forman 1987). However, the advantage of the buffer and transition in the biosphere reserve model is that the buffer does provide a zone in which it may be possible to filter the effects of inappropriate disturbance regimes (e.g., excess fires in managed forests) that occur in the transition area. Fires could be suppressed upon entry into the buffer zone unless they met a particular disturbance management prescription.

4. Reserve management principles

Disturbance management policy is affected by a variety of scientific, political, social, and economic concerns. The focus here is on the scientific aspects of disturbance management policy.

There are at least five major ways to manage disturbances in natural areas (Christensen 1990). These are to use surrogates, suppression, planned disturbance prescriptions, natural disturbance prescriptions, and a natural disturbance regime. I will discuss the first four of these in relation to the set of attributes of a disturbance regime (Table 1), presuming that the reserve management goal is to maintain as well as possible the characteristics of the natural disturbance regime. The last option, managing for a natural disturbance regime, has been discussed in the first part of the paper.

4.1. Surrogates

To use a surrogate means to simply substitute one type of disturbance for another (e.g., logging for fire). Surrogates may be necessary if the reserve protection goal is at the species or ecosystem level, disturbance is known to be required for these species or ecosystems, and it is not feasible to use other methods. It is possible to use surrogates to mimic some of the attributes of a disturbance regime, such as the size, shape, timing and spatial distributions of the original disturbance regime, but surrogates usually do not allow mimicry of the intensity distribution, and may have quite different physical effects on the landscape and biota.

4.2. Suppression

Only a few kinds of disturbances have been effectively controlled, and disturbance control is now known in some instances to have long-term effects that may be the reverse of the desired effect. A great deal is known about the effects of disturbance control on individual species and on certain communities, but very little is known about the effects of disturbance control on the structure of landscapes.

The effects of fire suppression on landscapes have been considered in several studies, but a comprehensive understanding is lacking. In the case of fires,-it has been suggested that fire control eventually results in 'replacement of a coarse-grained landscape mosaic containing large variable-sized patches by a more homogenous fine-grained mosaic' (Forman and Boerner 1981 p. 46). This may be the case where large fires can be prevented. More commonly, with fire control 'a fine mosaic has been replaced by a coarser one' (Minnich 1983 p. 1293). This latter trend is to be expected if most fires are controlled and thus remain small, but a few fires escape to become large, and if fire control increases the fuel loading, and consequently the intensity and size of subsequent fires. Fire suppression may also result in an increase in some components of landscape diversity (Romme 1982) and significant alterations in several other measures of landscape structure (Baker, in press). Dams and other water developments also primarily control the smaller floods. It is not known, however, whether flood control will eventually result in larger flood-disturbed patches. Although additional research is needed, it is clear that disturbance control has the potential to significantly alter spatial and temporal variation in the structure of landscapes.

In small reserves where the management'goal is to suppress very large disturbances that would destroy most of the reserve the long-term effect of disturbance control on landscape structure may be a lesser concern. In this situation, disturbance suppression may increase the probability of larger disturbances occurring, but all disturbances above some cutoff size will have to be suppressed in any case.

4.3. Planned and natural disturbance prescriptions

These two approaches are identical except that planned disturbances are intentionally started and natural disturbances start themselves. In both cases the disturbance is only allowed to continue if it meets a particular pre-determined need, or prescription. This is a common approach for fire management in cases where a natural fire regime is not feasible (cf. Wright and Bailey 1982).

A primary concern of planned disturbance efforts is often the physical feasibility of a planned burn, and the goal of planned disturbances is usually to modify species habitat or community structure. For example, in determining the size of prescribed fires, the main considerations are often the amount of acreage that can be burned within a single controlled burning period, the cost of establishing fire lines and monitoring the fire, and the size of fire that can effectively be controlled (Bunting *et al.* 1987). When species are the target of conservation, the goal may be to simply maintain a certain percentage of the landscape in a seral or climax stage favorable to a certain species (e.g., Garcia **1986),** with little concern for the effects on the landscape. On the community or ecosystem level disturbances may be used to restore community structure (White 1986) or maintain species diversity and community composition (Gibson 1988). In the few cases where the landscape ramifications of prescribed disturbances have been considered, the recommendations have often been to use small disturbances to create a heterogenous habitat (Foster 1980; Bunting *et al.* **1987),** presumably to increase species diversity or to favor certain species, rather than to manage the landscape itself.

