Forest Management Guidelines for Wildlife Management Areas

Abstract*: These guidelines advocate the use of sustainable forest management practices to conserve biological diversity on state-owned wildlife management areas (WMAs). Traditional use of WMAs for limited public recreation and wood production are consistent with the goal of maintaining native biological diversity in Massachusetts. At present, Massachusetts' forestlands as a whole lack adequate early- and late-seral forest habitat to meet this goal. Sustainable forestry practices can provide early-seral forest habitat while addressing increasing societal demands for wood products and biodiversity conservation. Sustainable practices also include designation of areas for development of late-seral forest conditions. Past human disturbance (e.g., agricultural conversion and/or cutting) has greatly modified tree species composition and simplified vegetation structure on many sites. Forest management on WMAs will focus on creating structurally diverse forest stands, and maintaining a range of seral forest stages across the landscape. Most upland forest will be designated for active management using a combination of even-aged and uneven-aged silviculture. Managers will incorporate elements of natural disturbance patterns into managed forestlands by extending conventional rotation lengths, increasing stand size, retaining clusters of mature trees, and fostering heterogeneity of tree species and size classes on all harvested sites. Priority natural communities, forested wetlands, riparian filters, and areas of adjacent upland forest will be designated for passive management using only occasional selection cutting to establish uneven-aged forest structure and late-seral forest character. Together, active and passive management will create extensive, heterogeneous forest patches with characteristics of unmanaged forest landscapes. Biological monitoring of plant and animal communities will be conducted at selected sites to document environmental response to both active and passive management. 'Green Certification' for Division forestry practices will be pursued to provide independent verification of sustainable management.*

Preface

This document reviews the Divisions statutory responsibilities and current EOEA policies, and places forest management decisions within a biodiversity framework to maintain native species, natural communities and ecological processes while addressing various cultural concerns, including public recreation and wood production. Global forest conservation issues are assessed from the local perspective of wood production in Massachusetts. The necessity, and the difficulty, of managing WMA forestlands in a landscape dominated by numerous, privately-held forestlands is addressed. The history of Massachusetts' forestlands is reviewed in the context of its impact on biodiversity conservation.

The development of sustainable forest management practices that conserve biodiversity is reviewed. Special attention is given to incorporating structural elements of unmanaged forests into managed landscapes. The difficulties and benefits of establishing forest reserves are reviewed.

The guidelines describe the current and desired future condition of WMAs, and advocates using sustainable forestry practices to achieve the desired condition. A simple methodology for prioritizing WMAs for management is presented. The importance of biological monitoring on managed forestlands is reviewed.

The document provides a template for landscape level decisions on WMA forestlands, and proposes both active and passive forest management practices to meet the Division's goal of biodiversity conservation while providing public recreation and wood products**.** The guidelines do not provide a management plan for an individual property, but rather provide a framework for generating site plans for multiple properties. The guidelines advocate that WMA forestlands are an appropriate setting to demonstrate how sustainable forest management can maintain biodiversity.

The guidelines do not suggest returning forest condition to any previous point in time. Rather, they advocate driving the future forest condition to achieve important cultural and biological goals. The guidelines recognize that availability of early-seral habitats has declined substantially over the last several decades, and that late-seral forest habitats are exceedingly rare in Massachusetts today. Management for early- and late-seral habitats is a fundamental part of these guidelines.

This document should be reviewed and modified periodically to reflect new information on forest management and biodiversity conservation (Noss 1993). An adaptive management approach is essential because conservation action is necessary, but knowledge of the ecological systems involved is always incomplete (see Walters and Holling 1990, Everett et al. 1994).

Acknowledgements

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Background

The Massachusetts Division of Fisheries and Wildlife (DFW) has statutory responsibility for the conservation (including protection, restoration, and management) of Massachusetts' flora and fauna (Darey and Jones 1997). Species of flora and fauna rarely exist in isolation, but often occur in assemblages, or natural communities. In turn, natural communities rely on ecosystem processes, such as nutrient cycling and energy flow, for their continued existence. This interacting complex of species, communities, and processes is expressed as biological diversity, or 'biodiversity'.

Given the Division's statutory responsibility, an appropriate litmus-test for management decisions on WMAs is conservation of biodiversity. Accordingly, the Division began a biodiversity initiative in July 1996 that seeks to combine management of upland habitats by the Wildlife section with restoration of degraded ecological communities by the Natural Heritage section. The goal of this coordinated effort is to enhance and maintain the biological diversity of Massachusetts.

This approach is consistent with current efforts by the Massachusetts Executive Office of Environmental Affairs to promote a biodiversity-based approach to managing public and private forestlands (Wickersham 2000). While the concept of a biodiversity-based approach to forest management is relatively new, the basis for this concept was established within the forestry profession long before the term 'biodiversity' was coined. In 1917, Gifford Pinchot (the first Chief of the U.S. Forest Service) noted that "The forest…takes its importance less from the individual trees which help to form it than from the qualities which belong to it as a whole" (Miller and Staebler 1999).

Diversity-driven management goals risk being too general to be effective (Lautenschlager 1997). Accordingly, these guidelines outline landscape composition goals and sustainable forest management practices that provide an array of successional stages (including both early- and late-seral forest conditions) to meet the Division's biodiversity goal. Sustainable forestry practices recognize and conserve biodiversity. Conservation of biodiversity is fundamental to ecosystem management, which can be seen as a logical extension of multiple-use, sustained yield management (Healy 1994).

It is possible and desirable to accommodate a variety of cultural demands on WMA forestlands, including traditional uses such as public recreation and wood production. However, the degree to which any cultural activity occurs on WMA's must be constrained by the goal of biodiversity conservation. For example, recreational activities are appropriate on public lands, but not without constraint. DFW has always encouraged a variety of recreational activities on WMA's, such as regulated hunting, fishing, and trapping, hiking, wildlife observation, and more recently, pedal-powered mountain biking. At the same time, DFW has generally prohibited use of motorized vehicles on WMA's, primarily because of the potential for degradation of soil and water quality by irresponsible use of ATV's and 4-wheel drive vehicles.

Likewise, wood production is appropriate on WMA forestlands, but not without constraint. DFW recognizes that public demand for wood products in Massachusetts far exceeds production. Sustainable forest management practices can provide wood products and a range of valuable wildlife habitat while maintaining the integrity of forest ecosystems. Harvest of renewable forest products from public lands also contributes to the state economy. Direct and indirect income from logging, trucking, and primary processing at sawmills in Massachusetts contributes over $100 million annually to the state economy (Campbell et al. 1999). However, public benefits from wood production should not, and do not, make timber management the primary objective on WMA forestlands[1].

Despite the fact that intensive use of forest resources has occurred in Massachusetts for nearly 300 years (Foster et al. 1998a), the concept of sound forest management was established in the Commonwealth less than 100 years ago with the passage of the Massachusetts Reforestation Act in 1908 (Rivers 1998). It is widely accepted that the post-European settlement view of forests as commodities to be exploited led to a dramatic and drastic alteration of the forest landscape throughout Massachusetts during the 18th and 19th centuries (Foster et al. 1998a). This alteration obscured regional

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[1] Over the past century, Massachusetts forestlands annually produced an average of 80-100 board feet of sawtimber per acre (the current average standing volume of 6,282 board feet per acre [USDA Forest Service 1998] occurs primarily on 60-80 year-old forestland). Based on this figure, timber management on WMA forestlands could sustainably produce 7.0-8.7 million board feet of timber annually (80-100 board feet/acre/year x 87,000 forested acres). However, the goal of biodiversity conservation demands complex forest structure, occurrence of very old (late-seral) as well as very young (early-seral) forest, and areas of passively managed forestland where trees are allowed to reach their full biological life span, then decay naturally without extraction of wood products. These constraints reduce the amount of wood production that can occur on WMA forestlands. Future annual harvests from WMA forestlands are expected to range from four to six million board feet, or 45-85% of potential timber yield (see 'Harvest Levels' on pg. 25).

forest patterns that corresponded to climate, substrate, and fire regime (Foster et al. 1998a, Fuller et al. 1998). Today, forest management must continue to evolve to include not just trees, but all aspects of the forest environment, including shrub, herb, and soil communities. This evolution is essential as managers struggle to balance increasing human demand for wood products with the pressing need for biodiversity conservation.

Global and Local Perspectives

In the decade between 1980 and 1990, the area of forest world-wide declined by 4%. A gain of about 20 million ha (78 square miles) of temperate forest (largely through re-forestation of former agricultural sites) during the decade was more than offset by a reduction of about 170 million ha (656 square miles) in tropical forest (Brooks 1993). Forest preservation (cutting exclusion) is commonly sought in North America to maintain a range of environmental values, but could lead to increased deforestation in threatened tropical forests (Sohngen et al. 1999). Forest reserves (areas without harvesting) in temperate North America are warranted for biodiversity conservation, but this action can shift wood harvest to other regions with less stringent environmental regulations. Establishment of reserves in Massachusetts should be carefully researched relative to both social and environmental concerns. Private/public partnerships may be a reasonable approach to establishing reserves in Massachusetts (see 'Forest Reserves', pg. 15).

North America currently supplies about 35% of world timber production, and reducing production here will increase harvests in tropical forests where there are already concerns over loss of biodiversity, undiscovered species, and plant biomass that removes carbon dioxide from the atmosphere (Sohngen et al. 1999). Reserving 20 ha (50 ac) of North American forest from harvesting could result in the loss of 1 ha (2.47 ac) of previously inaccessible tropical forest. A prudent course may be to stress sustainable harvests that maintain biodiversity in more resilient temperate forests in order to reduce economic pressures on more ecologically diverse tropical forests.

Massachusetts is a net importer of wood products. It is estimated that current harvests of timber and fuelwood from Massachusetts forestlands is equivalent to only 6% of annual per capita wood consumption in the state (Berlik 1999). Not only does Massachusetts import the great majority of the wood it uses, but that importation invariably utilizes fossil-fuel based transport which adds air pollutants to our atmosphere, and which may contribute to global warming.

Total removal of wood products from Massachusetts forestlands averages around 121 million board feet (MMBF) per year (USDA Forest Service 1998), but only about half (some 60 MMBF) of this total removal comes from timber harvests on land retained in forest use (MA DEM 1999). The remaining half comes from conversion of forestland to non-forest use through residential and commercial development. Currently, Massachusetts' forestlands could support a sustainable harvest rate of over 700 MMBF per year (vs. the current annual harvest of 60 MMBF), or about 41% of current consumption (Campbell et al. 1999). Over time, potential sustainable harvest could approach 1.5 billion board feet per year in Massachusetts, which would be equivalent to about 80% of current per capita utilization in the state (Berlik 1999). Should Massachusetts move toward becoming more self-sufficient in wood production, sustainable forest management practices that feature biodiversity conservation will become increasingly important.

Recycling and conservation of wood products are to be encouraged at all levels. Still, the renewable resource qualities of wood should make it a resource of choice over non-renewable fossil fuels in a world that now must support over six billion people. However, "To the average citizen, the prospect of harvesting trees triggers instant environmental concern notwithstanding the fact that carefully managed forests tend to provide more uses, values, and benefits than those lacking responsible stewardship actions" (Foster and Foster 1999:9).

