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RESEARCH ARTICLE

Simulating the cumulative effects of multiple forest management strategies on landscape measures of forest sustainability

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Abstract While the cumulative effects of the actions of multiple owners have long been recognized as critically relevant to efforts to maintain sustainable forests at the landscape scale, few studies have addressed these effects. We used the HARVEST timber harvest simulator to predict the cumulative effects of four owner groups (two paper companies, a state forest and non-industrial private owners) with different management objectives on landscape pattern in an upper Michigan landscape managed primarily for timber production. We quantified trends in landscape pattern metrics that were linked to Montreal Process indicators of forest sustainability, and used a simple wildlife habitat model to project habitat trends. Our results showed that most trends were considered favorable for forest sustainability, but that some were not. The proportion of all age classes and some forest types moved closer to presettlement conditions. The trend for the size of uneven-aged patches was essentially flat while the average size of patches of the oldest and youngest age classes increased and the size of patches of the remaining age classes decreased. Forest fragmentation generally declined, but edge density of age classes increased. Late seral forest habitat increased while early successional habitat declined. The owners use different management systems that cumulatively produce a diversity of habitats. Our approach provides a tool to evaluate such cumulative effects on other landscapes owned by multiple owners. The approach holds promise for helping landowner groups develop and evaluate cooperative strategies to improve landscape patterns for forest sustainability.

Keywords Timber management · Multiple owner landscapes · Landscape pattern · HARVEST simulation model · Sustainable forestry · Biodiversity · Forest products industry

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Introduction

Sustainable forestry involves the extraction of forest products while maintaining ecosystem integrity to conserve biodiversity and to provide other non-commodity benefits to society. The maintenance of biodiversity is complex because biodiversity is determined by the interactions of numerous population and ecosystem dynamics including disturbance, competition and predation (Reice 1994; Wootton 2001). Viable populations of many species cannot be maintained through the actions of a single landowner because these populations are sustained at larger scales than the holdings of even the largest landowner (Hansen et al. 1991; Saunders et al. 1991). Population viability is a function of the combined actions of multiple landowners, which create a dynamic mosaic of forest types, stand structures and age distributions. Consequently, it is necessary to understand how the actions of individual land owners interact with the actions of others to determine the spatial pattern of the landscape mosaic, and therefore its ability to maintain biodiversity (Polasky et al. 2005). Assessing cumulative impacts of management actions on federal lands is required by law (National Environmental Policy Act of 1969). Furthermore, developing protocols to practice effective environmental stewardship across ownership boundaries is recognized as one of the great challenges of our time (Knight and Landres 1998, Parkhurst et al. 2002). Although the cumulative effects of the actions of multiple owners have long been recognized as critically relevant to efforts to practice sustainable forestry at the landscape scale, few studies have addressed these effects because few analytical tools are available to do so.

Recognizing the importance of the landscape mosaic for the conservation of biodiversity, both large industrial forestland owners and conservation groups have begun to broaden management objectives and conservation strategies. While industrial forestland owners manage their forests to produce the mix of fiber and forest products needed to supply their mills and to sell on the open market, they are also committed to maintaining other forestland values, including biodiversity, through multiple mechanisms, including forest certification. In the United States, the two major certification organizations are the Sustainable Forestry Board (Sustainable Forestry Initiative or SFI) and the Forest Stewardship Council (FSC), and their standards require individual land owners to consider the landscape beyond their boundaries. Some conservation groups are now acting beyond political policy advocacy and land acquisition to address the challenges of multiple ownerships by facilitating cross-ownership planning efforts.

Because of the SFI and FSC criteria and the desire by the public for greater accountability, it is necessary to assess the cumulative impacts of multiple owners and multiple management objectives on biodiversity. However, managing and modeling such complex systems is very difficult, especially for large landscapes and diverse communities of species. The practice of ecosystem management has become widely adopted because it is based on the simple notion that functioning, sustainable ecosystems will preserve biodiversity as a natural consequence (Grumbine 1994). Our study follows this concept by assuming that biodiversity is more likely to be maintained if ecosystem health and sustainability are maintained.

The Montreal Process Working Group is a collaborative working to advance the development of internationally agreed upon criteria and indicators for the conservation and sustainable management of temperate and boreal forests at the national level (Montreal Process Working Group 1999). Twelve governments have endorsed the Montreal Process and have voluntarily begun to monitor the sustainability of their forests using the criteria and indicators. The seven criteria identified in the Montreal Process are the essential components of the sustainable management of forests. They include vital functions and attributes (biodiversity, productivity, forest health, carbon sequestration, and soil and water protection), socio-economic benefits (timber, recreation and cultural values), and the laws and regulations that constitute the forest policy framework of a nation. This study focuses on a subset of these indicators that is specifically related to the conservation of biodiversity and/or landscape composition and pattern. Trends in the indicators are used to evaluate whether progress is being made toward ensuring the sustainability of forest management.

Little research has been conducted to understand how the varying management objectives and strategies of multiple landowners interact to produce landscape patterns (but see Kurttila et al. 2001, 2002). Furthermore, it is difficult to predict the effects of these interacting objectives on biodiversity and ecosystem sustainability (Polasky et al. 2005). The HARVEST timber harvest simulator is well suited to predict the cumulative effects of multiple owner actions on forest spatial pattern (Gustafson and Rasmussen 2002). By providing researchers control over timber harvest parameters that are typically determined within strategic management plans, HARVEST can be used to conduct virtual experiments to provide insight into the interaction of the actions of multiple forest land owners to produce landscape-wide patterns. Included in the output of HARVEST are maps of future forest age and composition, which can be evaluated relative to the Montreal Process indicators.

The objective of our study was to predict the collective, landscape-wide effects of diverse management objectives in a working forest landscape to provide insight into the problem of practicing sustainable forestry in multi-owner landscapes. Our approach was to (1) combine datasets for multiple ownerships (including a generic, non-industrial private landowner) to produce complete initial condition maps for the entire study area, (2) develop HARVEST parameters to represent the current management strategies of each owner, (3) conduct replicated simulations of 100 years of implementation of these strategies to produce projections of future landscape patterns and (4) assess how the cumulative effects of the strategies is expected to impact ecosystem sustainability as inferred by trends in selected Montreal Process indicators and amounts of wildlife habitat.