In contrast, using prescribed disturbances to explicitly manage form and process in landscapes requires consideration of the historical variation in the attributes of the disturbance regime (Table 1). If the goal is to perpetuate the landscape disturbance process as well as possible, then it is desirable to try to vary disturbances so that the set of disturbances over some time and space scale has attribute distributions similar to those that occurred in the natural landscape (**Agee** and Huff 1986). Prescribed fires, for example, should vary in size, be initiated more frequently in favored initiation zones, and vary in intensity and time of burn in a manner similar to fires that occurred in the natural landscape. This kind of disturbance management requires substantial knowledge about the natural disturbance regime and a simultaneous consideration of effects on species, ecosystems, and the landscape.

From the landscape perspective it is clear that multiple small disturbances, which are easier to control, cannot substitute for a single large disturbance. It has been proposed that very large fires, such as burned in Yellowstone National Park in

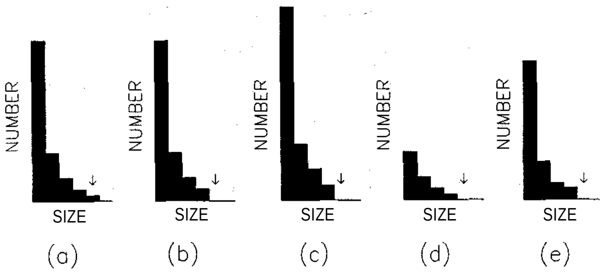

Fig. 3. Four options for mimicking the size distribution when using prescribed disturbances in a reserve smaller than maximum historical disturbances. The arrow indicates the size of the reserve. (a) the natural size distribution, (b) truncated size distribution, (c) truncated and redistributed size distribution, (d) shifted size distribution, (e) shrunk size distribution.

1988, are unnatural, and that “scientific management can restore a more natural condition and create higher biotic diversity on the burned areas than ‘letting nature take its course’ while also reducing the size and destructiveness of future fires” (Bonnicksen 1989). Scientific management goals may vary, but excluding large fires from landscapes that historically had large fires (Romme and Despain 1989) is one that will significantly alter landscape structure from its natural condition. Based on a simulation analysis of fires in the Boundary Waters Canoe Area in Minnesota, a disturbance regime with small prescribed fires, compared to a natural disturbance regime, can be expected to significantly alter landscape structure (Baker, in press). An essential component of a scientific landscape management approach guided by history is to mimic the size distribution, as well as perhaps other attribute distributions, of historical disturbances. Multiple small disturbances are not a substitute for large disturbances.

Where the size of the reserve is significantly smaller than the size of historical disturbances, and simultaneous disturbance of the whole reserve is not desirable due to species-level or ecosystem-level concerns, then what aspect of the natural size distribution should be mimicked in a prescribed disturbance regime? There are at least four options (Fig. 3). With the first option the size distribution is simply truncated below the reserve size (Fig. 3b). The result is that none of the structure that was present in the natural landscape will be altered, except that structure produced by very large disturbances. With the second option (Fig. 3c), the truncated disturbances are redistributed so that the curve has the same general shape, but is taller. With the third option, part of the curve is preserved, but it is shifted to the left so that maximum prescribed disturbance sizes are smaller than the reserve size (Fig. 3d). This rescales the whole patch mosaic, decreasing the number of small disturbances, which means that the distance between patches will increase. In so doing this option will create a landscape mosaic which is perhaps different than has existed in the landscape in the past. The last option shrinks the curve down, but preserves its shape (Fig. 3e). This again rescales the patch mosaic, but in a more equitable manner. There is no perfect choice among these options, and it is likely that disturbance prescriptions will be imprecisely achieved in any event. Nonetheless, truncation (Fig. 3b) alters landscape structure the least, and may be comparatively easy to achieve if fixed disturbance breaks that establish a maximum disturbance size can be used.

