# The Myth of Nature's Constancy—Preservation, Protection and Ecosystem Management

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## Introduction

Two recent essays by noted ecologists Daniel Botkin and Jared Diamond throw light on certain issues in natural resource conservation today. Botkin (1992), in *A Natural Myth*, relates his experience with the Hutcheson Memorial Forest, an oak forest reserve on the New Jersey Piedmont. The forest had been set aside with private funds in the 1950s to preserve the "state of harmonious balance" that would perpetuate itself for centuries if left undisturbed. Within two decades, however, it became obvious that the oaks were not regenerating and a dense maple understory had developed. So much for harmonious balance. Later studies revealed that fires, probably set by Indians, had occurred at about 10–year intervals prior to, but not after, European settlement in 1701. Fires removed the understudy and favored oaks, creating the tall, open forest which naturalists in the 1950s and 1960s thought to be original and unaffected by human influence.

Jared Diamond (1992), in *Must We Shoot Deer to Save Nature?*, describes changes in Fontenelle Forest, a mature oak/hickory reserve on the floodplain of the Missouri River near Omaha. A 1,300–acre fragment of the once vast floodplain forests of the Missouri drainage, Fontenelle Forest was privately set aside to preserve it in its natural state—all plant and animal life is protected, no hunting is allowed. The philosophy for operating the reserve is one of no management, no human interference.

After describing the beauty of the place, Diamond recounts his dismay after a closer look: no oak or hickory seedlings, few acorns and nuts. The few understory stems were of ironwood and hackberry, plants of disturbed areas which disperse by wind-blown seeds or tiny fruits. Herbs such as snakeroot and stinging nettle had replaced oak and hickory seedlings on the forest floor. The forest was undergoing reverse succession. White-tailed deer (*Odocoileus virginianus*) were the culprits and heavy browsing indirectly affected understory birds, butterflies and wintering jays. The rules of non-interference had frustrated the goal of preserving the forest "in its natural state," i.e., the way it looked when the decision was made to preserve it.

Botkin and diamond have pointed out the paradox that the goals of non-interference with nature and preservation of natural habitats can be incompatible. Examples ranging from elephant damage to Kenya's Tsavo National Park to deer overbrowsing the Gettysburg National Historical Monument show that nature reserves probably can't be left to nature to manage. The same largely is true for designated wilderness areas, where it is becoming evident that fire suppression, changes in surrounding landscapes and environmental contaminants have profound effects on the wilderness itself. For example, many wilderness areas were not designated with ecosystem or biodiversity goals in mind,

and, while the intent is for management by natural forces, normal natural disturbance patterns are rarely achieved.

Why does simple preservation often fail to achieve our expectations for ecosystems protection? We suggest that it is the set of relevant ecosystem processes rather than a given stage of development that must be preserved. We will emphasize the importance of regional differences in disturbance regimes and ecosystem processes, and the potential values of active management in preserving ecosystems based on our experiences in New England.

## Preservation—States or Processes?

The idea of preserving a landscape in its natural state flows logically from the idea of natural succession to a climax community. Given enough time, ecosystems will tend toward a steady state of dynamic equilibrium (Bormann and Likens 1979). Consequently, the obvious strategy for achieving or preserving a climax state would seem to be to leave things alone.

Recent evidence, however, suggests the climax model may be inappropriate and that constant change is the rule for North American ecosystems (Botkin 1990, Pielou 1991). Furthermore, there is ample evidence that forests in eastern North America are still responding to the last glacial cycle (for a summary, *see* Davis 1976). If natural systems change constantly, preservation alone will rarely, if ever, maintain a particular ecosystem condition. Instead of preserving certain ecosystem states, we need to think in terms of maintaining ecosystem processes that are within our control. This entails taking the long view, realizing that plant and animal communities at a given site will change, sometimes dramatically, over time.