Methods

Description of study area

The study area is a 68,152 ha block in Menominee County, Michigan, USA (Fig. 1). The study area is almost completely contained within the West Green Bay Till Plain subsection (Keys et al. 1995). Topography is of glacial origin, featuring low moraines and eskers embedded in a matrix of relatively flat, moist lowlands. Northern hardwoods are predominant on the uplands and cedar and native conifers in the lowlands. This landscape is somewhat unique in the area, as a greater proportion of agricultural land is found to the south and east, and more upland forest to the west. The area is a regional deer wintering area. White-tailed deer migrate south from the Lake Superior snow belts to seek thermal cover in the conifer swamps that are sheltered from wind by eskers and moraines.

Two industrial landowners own large, relatively contiguous holdings that together dominate the study area (Fig. 1). The industrial owners have different management objectives that are determined by the particular product manufactured at their mills. Escanaba Timber LLC (ET) owns 22,002 ha that are managed primarily for softwoods while International Paper (IP) owns 7,828 ha that are managed primarily for hardwoods. The Michigan Department of Natural Resources (MDNR) owns 4,426 ha that are managed for both timber and wildlife. Non-industrial private forest (NIPF) tracts total 33,896 ha and are scattered throughout the study area. Based on the consensus of foresters working in the study area and the USDA Forest Service Woodland Owner Survey (Brett Butler, unpublished data), we assumed that 40% of NIPF land is not managed for timber (i.e., no timber is cut), and that the remainder is managed for generic timber objectives as described below.

Spatial timber harvest simulation

The timber harvest simulator HARVEST was designed as a strategic research and planning tool, allowing assessment of the spatial pattern consequences of broad timber management strategies (Gustafson 1999). The model is well suited to evaluate how the spatial pattern of age classes and forest composition change over time under specific management scenarios, providing mapped predictions of the spatial distribution of seral stages and cover types that are amenable to spatial analysis. With HARVEST, the object is not to find a scheduling solution (i.e., determining the sequence of harvest activities to optimize the

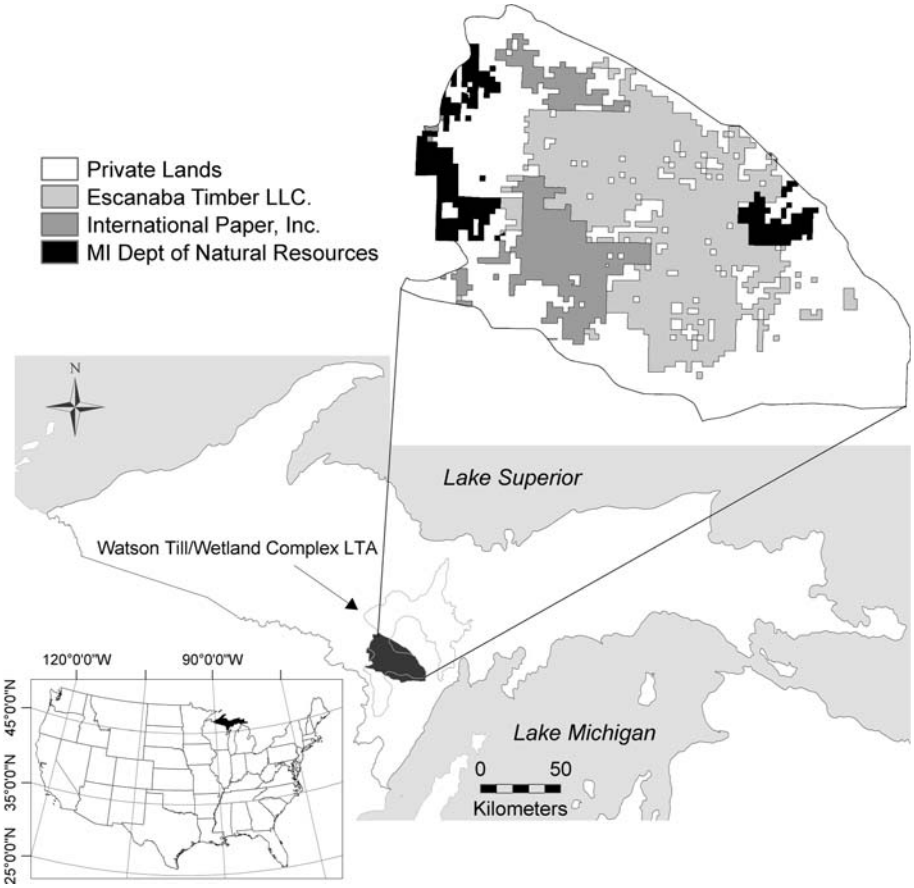

Fig. 1 Location of the study area and the ownership pattern

achievement of a specific objective), but to predict the spatial pattern consequences of a management strategy. It has been verified that HARVEST can mimic patterns produced by past timber management activity (Gustafson and Crow 1999). Because HARVEST targets management strategies to mapped spatial zones, it can readily simulate the strategies of multiple owners on complex ownership patterns.

HARVEST version 6.1 was developed for this study by adding to v6.0 (Gustafson and Rasmussen 2002) the capability to simulate forest type conversions (either by planting, release of advance regeneration or deterministic succession) and allowing the user greater control over the effects of harvest actions on forest age. In prior versions of HARVEST, simulated harvests always reset stand age to 1 year of age, which represented clearcutting or other even-aged silvicultural treatments. In v6.1, the user can specify any value for the residual age, or can decrease the existing age by a specific amount (Gustafson and Rasmussen 2005). This allows simulation of a greater variety of silvicultural treatments, including uneven-aged treatments. For example, if the cutting of a given forest type typically releases the advance regeneration of a different species found in the understory, v6.1 can convert the forest type to the understory species, and assign the age of the advance regeneration. The simulation of uneven-aged treatments requires a slight modification of the common conceptual model of stand age. If we think of stand age as a surrogate for the development of stand structure through time, rather than the time since the stand was established, age can be viewed as an index representing the structural characteristics of a stand as it would develop from a regeneration event in the past. For example, if the typical treatment of northern hardwood stands is to thin them every 15 years, keeping the stand structure relatively constant through time, we would decrease the "age" of those stands by 15 years at each thinning entry. If our strategy was to gradually produce stands with more mature structure by thinning less intensively, we would decrease the age at each entry by some value less than 15, say 10 years. Over time the average "age" (structural development) of northern hardwood stands across the landscape would increase.