The Landscape Context

Forest ownerships do not exist in isolation, but are part of the larger landscape. The goals of any forest landowner, and the opportunity to achieve those goals, are affected by the nature of the surrounding lands (New Hampshire Forest Sustainability Work Team 1997). Management planning for each WMA should consider the protection status and ownership pattern of lands within the watershed sub-basin(s) where the WMA occurs. When other public land and private land with permanent protection status (e.g., Article 97 or conservation restriction, respectively) occurs near a WMA, a stable planning horizon exists. Adjacent private forestlands with non-permanent protection (e.g., Chapter 61), or with no protection create increasingly uncertain futures since these landscapes could become fragmented by development.

Forest planning at the landscape level seems difficult if not impossible to implement with the various landowner types and landowner objectives found in Massachusetts. A de facto form of diversity will be maintained as a result of such a fragmented ownership pattern (Hunter 1990). In the absence of a large-scale landscape plan, management for a variety of goals that include income, wildlife, aesthetics, and preservation of open space, will inherently produce a diversity of species, forest age classes, and habitat types. However, de facto diversity is not enough to maintain the most sensitive species and communities (Hunter 1990). Even though WMA forestlands account for <3% of commercial forestland in Massachusetts, they should not be managed as discrete entities, but rather as integral parts of the local landscape. A landscape perspective can best account for sensitive species and communities.

A primary challenge in landscape planning is to accommodate change while protecting physical, biological and cultural resources. Basic approaches include separation of incompatible uses, clustering of complementary uses, protection of sensitive areas, and limiting infrastructure (Ahern 1997). In terms of forest management, examples of landscape planning include separating fragmented forestland from extensive, unfragmented forests, clustering of early-seral habitats to achieve maximum benefits and minimum detriments of edge, and limiting permanent access roads to the minimum necessary to allow sustainable management activities.

Massachusetts Forestlands

Sixty-three percent of Massachusetts is forested, and this forest has matured substantially in recent times (Dickson and McAfee 1988, USDA Forest Service 1998). Presently, Massachusetts forestlands are comprised of 5% seedling-sapling forest, 23% pole forest, and 72% sawtimber forest (USDA Forest Service1998). A dramatic decline in young (seedling-sapling) forest has occurred throughout Massachusetts during the past half-century (Fig.1), primarily because cutting on private forestlands does not completely regenerate stands to the seedling stage. This is evidenced by the fact that harvests from Massachusetts forestlands have averaged nearly 60 MMBF from some 30,500 acres annually since 1985 (total harvest of over 920 million board feet from nearly 490,000 acres)(MA DEM 1999), yet the percentage of seedling-sapling forest declined in Massachusetts from 7% to 5% between 1985 and 1998 (USDA Forest Service 1998*).*

The level of harvesting occurring in Massachusetts is relatively light (average removal of about 2 mbf per acre from an average standing volume of about 6 mbf per acre [MA DEM 1999, USDA-Forest Service 1998]), is widely dispersed, and generally does not provide substantial early-seral forest habitat. A lack of markets for low quality hardwoods limits true regeneration cutting on most private forestlands in Massachusetts (Commonwealth of Massachusetts 1996).

Even-aged forests that now dominate the Massachusetts landscape are the result of historic land use practices and farm abandonment (Litvaitis 1993, Foster et al.1998a). Various bird and mammal species commonly associated with early-seral habitats have declined consistently since the 1950's in response to the limited availability of these habitats (Hill and Hagan 1991, Litvaitis 1993). This reduction in the number of early-seral bird and mammal species represents a trend that may be extended in space and time beyond the previously described effects of forest maturation. Current land uses fragment and isolate habitat patches, thus potentially reducing the viability of some local populations (Litvatis 1993).

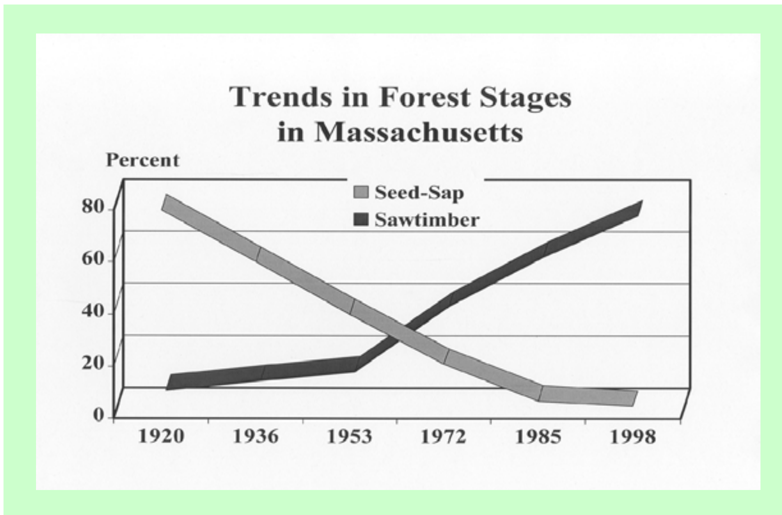

Fig. 1. General Trends in Forest Stages on Massachusetts Forestlands.

As a result, relatively large tracts of early-seral habitat may be needed to sustain local and regional populations of species associated with these habitats (Askins 1993, Litvaitis 1993, Litvaitis et al. 1999). Maintaining early-successional forest is recommended by Partners in Flight, a program of the National Fish and Wildlife Foundation (Pashley et al. 2000). More than 5% of forestland in portions of Massachusetts (areas west of the Connecticut River and northern Worcester county) would have to maintained in early-successional forest to meet Partners in Flight goals for the Northern New England physiographic region.

At the same time, large blocks of unfragmented forest have become relatively rare in Massachusetts. The Nature Conservancy (TNC) is presently conducting an ecological land unit analysis in the Northeastern states to identify the most viable large forest blocks with highest conservation value. Protection efforts would likely focus on large, viable areas of matrix forest embedded with unique communities and rare species occurrences (Anderson et al. 1998).

Biodiversity Concerns

The primary concerns for biodiversity conservation in Massachusetts are retaining land in forest use, and containing the spread of invasive, exotic organisms (Wickersham 2000). Additional concerns include diversifying forest successional stages, addressing high-grade cutting, reversing both the proliferation of red maple (*Acer rubrum*) and the trend in regeneration failure of red oak (*Quercus rubra*).

The configuration of habitat becomes increasingly important in landscapes fragmented by permanent development (Andren 1994), and forest management options to promote biodiversity will likely be more constrained in developed landscapes. Each year, several square miles of forestland are developed in Massachusetts, resulting in both the outright loss of habitat, and degradation of remaining habitat as forestlands are fragmented into smaller patches. In addition, remaining forestland is parceled into smaller ownerships (see ‘Private Forestlands’, below).

The continued loss of forest to development in Massachusetts can perhaps be curtailed through changes in tax policy, which currently provide an economic disincentive for private landowners to retain land in forest use (Campbell et al. 1999). "Green Certification" (an independent review to verify sustainable forest management practices [Teisl and Roe 2000]) offers potential for landowners to realize a higher price for wood products, and thus create an economically sound alternative to development. The MDC's Quabbin Reservation was the first public land in the nation to earn Green Certification (Kyker-Snowman 1997), and other public lands followed (Brown 1998). In the absence of changes in tax policy and premiums for sustainably produced forest products, on-going efforts to identify the highest priority lands for conservation protection are extremely useful (McGarigal et al. 1999, NHESP 2000a, Poiani and Richter 1999).

Within fragmented landscapes, forest area is the best predictor of density and species richness for birds associated with interior forest habitats. The most area-sensitive bird species reached a 50% probability of occurrence in forest patches ≥1,000 ha (2,700 ac) (Robbins et al. 1989). In extensive (unfragmented) forest environments, isolation (distance from the nearest forest edge) is the best predictor of density and richness for interior forest birds (Askins et al. 1987, Askins et al. 1991).

Nesting success for many birds appears to be considerably greater in large forest tracts than in small forest patches (Gibbs and Faaborg 1990, Porneluzi et al. 1993) due to high rates of predation on eggs and nestlings in small forest fragments (Wilcove 1985, Robinson 1988, Small and Hunter 1988). Extensive forest cutting can cause a temporary reduction in habitat for area-sensitive birds, but provides important habitat for declining bird species associated with early-successional forest (Rosenberg et al. 1999). Effective conservation strategies create early-successional forest while maintaining extensive areas of mature forest at any point in time.

Several tree species present in the original forest of southern New England have been greatly reduced by invasive, exotic pathogens. Oaks and other hardwoods have been stressed by the gypsy moth (*Lymantria dispar*). The American chestnut (*Castena dentata*) has been rendered an understroy tree or shrub by the chestnut blight (*Eodothea parasitica*). Beech bark disease (*Nectria*) has greatly reduced numbers of large American beech (*Fagus americana*). Currently, the hemlock wooly adelgid (*Adelges tsugae*) is causing widespread mortality of eastern hemlock (*Tsuga canadensis*) (DeGraaf and Miller 1996). Other exotic pathogens exist, and more are likely to arrive.

About 45% (1,276 of 2,814) of all vascular plant species in Massachusetts are exotic (Sorrie and Somers 1999). The vast majority of these plants do not threaten native biodiversity, but a handful are of serious concern (Weatherbee et al. 1998). Invasive exotic plants are quick colonizers of disturbed areas. Forest cutting activities are a form of disturbance, and harvested sites can be readily invaded by invasive plants. Logging machinery can carry invasive plant seed from previous work sites into forested areas that were previously free of invasives. The faster growing rates of invasive plants, coupled with efficient dispersal mechanisms, and tolerance for a wide range of environmental conditions often allow invasive exotics to out-compete native species (Hoffman and Kearns 1997).

Generally, as the populations and the distribution of invasive exotics increase, the diversity and populations of native plants declines, as does the diversity of habitats available for wildlife (Braton 1982, Harty 1986, Hoffman and Kearns 1997). Invasive exotics have also been implicated in contributing to the decline of 42% of those species listed as threatened or endangered by the US Fish and Wildlife Service (Stein and Flack 1996). Invasive exotics not only have negative environmental effects, but also negative economic effects. The Congressional Office of Technology Assessment noted in a 1993 report that 79 problem plants and animals caused losses of $97 billion between 1906 and 1991.

Natural disturbances will likely increase structural diversity of presently even-aged forests in Massachusetts over time, but it is unlikely that natural disturbance will provide a variety of seral stages for many decades (Litvaitis 1993). In the mean time, forest management that incorporates modified silvicultural practices to increase structural diversity has the potential to conserve species and processes, within our mostly even-aged forests (Litvaitis 1993). Partial cutting practices tend to increase within-stand bird species diversity in New England forestlands, but if applied exclusively may reduce diversity at the landscape scale because they do not accommodate either mature or early-seral forest specialists (e.g. Brown Creeper (*Certhia americana*) and Alder Flycatcher (*Empidonax alnorum*), respectively) (King and DeGraaf 2000). It is unnecessary and undesirable to maximize species diversity in all forest stands, rather, maintaining viable populations of all native species characteristic of the forest ecosystem is paramount (Welsh and Healy 1993). Overall, a variety of silvicultural practices is warranted to maintain bird species diversity in New England forested landscapes (Welsh and Healy 1993, King and DeGraaf 2000). Forest management practices that conserve biodiversity are discussed below (see 'Biodiversity and Forest Management', pg. 9).

High-grade cutting (cutting of individual timber-quality trees with high economic value and retention of poorly-formed trees of low vigor) is a problem throughout Massachusetts (Mauri 1998). High-grade cutting can reduce the future economic value of forestland, and in turn, can reduce the likelihood of land being retained in forest use. Development pressure is generally greater when alternative land use, such as growing forest products, is unprofitable. In addition, high-grading is generally unfavorable to regenerating oak because the selective removal of high-value trees fails to provide adequate sunlight for survival of oak seedlings. High-grading can best be addressed by development of improved markets for what are currently low value wood products, such as firewood and pulpwood (Commonwealth of Massachusetts 1996).