An additional concern for determining disturbance prescriptions is the effect of patch relaxation (Baker 1989a). If reserves become isolated, so that disturbances no longer spread into the reserve area from the outside, then the disturbance regime will be altered even if all disturbances that originate within the reserve are allowed to spread naturally. This effect usually increases as reserve size decreases. In small isolated reserves prescribed disturbances thus could be necessary even if there is a no-suppression policy within the reserve.

The Konza Prairie Research Natural Area is an example of a reserve in which natural disturbances are managed entirely by prescription (Marzolf 1988) in a manner inconsistent with perpetuating landscape process and form. The 3487 ha reserve is divided into about 50 treatment areas, that are primarily small watersheds. Fire treatments and grazing treatments are superimposed, so that some watersheds are burned and grazed, others are unburned but grazed, etc. A suite of treatments includes different year-to-year fire intervals, but all fires are ignited in the Spring, so seasonal variation is not included. The entire watershed is apparently burned at once in all cases. Although ostensibly a nature reserve, the Konza is actually managed as an experimental area in which the goal of the experiments is to understand the effects of different fire intervals and grazing on species and ecosystems. Other aspects of the disturbance regime are ignored. There is no question that these experiments are invaluable, but the natural disturbance regime in tallgrass prairies undoubtedly was not one with fixed patch sizes at regular intervals in the same location. Such a disturbance regime can be expected to alter the populations of organisms in the tallgrass prairie that are sensitive to the natural pattern of fire-induced variation in landscape structure, particularly those organisms that are sensitive to patch-to-patch distance or patch size (both of which are now fixed at one scale), or depend upon irregularities in the timing of fires.

4.4. Structure and process goals in restoration

Some researchers believe that disturbance suppression and other human effects have so altered the composition and structure within some reserves that it is first necessary to restore natural structure before natural disturbance processes can be reintroduced (Bonnicksen and Stone 1982, 1985). In the case of fires, these authors argue that if natural structure is not first restored to what it would have been today if European settlers had not interfered with natural processes (identified using their 're-construction-simulation approach'), then reintroduction of the process may result in unnaturally large fires, due to fuel buildup over decades of fire suppression. It has been the policy of the National Park Service, since the recognition of the undesirable effects of fire suppression, to use prescribed fires as a means to reduce unnatural fuel accumulations prior to reintroducing a natural fire policy (Parsons *et al.* 1986). Nonetheless, Parsons *et al.* argue that there are too many uncertainties and too many interacting sources of landscape change in the period since settlement to attempt to precisely re-create the landscape as it would have developed had European settlement never occurred. Thus these authors suggest allowing '. . . process-structure interactions to equilibrate on their own after one or more prescribed fires' (Parsons *et al.* 1986, p. 23).

There are two reasons that both the Parsons *et al.* perspective and the Bonnicksen and Stone perspective may be inconsistent with the landscape management perspective that I have presented. First, prescribed burning is not a panacea, because prescribed burning that is inconsistent with the historical disturbance regime's attribute distributions may further alter landscape structure, rather than restore it. In particular, using small prescribed fires to '. . . break up large areas of homogenous heavy fuels, thus reducing the chances of a high intensity wildfire . . . ' (Parsons *et al. 1986)* may impose a smaller mean patch size on the landscape than has ever been present historically, as well as prevent high intensity wildfires that may historically have been an important part of the disturbance regime (Turner and Romme, in press, Parsons *et al.,* in press). These effects are not consistent with management for natural landscape-level process and form. Fluctuation in landscape structure due to variation in disturbance size and timing may allow parts of natural landscapes to accumulate large amounts of fuel at certain times. This accumulation in itself is not necessarily unnatural. Its naturalness can only be assessed by comparing recent characteristics of the disturbance regime's attribute distributions with historical characteristics.