Description of simulation parameters

The current management strategies of all the major owners on this landscape reflect both best management practices (BMPs, Peterson et al. 1998) and the specific objectives of each owner for forest and game commodities and forest and aquatic habitat conditions. Escanaba Timber manages primarily for softwoods in even-aged stands and plantations, International Paper manages primarily for hardwoods in uneven-aged stands, the MIDNR uses a mixture of even- and uneven-aged techniques, and the NIPF owners have varied objectives. We worked with representatives of each major landowner to convert their timber management strategies into HARVEST parameters (available on request). These representatives also estimated HARVEST parameters for NIPF landowners, based on their many years of experience working in this landscape. The parameters represent an average NIPF landowner who does allow timber cutting. We used HARVEST to simulate six replicates of the timber cutting practices of all owners for 100 years, using a 5-year time step and producing maps of forest age and forest type at each time step. Harvest rates for NIPF were chosen so that only 60% of the NIPF land base was harvested. The specific stands chosen for harvest were randomly selected by the model at runtime using the 'dispersed' dispersion method of HARVEST. Two deterministic succession processes were simulated on all ownerships at each time step. We assumed stable land ownership and objectives throughout the simulation to allow the cumulative effects of current management strategies to manifest themselves, recognizing that ownership and management strategies will almost certainly change over a century.

The three major landowners maintain stand maps in digital form that contain the information needed to create HARVEST input maps (stand boundaries, stand age and forest type). Stand maps were not available for the NIPF owners, so we estimated maps for these lands using a combination of a Landsat Thematic Mapper (TM) image classification created by MDNR (Michigan Department of Natural Resources 2001) (for forest type) and USDA Forest Service Forest Inventory and Analysis (FIA) data (for forest age, by type). To create the forest type map for NIPF lands, we created a subset of the classified image for the study area and used a 3×3 majority kernel filter to reduce the number of single-cell patches. We then reclassified the smoothed image to match the forest types recognized by the three major landowners. To create the stand age map, we delineated stand boundaries by assuming that all contiguous cells (adjacent in an 8-cell neighborhood) of a single forest type formed a stand. We probabilistically assigned a stand age based on the age distribution on the FIA plots that fell within the Landtype Association (LTA) (Jordan et al. 2002) encompassing the study area (Watson Till/Wetland Complex LTA). For each stand, we randomly selected (with replacement) an FIA plot having the same forest type as the stand, and assigned the age estimated for the FIA plot. These forest type and age maps were intersected with the corresponding maps created using stands data from the three major landowners to create the final input maps.

Calculation and analysis of indicators

Assumptions about indicators and biodiversity

Our study focused on three Montreal Process indicators under Criterion 1 (conservation of biological diversity), namely (1.1.a) proportion of area by forest type, (1.1.b) proportion of area by age class and (1.1.e) fragmentation of forest types. We calculated one indicator under Criterion 2 (maintenance of productive capacity of ecosystems), namely (2.c) the area of plantations of native and exotic species. The Montreal Process does not specify explicitly how trends in the indicators under Criterion 1 relate to trends in biodiversity. To assess trends in forest type, we compared current and predicted forest cover distributions to presettlement forest cover distributions. We assumed that (1) native species are adapted to presettlement conditions (Moore et al. 1999; Swanson et al. 1994), and (2) the available presettlement data are a valid representation of the environment to which native species are adapted. Although forest composition and structure at a particular site may vary in time, the ranges of the major trees (maples, pines and oaks) in the Lake States have moved only 4–10 km/century over the last 10,000 years (Davis 1981; Frelich 2002). We used estimates of presettlement vegetation derived from General Land Office (GLO) notes collected by surveyors in the early to mid 1800s (Comer et al. 1995, Schulte and Mladenoff 2001).

Calculation of indicators from the output maps

Response variables relevant to Montreal Process indicators were calculated using the analytical functions of HARVEST and APACK (Mladenoff and DeZonia 2004). Forest type classes were analyzed directly from the forest type output maps generated by HARVEST. Age class maps for analysis were produced by recoding the age map into five age classes (1–15, 16–30, 31–55, 56–70, >70 yrs) and an uneven-aged class consisting of all northern hardwood, aspen or hemlock cells with an age >70 yrs, and all upland softwood cells >60 years of age. These types tend to develop an uneven age structure by age 70 when actively managed. Indicators were calculated by forest type and by age class. Indicators for Criterion 1 were landscape proportion, and measures of forest fragmentation (mean patch size, overall edge density, contagion, area of forest interior habitat (forest >150 m from an opening (cut within 20 yrs) or non-forest edge), and forest edge habitat (all non-interior forest). The indicator for Criterion 2 was the area of softwood plantations (all of the European larch and red pine).

Estimating trends in wildlife habitat

Direct effects of landscape composition on animal species diversity were determined using the non-spatial MI Wild software tool (Doepker et al. 2000). This tool, developed and used by the Michigan Department of Natural Resources, estimates the amount of habitat for Michigan animal species based on species-specific habitat models, published species–habitat studies and expert opinion. The model relates the habitat requirements of species to the forest types in different age classes that provide those habitat conditions. MI Wild calculates the total area of habitat for a species that is present on a landscape, but considerations such as fragmentation, type juxtaposition, topographic position, minimum home range size and population viability are not included. For this reason, predictions by MI Wild should be considered approximate.

We calculated the area in each combination of forest type and age class (including uneven age) and input these values into MI Wild to predict the percent change in habitat area for 153 wildlife species over the 100 simulated years. We did not include species that have habitat requirements determined primarily by aquatic environments (e.g., common goldeneye, Northern waterthrush), which our model did not simulate.

To apply MI Wild, the forest types used in our study were correlated with those used in MI Wild. The cedar, hemlock, aspen, red pine, and northern hardwood types were the same in both classifications. We converted our lowland softwood, lowland hardwood, and upland conifer types to mixed swamp conifer, mixed lowland hardwood, and mixed upland conifer, respectively. Since European larch plantations do not correspond to the ''larch'' type in MI Wild, which is a swamp type, this type was called ''plantation.'' The age classes used in our study (1–15, 16–30, 31–55, 56–70, and >70 years) were assumed to correspond to the MI Wild size classes regeneration, sapling, pole, small saw, large saw, respectively, and ''uneven'' was the same in both. The only exception was that at 100 years our simulations produced an average of 258 ha of uneven-aged aspen, which MI Wild does not allow. This class was converted to the >70 year age class.

Estimating the slope of the trends

For Montreal Process indicators, we plotted the mean of each indicator against simulated time and fitted a linear regression line to the points to estimate the direction and significance of any trend. For trends in proportions that were deemed significantly different than zero ($\alpha = 0.05$, two-tailed test), we evaluated whether the proportion was converging or diverging from the pre-settlement condition.

Results

Montreal Process indicators

The proportion of cedar, lowland conifer, European larch and aspen changed significantly through time (Fig. 2). The abundance of cedar, lowland conifer and aspen moved closer to pre-settlement conditions, and red pine remained near pre-settlement abundances. Northern hardwood and hemlock persisted at levels below pre-settlement levels, while European larch (an exotic), and upland and lowland hardwoods were above pre-settlement levels. The proportion of all age classes except the youngest changed significantly through time (Fig. 3). The abundance of each age class moved closer to the putative pre-settlement condition.