The oak genera provide the greatest amount of mast for wildlife in southern New England. Oaks and acorns play a fundamental role in the organization and dynamics of eastern wildlife communities, and these relationships have been developing for millennia (Healy et al. 1997:251). Oak forests are generally not regenerating successfully in the northeastern U.S. on mesic sites amenable to growing oak, and are gradually being replaced by more shade tolerant red maple and black birch (*Betula lenta*) (Lorimer 1993 and Healy et al. 1997). This is true in Massachusetts, where red maple and black birch are increasing in number (USDA Forest Service 1998).

The increase in red maple dominance represents one of the most dramatic changes in eastern US forests during the 20th century, and may have profound economic and ecological consequences. Oaks and hickories (*Carya spp*.), for example, supply many small mammals and birds with nuts and acorns. And the oak's rough bark, unlike the maple's smooth bark, houses bark-dwelling insects for insect-eating birds. A shift in wildlife populations is likely to parallel this shift in tree species. Red maple's proliferation also poses a biodiversity concern. Fairly diverse eastern forests, with 10 or more tree species in the overstory, may be changing to red maple-dominated stands. Single-species stands are more susceptible to insects and disease outbreaks (Abrams 1999).

A primary challenge to forest managers interested in maintaining biodiversity on the Massachusetts landscape is to establish and maintain structurally complex multi-aged stands, adequate early-seral forest, and extensive late-seral forest for species that are closely associated with these habitats (Askins et al. 1987, Litvaitis 1993). Most private parcels of forestland are relatively small (around 10 acres), are susceptible to conversion to non-forest use, and are unlikely to support extensive early-seral habitat. Therefore, the "challenge" to create complex multi-aged stands as well as adequate early- and late-seral forest can most realistically be met by cooperation between public land managers, private forestland owners, land trusts and other private non-profit groups who are willing to work at the landscape level.

Private Forestlands

Forest management decisions in a landscape context must consider the status of private forestlands because 83% (2.4 of 2.9 million acres) of all commercial forestland in Massachusetts is in private ownership (Dickson and McAfee 1988). Average private forestland ownership size has decreased from 9.5 ha (23.4 ac) in 1972 (Kingsley 1976) to 4.3 ha (10.6 ac) in 1985 (Brooks et al. 1993). These smaller forested parcels are distributed among an increasing number of owners, and may result in fragmented habitat, limits to commercial harvesting feasibility and access problems (Brooks 1989, Brooks and Birch 1988, Broderick et al. 1994). The trend toward smaller forest parcels could ultimately reach a point where commercial loggers cannot operate profitably on some private lands (Kittredge et al. 1996).

In addition to smaller parcel sizes, Massachusetts experienced a 2.9% loss in forestland (conversion to non-forest use) between 1971 and 1985 (MacConnell et al. 1991), followed by a 3% loss between 1985 and 1998 (USDA Forest Service 1998). Presently, Massachusetts loses 25 square miles of open space annually to development (Steel 1999), of which more than 10 square miles is forest (USDA-Forest Service 1998).

Private, non-profit conservation groups and land trusts control more than 120,000 acres in Massachusetts (Wickersham 2000). Forest management practices on these lands are varied, and range from a no harvest policy on Massachusetts Audubon Society lands to active, sustainable forest cutting practices on New England Forestry Foundation lands.

Public Forestlands

Over 500,000 acres in Massachusetts are owned by state agencies, primarily including the Department of Environmental Management (DEM), the Metropolitan District Commission (MDC), and DFW (Wickersham 2000). DEM is the largest public landowner in Massachusetts, managing >285,000 acres for multiple benefits. These benefits include improving the quantity and quality of the forest crop, providing ecological diversity, and a sustained flow of raw material to the Commonwealth's wood using industries in a manner that enhances wildlife, water, recreation and amenity values (DEM 1992). Overall, priority is given to regeneration methods that remove the overstory in stages, which will generally limit the amount of early seral habitat produced on DEM land. However, special provisions can be included in timber sale contracts to create specific wildlife habitats, including habitats for rare and endangered species, and early-seral habitats (DEM 1992).

DEM has also committed to preserving and maintaining the integrity of existing old growth stands on state forest and park lands (DEM 1999). While these scattered old growth stands likely account for less than a few thousand acres state-wide, DEM plans to buffer old growth sites by establishing areas of "protection forest" surrounding old growth stands. Within protection forest, human-caused disturbance will be either precluded or minimized (DEM 1999). Total acreage of protection forest is not yet determined, but will likely exceed 20,000 acres state-wide (B. Rivers, pers. comm.).

MDC administers approximately118,000 acres state-wide, including 58,000 acres associated with the Quabbin watershed that is managed primarily to maintain water quality in the 24,529-acre Quabbin reservoir, metropolitan Boston's major drinking water supply (O'Connor et al. 1995). Uneven-aged silviculture, primarily group selection cuts, are favored to create and maintain a stable, multi-layered canopy to ensure adequate filtering of water entering Quabbin Reservoir. Some even-aged management occurs, primarily to convert softwood plantations to more structurally diverse forest. Special attention is directed towards rare and endangered species habitats, but overall wildlife habitat at Quabbin will favor those species adapted to extensive, mature forest dynamics, with relatively little early seral habitat produced (O'Connor et al. 1995).

MDC lands on the 23,000-acre Ware River watershed forest share the same watershed protection goal of the Quabbin forest. However, the Ware River forestlands do not drain directly into a reservoir, but rather collect into the Ware River, and can then be diverted periodically into the Quabbin Reservoir via an underground aqueduct. Portions of the Ware River forest occur in uneven-aged, even-aged, and unmanaged forest conditions. A portion of the Ware River watershed lands are cooperatively managed by the MDC, the Division, and the U.S. Army Corp of Engineers as the 10,557 acre Barre Falls WMA. The MDC also administers some 16,000 acres at Wachusett Reservoir. Management plans are currently being developed for both the Ware River and Wachusett properties (Clark, pers. comm.).

DFW administers about 100,000 acres on 110 WMAs. Individual WMAs range in size from less than 100 acres to 5,500 acres, and average about 900 acres (MDFW 2000). Approximately 87% (87,00 ac) of WMA lands are forested (MDFW unpubl. data). WMA forestlands account for <3% of the 3.1 million acres of forest in Massachusetts (USDA-Forest Service 1998), and should be managed in a landscape context to meet the Division's biodiversity goal.

Public lands managers who look toward balancing forest age class distribution are traditionally cautioned about management for early seral species that are lacking on public lands, but may be plentiful on surrounding or adjacent private forest industry land (Hunter 1994). This is not the case in Massachusetts, where the forest industry accounts for only 2% of all forestland, and relatively mature, private, non-industrial forestlands dominate the landscape (Dickson and McAfee 1988).

Biodiversity and Forest Management

Forest management policy should recognize the role of disturbance in maintaining biodiversity (DeGraaf and Miller 1996). Preserving biodiversity in temperate forest regions requires the maintenance of all seral stages, including the creation of early-seral habitats and the preservation or re-creation of late-seral or old-growth forests (Franklin 1988). Limiting or eliminating the effects of disturbance are not constructive practices if they preclude the occurrence of species that would naturally recolonize an area, and thus reduce plant and animal diversity over time (DeGraaf and Miller 1996).

Managing forested ecosystems to maintain biodiversity requires protecting some natural ecosystems in reserves (see 'Forest Reserves', pg. 15), and combining biodiversity conservation and commodity production in modified, semi-natural ecosystems (Hunter 1996a, Irland 1999). Successful strategies for conservation of biological diversity in temperate forest regions must effectively address both the implementation of networks of reserves and stand conditions in the managed matrix (Lindenmayer and Franklin 1997).

Forest managers are encouraged to examine the unmanaged forest landscape, to observe natural disturbance regimes and resulting forest structural patterns, and modify traditional cutting practices to incorporate natural structural patterns into managed forests (Spur and Cline 1942, Franklin and Forman 1987, Hansen et al. 1991, Rowe 1992, Aplet et al. 1993, DeGraaf and Healy 1993, Franklin 1993, Mladenoff and Pastor 1993, Mladenoff et al. 1993, Noss 1993, Alverson et al. 1994, Lorimer and Frelich 1994, deMayndier and Hunter 1995, Meier et al. 1995, Yahner 1995, Hunter 1996b, Rogers 1996, Lindenmayer and Franklin 1997, Foster and Foster 1999, Seymour and Hunter 1999). Managers need to evaluate current conditions and anticipate future changes with an understanding of the extent, magnitude, and consequences of past dynamics (Foster et al. 1998a).

Human impact (landuse change, introduction of exotic pathogens, etc.) makes it is impractical to implement management scenarios that mimic landscape conditions under pre-settlement disturbance regimes. However, it is still prudent to use the range of pre-settlement conditions to evaluate current and future management scenarios. An understanding of the background rates and causes of change in forested landscapes can help to guide conservation efforts on many scales (DeGraaf and Miller, 1996). Forest cutting practices that incorporate structural patterns associated with natural processes will help sustain the long-term productive potential of forests, maintain biodiversity, and provide a buffer against future uncertainties such as climate change (Mladenoff and Pastor 1993), natural disturbance (e.g., wind, fire, and insect infestations) (Foster and Foster 1999), and economic shifts in market conditions.

Biological diversity is usually best served when there is management at a number of different scales (Hunter 1990). Overall, a range of managed stand sizes is warranted, but the majority of forest area should occur in relatively large stands. Even large clearcuts, which are often criticized by the public, play an immediate role in providing habitat for early-seral species, and will eventually become large stands of mature forest (Hunter 1990). Boundaries between stands should be considered fluid to mimic overlapping natural disturbances (Mladenoff and Pastor 1993).

Natural Disturbance Regimes and Unmanaged Forest Landscapes

Natural disturbance processes impacting Massachusetts forestlands are characterized by frequent local events, such as windstorms, lightning, pathogens and fire, and by occasional broad-scale damage by hurricane winds (Foster 1988). Large, infrequent disturbances such as hurricanes result in massive structural alteration of the forest, yet net energy and nutrient processes remain largely intact (Foster et al. 1997). Hurricanes produce enduring legacies of physical and biological structure that influence ecosystem processes for decades or centuries. These legacies include undisturbed remnants of the previous forest, as well as uproot mounds, downed boles, and standing snags. Ecosystem recovery following large disturbance appears related to the mean distance within the disturbed area to an undisturbed patch (Foster et al. 1998b).

The original deciduous forests of eastern North America displayed great variation in tree species composition among stands within a given forest type, and most individual stands contained a mixture of shade-tolerant and shade-intolerant species (Braun 1950). In old-growth forests of eastern North America characterized by small-scale disturbance, an average of 9.5% (range of 3.2-24.2%) of forest area occurred in natural canopy gaps. Forests in the Northeast tended to have less forest area in canopy gaps (3.2%) than forests in the Southeast (up to 24.2%). Overall, new gaps were formed at an average rate of 1% of total land area per year. Most natural gaps were <0.1 acres, and the largest natural gaps were about 0.4 ac (Runkle 1982).