Eliminating human activity from a landscape may not produce a climax state, but it may protect the ecosystem if enough of it can be reserved. Presumably, if a reserve encompassed an entire ecosystem type, no management would be necessary because the full range of natural disturbance regimes, successional stages and species would be included. Few, if any, reserves or management areas encompass an entire ecosystem type; most include only a small fraction of the ecosystem type. While entire ecosystems need not necessarily be preserved to have the full range of disturbance regimes, stages and species, the extent does depend on the types of disturbance that characterize the ecosystem and the area requirements of particular species. Most reserves are too small to have disturbances that are frequent or large enough to maintain viable populations of early successional species. The smaller the portion of an ecosystem type that is protected, the more likely that native species, including key species such as top predators and large herbivores, will be missing, and natural processes will be interrupted and exotic species will be present. When only part of an ecosystem type is under protection, it is likely that management will be necessary to maintain the disturbance regimes, species and processes that shaped the original ecosystem.

## Regional Differences

It follows that management must be conducted in a regional context that recognizes the disturbance and climatic regimes and geological factors that shaped the ecosystem in the recent past. Forces that shaped the presettlement forests of the Atlantic coastal plain, the Ohio River valley and New England are very different (Figure 1). Yet, concerns

founded in one region often influence public opinion and management decisions in another, sometimes with little biological justification. For example, concerns about clear-cutting and brown-headed cowbird (*Molothrus ater*) parasitism voiced in mid-western woodlots, where gaps are the major forest disturbance, are echoed in northern New England where forests are extensive, big blowdowns are a major forest disturbance and cowbirds are uncommon. Obviously, management that would be appropriate in one region may not be in another.

Much of the concern about forest management in the eastern United States has focused on migratory birds since it became obvious that many of these species had undergone severe population declines at several widely scattered locations. Precipitous population declines occurred in the 1960s and 1970s at particular sites in the Middle Atlantic area (Briggs and Criswell 1978, Robbins 1979), New Jersey (Leck et al. 1988), upstate New York (Litwin and Smith 1992), Connecticut (Butcher et al. 1981) and Wisconsin (Ambuel