The average size of patches of lowland conifer, European larch and aspen increased significantly through time, while the size of cedar patches decreased (Fig. 4). Patch sizes of the remaining types were remarkably stable because there was little conversion of these types. The average size of patches defined by age class varied considerably through time (Fig. 5). The trend for the size of uneven-aged patches was essentially flat while the average size of patches of the oldest and youngest age classes increased and the size of patches of the remaining age classes decreased.

The fragmentation of forests generally declined through time. The area of forest interior habitat increased significantly through time, while the area of edge habitat was stable (Fig. 6). Interior conditions increased where no-cutting and uneven-aged practices were aggregated within the ownerships where they were used, and even-aged practices maintained edge habitat elsewhere. Overall density of edges among age classes increased through time, while the density of edges among forest types decreased (Fig. 7). The measure of forest productivity increased, as the area of plantations doubled in the first 20 years, producing a significant increasing trend over the 100 years of simulations (Fig. 8).

Wildlife habitat

Because MI Wild predictions are approximate, it was judged that changes over the 100-year simulation of less than ±10% are not large enough to

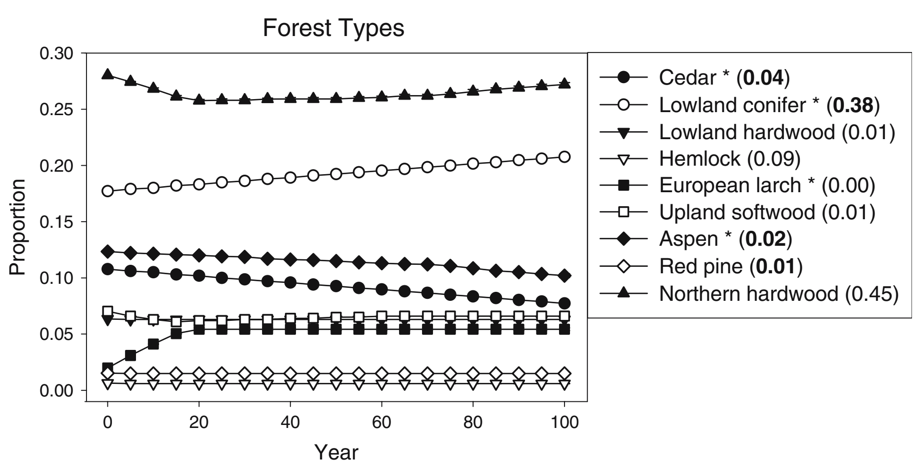

Fig. 2 Proportion of forest types through simulated time. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). The value in parentheses gives the pre-settlement proportion (Comer et al. 1995), and it is in boldface if the trend is at or moving toward the pre-settlement value

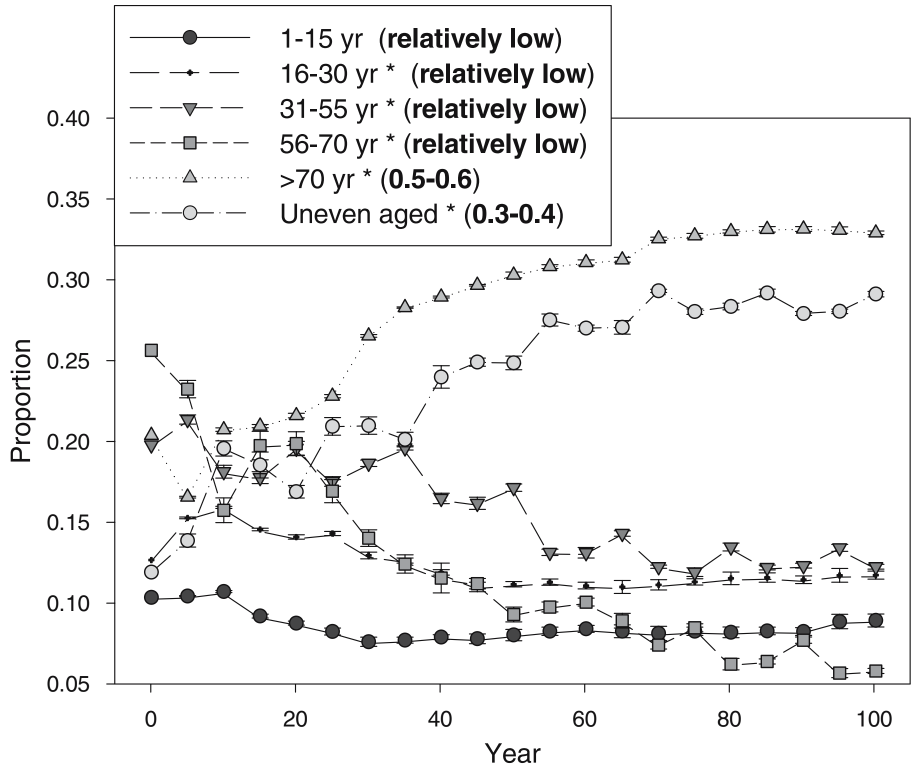

Fig. 3 Proportion of age classes through simulated time. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). The value in parentheses gives the pre-settlement proportion (Comer et al. 1995), and it is in boldface if the trend is at or moving toward the pre-settlement value

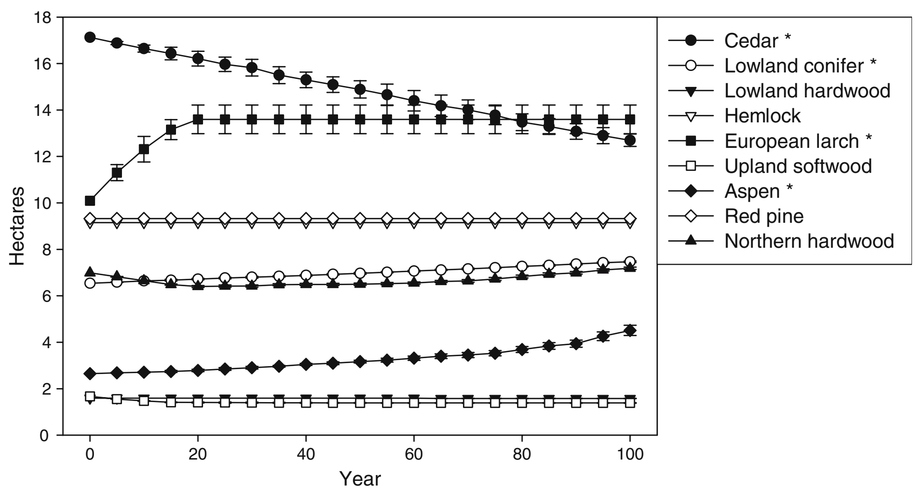

Fig. 4 Mean size of patches defined by forest type through simulated time. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). Sustainability was assumed to be enhanced as patch size increases