The age-structure and composition of old-growth forest in southern New England suggest that broad-scale disturbances are most important in stand establishment and organization, whereas more local events are recorded in tree-growth response (Foster 1988). Isolated blowdowns from localized wind events occur every 10-110 years in discrete patches averaging about 30 m^2 (<0.01 ac), and ranging up to 1490 m2 (about 0.4 ac). Large-scale blowdowns from hurricanes occur at 95 to >350 year intervals and range from .03-3.25 km2 (<10-400 ac)(Foster 1988, Turner 1989, Boose et al. 1997).

Characteristic landscape structural patterns distinguish temperate old-growth (unmanaged) forest from managed forest in North America (Mladenoff et al. 1993). Old-growth tends to be heterogeneous in character, with various size patches (small to large) of mixed tree species composition commonly occurring throughout the forest. While most individual patches (>50%) tend to be relatively small (<5 ha [12 ac]), the majority of old-growth forest area (>50%) occurs in relatively large patches of >40 ha (100 ac). In contrast, managed forests are characterized by smaller patches (stands of <5 ha [12.5 ac]) in both the percentage of individual patches and the percentage of total forest area. Managed forests exhibit fewer large matrix patterns than old-growth forest, and a subsequent reduction in regional diversity (Mladenoff et al. 1993).

The amount and patterns of forest age class distributions most likely vary over time and do not appear to be constant (Spies and Turner 1999:115), however, it does appear that pre-settlement forests in New England had substantially more late-seral forest than exists today. In northern New England, unmanaged forest has twice the mean age as the managed forest. In fact, more than 37% of unmanaged forest is older than 100 years, while managed forest in this region rarely reaches or exceeds 100 years

(Seymour and Hunter 1999:48).

Unmanaged, old-growth forest vegetation in southern New England is organized along a complex gradient of soil depth, slope position, and frequency and intensity of disturbance (Foster 1988). Old-growth forests in Massachusetts carry basal areas about 23% higher than nearby second-growth forests. Old-growth tends to be more mesic and second-growth more xeric. A diverse association of bryophytes (non-vascular plants including mosses and liverworts) tends to develop on lower portions of tree trunks and as a ground cover in southern New England old-growth (Dunwiddie et al. 1996). More bryophytes occur in Massachusetts old-growth than in nearby second-growth forest. Many of these species are epiphytes with slow dispersal mechanisms that need vigorous source populations for colonization of disturbed sites (Cooper-Ellis 1998). While there is no evidence of old-growth dependence among vertebrate wildlife and vascular plants in New England, there are species of lichen that grow almost exclusively within old-growth forests (Selva 1996). Also, some beetle species which occupy the forest-floor appear to be more abundant in old-growth than in managed forests (Flatebo et al. 1999). Flora of mosses in Massachusetts has not been thoroughly studied (Anderson et al. 1997).

While no vascular plants appear to be restricted to old-growth forests in southern New England, there are significant differences in communities of understory flora in old-growth versus secondary (old-field) woods (Whitney and Foster 1988). A variety of factors, including microclimatic conditions and the low seed production and poor colonizing ability of many species are probably responsible for the distinctive flora of central New England's old-growth forests. Wild sarsasparilla (*Aralia nudicaulis*), spotted pipsisiwa (*Chimaphila umbellata*), Mayflower, or trailing arbutus (*Epigaea repens*) and Indian cucumber-root (*Medeola virginiana*) are indicative of primary woodlands. New England's old-growth forests were characterized by a unique assemblage of shrubs, seldom encountered in the region's second growth forest. Hobblebush (*Viburnum lantanoides*), striped maple (*Acer pensylvanicum*) and yew (*Taxus canadensis*) were the dominant representatives of New England old-growth shrub communities. All are mesophytic species that reach their southern limits in central New England, and require mesophytic conditions (like those found in old-growth forest) to germinate and grow. Hobblebush has poor colonizing ability, with invasion of new areas by chance via long-distance dispersal of seed by small mammals (Nichols 1913 and Egler 1940, in Whitney and Foster 1988).

The more aggressive nature of species characteristic of secondary woodlands is in striking contrast to the poor seed production and colonizing ability of many primary woodland species. Secondary species including clubmosses (*Lycopodium clavatum* and *L. obscurum*), hair-cap moss (*Polytrichum commune*), hay-scented fern (*Dennstaedtia punctilobula*), bracken fern (*Pteridium aquilinum*), and shinleaf (*Pyrola* spp.) produce large quantities of light, wind-dispersed spores or seeds, and expanses of mineral soil in early phases of old-field succession probably provided favorable conditions for the establishment of gametophytes of many of these spore-bearing species. Two other secondary species, dewberry (*Rubus fragilaris*) and Canada mayflower (*Maianthemum canadense*) bear conspicuous fleshy fruits that are dispersed over long distances by birds and mammals (Whitney and Foster 1988).

<u>Forest Cutting Practices</u>

There is a clear need for greater integration of biodiversity enhancement and commodity production on managed forest landscapes (Mladenoff et al. 1994, Hunter 1996a). Biodiversity, hydrology, and climatic values of forests have become important, and are no longer secondary to timber and other commodities (Wilson 1993). Strict even-aged forest cutting practices that focus primarily on timber production at the stand level can potentially reduce forest structure to the point of biotic impoverishment through the continual replacement of older forests with younger ones; the replacement of complex, uneven-aged stands with simplified even-aged stands; and the reduction of large forest patches to ever smaller ones (Noss 1993).

1) Patch Size

Extensive, heterogeneous patches of mature forest should be retained on the landscape at all times

(Turner 1989). Species associated with early seral habitats, including some migratory songbirds, are often adept at dispersing between isolated patches of habitat. However, some species of late seral habitats, such as flightless invertebrates, are poor dispersers and are more vulnerable to habitat isolation (Noss 1993). While several small forest patches may have greater bird species richness than a single large patch, certain species are never found in small patches (Askins et al. 1987, Robbins et al. 1989).

The heterogeneous nature of extensive, unmanaged forest in the Northeast results in part from the routine formation of small gaps discussed above. On managed forestlands, uneven-aged silviculture using group selection cuts with no more than 0.2 ha (0.5 ac) openings best emulates natural disturbance size for typical forest management intervals of 25-50 years in Northeastern forests (Seymour 1999). Small-scale gaps occur frequently in the Northeast, while large-scale, catastrophic disturbances that result in an entirely new stand are relatively rare. Because natural forest composition is partly or primarily a function of small-scale disturbance patterns, management based on light partial harvests occasionally interspersed with small clearcut patches should favor native forest communities (Bryan 1999).

Consideration of patch size relates to tree species of high wildlife value such as black cherry and red oak, both of which are currently not regenerating well in eastern forests (Abrams 1999). Group selection cuts that create openings of <0.5 acres generally do not favor black cherry or red oak regeneration because these species are relatively intolerant of shade. Group cuts that create openings of at least 0.5-1.0 acres are generally better for securing regeneration of species like cherry and oak (Leak and Gottsacker 1985). Still larger cuts are warranted for providing adequate early-seral forest for wildlife species associated with this habitat (Askins 1993).

A range of cut patch sizes is warranted at the landscape level (Hunter 1990). In any size cut, structural diversity can be increased through the retention of cavity bearing stems, understory plants, coarse woody debris, as well as groups of live trees (Gillis 1990, Mladenoff and Pastor 1993, Hunter 1996a, Bryan 1999, Seymour 1999). Retained structural features should ideally persist through future cutting operations (Hansen et al. 1991, Mladenoff and Pastor 1993, Lindenmayer and Franklin 1997).

Recent land-use history can be used to determine appropriate cut size. For example, previously high-graded forest with reduced tree species diversity and trees of low merchantability may be amenable to more extensive cutting, while stands with high species diversity and merchantability may be amenable to smaller openings. Likewise, existing softwood plantations can be cut extensively to create a more structurally diverse forest.

The form, or merchantability of individual trees is unimportant for biodiversity conservation, and no particular tree species has to occur on a particular site. In fact, a wide range of tree species may have occurred on any given acre of forest in Southern New England over the past few thousand years due to the dynamic nature of natural disturbance regimes that impact the region. Species that comprise a single forest type today (e.g., northern hardwoods) have followed quite different historical migration routes (DeGraaf and Miller 1996). Still, different tree species grow more vigorously on different sites, so there is economic, aesthetic, as well as habitat value in using silviculture to match tree regeneration to inherent qualities of a given site on actively managed forestland.

The oak regeneration failure discussed earlier provides a case in point. The high habitat (acorn mast), economic (sawtimber and veneer), and aesthetic (large, spreading crowns) values of red oak are all valid cultural reasons to regenerate this species on good oak sites (generally mid- and lower slope positions on relatively fine textured soil [Fowells 1965]).

2) Retention Harvesting

The concept of retention harvesting is based on the understanding that biological legacies-remnants of the previous forest ecosystem-persist following a natural disturbance regardless of whether the event is large or small, frequent or infrequent (Foster et al. 1998b). The amount of retention may vary, and includes two basic patterns: dispersed and aggregate retention (Franklin et al. 1996) (Fig. 2). Dispersed retention relies on a relatively even distribution of retained structural features over the entire harvest area. Aggregate retention reserves patches of forest of various size and shape which are representative of initial stand composition.

Large live trees, large cavity-bearing trees, understory shrubs and large downed logs are specifically retained during harvests. Variable retention harvesting can be used to produce structurally complex stands for the conservation of species, processes, and connectivity within the managed landscape. Variable retention harvesting provides infinite prescription flexibility to fit a mosaic of dispersed and aggregate retention to local site dynamics and landscape objectives (Franklin et al. 1996).

Aggregate retention may conserve, promote or restore elements of biological diversity to a higher degree than dispersed retention (FEMAT 1993). Aggregate retention on harvested sites can maintain abundance of cavity-nesting woodpeckers (Gunn and Hagan 2000), and is recommended to provide cool, moist microhabitats for various amphibians (Dupuis et al. 1995). Forest management alters forest structure and thereby influences ecological processes; how much structure must be retained to maintain ecosystem integrity is still an open question (Perry 1998).

While retention serves an important biodiversity function, it can also be a sound economic strategy on private forestlands where retained trees may eventually be harvested. Environmental damage to retained trees that reduces economic value (i.e., wind throw, epicormic branching, etc.) is off-set by increased growth of surviving retained trees. Economic damage to retained trees can be further ameliorated by avoiding tree-length skidding (which often skins bark from retained trees), and avoiding spring logging (when tree bark is loose and easily damaged by logging machinery) (Smith et al. 1989, Johnson et al. 1998).

3) Rotation Length

Rotation length is commonly defined as the number of years required to grow a stand of trees to some specified condition of either economic or biological maturity (Smith et al. 1996). Conventional rotation lengths are based on economic maturity, and extending conventional rotations is necessary for biodiversity conservation because forests reach economic maturity long before they reach biological maturity (Noble et al. 1980, Hunter 1990, Duffy and Meier 1992, Patton 1992). Biodiversity can be reduced by cutting practices that repeatedly disturb herbaceous flora that develops over long periods of time on the forest floor (Meier et al. 1995).

For private forestlands that must consider economic return from wood products, use of extended rotations to conserve biodiversity may result in substantially less short-term profit (Kuusipalo and Kangas 1994). This is because extended rotations retain higher volumes of trees than traditional sustained-yield management, where rotation length coincides with the culmination of mean annual increment (the point where the average annual increase in basal area or volume is maximized on a per acre basis) (Kuusipalo and Kangas 1994). Use of extended rotation lengths results in larger sawlog material and lower environmental impacts, but increases the likelihood that windthrow and/or pathogens will reduce economic return (Leak 1999).