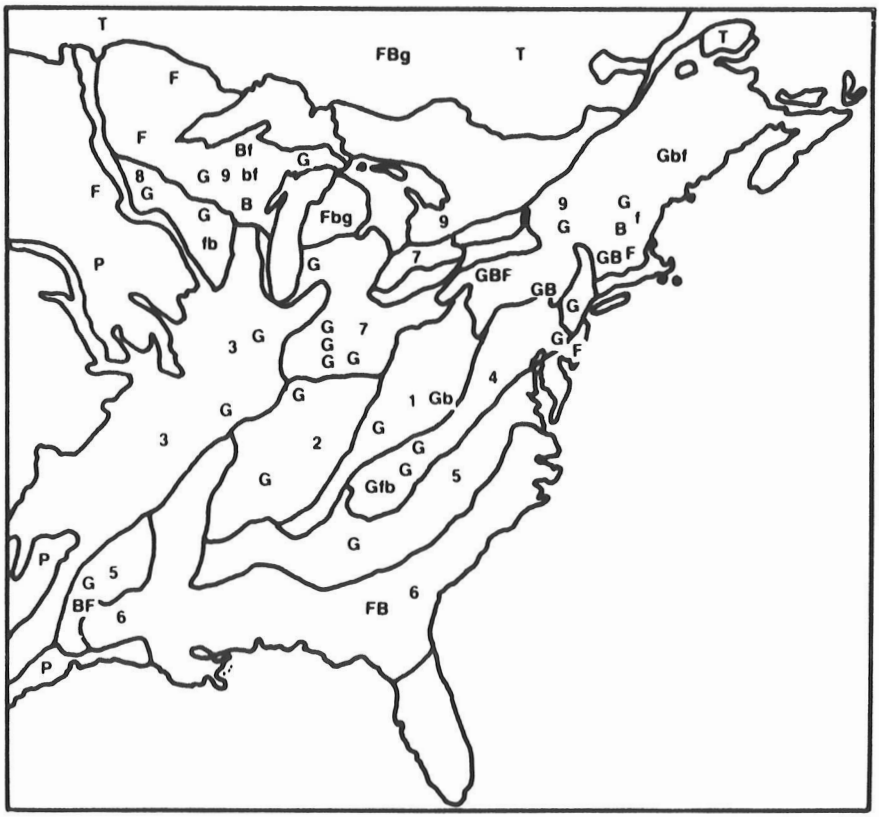


Figure 1. Geography of disturbance for the eastern deciduous forest (from Runkle 1990). F. and *f* locations indicate where fire was a major and minor importance, respectively; B and *b*, where big blowdowns were of major or minor importance, respectively; G and *g*, where gaps were of major or minor importance, respectively. The literature on which this figure is based and the names of the forest regions numbered on the figure, are given in Braun (1950).

and Temple 1982). Most of the species involved are so-called forest interior birds, species not usually found in open or early successional habitats. Furthermore, most are neotropical migrants. These declines have been hypothesized to have resulted from the fragmentation of forest near the study sites, resulting in the increasing isolation of the small patches (generally <100 ha) due to rapid suburbanization since 1950 (Askins et al. 1990).

It is well established that many species of forest migrants have low reproductive rates in small isolated forest fragments (Wilcove 1985, Robinson 1988, Small and Hunter 1988). In Missouri, male ovenbirds in small forest patches were less likely to be mated than males in large forests (Gibbs and Faaborg 1990). In small woodlots in Illinois, Robinson (1988) estimated that 80 percent of cup nests were destroyed by predation and 65 percent were parasitized by cowbirds. In large and small forests in the eastern U.S., the overall bird abundance and species richness are similar, but species composition is different: more forest interior migrant species occur in large forests and more generalist species occur in small forests (Whitcomb et al. 1981, Lynch and Whigham 1984, Freemark and Merriam 1986).

The brown-headed cowbird has long been known to parasitize nests of neotropical migratory birds, especially in the Midwest where the eastern deciduous forest gives way to the Great Plains (Leopold 1924, Gates and Gysel 1978). The Midwest is a farm/woodlot landscape (Whitcomb et al. 1981), where cowbirds penetrate to the interiors of forest islands and parasitize nests of forest-breeding birds (Brittingham and Temple 1983, Freemark and Merriam 1986).

Rates of nest predation and cowbird parasitism probably are higher in small isolated forest patches because even the center of a small fragment is close to the forest edge. Edges of forests in agricultural landscapes have higher densities of generalist mammalian predators (Wilcove 1985, Andren and Angelstam 1988) and nesting success has been shown to be lower at the forest edge than in the interior (Gates and Gysel 1978, Temple and Cary 1988).

In contrast to many parts of the Midwest, New England, since the 1840s, has experienced a steady, inexorable decline in agriculture that started with the opening of the Erie Canal and has continued to the present day. Once covered by the primeval forest, New England was cleared for family farms after European settlement in the seventeenth century. By 1840, 75 percent of the landscape was in crops and pasturage (Raup 1966). However, New England today is at least 75 percent forested and northern New England is more than 90 percent forested (Frieswyk and Malley 1985, Brooks and Birch 1988). The reversion of farmland to forest has resulted in extensive, mature forest cover, although species composition is different from that prior to European settlement (Foster et al. 1992). The pileated woodpecker (*Dryocopus pileatus*), as revealed by the Breeding Bird Survey, has shown a significant increase for the period 1982–1991 in eastern North America; the brown-headed cowbird has shown a significant decline for the same period. New England's forests become older and more extensive every year (Waddell et al. 1989), but miles of stone fences and thousands of old cellar holes give mute testimony to a history of intensive land use.