Second, while Bonnicksen and Stone's (1982) 'reconstruction-simulation approach' appears to produce a reasonable target for landscape restoration, their simulator produces estimates of proportions of the landscape that are in a particular state, but ignores the spatial distribution of these states. There is increasing evidence that the spatial structure of landscape states is important to many plants and animals, and itself influences the future pattern of disturbances (Forman and Godron 1986; Turner and Romme, in press). Bonnicksen and Stone's reconstruction-simulation approach needs to replicate spatially explicit changes in landscape structure before it can be used to produce a precise structure target. Landscape modelling is not, however, currently sufficiently precise to produce this kind of reconstruction (Baker 1989c).

If a spatially accurate reconstruction could be produced, then the structural restoration goal presumably would be to restore the spatial pattern of each of the landscape states to what it would have been now had European settlement not occurred. But one cannot restore what is now a 150 year old patch at a particular point in the landscape to a 75 year old state, if that is what the reconstruction suggests would have been present without European settlement.

Where restoration of the disturbance-induced landscape structure is necessary, the best approach may be to design a transitional restoration-oriented disturbance regime that will move the landscape back toward the expected structure. This regime might be a quite different disturbance regime than ever occurred in the landscape historically. In fire-prone landscapes, this transitional regime would probably seldom consist of the small prescribed fires that have traditionally been used to restore ecosystems subject to fire suppression, as small fires may further alter the structure of landscapes that historically experienced large fires (Baker, in press). A simulation approach can be used to develop the transitional disturbance regime (Baker *et al.* 1991). Until this transitional regime is designed it is premature to undertake extensive manipulative restoration action using either prescribed disturbances or mechanical means, as these may only produce undesirable alteration.

4.5. Landscape modelling

Reserve managers often must make decisions that have unclear long-term ramifications for landscapes, ecosystems, and species. Spatial models of landscape change are now becoming available that may enable simulation of future landscape structure under a variety of management scenarios (e.g., Costanza *et al.* 1990, Baker *et al.* 1991). Such models could be used to examine how disturbances will interact with a landscape as an aid to designing reserves, to determine what the effects of external human activities might be, and to analyze how the landscape structure in the reserve may fluctuate over time. Models with explicit links to climate could be used to simulate future landscape dynamics given particular climatic change scenarios (Baker *et al.* 1991; Turner and Romme, in press).

5. Conclusions

Natural disturbance is an important process that affects species, ecosystems, and landscape structure. More information is needed on natural disturbance regimes if these regimes are to be effectively managed in nature reserves. In designing a reserve it is important to consider needed reserve size, reserve location, reserve boundaries, and reserve buffer in relation to the attributes of the disturbance regime. The ability to perpetuate the natural disturbance regime within the reserve is dependent upon how well designed the reserve is. Natural disturbance regimes are probably only feasible within reserves that are: (1) several times the maximum disturbance size typical of the region, (2) located so that disturbance initiation and export zones are contained within the reserve, and (3) have boundaries along natural or artificial disturbance breaks or have buffer zones within which disturbances can be controlled. In reserves that do not have these attributes a variety of alternative management strategies can be considered. Significant manipulative restoration actions on the landscape scale, using prescribed disturbances or mechanical means, are premature given the state of knowledge about spatial aspects of disturbances in landscapes. Landscape-level management may conflict with species or ecosystem-level management in some instances, but it is essential that some of our large reserves focus upon perpetuating a natural disturbance regime, as the role of natural disturbances on the landscape scale is only beginning to be understood.

6. Acknowledgments

This research was completed in part with funds from the Ecological Research Division, Office of Health and Environmental Research, U.S. Department of Energy under Grant DE-FGOZ 90ER60977. This support does not constitute an endorsement by DOE of the views expressed in this article. I thank Curt Soper for the opportunity to speak with the western regional land stewards of The Nature Conservancy about this subject. I appreciate the comments of James Agee and Thomas Bonnicksen on an earlier version of this manuscript.