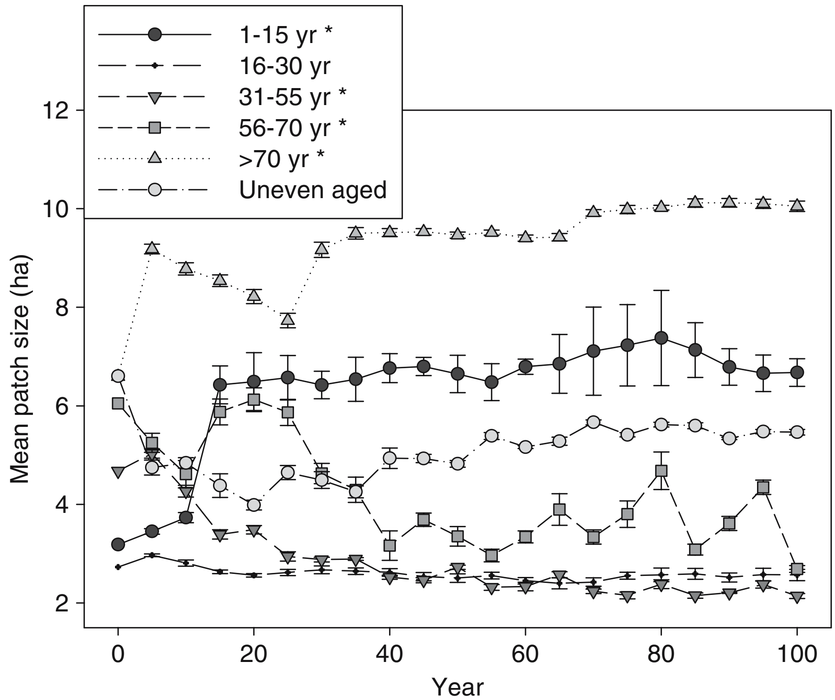

Fig. 5 Mean size of patches defined by age class through simulated time. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). Sustainability was assumed to be enhanced as patch size increases

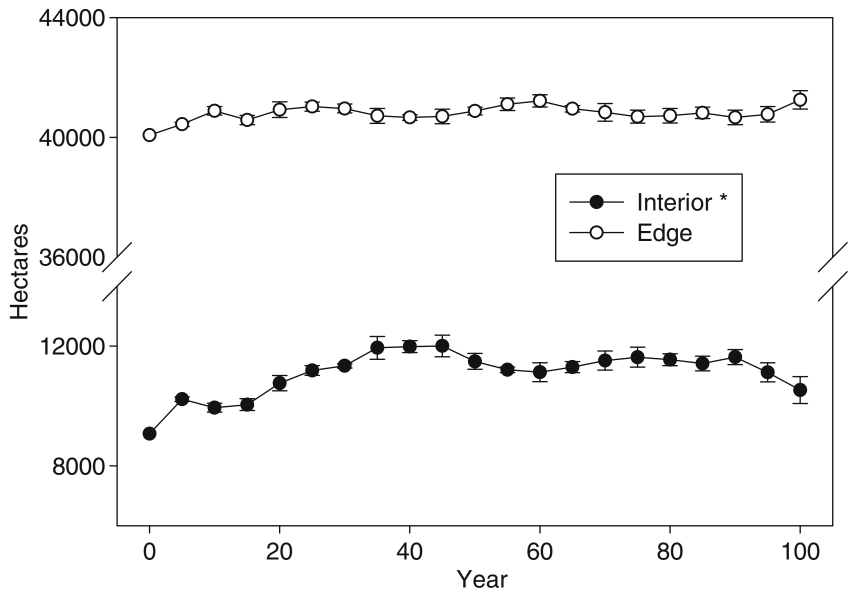

Fig. 6 Mean area of forest edge and interior habitat through simulated time. Interior was defined as forest >150 m from an opening (cut within 20 yrs) or non-forest edge, and forest edge habitat was all non-interior forest. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). Sustainability was assumed to be enhanced as forest interior increases, edge decreases

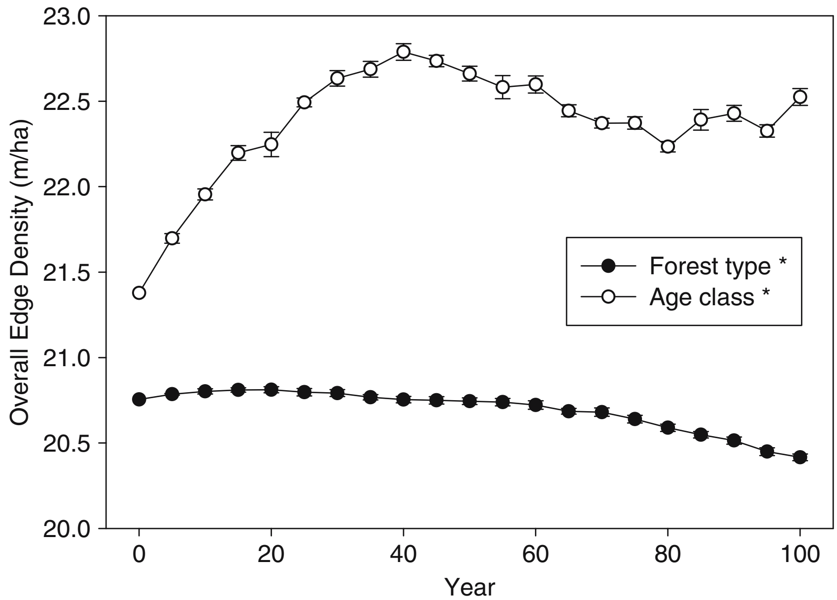

Fig. 7 Mean overall edge density where edges are delineated by either forest type or age class. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). Sustainability was assumed to be enhanced as edge density decreases