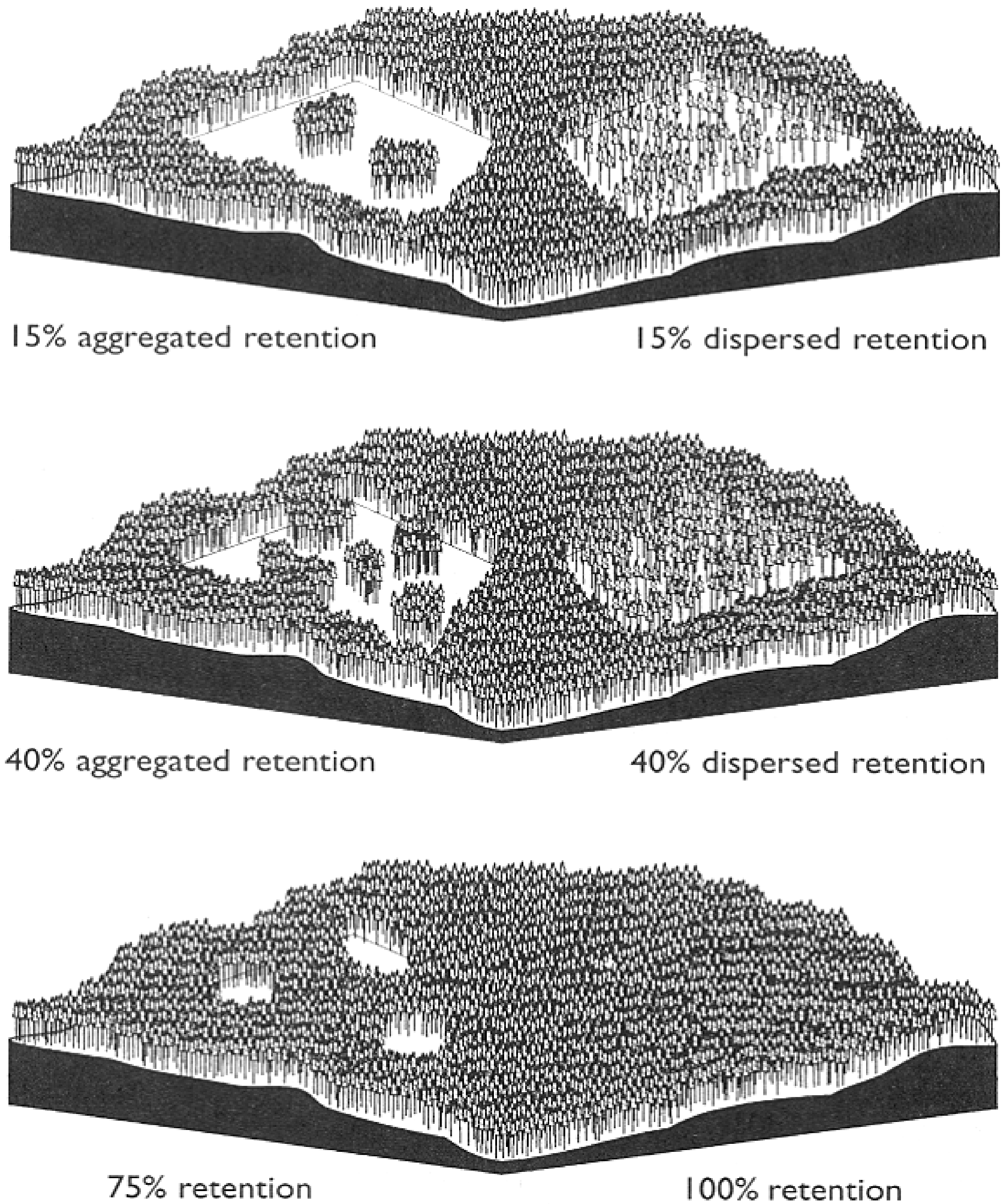

Fig. 2. Retention harvest options: aggregate vs. dispersed retention (reproduced from Franklin et al. [1997]).

Forest Reserves

The term "forest reserve" is commonly used, but a variety of meanings have been associated with this term. From DFW's perspective, forest reserves are portions of state wildlife lands where commercial harvesting of wood products is excluded in order to capture elements of biodiversity that are missing from harvested sites, including an abundance of 200-500 year old trees, and extensive accumulations of large woody debris with associated "pit and mound" micro-topography that occurs when old trees *are uprooted and blown over, then remain in the forest. Reserves would likely need to be a few thousand acres in size to capture these elements. A substantially different approach to reserves utilizes multiples of natural disturbance patch size, and the area needed to maintain genetic diversity of viable wildlife populations to determine reserve area. Anderson et al. (2004) recommends reserve areas based on 4x the size of major natural disturbance patch size (e.g., 5,000 ac patch x 4 = 20,000 ac), and/or 25x the size of a breeding territory for area-dependent wildlife species such as the Broad-winged hawk (e.g., 569 ac territory x 25 = 14,225 ac). Similarly, the concept of "Minimum Dynamic Area" described by Alverson et al. (1994) favors reserves large enough to sustain landscape processes, which might require reserves that approach 20,000 ha for Northeastern forests.*

DFW maintains that viable populations of area dependent wildlife populations should be sustained within forested landscapes dominated by working forest open to harvesting of renewable wood products, and with limited reserves that represent the diversity of native forest communities. DFW supports reserve area based on major natural disturbance patch sizes of ±5,000 ac. While it is important to have the great majority of forestland open to the sustainable harvesting of wood products that support human society, it is equally important to retain limited portions of our forested landscapes in a condition where all components of the ecosystem remain in place. Forest reserves allow us to more fully assess human impacts on harvested sites, and may provide insights into how extractive management of harvested forestlands can be improved[2].

While no forestland is free of human impact from ubiquitous influences such as air pollution and invasive, exotic organisms, reserves can still help ensure that representative examples of biodiversity indigenous to an area are protected. Forest reserves provide reference sites for objective assessment of the sustainability of forest management practices (Norton 1999), and are essential for practicing adaptive resource management (Walters and Holling 1990). Reserves create opportunities for connectivity within the landscape, conservation of species and processes, buffering against future uncertainty, and other hard to measure but valuable functions (Hunter 1996a).

Effective reserves must include the full range of ecosystems and landforms that occur in an area. Because human activities have dramatically altered many fertile, lowland ecosystems (e.g., floodplain forest), it may be prudent to consider degraded sites that are suitable for ecological restoration for inclusion in reserves (Norton 1999). While establishment of reserves can provide multiple benefits, reserve function depends on adequate buffer from development. Extensive areas of forest buffer retained in combination with forest reserves are critical elements of biodiversity conservation.

The process for determining candidate sites for reserves in New England has not been well studied, but reserve design principles have emerged that can be applied here. It is generally accepted that large reserves are better than small reserves because they can absorb infrequent large-scale disturbances and allow re-colonization of disturbed sites from adjacent, undisturbed portions of the reserve. However, it is also recognized that small reserves can form key components that protect particular combinations of biodiversity that are not present elsewhere in the landscape. Small reserves can facilitate migration that sustains viability of metapopulations (Norton 1999). Overall, both occasional large and multiple small old-growth areas representing the full range of ecological diversity are needed (Vora 1994).

Candidate sites should have high biodiversity value and will likely include attributes such as priority natural communities (e.g., rich mesic forest, floodplain forest), rare species habitat, concentrations of wetland and riparian habitats, areas of primary forest, and a paucity of maintained roads, power lines and other rights-of-way corridors. Specific benefits of forest reserves could include development of primary herbaceous communities on the forest floor (Meier et al. 1995), unique assemblages of lichens and bryophytes (Dunwiddie et al. 1996, Flatebo et al. 1999), and possible development of unique

[2] Italicized text inserted in February 2007 in order to present DFW perspective on function and extent of forest reserves.

micro-invertebrate communities with accumulated woody debris and intact forest soils. Control of invasive, exotic plant species is critical within reserves (Hunter 1996a).

While timber harvesting would generally not occur on reserves, public hunting should occur within Massachusetts' reserves, which will lack large natural predators extirpated by humans (e.g., timber wolf (*Canis lupus*) and mountain lion (*Felis concolor*)). Overabundance of white-tailed deer (*Odocoileus virginianus*) affects forested ecosystems in many parts of North America (Warren 1997). Sustained high deer densities can negatively affect habitat composition and structure for various species (Waller and Alverson 1997). Without predation, herbivory by white-tailed deer can substantially reduce abundance and richness of native forest herbs and shrubs (Rooney and Dress 1997). Heavy deer browsing can subsequently decrease occurrence of some bird species that are negatively associated with high deer numbers in southern New England forests (e.g., Canada warbler [*Wilsonia canadensis*] and chestnut-sided warbler [*Dendroica pensylvanica*]) (DeGraaf et al. 1991).

Division forestlands are too small (average size about 900 acres each) to provide an effective reserve system. Also, the Division's landscape composition goal for 5-10% early-seral forest cannot be adequately addressed within reserves. With >80% of Massachusetts' forestland in private ownership, effective reserves will likely involve private/public partnerships. Legislative support to create financial incentives for private landowner participation in reserve establishment may be warranted. The Division, with its focus on biodiversity protection, should work with other conservation organizations to evaluate candidate sites on an eco-region basis (EPA 1994). This approach would be consistent with biodiversity conservation efforts by the Division's Natural Heritage & Endangered Species Program (Barbour et al. 1998), and fits well with the Division's existing Nature Preserves Program.

Forest Management on WMAs

Forest management on WMAs should maintain native species and natural communities. Silvicultural activities should be appropriate for each WMA based on its landscape setting, the occurrence of rare species habitat, and the presence of priority natural communities (NHESP 1998, NHESP 2000b). Identifying and protecting rare species populations and uncommon natural communities is a critical component of conserving native biodiversity (NH Forest Sustainability Standards 1997). Maintaining natural communities provides for many species whose ecology and habitat needs are poorly understood (Flatebo et al.1999). Inventory of natural community types (Swain and Kearsley 2000) will occur through a combination of remote sensing and subsequent field work.

<u>Inventory</u>

Approximately 20,000 ac of WMA lands were inventoried from 1985-1997 using a combination of aerial photo interpretation and ground-based vegetation sampling to delineate forest cover types (Mawson and Rivers 1995). In 1998, a comprehensive landcover mapping effort was undertaken to interpret natural community types on over 325,000 acres, including about 90,000 acres of WMA lands and about 235,000 ac of adjacent lands within ½ mile of WMA boundaries. Color-infrared aerial photo coverage was flown in 1999, interpretation and accuracy assessment occurred in 2000, and final GIS coverages of natural community types are expected by June 2001. Subsequent ground-based inventory will occur in 2001 and beyond based on results from the natural community mapping effort.

WMAs should be prioritized for subsequent inventory based on their landscape context, with larger WMAs in primarily forested landscapes, and WMA's with unique natural communities identified first. Ground-based inventory procedures should include parameters useful for assessing biodiversity value (Allen and Plantinga1999), and should be consistent with biological assessments conducted by the Division's Natural Heritage & Endangered Species section.