References

Agee, J.K. and Huff, M.H. 1986. Structure and process goals for vegetation in wilderness areas. *In* Proceedings-national wilderness research conference: current research [Fort Collins, Colo.-July 23-26, 1985]. pp. 17-25. Edited by R.C. Lucas. USDA Forest Service General Technical Report INT-212, Intermountain Research Station, Ogden, Utah.

Arriaga, L. 1988. Gap dynamics of a tropical cloud forest in northeastern Mexico. Biotropica 20: 178–184.

Austin, R.F. 1984. Measuring and comparing two-dimensional shapes. In Spatial statistics and models. pp. 293–312. Edited by G.L. Gaile and C.J. Willmott. D. Reidel Publ. Co., Boston.

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

Baker, W.L. 1989b. Effect of scale and spatial heterogeneity on fire-interval distributions. Can. J. For. Res. 19: 700-706.

Baker, W.L. 1989c. A review of models of landscape change. Landscape Ecol. 2: 111-133.

Baker, W.L. 1990. Climatic and hydrologic effects on the regeneration of *Populus angustifolia* James along the Animas River, Colorado. J. Biogeogr. 17: 59-73.

Baker, W.L., in press. Effects of settlement and fire suppression on landscape structure. Ecology, in press.

Baker, W.L., Egbert, S.L. and Frazier, G.F. 1991. A spatial model for studying the effects of climatic change on the structure of landscapes subject to large disturbances. Ecol. Modelling 56: 109-125.

Bonnicksen, T.M. 1989. Fire gods and federal policy. Am. Forests 95(7–8): 14-16, 66-68.

Bonnicksen, T.M. and Stone, E.C. 1982. Managing vegetation within U.S. national parks: a policy analysis. Environ. Manage. 6: 109-122.

Bonnicksen, T.M. and Stone, E.C. 1985. Restoring naturalness to national parks. Environ. Manage. 9: 479-486.

Bormann, F.H. and Likens, G.E. 1979. Catastrophic disturbance and the steady state in northern hardwood forests. Am. Sci. 67: 660-669.

Bunting, S.C., Kilgore, B.M. and Bushey, C.L. 1987. Guidelines for prescribed burning sagebrush-grass rangelands in the northern Great Basin. U.S. Department of Agriculture, Forest Service General Technical Report INT-231, Intermountain Research Station, Ogden, Utah.

Butler, D.R. 1980. Terminal elevations of snow avalanche paths, Glacier National Park, Montana. Northwest Geol. 9: 59-64.

Canham, C.D. and Loucks, O.L. 1984. Catastrophic windthrow in the presettlement forests of Wisconsin. Ecology 65: 803-809.

Chou, Y.-H., Minnich, R.A., Salazar, L.A., Power, J.D. and Dezzani, R.J. 1990. Spatial autocorrelation of wildfire distribution in the Idyllwild quadrangle, San Jacinto Mountain, California. Photogr. Eng. & Rem. Sens. 56: 1507-1513.

Christensen, N.L. 1988. Succession and natural disturbance: paradigms, problems, and preservation of natural ecosystems. *In* Ecosystem management for parks and wilderness. pp.62–86. Edited by J.K. Agee and D.R. Johnson. University of Washington Press, Seattle.

Christensen, N.L. 1990. Variable fire regimes on complex landscapes: ecological consequences, policy implications, and management strategies. Paper presented at the International symposium on fire and the environment: ecological and cultural perspectives [Mar. 20-24, 1990-Knoxville, Tennessee].

Costanza, R., Sklar, F.H. and White, M.L. 1990. Modeling coastal landscape dynamics. Bioscience 40: 91-107.

Dyer, M.I. and Holland, M.M. 1991. The biosphere-reserve concept: needs for a network design. BioScience41: 319-325.

Emanuel, W.R., Shugart, H.H. and Stevenson, M.P. 1985. Climate change and the broad-scale distribution of terrestrial ecosystem complexes. Climatic Change 7: 29-43.

Forman, R.T.T. and Boerner, R.E. 1981. Fire frequency and the pine barrens of New Jersey. Bull. Torr. Bot. Club 108: 34-50.