Most woodlands in New England are privately owned. Would the aging, extensive forests of New England, particularly northern New England, be fragmented by even-age management, specifically clear-cut harvesting? Two lines of evidence from managed public lands suggest that they would not. First, regeneration occurs rapidly and closed canopy sapling stands form within 7–10 years after clearcutting. The interfaces between even-aged stands (internal edges) are ephemeral and do not support distinct bird com-

munities as do field/forest edges (DeGraaf 1992). We have found no evidence for increased rates of predation on artificial nests along these internal edges (DeGraaf unpublished). Predation rates on artificial nests, which are elevated in fragmented forests (Wilcove 1985, Angelstam 1986), were not elevated in seedling/sapling or poletimber stands compared to rates in mature northern hardwood stands in extensive forest (DeGraaf and Angelstam in press).

Second, all species of birds found in old-growth or virgin northern hardwood stands also are found in mature managed stands (DeGraaf 1987, Absalom 1988). In New England northern hardwood forests, four distinct breeding avifaunas occur in seedling, sapling, poletimber and mature stands; no species are unique to old-growth stands, nor are there differences in breeding bird composition among even-aged sawtimber, old growth or uneven-aged stands (DeGraaf 1987). Furthermore, a distribution of size classes ranging from regenerating to mature stands provides breeding habitat for approximately twice as many bird species as does an extensive, uneven-aged hardwood forest (DeGraaf 1987). Among small mammal communities, all species found in mature stands also are found in younger stands (Healy and Brooks 1988, DeGraaf et al. 1992). In our opinion, the main negative impact of logging probably is the resultant haul roads that are large enough to create permanent corridors or promote human access rather than the logging itself. In Maine, marten (*Martes americana*) occupy logged as well as mature conifer forest, but are taken in disproportionally high numbers in logged stands due to trapper access via the logging roads (D. Harrison personal communication). Megafaunal species that have shown declines or avoidance of habitats as road densities increased include black bear (*Ursus americanus*) in the Adirondacks (Brocke et al. 1990), wolf (*Canis lupus*) in Minnesota and Wisconsin (Theil 1985, Mech et al. 1988), and mountain lion (*Felis concolor*) in Utah (Van Dyke et al. 1986). Increased vulnerability to hunter harvest has been related to road density for moose (*Alces alces*) in Canada (Fraser 1976, Crete et al. 1981) and white-tailed deer (Sage et al. 1983).

The less-frequent once or twice per century stand entry associated with even-aged management may result in fewer roads, or roads that grow over more quickly than roads needed for the frequent entry (every 10–15 years) under uneven-aged management. As many roads as possible should be closed after logging, especially to vehicular traffic (Brocke et al. 1990).

## Multiple Use and Old Growth

Can multiple-use management also accommodate the need for old-growth forest? We think it can. First, and most directly, large blocks can be managed for old-growth by excluding most vegetation management practices. This is the approach used in New England's national forests, where about half the forest area has been designated for old-growth. Even with intensive timber management in the remaining forest, mature and old-growth forest forms a contiguous block with patch size nearly equal to the total forest area. Seedling/sapling stands will be ephemeral islands in the forest landscape.

We also are optimistic that some old-growth values can be provided in stands and forests managed for other values. In northern hardwoods, it is possible to achieve an old-growth age structure (*sensu* Hayward 1991) and harvest some timber using uneven-age silvicultural systems. We also think even-aged and two-aged silvicultural systems can be used to provide some old-growth values, provided we can define the desired age and stand structures. Modifying silvicultural systems to provide commodities and old-growth

values deserves more attention; it will require a clear definition of old growth for eastern forest types.

Much concern has been expressed about forest birds, but that concern should not be limited to any one species, guild or habitat type. We need to save all the pieces of the regional mosaic—early successional, late successional and everything in between. We need to provide these components at spatial and temporal scales that meet the needs of wildlife and reflect the natural patterns of disturbance.