Landscape Composition Goals

Vertebrate wildlife species in New England benefit when primarily forested landscapes contain a mix of forest size classes, generally 5-15% seedling (or early-seral forest), 30-40% sapling-pole, 40-50% sawtimber, and <10% large sawtimber (DeGraaf et al. 1992:17). In addition to these seral stages, establishment of late-seral forest habitat is warranted to meet the Divisions' biodiversity goal, since late-seral habitat will likely benefit invertebrate wildlife, and understory plant communities. Maintaining 10% or more of forestland in a late-seral condition may be desirable (Vora 1994).

Late-seral forest is defined here as having attained >50% of its maximum expected biological age – generally >150 years for the range of tree species native to Massachusetts. Late-seral forest is uncommon throughout New England because trees generally reach economic maturity long before they reach biological maturity (60-90 years, vs. >150 years, respectively). To approximate a natural landscape age structure in New England, a portion of forest area should reach 300 years of age (Seymour and Hunter 1999).

Establishing landscape composition goals is a decidedly inexact science, but it is prudent to determine a desired future condition for WMA forestlands based on available knowledge. After considering habitat requirements for both vertebrate and invertebrate wildlife, landscape composition goals for WMA forestlands presently include 5-10% early-seral (seedling) forest, 10-15% sapling/small pole, 35-40% large pole, 35-40% sawtimber forest, and 10-15% late-seral forest (Fig. 3).

Forestlands on WMAs are presently <1% seedling (dbh <1"), <2% sapling-small pole (dbh 1-6"), nearly 40% large pole, nearly 60% sawtimber, and <1% large sawtimber (dbh >22") (Fig. 3)(MDFW unpubl. data). The large component of sawtimber forest is approaching, or is at economic maturity. The limited component of large sawtimber forest is at or is past economic maturity, is scattered in occurrence, and is generally 80-110 years old. WMAs currently support little or no true late-seral forest (trees>150 years of age).

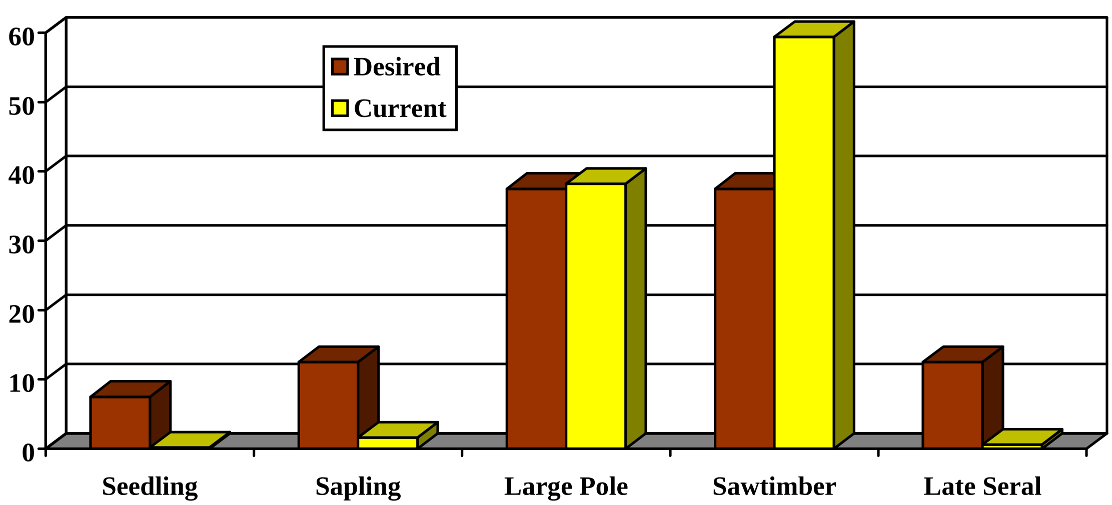

Fig. 3. Current vs. Desired Distribution of Forest Stages on WMAs.

Additional early- and late-seral forest habitat need to be secured on WMAs to meet composition goals described above. Maintaining the 'tails' of the age class distribution helps to provide temporal and spatial heterogeneity in landscapes to conserve biological diversity (Spies and Turner 1999). This can be accomplished by converting some sawtimber stands to early seral forest through active management, and by growing other sawtimber stands until they develop characteristics of late-seral forest. Late-seral character will be obtained on WMA forestlands by leaving groups of trees uncut on all actively managed forestland, and by designating areas of forest for passive management with little or no cutting.

Prioritization for Management

WMAs will be prioritized for management based on their landscape setting and forest condition (Fig. 4). Forest management practices should follow a checklist of procedures that includes a review of various data sources, conferences with District and Natural Heritage staff, and development of site plans that detail proposed management activities (Table 1). The site plan should contain specific recommendations by Division foresters regarding the type and extent of silvicultural activities designed to achieve landscape composition goals. Both active and passive management will be applied.

Active & Passive Management

Active management involves a variety of silvicultural practices that range from selection cuts with small (<1 acre) openings, to regeneration cuts with large (up to 25 acres) openings that feature retention of mature forest patches. Early- and mid-seral forest habitat will occur primarily on actively managed sites, while areas of late-seral forest will develop on passively managed sites.

Passive management involves either no cutting, or occasional single tree or small group (<0.25 ac) selection cutting to establish and maintain uneven-aged forest structure. Overall, most trees within passively managed areas will be left uncut. Over time, these areas will develop late-seral forest conditions. Passive management will occur on a variety of sites, including priority natural communities, forested riparian filters, steep slopes and ledges, and various upland forest areas that tie disparate sites into continuous patches of potential late-seral forest (Fig. 5). Most forested wetlands and floodplain forests will also be included in passively managed areas, and will generally be excluded from any cutting. Passive management should occur across a range of sites, from high to low productivity, in order to create a diversity of late-seral forest conditions. Designation of areas for passive management on each WMA should be made jointly by District, Natural Heritage, and Forestry staff.

About 85% (74,000 of 87,000 acres) of WMA forestlands will be actively managed using a combination of even-aged and uneven-aged silviculture. About 15% (13,000 of 87,000 acres) of WMA forestlands will be passively managed, with occasional selection cutting geared toward structural enrichment of presently even-aged forest.

Two-thirds of actively managed forestland will be devoted to even-aged silviculture, primarily including natural community types dominated by pine and oak, and where regeneration of shade intolerant northern hardwoods is desired. One-third of actively managed lands will be devoted to uneven-aged silviculture, primarily including natural community types dominated by shade tolerant northern hardwoods such as Eastern hemlock and sugar maple (*Acer saccharum*).

Rotation lengths on actively managed lands will be extended beyond the approximately 75 years typically employed in southern New England. Rotations are presently modeled at 100 and 120 years for even-aged and uneven-aged management, respectively, to meet landscape composition goals. Even-aged management will include some intermediate thinnings (at 25-50 year intervals) designed to increase structural diversity of even-aged forests that currently dominate WMAs. Uneven-aged management proposed here assumes a 30-year cutting cycle where one-quarter of stand area is treated

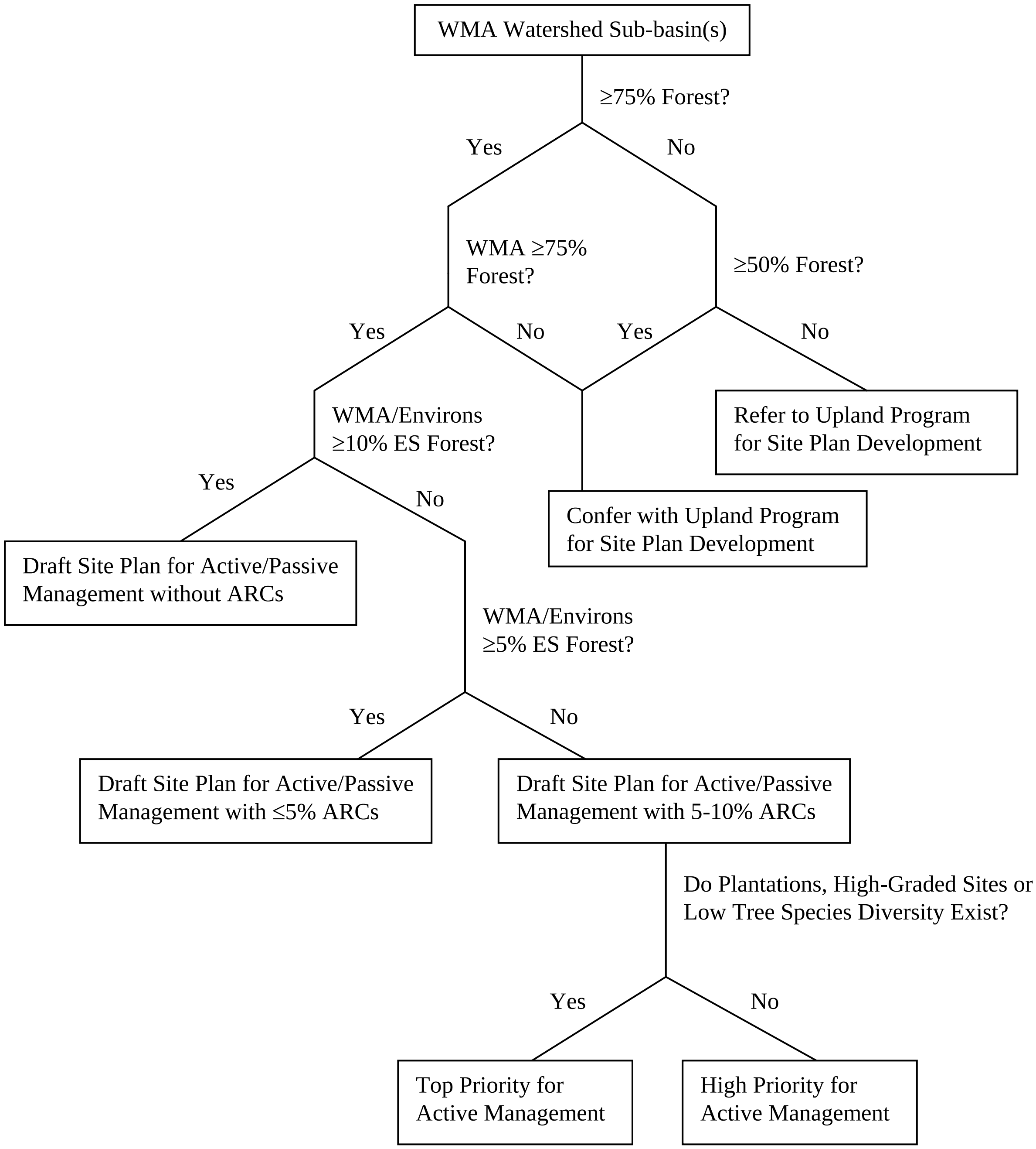

Fig. 4. Prioritization of WMA forestlands for management. High priority sites for management are those without adequate early-successional (ES) forest to meet landscape composition goals. Prioritization assumes that late-seral forest is lacking on all primarily forested landscapes in Massachusetts. Passive management for late-seral forest conditions will occur on all WMAs in primarily forested landscapes.

Table 1.Checklist of Procedures for Developing Site Plans and Conducting Timber Sales on WMAs in Primarily Forested Landscapes.