Forman, R.T.T. and Godron, M. 1986. Landscape ecology. John Wiley and Sons, New York.

Foster, D.R. 1988. Disturbance history, community organization and vegetation dynamics of the old-growth Pisgah Forest, south-western New Hampshire, U.S.A. J. Ecol. 76: 105-134.

Foster, J.R. and Reiners, W.A. 1986. Size distribution and expansion of canopy gaps in a northern Appalachian spruce-fir forest. Vegetatio 68: 109-l 14.

Foster, R.B. 1980. Heterogeneity and disturbance in tropical vegetation. *In* Conservation biology. pp. 75-92. Edited by M.E. Soulé and B.A. Wilcox. Sinauer Associates, Inc., Sunderland, Massachusetts.

Fowler, P.M. and Asleson, D.O. 1984. The location of lightning-caused wildland fires, northern Idaho. Phys. Geogr. 5: 240-252.

Frankel, O.H. and Soulé, M.E. 1981. Conservation and evolution. Cambridge University Press, Cambridge.

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

Garcia, E.R. 1986. Grizzly bear direct habitat improvement on the Kootenai National Forest. *In* Proceedings-grizzly bear habitat symposium [Missoula, Montana-April 30-May 2, 1985]. pp. 185-189. Edited by G.P. Contreras and K.E. Evans. U.S. Department of Agriculture, Forest Service General Technical Report INT-207, Intermountain Research Station, Ogden, Utah.

Gibson, D.J. 1988. Regeneration and fluctuation of tallgrass prairie vegetation in response to burning frequency. Bull. Torr. Bot. Club 115: 1-12.

Gregg, W.P., Jr., Krugman, S.L. and Wood, J.D., Jr., editors. 1989. Proceedings of the symposium on biosphere reserves, fourth world wilderness congress [Sept. 14-17, 1987-Estes Park, Colorado]. U.S. Department of the Interior, National Park Service, Atlanta, Georgia.

Harris, L.D. 1984. The fragmented forest. University of Chicago Press, Chicago.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quat. Res. 3: 329-382.

Hubbell, S.P. and Foster, R.B. 1986. Canopy gaps and the dynamics of a Neotropical forest. *In* Plant ecology. pp. 77-96. Edited by M.J. Crawley. Blackwell Scientific Publications, Oxford.

Janzen, D.H. 1983. No park is an island: increase in interference from outside as park size decreases. Oikos 41: 402-410.

Johnson, E.A. 1979. Fire recurrence in the subarctic and its implications for vegetation composition. Can. J. Bot. 57: 1374-1379.

Johnson, E.A. and Van Wagner, C.E. 1985. The theory and use of two fire history models. Can. J. For. Res. 15: 214–220.

Lovejoy, T.E., Rankin, J.M., Bierregaard, R.O., Brown, K.S., Emmons, L.H. and Van der Voort, M.E. 1984. Ecosystem decay of Amazon forest fragments. *In* Extinctions. pp. 295-325. Edited by M.H. Nitecki. University of Chicago Press, Chicago.

Malanson, G.P. and Butler, D.R. 1984. Avalanche paths as fuel breaks: implications for fire management. J. Environ. Manage. 19: 229-238.

Martin, R.E. 1982. Fire history and its role in succession. *In* Forest succession and stand development research in the Northwest. pp. 92-99. Edited by J.E. Means. Forest Research Laboratory, Oregon State University, Corvallis, Oregon.

Marzolf, R. 1988. Konza prairie research natural area of Kansas State University. Trans. Kans. Acad. Sci. 91: 24-29.

Minnich, R.A. 1983. Fire mosaics in southern California and northern Baja California. Science 219: 1287-1294.

Noss, R.F. and Harris, L.D. 1986. Nodes, networks, and MUMs: preserving diversity at all scales. Environ. Manage. 10: 299-309.

Paine, R.T. and Levin, S.A. 1981. Intertidal landscapes: disturbance and the dynamics of pattern. Ecol. Monogr. 5: 145-178.