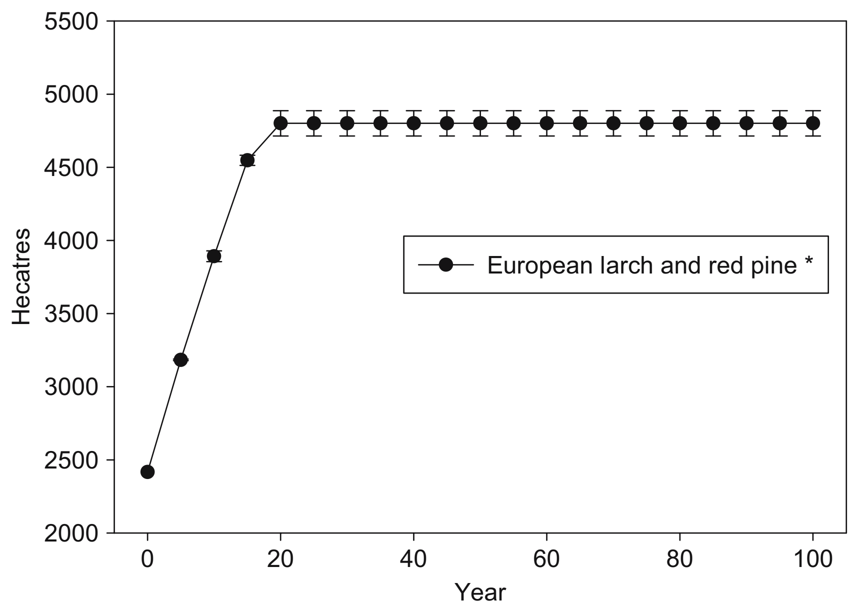

Fig. 8 Mean area of plantations of European larch or red pine. Points represent the mean of six replicates and the error bars indicate one standard error (which may be smaller than the width of the symbols). An asterisk in the legend indicates a trend line significantly different than zero ($\alpha = 0.05$). Forest productivity was assumed to be enhanced as the area of plantations increases

be significant and these are considered to be ''no change'' below. In general, habitat for species requiring older forests increased while habitat for species living in early successional forest declined (Table 1).

Birds

Overall, 16 bird species had decreased habitat, 53 had stable habitat, and 24 had increased habitat over the 100-year simulation. Three species had large declines and eight had large increases in habitat. Species showing more than a 40% habitat decline were Chestnut-sided warbler, Pine warbler, and Red crossbill. Species showing more than a 40% increase in habitat were American kestrel, Merlin, Mourning dove, Common nighthawk, Black-backed woodpecker, Winter wren, Clay-colored sparrow and Field Sparrow, although most of these had relatively small amounts of habitat on the landscape.

Mammals

Overall, 2 species had decreased habitat, 33 species showed no change and 9 had increased habitat over the 100-year simulation. The MI Wild results showed essentially stable white-tailed deer habitat, which coupled with increasing edge density, suggests that deer populations are likely to remain high. American marten habitat increased 8%, which while not above our criterion for significance, is more likely to be positive than negative. Another fur-bearer, Ermine, showed an 11% increase. The greatest increases in mammal habitat were for the two squirrel species. The only major decline was –39% for the little brown bat.

Herpetofauna

Overall, 1 species (Brown snake) had reduced habitat, 13 had no change, and 4 had increased habitat over the 100-year simulation.

Discussion

Our objective was to assess the cumulative effects of very different landowner management strategies in a 68,000 ha industrial forest landscape on measures of forest sustainability over time. Our results were somewhat equivocal because some measures showed trends considered favorable for sustainability while others did not. For example, when edge density is measured among forest types, the effect of forest management on measures of sustainability is likely to be positive (i.e. edge density declines), but when measured among age classes, the effect is likely to be negative (i.e. edge density increases; Fig. 7). In addition, the response of biodiversity to changes in the Montreal Process indicators is difficult to quantify, and our assessment of a positive change is based on the assumption that biodiversity responses to landscape characteristics will be linear (Harper et al. 2005). One of us (R. Swaty) will test this assumption using field studies of Michigan landscapes with different spatial configurations. Until these data are in hand, our statements regarding sustainability must remain qualitative. Taken together, however, our results suggest that this landscape is being managed sustainably, and is projected to show improvement in most categories. This result is consistent with the findings of Polasky et al. (2005), who found that a large fraction of conservation objectives in a working forest landscape can be achieved without compromising economic objectives.

The most pronounced effect of forest management on the study landscape is an increase in the area occupied by mature and uneven-aged forest stands. By the end of the 100-year simulation, approximately 65% of the landscape is occupied by these age classes, up from approximately 35% at the beginning. These gains come at the expense of stands between 30 and 70 years of age, which decrease from 45% to 25% of the landscape. These trajectories are consistent with a movement of the forest stand age structure towards pre-settlement conditions. Increases in stand age, however, are not uniform across stand types or ownerships. About 86% of the uneven-aged stands consist of northern hardwoods, and over 80% of stands >70 years consist of cedar and lowland conifers. While uneven-aged northern hardwood stands occur on all ownership classes, the majority of mature cedar and conifer stands occur on the NIPF ownership class.

Table 1 Results of MI Wild calculations of changes in habitat for forest-dwelling species within the study area. Water-dependent bird species were omitted. Species with habitat area <5% of the landscape are not shown