- ♦ Determine ecological region (EPA 1994), watershed basin and sub-basins (Bickford and Dymon 1990), and soil types (various NRCS publications).
- ♦ Review WMA and environs for rare species habitat (NHESP 2000b), priority natural communities (NHESP 1998), BioMap polygons (NHESP 2000a), vernal pools, and seeps. Determine the status of invasive, exotic plants in and near the WMA.
- ♦ Prepare landcover maps for the WMA and environs depicting natural community types and seral stages based on 1999-2000 data.
- ♦ Identify ownership and protection status of non-WMA properties within the watershed sub-basin(s) where WMA occurs.
- ♦ Summarize disturbance history of the WMA and environs. Include any current disturbance issues (e.g., excessive browsing by white-tailed deer and/or moose).
- ♦ Plan active and passive forest management to achieve landscape composition goals of 5-10% seedling forest (avg. dbh <1”), 30-35% sapling/pole forest (avg. dbh 1-9”), 40-50% sawtimber forest (avg. dbh 10-20”), and 10-15% late-seral forest (avg. dbh >20”). Map present and future cutting units by year and type of cut. Propose mitigation to limit establishment of invasive, exotic plants on planed cutting units.
- ♦ Outline pre-treatment biological monitoring as appropriate.
- ♦ Compile above items into a draft site plan. Request District and NHESP review of draft site plan. Modify active and passive management based on District and NHESP review. Reflect modifications in an amended site plan.
- ♦ Locate and mark all vernal pools within present cutting units. Plan harvest near pools according to guidelines for certified vernal pools in the Massachusetts Forest Best Management Practices Manual (the BMP manual) (Kittredge and Parker 1995), District, and NHESP recommendations.
- ♦ Locate and mark all seeps within present cutting units. Operate according to guidelines by Healy and Casalena (1996).
- ♦ Establish access roads, skid trails, and landing areas according to specifications in the BMP manual.
- ♦ Establish buffer strips along roads, and filter strips along riparian areas as per the BMP manual.
- ♦ Avoid wetland resource area crossings when planning skid trails and access roads whenever possible. Establish and maintain necessary stream crossings for logging machinery as indicated in the BMP manual.
- ♦ Obtain additional input from District and NHESP as appropriate after field work. Modify draft site plan based on field work. Submit modified site plan to the Forest Project Leader for review.
- ♦ Incorporate Forest Project Leader’s comments into final site plan. Complete timber sale marking for present cutting units as prescribed in site plan. Reserve groups of standing trees, including sound live trees, den trees and/or snag trees in all cuts through at least 2 full rotations. Reserve 10-25% tree cover in 2-3 groups per acre.
- ♦ File all required forms to obtain an approved Chapter 132 Forest Cutting Plan through DEM.
- ♦ Contract timber sale through a public, competitive bid process.
- ♦ Lop and locate slash to meet or exceed the Massachusetts Slash Law. Retain large downed dead material, such as felled cull trees, in all cuts, especially within reserved groups of standing trees.
- ♦ Establish post-treatment biological monitoring as appropriate. Use monitoring results to modify future harvesting practices as appropriate.

through group selection cutting. Under these conditions the term 'rotation' is applicable to uneven-aged management (Leak 1999). Group size will range between 0.1 and 1.0 acres. Passively managed lands will not have a rotation period applied since most trees in these areas will go un-harvested.

Overall, silvicultural practices on actively and passively managed forestlands will create extensive, heterogeneous forest patches with characteristics of unmanaged forest landscapes. While most of the Division's WMAs occur within primarily forested landscapes, some WMAs located in the Northeast and Southeast Districts are located within landscapes fragmented by development. The Division's landscape composition goals may not apply to WMAs that occur as fragmented forestland within a developed landscape.

1) Active management

On actively managed sites, cutting area boundaries will be determined by landform and forest condition. Within a cutting area, various size openings may occur. Uneven-aged silviculture using small openings of <0.5 ac will generally occur where shade tolerant tree species (e.g. sugar maple, and Eastern hemlock) are well adapted to the site. Even-aged silviculture with openings of one acre or more will occur on sites amenable to growing tree species that are intermediate or intolerant of shade.

The even-aged practice of shelterwood cutting will generally be used to regenerate tree species of intermediate shade tolerance (e.g. red oak and white pine). The shelterwood approach removes the overstory in two or three cuts spaced several years apart to secure adequate seed production, and subsequent seedling development (Smith et al. 1996). Large openings of up to 25 acres will occur on previously high-graded sites where tree species diversity and merchantability are relatively low, and where regeneration of shade intolerant northern hardwoods (e.g., black cherry and white ash) is desired.

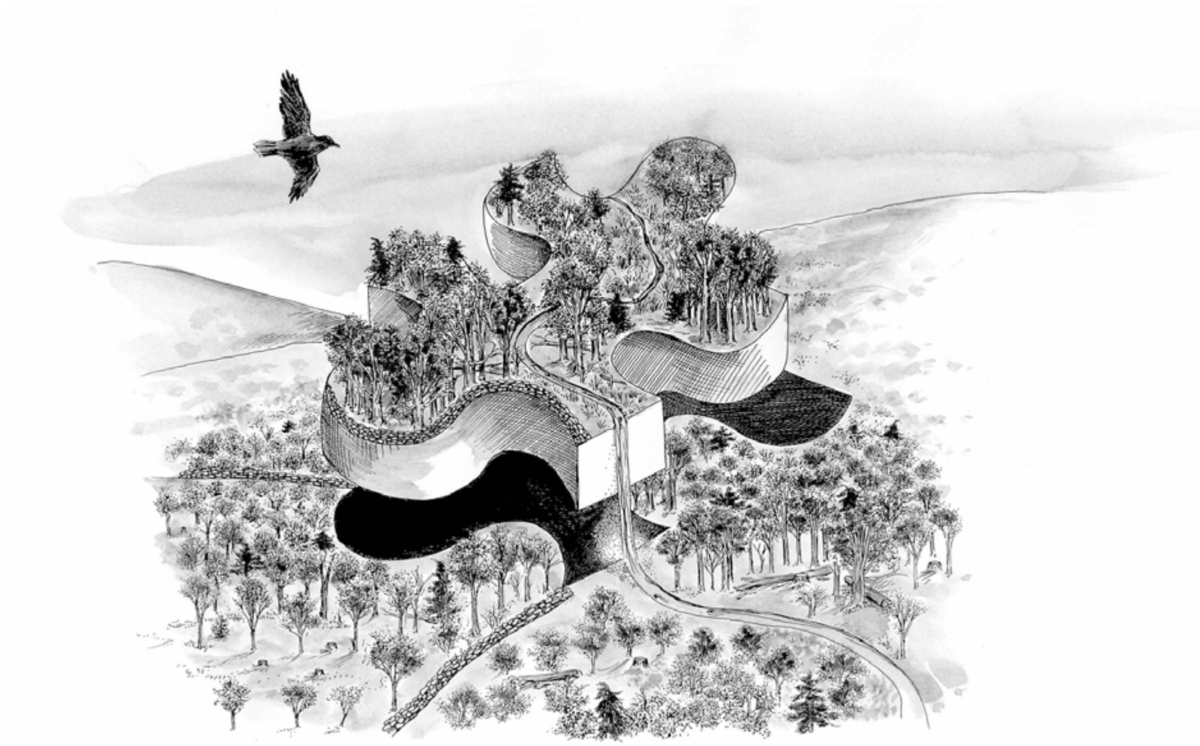

Fig. 5. Passive management for late-seral forest conditions will occur on portions of WMAs in primarily forested landscapes.

All large openings will feature clustered, or aggregate retention of mature trees. These aggregate retention cuts (ARCs) will produce two-aged stands with abundant early-seral habitat, while retaining important structural attributes associated with natural disturbance process (Table 2, Fig. 2). Within an ARC, 10-20% of existing forest cover is retained in clusters of live trees and snags. Retained groups will include both deciduous, mast-producing trees, and coniferous stems to provide thermal cover whenever possible. Existing den and cavity trees provide ideal nuclei for retained groups. Retained trees will generally not be cut during subsequent operations, but will be allowed to break up and decay naturally over time. Even where small openings occur, some trees will go un-harvested, and will eventually become a source of large, woody debris as they decay upon reaching their maximum biological life span.

To a degree, retained groups within ARCs act as miniature reserves, from which resident flora and fauna can repopulate harvested sites. ARCs will foster regeneration of both shade intolerant tree species (primarily on the south and west sides of retained groups), as well as shade tolerant species (primarily on the north and east sides of retained groups). Occurrence of both shade tolerant and shade intolerant tree species in the same stand is a characteristic of original eastern forests (Braun 1950).

ARC's are structurally diverse, and replace traditional clearcutting practices. While disturbance from clearcutting is beneficial to many wildlife species in New England (Tubbs et al. 1987), traditional clearcutting (removal of all woody stems) is generally detrimental to more wildlife species than it benefits (McAninch et al. 1984). Division foresters should prescribe and apply wildlife guidelines for ARCs to convert some small and medium sawtimber stands to early-seral forest (Table 2).

2) Passive management

While passively managed forestlands will eventually assume structural characteristics of old-growth forest, these areas will not be static. Old-growth forest ecosystems are often highly varied and dynamic in structure, composition and landscape pattern due to major structural or compositional reorganization following periodic natural disturbance by fire, wind, or pathogens (Foster et al. 1996). Ironically, strategies for the passive management of forestlands must be based on the acceptance and anticipation of change. Occasional, catastrophic disturbance will occur at some point in passively managed areas. Following such disturbance, active management is anticipated only for public safety, and to minimize potential for damage to adjacent lands from fire, insects, and disease. Major disturbance in passively managed areas will likely reduce planned establishment of early-seral forest on adjacent, actively managed forestland.

Passively managed areas will include some of the primary forestlands that occur on Division property. Primary forestlands are areas that may have been cut repeatedly in the past, but were never converted to agricultural use such as cropland or pasture. As such, original communities of forest soil micro-organisms (e.g., mycorrhizae) may be more intact in primary than secondary forest. However, since only 20-25% of Massachusetts forestlands occur as primary forest, and since these lands generally represent less productive and inaccessible areas, passive management should also occur on secondary forestlands with higher site productivity.