## Seral Communities

Declines of grassland and shrubland birds in eastern North America are more alarming and more consistent than those reported for forest migrants (Askins 1992). Many species of grassland birds have declined significantly since 1966 and these declines have occurred in the Midwest as well as the Northeast (Robbins et al. 1986, Bollinger and Gavin 1992). Compared with birds of mature forests, which, in the East, have been shown to be quite tolerant of disturbance and successional changes beyond the poletimber stage (e.g., Webb et al. 1977, Maurer et al. 1981, DeGraaf 1987), grassland birds are specialists that quickly disappear from a site as the vegetation changes. For example, grasshopper sparrows (*Ammodramus sauannarum*) need grassland interspersed with bare ground (Smith 1963, Whitmore 1981); Henslow's sparrows (*A. henslowii*) need fields with a deep litter layer, standing dead forbs and tall, dense grass (Zimmerman 1988), and bobolinks (*Dolichonyx oryzivorus*) need hayfields with low proportions of alfalfa (Kantrud 1981, Bollinger and Gavin 1992).

As grasslands and abandoned fields are invaded by shrubs and small trees, grassland specialists are replaced by shrubland specialists, which, like the grassland species, are dependent on transitory, even ephemeral habitats. Shrublands quickly become unsuitable habitat for species such as golden-winged warblers (*Vermivora chrysoptera*) (Confer and Knapp 1981) or yellow-breasted chats (*Icteria virens*) (Shugart and James 1973, Thompson 1977, Andrle and Carroll 1988). A shrubland/forest edge generalist, the rufous-sided towhee (*Pipilo erythropthalmus*), has declined steadily 8–10 percent per year in New England since 1966 (John Hagan personal communication).

## Grasslands and Shrublands as Natural Habitats

Do the declines of grassland and shrubland birds (and possibly other species) in eastern North America reflect a return to presettlement conditions? Clearly, some species spread eastward from the Great Plains as the East was cleared for farmland—horned lark (*Eremophila alpestris*), dickcissel (*Spiza americana*), western meadowlark (*Sturnella neglecta*) and brown-headed cowbird are examples (Lanyon 1956, Hurley and Franks 1976). But there is ample evidence that grasslands and other open habitats were common in eastern North America before Europeans arrived. Large natural prairies occurred on Long Island (Niering and Dreyer 1989); open habitats occurred in southern New England, possibly maintained by Indian burns (Bromley 1935). In the period 500–1000 AD, Indians shifted from food gathering to food production and storage—maize, beans and pumpkins were planted in fields (Likens 1972). The interior of the eastern deciduous forest biome (present-day Ohio River Valley) was primarily influenced by small-scale disturbances, i.e., gaps, but large-scale disturbances occur throughout the biome. Hurricanes affect coastal areas primarily (Nelson and Zillgitt 1969; Foster 1988a, 1988b).

Fires are major sources of disturbance at the edges of the biome, probably for different reasons in different locations (Runkle 1990). In the Southeast, sandy soils and high temperatures make fires more likely (Nelson and Zillgitt 1969); toward the Great Plains, low precipitation increases fire frequency. In northern forests, fire frequency may be related to increased proportions of flammable conifers such as pine and spruce (Whitney 1986). Low-intensity fires have maintained open habitats in Maine for at least the past 900 years (Winne 1988). The health hen, an extinct subspecies of the greater prairie chicken (*Tympanuchus cupido*), restricted to grassland and other open habitats (Forbush 1927), was abundant in the 17th century from Massachusetts to Maryland (Gross 1932), indicating that there were extensive grasslands, but most East Coast grasslands were destroyed long before their bird communities were described (Askins in press).

The nomadic habits of grassland birds (Wiens 1969, Fretwell 1986, Whitmore and Hall 1978) and the tolerance to disturbance of mature forest birds (Webb et al. 1977, Maurer et al. 1981, DeGraaf 1987) likely reflect avian responses to disturbance regimes in eastern North America.

Clearly, grassland and other early successional habitats were historically present in presettlement New England, and it is reasonable to maintain and manage grasslands using fire, mowing and grazing to prevent invasion by forest vegetation. Shrublands can be maintained by applying methods used to produce stable shrub communities on powerline rights-of-way (Niering and Goodwin 1974, Bramble et al. 1990).