Common name	Scientific name	Habitat area at time 0 (ha)	% change
Birds			
Bald Eagle	*Haliaeetus leucocephalus*	26,133.6	31.0
Sharp-Shinned Hawk	*Accipiter striatus*	19,864.0	2.1
Cooper's Hawk	*Accipiter cooperii*	23,172.4	–1.0
Northern Goshawk	*Accipiter gentilis*	19,864.0	2.1
Red-Shouldered Hawk	*Buteo lineatus*	19,730.8	–1.5
Broad-Winged Hawk	*Buteo platypterus*	19,158.0	3.0
Red-Tailed Hawk	*Buteo jamaicensis*	23,541.6	2.8
Merlin	*Falco columbarius*	8,899.2	86.4
Ruffed Grouse	*Bonasa umbellus*	8,123.2	–0.6
Great Horned Owl	*Bubo virginianus*	28,400.0	–3.6
Barred Owl	*Strix varia*	20,159.6	–1.4
Long-Eared Owl	*Asio otus*	16,053.6	37.2
Northern Saw-Whet Owl	*Aegolius acadicus*	16,053.6	36.1
Whip-Poor-Will	*Caprimulgus vociferus*	23,668.0	2.1
Ruby-Throated Hummingbird	*Archilochus colubris*	22,938.0	36.3
Red-Headed Woodpecker	*Melanerpes erythrocephalus*	16,639.6	4.2
Red-Bellied Woodpecker	*Melanerpes carolinus*	18,199.6	1.0
Yellow-Bellied Sapsucker	*Sphyrapicus varius*	19,158.0	3.0
Downy Woodpecker	*Picoides pubescens*	26,759.2	–1.8
Hairy Woodpecker	*Picoides villosus*	26,987.6	28.7
Black-Backed Woodpecker	*Picoides arcticus*	14,245.2	45.2
Northern Flicker	*Colaptes auratus*	22,318.4	0.5
Pileated Woodpecker	*Dryocopus pileatus*	18,199.6	1.0
Eastern Wood-Pewee	*Contopus virens*	16,639.6	4.2
Acadian Flycatcher	*Empidonax virescens*	18,199.6	1.0
Least Flycatcher	*Empidonax minimus*	21,020.0	–0.6
Eastern Phoebe	*Sayornis phoebe*	4,902.8	–0.3
Great Crested Flycatcher	*Myiarchus crinitus*	21,596.8	–1.3
Gray Jay	*Perisoreus canadensis*	19,327.2	7.5
Blue Jay	*Cyanocitta cristata*	21,325.6	–0.4
American Crow	*Corvus brachyrhynchos*	29,008.8	3.4
Common Raven	*Corvus corax*	49,536.0	0.2
Black-Capped Chickadee	*Poecile atricapillus*	44,497.2	4.8
Boreal Chickadee	*Poecile hudsonicus*	12,798.0	30.2
Red-Breasted Nuthatch	*Sitta canadensis*	13,230.8	29.3
White-Breasted Nuthatch	*Sitta carolinensis*	18,909.6	–3.2
Brown Creeper	*Certhia americana*	34,848.0	16.3
Winter Wren	*Troglodytes troglodytes*	13,852.8	41.7
Golden-Crowned Kinglet	*Regulus satrapa*	20,790.0	10.0
Ruby-Crowned Kinglet	*Regulus calendula*	19,327.2	7.5
Blue-Gray Gnatcatcher	*Polioptila caerulea*	18,199.6	1.0
Veery	*Catharus fuscescens*	21,841.2	0.8
Swainson's Thrush	*Catharus ustulatus*	17,403.6	7.2
Hermit Thrush	*Catharus guttatus*	3,386.4	24.5
Wood Thrush	*Hylocichla mustelina*	19,730.8	–1.5
American Robin	*Turdus migratorius*	29,151.2	–7.8
Cedar Waxwing	*Bombycilla cedrorum*	36,044.8	–5.0
Warbling Vireo	*Vireo gilvus*	26,744.0	0.6
Red-Eyed Vireo	*Vireo olivaceus*	21,596.8	–1.3
Nashville Warbler	*Vermivora ruficapilla*	15,032.8	–11.7
Chestnut-Sided Warbler	*Dendroica pensylvanica*	4,760.4	–44.4
Magnolia Warbler	*Dendroica magnolia*	4,403.2	–29.1
Black-Throated Blue Warbler	*Dendroica caerulescens*	16,639.6	4.2

Table 1 continued

Common name	Scientific name	Habitat area at time 0 (ha)	% change
Yellow-Rumped Warbler	*Dendroica coronata*	3,386.4	19.5
Blackburnian Warbler	*Dendroica fusca*	3,386.4	19.5
Cerulean Warbler	*Dendroica cerulea*	18,199.6	1.0
Black-And-White Warbler	*Mniotilta varia*	25,427.6	32.4
American Redstart	*Setophaga ruticilla*	19,294.8	3.9
Ovenbird	*Seiurus aurocapillus*	21,020.0	–0.6
Mourning Warbler	*Oporornis philadelphia*	15,032.8	–11.7
Canada Warbler	*Wilsonia canadensis*	16,639.6	4.2
Scarlet Tanager	*Piranga olivacea*	16,639.6	4.2
Northern Cardinal	*Cardinalis cardinalis*	8,825.6	–19.2
Rose-Breasted Grosbeak	*Pheucticus ludovicianus*	18,909.6	–3.2
Chipping Sparrow	*Spizella passerina*	3,928.0	21.0
White-Throated Sparrow	*Zonotrichia albicollis*	5,318.8	–39.8
Dark-Eyed Junco	*Junco hyemalis*	4,788.8	–6.4
Brown-Headed Cowbird	*Molothrus ater*	4,602.4	5.8
Baltimore Oriole	*Icterus galbula*	19,730.8	–1.5
Purple Finch	*Carpodacus purpureus*	17,836.4	7.1
Pine Siskin	*Carduelis pinus*	20,051.6	9.4
Mammals			
Virginia Opossum	*Didelphis virginiana*	26,491.2	–1.4
Arctic Shrew	*Sorex arcticus*	19,397.6	–0.1
Masked Shrew	*Sorex cinereus*	61,220.4	–3.0
Pygmy Shrew	*Sorex hoyi*	48,027.2	–1.9
Water Shrew	*Sorex palustris*	17,403.6	7.2
Northern Short-Tailed Shrew	*Blarina brevicauda*	62,566.8	0.8
Star-Nosed Mole	*Condylura cristata*	16,111.2	37.8
Northern Myotis	*Myotis septentrionalis*	39,866.8	9.3
Little Brown Myotis	*Myotis lucifugus*	5,018.8	–39.1
Silver-Haired Bat	*Lasionycteris noctivagans*	34,142.0	17.1
Eastern Red Bat	*Lasiurus borealis*	20,325.6	–2.9
Hoary Bat	*Lasiurus cinereus*	34,848.0	16.3
Eastern Cottontail	*Sylvilagus floridanus*	4,602.4	5.8
Snowshoe Hare	*Lepus americanus*	28,114.4	1.8
Eastern Chipmunk	*Tamias striatus*	19,093.6	–3.0
Least Chipmunk	*Tamias minimus*	24,186.4	–1.4
Eastern Gray Squirrel	*Sciurus carolinensis*	6,942.0	114.2
Eastern Fox Squirrel	*Sciurus niger*	18,199.6	1.0
Red Squirrel	*Tamiasciurus hudsonicus*	10,459.2	70.3
Northern Flying Squirrel	*Glaucomys sabrinus*	32,416.0	20.6
Southern Flying Squirrel	*Glaucomys volans*	18,199.6	1.0
American Beaver	*Castor canadensis*	7,849.2	–11.3
White-Footed Mouse	*Peromyscus leucopus*	33,196.0	–5.3
Deer Mouse	*Peromyscus maniculatus*	62,566.8	0.8
Southern Red-Backed Vole	*Clethrionomys gapperi*	62,566.8	0.5
Southern Bog Lemming	*Synaptomys cooperi*	33,126.0	14.9
Woodland Jumping Mouse	*Napaeozapus insignis*	22,814.0	–0.3
Common Porcupine	*Erethizon dorsatum*	42,736.4	3.4
Coyote	*Canis latrans*	53,888.0	1.8
Gray Wolf	*Canis lupus*	27,269.2	–3.1
Red Fox	*Vulpes vulpes*	53,888.0	2.1
Common Gray Fox	*Urocyon cinereoargenteus*	19,730.8	–1.5
Black Bear	*Ursus americanus*	53,888.0	1.8
Common Raccoon	*Procyon lotor*	27,693.6	27.4
American Marten	*Martes americana*	23,517.2	7.9
Fisher	*Martes pennanti*	22,011.6	6.4
Ermine	*Mustela erminea*	41,202.0	11.1