Many riparian areas will be passively managed. Functionally, the importance of riparian areas is disproportionately greater than other habitats of similar size because they are ecotonal in nature, and generally support greater density and diversity of species than adjacent uplands (Melton et al. 1984). The use of modified uneven-aged silvicultural practices in riparian areas will retain a shade-producing overstory canopy at all times which in turn maintains riparian area functions such as providing wildlife travel corridors, filtering debris from surface runoff, and insulating stream water temperatures (Melton et al. 1984). Retention of structural features such as large cavity-bearing trees during selection cutting in riparian areas is especially important because the habitat features provided by

Table 2. Checklist of Wildlife Guidelines for Aggregate Retention Cuts (ARCs) on WMA's

- ARCs should always be established under an approved Forest Cutting Plan through the Massachusetts Department of Environmental Management (DEM).
- ARCs should occur only on forested areas that can be successfully regenerated to desired tree species under nearly full sunlight (Sweeny et al 1984) (Rudnicky and Hunter 1993), and should be restricted to stable upland soils with slopes of <60%. Individual ARCs should be relatively large (3-25 ac) to provide adequate habitat for those wildlife species that prefer early successional forest (Thompson and Dessecker 1997). Individual ARCs should ideally be elongate and irregular in shape (McAninch et al. 1984).
- Establish ARCs in clusters to provide adequate habitat for wildlife species that are closely associated with early-seral stage forest. Retained strips of mature forest between ARCs to provide cover and travel corridors for wildlife, and vantage points for people to observe animals that are attracted to these sites. Retained strips can often be centered along or around cultural (e.g., stone walls and cellarholes (Sanford et al. 1994)) or natural (e.g., intermittent streams and seeps) features. Daylighted skid trails can be created which pass through retained strips and connect individual ARCs.
- Within each ARC, retain tree cover of >10 sq. ft. of basal area per acre. This level of retention meets or exceeds current recommendations of two 0.3 acre groups per 5 acres (Thompson and Dessecker 1997). Retained trees should generally occur in 2 or more groups per acre, and include existing cavity/den trees, live, wind-firm, mast-producing trees (such as oak, beech, or black cherry)[1], and conifer trees (such as hemlock, spruce, or white pine) for cover whenever possible (NH Forest Sustainability Standards 1997). U.S. Occupational Safety and Health Administration (OSHA) regulations regarding danger trees may be in conflict with retaining snags and/or live trees with dead or dying branches. The contractor must demonstrate that working within two tree lengths of danger trees will not create a hazard for an employee.
- All stems within an ARC >1" in diameter should be cut, except for trees within retained groups and individual stems of native species of valuable wildlife shrubs including viburnums, hawthorn, serviceberry, dogwood, hazelnut, cherries, grape and hophornbeam (Gill and Healy 1974). Retain wild apple trees and treat according to Olson and Langer (1970). Merchantable stems >4" dbh should be removed from the site. However, it is beneficial to have some large diameter (10-20+") downed logs available as habitat for over 30% of the regions mammal species (primarily rodents, shrews and carnivores), 45% of amphibians (primarily salamanders), 50% of reptiles (primarily turtles and snakes) (DeGraaf et al. 1992), and for numerous invertebrate species of wildlife that utilize downed woody debris. Downed logs are especially desirable within retained groups of mature trees. All branches and unmerchantable sections of cut trees should be retained on site.
- Slash should generally be reduced to <2' height for aesthetic purposes, but scattered brush piles are desirable. Construct 2 or 3 piles per acre atop rocks or stacked logs. Place large brush on the pile first, then stack smaller brush on top to a height of 4-6'. Do not construct piles adjacent to roads or property boundary lines.

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[1]Black cherry is a primary soft mast producer, with peak production at 30-100 years of age. Individual black cherry trees vary widely in fruit production, making the production history of individual trees an important consideration in selecting trees for retention. Likewise, individual oak trees tend to produce consistently good or poor acorn crops. Observe fruit production of individual oak and cherry trees for 2-3 years whenever possible before establishing an ARC to determine which individual trees are the best producers. Individual beech trees with recent bear claw marks and/or clumps of broken branches in the crown should also be retained.

living, dead and dying trees within riparian areas generally receive greater use by wildlife than the same features found within upland conditions (Hoover 1984).

Individual areas identified for passive management will generally be less than a few hundred acres. While the late-seral forest conditions that will develop in passively managed areas will not be extensive in acreage, they will include a range of natural community types and ecological conditions. In addition, these areas of late-seral forest will be well distributed throughout Massachusetts, and some or most of these areas will likely survive any single, occasional, catastrophic natural disturbance.

Harvest Levels

Harvest levels will follow an area-based approach on actively managed lands that, over time, moves toward a landscape structure containing both early- and late-seral forest. An area-based approach is the best way to achieve an ecologically sustainable harvest level (Seymour and Hunter 1999).

As presently modeled, publicly bid timber sales on actively managed forestland would initially generate about 4 million board feet (MMBF) from 1,500 acres annually. Eventually, after area regulation is achieved, commercial timber sales would harvest about 6 MMBF from 2,200 acres annually. This approach would maintain 5-10% of WMA forestland in early-seral forest (MDFW unpubl. data).

Harvest area increases over time because shelterwood cuts are repeated in two or three phases on the same site over a period of several years (Smith et al. 1996). Harvest volume increases over time for two reasons: one is that harvest area increases, and the other is that forestland cut in future years will have accrued additional volume relative to forestland cut today. WMA forestlands are generally 50-75 years old at present, and acreage cut today will support less volume than acreage cut in the future. Net income from commercial timber sales will vary with market conditions, but will be negatively impacted in the short run by the cost to remove trees of low merchantability (the result of previous high-grade cutting), and by the cost associated with upgrading existing access roads.

Economic value of sawtimber forest on some WMAs is relatively low due to past cutting of high-grade timber trees prior to public acquisition. This past cutting removed trees that made good lumber, and retained trees of poor form and vigor that now dominate many sites. High-grading is a common problem throughout Massachusetts (Mauri 1998). In some cases, dominant trees on high-graded sites are not only of poor form (low merchantability), but also occur on sites that are more amenable to growing other tree species. Examples include old field white pine (*Pinus strobus*) on hardwood sites following agricultural abandonment, and red maple growing on good red oak sites following high-grade cutting and/or fire exclusion by humans. Past human disturbance (e.g., agricultural conversion and/or cutting), as well as human constraint of natural disturbance (most notably fire) has greatly modified overstory tree species composition on many sites (Foster et al. 1998a, Abrams 1999). Active forest management can re-establish a mixture of native tree species well-adapted to these sites.

Should landscape conditions indicate that additional early-seral forest is desirable on a WMA, a portion of upland forest could be cut at 40-60 years as opposed to full rotations of 100-120 years or more. Regenerating some stands at a relatively young age of 40-60 years, and maintaining other stands beyond 150 years invokes a quasi-random spatial pattern of disturbances inherent in natural forests (Seymour and Hunter 1999). Any forest area cut on a shortened interval would subsequently receive an extended rotation.

Access Roads & Logging Machinery

The greatest potential for environmental degradation associated with any forest cutting, including clearcutting, is not from felling of trees, but rather from the transportation of the cut trees over improperly located and maintained skid trails and truck roads (Haussman and Pruett 1978). Soil exposure, compaction, and rutting from logging operations can combine to reduce biodiversity by disrupting the recycling of soil nutrients, reducing groundcover, limiting regeneration, eliminating habitat opportunities for soil biota, and creating opportunities for increased soil erosion (Flatebo et al.1999). Excessive disturbance to forest soils during logging can prevent re-establishment of primary herbaceous conditions due to the poor dispersal ability of many vernal forest herbs (Duffy and Meier 1992, Meier et al. 1995). Permanent edges of access roads can potentially serve as a convenient route for small predators that travel from open areas into the interior forest in order to feed on the eggs of ground-nesting birds (Askins 1994).

Truck roads should be restricted to the minimum extent necessary to provide access to a given WMA. Permanent truck roads capable of supporting logging trucks and tandem trailers (for removal of pulpwood) should generally be limited to 1.0 miles per square mile of forest in flat terrain, and <1.9 miles per square mile in rugged terrain (Maine Council on Sustainable forest Management 1996). Major skid roads and landings should generally occupy <10% of a harvested area (Flatebo et al. 1999). Location of permanent truck roads and temporary skid roads should be determined using topographic and soils maps to evaluate potential for erosion and soil compaction. Heavy logging equipment can compact soil air spaces, especially during wet conditions, impeding root growth and allowing entry of root diseases (Martin 1988).

Because it is generally feasible to transport wood products via forwarder along temporary skid trails for one-half mile or more to a landing site accessible to logging trucks, substantial portions of WMAs can and should remain without maintained roads. Ruts on both truck roads and skid trails should be repaired after completion of a harvesting operation according to standards established in the Massachusetts Forestry Best Management Practices Manual (Kittredge and Parker 1995).

Forest access roads that contain ruts can attract amphibians that breed in vernal ponds. Rut ponds may in effect act as ecological traps due to potentially elevated drying rates relative to naturally occurring vernal ponds (deMandyier and Hunter 1995). Forest access roads should generally not be located within a 100 foot buffer zone of a vernal pond. Some vernal pond species spend a great deal of time in upland habitats and travel more than 0.25 miles to a pond during the spring breeding season. While forest access roads can act as migration barriers, road maintenance and improvement practices will remove these barriers to migration (deMandyier and Hunter 1995).

Timber harvest operations can be extremely disruptive. However, planning elements described above combined with seasonal timing and the following equipment considerations can minimize the negative aspects of logging disturbance. The use of forwarders and directional feller-bunchers (which largely replace manual felling of trees by chainsaw operators) with “cut-to-length” operators are the preferred method of operation because they minimize soil disturbance and retain unmerchantable sections of trees on site as downed woody debris. Whole-tree harvesting and tree-length skidding should not be allowed on WMAs because these practices remove nutrients and elements of micro-habitat from forested sites (Perry 1998 and Flatebo et al.1999).

Biological Monitoring

Monitoring is a key element of wildlife habitat management (Gray et al. 1996) and of an ecosystem approach to sustainable forestry (Bordelon et al. 2000). A great variety of biological resources could potentially be monitored. On WMA forestlands, monitoring data should allow observers to determine if landscape composition goals, silvicultural goals, and biodiversity conservation goals are being achieved.

The majority of monitoring efforts on WMA forestlands to date have focused on breeding birds because they respond relatively quickly to changes in forest composition, and because various bird species that exhibit long-term population declines are of major conservation interest in the Northeast. Monitoring of additional taxonomic groups, including invertebrate wildlife and plants, is warranted. Of special concern are less mobile groups that may recolonize developing habitats more slowly than birds (Welsh and Healy 1993). Monitoring species composition and condition of tree, shrub, and herbaceous vegetation (Peet et al. 1998), as well as invertebrate and vertebrate occurrence in vernal pools (Kenney and Burne 2000) could help determine if silvicultural and biodiversity conservation goals are being met. Small mammals, while not unimportant for monitoring, respond to forest understory vegetation structure at a scale smaller than the stand level. Most forest stands will provide a variety of small mammal habitats at any stage of stand development (Healy and Brooks, 1988).

Division foresters should work cooperatively with personnel from the District offices, the Biodiversity Initiative's Upland Habitat Management and Ecological Restoration Programs (Biodiversity Initiative 1999), and the Division's Natural Heritage & Endangered Species Program to utilize existing sampling designs for monitoring both plant and animal communities. Monitoring should generally occur on both treated (cut) and control (uncut) sites. Monitoring efforts should generally be established prior to treatment, and continue until the impacts of treatment are adequately characterized (generally a few or several years). Results of monitoring efforts should be evaluated against the stated objective to maintain biodiversity. Management practices that reduce biodiversity at the landscape level should be amended or discontinued.

It is probably not feasible or necessary to monitor every forest management activity. It may be reasonable to monitor some actively and passively managed sites on WMAs in each ecoregion, and in each major natural community type within an ecoregion. All sites identified for active or passive forest management should be reviewed annually with District and Natural Heritage staff to select the most appropriate sites for monitoring.

Summary

This document presents statewide generic guidelines for all WMAs occurring in primarily forested landscapes, and is subject to periodic revision. The various management practices advocated in this document have specific applications, benefits and limitations. The successful application of the guidelines depends on the use of multi-scale management strategies to protect and maintain a diversity of species and processes, and to provide an element of risk-spreading against uncertainties that can be associated with the use of any single strategy.

Each of the forest management strategies contained in this document:

- Is designed to achieve the Division's goal to maintain biological diversity in Massachusetts.
- Considers populations and communities that occur across the entire landscape of natural and unnatural features owned by a diverse group of landowners.
- Will be enhanced through coordination, information sharing, and inter-organizational trust between all landowner types.

The Division recognizes that Massachusetts is a net importer of wood products, and that current state-wide harvests account for less than 10% of the per capita use of forest products in the state. To meet the Division's biodiversity goal in the face of increasing societal demands for wood products, foresters need to manage sustainably in a landscape context. Sustainable management must incorporate important structural elements of natural forest into actively managed forestlands.

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