Parsons, D.J., Graber, D.M., Agee, J.K. and Van Wagtendonk, J.W. 1986. Natural fire management in national parks. Environ. Manage. 10: 21-24.

Parsons, D.J., Stephenson, N.L. and Swetnam, T.W. (in press). Restoring natural fire to the sequoia-mixed conifer forest: should intense fire play a role? Proceedings of the Tall Timbers Fire Ecology Conference 17, in press.

Pickett, S.T.A., Kolasa, J., Armesto, J.J. and Collins, S.L. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. Oikos 54: 129-136.

Pickett, S.T.A. and Thompson, J.N. 1978. Patch dynamics and the design of nature reserves. Biol. Cons. 13: 27-37.

Reiners, W.A. and Lang, G.E. 1979. Vegetational patterns and processes in the balsam fir zone, White Mountains, New Hampshire. Ecology 60: 403-417.

Romme, W. 1980. Fire history terminology: report of the ad hoc committee. *In* Proceedings of the fire history workshop [Tucson, Arizona-Oct. 20-24, 1980]. pp. 135-137. Edited by M.A. Stokes and J.H. Dieterich. USDA Forest Service General Technical Report RM-81, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Romme, W.H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecol. Monogr. 52: 199-221.

Romme, W.H. and Despain, D.G. 1989. Historical perspective on the Yellowstone fires of 1988. Bioscience 39: 695-699.

Romme, W.H. and Knight, D.H. 1982. Landscape diversity; the concept applied to Yellowstone Park. Bioscience 32: 664-670.

Runkle, J.R. 1982. Patterns of disturbance in some old-growth mesic forests of eastern North America. Ecology 63: 1533–1546.

Rykiel, E.J., Jr. 1985. Towards a definition of ecological disturbance. Austr. J. Ecol. 10: 361-365.

Schonewald-Cox, C.M. 1988. Boundaries in the protection of nature reserves. Bioscience 38: 480-489.

Schonewald-Cox, C.M. and Bayless, J.W. 1986. The boundary model: a geographical analysis of design and conservation of nature reserves. Biol. Conserv. 38: 305-322.

Shugart, H.H., Jr. and West, D.C. 1981. Long-term dynamics of forest ecosystems. Am. Sci. 69: 647-652.

Shukla, J., Nobre, C. and Sellers, P. 1990. Amazon deforestation and climate change. Science 247: 1322– 1325.

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

Turner, M.G. and Romme, W.H. (in press), Landscape dynamics in crown fire ecosystems. *In* Pattern and process in crown fire ecosystems. Edited by R.D. Laven and P.N. Omi. Princeton University Press, Princeton.

Urban, D.L., O'Neill, R.V. and Shugart, H.H., Jr. 1987. Landscape ecology. Bioscience 37: 119-127.

van Wagtendonk, J.W. 1986. The role of fire in the Yosemite Wilderness. *In* Proceedings-national wilderness research conference: current research [Fort Collins, Colo.-July 23-26, 1985]. pp. 2-9. Edited by R.C. Lucas. USDA Forest Service General Technical Report INT-212, Intermountain Research Station, Ogden, Utah.

White, A.S. 1986. Prescribed burning for oak savanna restoration in central Minnesota. U.S. Department of Agriculture, Forest Service Research Paper NC-266, North Central Forest Experiment Station, St. Paul, Minnesota. 12 pp.

White, P.S. 1979. Pattern, process, and natural disturbance in vegetation. Bot. Rev. 45: 229-299.

White, P.S. 1987. Natural disturbance, patch dynamics, and landscape pattern in natural areas. Nat. Areas J. 7: 14-22.

White, P.S. and Pickett, S.T.A. 1985. Natural disturbance and patch dynamics: an introduction. *In* The ecology of natural disturbance and patch dynamics. pp. 3-13. Edited by S.T.A. Pickett and P.S. White. Academic Press, New York.

Wright, H.A. and Bailey, A.W. 1982. Fire ecology. John Wiley and Sons, New York.

Yancik, R.F. and Roussopoulos, P.J. 1982. User's guide fo the national fire occurrence data library. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.