Table 1 continued

Common name	Scientific name	Habitat area at time 0 (ha)	% change
Long-Tailed Weasel	*Mustela frenata*	61,220.4	–3.0
Striped Skunk	*Mephitis mephitis*	34,542.4	1.8
Bobcat	*Lynx rufus*	52,364.8	–3.7
White-Tailed Deer	*Odocoileus virginianus*	21,200.0	1.4
Moose	*Alces alces*	39,106.8	1.8
Herpetofauna			
Eastern Newt	*Notophthalmus viridescens*	34,848.0	16.3
Spotted Salamander	*Ambystoma maculatum*	18,909.6	–3.2
Blue-Spotted Salamander	*Ambystoma laterale*	19,730.8	–1.5
Tiger Salamander	*Ambystoma tigrinum*	31,256.0	–4.8
Redback Salamander	*Plethodon cinereus*	18,992.0	4.7
Four-Toed Salamander	*Hemidactylium scutatum*	16,639.6	4.2
American Toad	*Bufo americanus*	41,822.8	–4.4
Spring Peeper	*Pseudacris crucifer*	45,118.0	3.5
Gray Treefrog	*Hyla versicolor*	33,976.0	18.1
Wood Frog	*Rana sylvatica*	33,976.0	18.1
Eastern Box Turtle	*Terrapene carolina*	16,639.6	4.2
Five-Lined Skink	*Eumeces fasciatus*	16,639.6	4.2
Common Garter Snake	*Thamnophis sirtalis*	37,509.2	–4.8
Brown Snake	*Storeria dekayi*	42,143.6	–47.1
Redbelly Snake	*Storeria occipitomaculata*	9,313.6	–9.5
Ringneck Snake	*Diadophis punctatus*	18,563.2	4.8

While Montreal Process indicators suggest a slight net improvement in conditions conducive to the maintenance of biodiversity through time, these measures alone do not provide a complete assessment of biodiversity responses to forest management. However, additional information can be drawn from our simulation results. For example, although stands between 1 and 15 years of age are remarkably stable throughout the simulation (covering approximately 10% of the landscape), the composition of these stands changes greatly through the simulation. Initially they are approximately equally divided among six stand types, but by year 100 nearly 85% of young stands consist of European larch and aspen located on industrial ownerships, and there are no acres of young cedar, hemlock, or northern hardwoods. The concentration of young stands in a few cover types, especially when one is an exotic, may be cause for concern about the composition of the forest landscape beyond the 100 year time horizon of this simulation. However, these data should not be interpreted as a general lack of forest regeneration on the study landscape. The species that comprise the most common stand types (northern hardwood, cedar, and lowland conifers) at the end of simulation are capable of regenerating under a mature forest canopy, and thus young trees are likely to occur under mature and uneven-aged conditions in these types.

Of the nine forest cover types simulated in this study, the future of hemlock is probably of most concern. On the pre-settlement landscape, hemlock was capable of self-replacement in the western Great Lakes region, and some hemlock stands persisted for thousands of years (Davis et al. 1998). However, a variety of factors (loss of seed sources, increased deer densities, and climate changes (Alverson et al. 1988; Mladenoff and Stearns 1993)) have contributed to widespread regeneration failure, and hemlock stands are much less common now than on the pre-settlement landscape. The simulation results for this study show the hemlock cover type as stable, but only because stands extant at the beginning of the simulation persist, and not because of a balance between stand destruction and formation. If regeneration of hemlock in these stands remains rare (regeneration is a process not simulated by HARVEST), eventually hemlock stands will succeed to other forest types, further reducing the abundance of this type on the landscape. In addition, because no ownership group has developed or implemented a hemlock restoration strategy, it is unlikely that any particular ownership block will serve as a refuge in which hemlock will persist.

Many industrial forest landowners seek independent certification of their forestry practices to enhance the marketability of their products and to improve the sustainability of their forests (Lucier and Shepard 1997). Our approach allows such owners to objectively predict the effects of all owners (including themselves) on landscape pattern, and to make inferences about its effect on sustainability. In the absence of information on cumulative effects such as we have documented here, forest managers may have some confidence in their predictions of the effects of their actions on their own lands, but they may have little assurance that their management goals will be achieved on the greater landscape because they cannot account for the actions of other landowners. The approach can also be used to help multiple landowners develop and evaluate collaborative management strategies to achieve common objectives. Examples include the development of conservation plans for endangered species and the control of deer populations, which may migrate across the landscape. In some cases, different parts of a landscape may support different life history stages for a species, such as breeding versus winter range. Entire landscapes must be considered to develop effective management plans. Our results suggest that habitats supporting some species are maintained by a subset of owners using cutting practices that typically provide such habitat, while habitats for other species are maintained by owners with different practices. It would be unreasonable to expect single land owners to support all species when the entire landscape does in fact support them. If the diversity of ownership objectives was less in our study area, there would be less diversity of habitat and biodiversity would be lowered. Encouraging complimentary contributions by various ownerships can be an effective approach for species conservation planning (e.g., Kurttila and Pukkala 2003), and may produce superior results to thinking only in terms of ''reserves.''

Conclusion

Our results challenge the conventional wisdom that uncoordinated commodity extraction activities by landowners with different objectives will lead to fragmentation, ecological simplification and an erosion of biodiversity. We found that the cumulative effects of the landowner management strategies were generally favorable for indicators of forest sustainability. Each owner provides habitat conditions that cumulatively produce a positive result. Our approach provides a tool to evaluate such cumulative effects on other landscapes owned by other types of owners (e.g., national forest, timber investment companies, conservation groups). The model itself (HARVEST) is easy enough to use that conservation or landowner groups can apply it to develop and evaluate cooperative strategies to improve landscape patterns to conserve ecosystem diversity. While our findings may be unique to this landscape, our study represents a modest but important first step in addressing the critical management issue of cumulative impacts in multiple-owner landscapes.

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