Management to provide early successional habitats is necessary in view of recent declines of such habitats. In 1950, about 30 percent of the New England forest was in the seedling or sapling stage (Black 1950); by the 1970s, these stages represented 14 percent; and by the 1980s, 8 percent of the forest cover (Brooks and Birch 1988). Hay crop acreage has declined 46 percent in New England since 1966 (U.S. Department of Agriculture 1967, 1987). The decline of early successional habitats and the aging of forests in the Northeast have implications for all wildlife species.

In sum, concerns about forest migrants are valid, but their breeding habitats are increasing in parts of the Northeast. Early successional species and habitats are declining acutely in New England and in eastern North America in general (Askins 1992). Natural disturbance regimes vary regionally. Wildlife communities reflect these disturbance patterns and management practices should acknowledge, if not mimic, these regimes.

## Habitat Relationships—Effects of Scale

Increasingly, natural resource management is being viewed in a landscape or ecosystem context (Forman and Godron 1986, Rodiek and Bolen 1991, DeGraaf et al. 1992). Most forest management activities are applied at the stand level, but many species have territories or home ranges that are much larger and also include nonforest habitats: red-tailed hawk (*Buteo jamaicensis*), wild turkey (*Meleagris gallopavo*), black bear (*Ursus americanus*) and moose (*Alces alces*) are examples. Traditional wildlife habitat management has focused on single or featured species approaches to providing and manipulating the target species' habitat requirements of food, water, cover and their spatial distribution in a given area (Schemnitz 1980). Incorporating the needs of all wildlife species in a management plan requires a hierarchical approach to habitat relationships. Such an approach has been proposed for management of New England wildlife associated with forest habitats (DeGraaf et al. 1992).

Today, resource management professionals are faced with several new philosophical

outlooks, described as "new forestry" and "ecosystem management." New habitat management approaches are concerned with addressing the complexity of natural and managed systems and the task of managing lands for biological diversity (Trauger and Hall 1992). Much greater emphasis is being placed on spatial distribution of habitats and changing habitat patterns across landscapes and over time. Basic ecological approaches to forest land management must consider forest area, species diversity related to habitat scales, predictable patterns of vegetative structure, natural disturbance patterns and human impacts (DeGraaf et al. 1992, Hunter 1990). No single management system on any one scale will meet the needs of all wildlife at any given time or place. It is important to have a suite of ecologically based management strategies to address all species needs in view of the changing cultural demands placed on forests today.

In new England, extensive forests of uniform age or vegetative structure provide habitat for relatively few species. When a variety of upland openings and aquatic habitats are present, the number of species increases dramatically. For example: landscapes of unbroken mature forests have approximately 100 vertebrates; forests and early successional habitats, about 200; and forests with early successional and aquatic habitats, more than 300 vertebrates.

## Conclusions

Management for both societal and biological goals must be planned and conducted in a regional context. Forest products vary regionally in economic importance. Natural disturbance regimes, to which endemic communities are adapted, also vary regionally.

Human activities dominate the landscape; we are "managing" vegetation and wildlife whether we are aware of it or not. The forests have returned after extensive clearing started in the 18th century, but introduced pests have altered the forests forever. The introduction of the chestnut blight (*Endothia parasitica*) and gypsy moth (*Lymantria dispar*) probably had a greater impact on forests and wildlife in the eastern deciduous forest than any other event since land clearing in the 17th and 18th centuries.

In eastern North America, forest management and wildlife management are not at biological odds. Many vertebrate species depend on early successional habitats, while none are unique to old-growth forests. Early successional habitats and species are declining, while forests are becoming older and more extensive.

We emphasize the need to reach a middle ground between protecting ecosystems and producing goods and services. These activities are not mutually exclusive and, as professionals, we need to continue to seek ways to accommodate both protection and production. We will be able to preserve only small parts of ecosystems. Most of the landscape will be used but that fact does not diminish our responsibility to protect the land by preserving ecosystem processes. That requires management and would produce commodities as a by-product. We think the idea that commodities are by-products of the ecosystem ought to prevail in resource management.

We can consciously manage landscapes to provide habitat for endemic communities or let nature be and accept the consequences. In the heavily altered landscapes of the 20th century, nature is what we make it.

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