How Will the Changing Industrial Forest Landscape Affect Forest Sustainability?

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ABSTRACT

Large-scale divestiture of commercial forestlands is occurring in the United States. Furthermore, increasing demand for cellulose for bioenergy may modify forest management practices widely enough to impact the spatial characteristics of forested landscapes. We used the HARVEST timber harvest simulator to investigate the potential consequences of divestiture and increased harvest from existing stands for bioenergy on landscape indicators of sustainability in a working landscape in upper Michigan. Divestiture tended to reduce the amount of older forests, increased fragmentation, reduced public access, and decreased the volume of wood extracted from the landscape. Increasing bioenergy production also reduced older forests, increased fragmentation of age classes, and reduced fragmentation of forest types, while increasing wood volume extracted. Our results suggest that divestiture and increased harvest for bioenergy will have negative effects on most indicators of ecological sustainability studied, although it is less clear whether these effects are ecologically significant because the slopes of the negative relationships are relatively small at the divestiture rates studied.

Keywords: divestiture, sustainable forestry, forest products industry, bioenergy, HARVEST simulation model

Human societies have long depended on forests for wood products, fuel, wildlife, water, and other ecosystem services. The specific forest benefits most prized by people have evolved over time, driven by changes in technology, lifestyle, and personal wealth. In the United States, forests have become less valued for commodities and more valued for recreation, biodiversity, ecosystem services (Kohm and Franklin 1997), and future land development (Wear and Newman 2004). The management of publicly owned forests has gradually followed this evolution of public values (Salwasser 1991). Forest products industries, particularly the paper industry, have also traditionally owned vast areas of forested land. However, the primary objective of these owners is to generate wood fiber, and although they have been willing to modify their management practices in response to changing public values, the resulting decline in efficiency is a major threat to profitability (Weigand and Haynes 1991, Loehle et al. 2002, Zobrist and Lippke 2007). Furthermore, tax policies are less favorable toward paper company forest owners (C-corporations) than toward other corporate owners (Yin et al. 2000).

The combination of public pressure to adopt less efficient management practices and taxation policies and increased competition in a global marketplace is resulting in large-scale divestiture of forestland by the forest products industry in the United States (Mehmood and Zhang 2001). Many of these lands are being purchased by timber investment groups, but some are sold to private citizens. In many cases these lands will continue to be managed for forest products, although the specific products may change and may soon also include wood for consumer energy supplies. However, the new owners are less tied to forest products and are likely to sell parcels when that produces a greater return on investment than keeping them (Gobster and Rickenbach 2004). The ultimate use of lands sold to private citizens varies by owner. Some parcels may be converted to a nonforest use, but the most common use in many regions is for recreation, specifically hunting (Craig Albright, personal communication, Mar. 13, 2007). This

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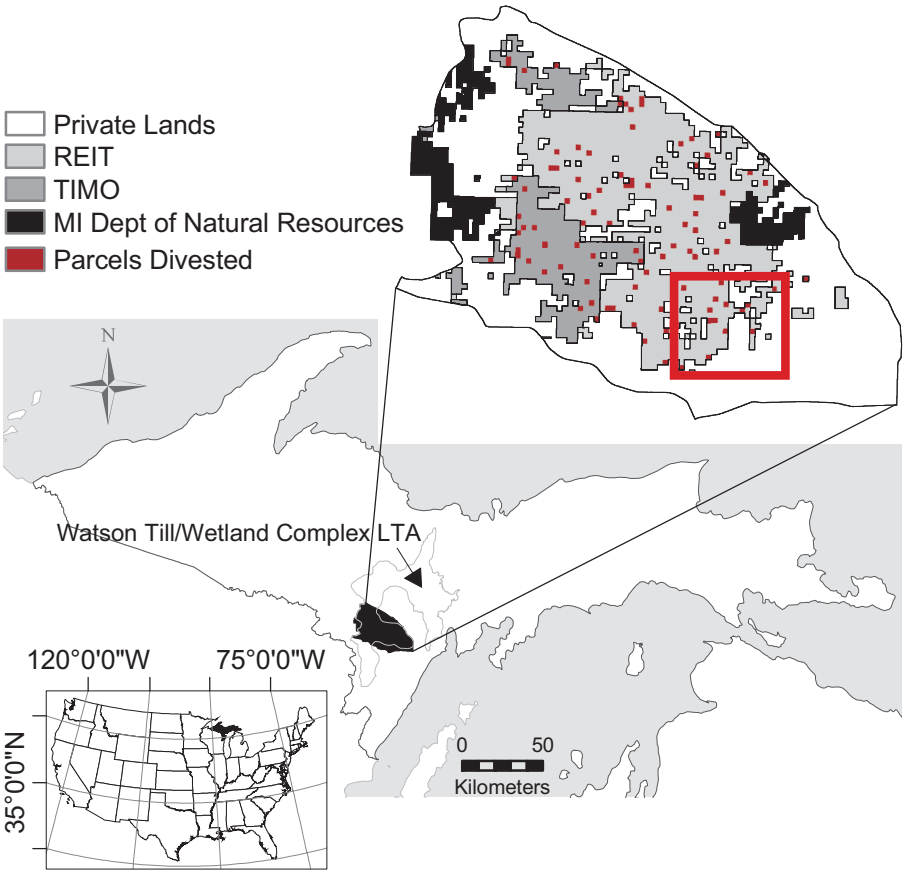

Figure 1. Location of the study area and the ownership pattern. Pattern of simulated divested parcels shows the 5% divested, replicate case 2. The red square shows the location of the maps in Figure 2.

land-use conversion does not always preclude timber harvest, but in the upper Midwest it almost always includes the development of a hunting camp for use by friends and family of the owner (Craig Albright, personal communication, Mar. 13, 2007). This development results in permanent openings and buildings that may fragment the forest such that it is less suitable for species that avoid forest edge habitat (Saunders et al. 1991).

A primary goal of most forest policies is to ensure that the use of forests is sustainable. Although the definition of sustainability is subject to debate (Gatto 1995), at its core is the idea that the use of natural resources by the current generation should not preclude their use by future generations. Sustainable forest management involves the extraction of forest products while maintaining ecosystem integrity to conserve biodiversity and to maintain other ecosystem services to society (Polasky et al. 2005). 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). The practice of ecosystem management has become widely adopted because it is based on the simple notion that functioning, healthy ecosystems will sustain commodity and noncommodity benefits as a natural consequence (Grumbine 1994).

The total area of land available for new hunting camp development, the reduction in land accessible to the public, and changes in timber supply are directly related to the divestiture activities of industrial owners. However, the equally important effects of divestiture on the spatial pattern and ecological functioning of forested landscapes are more difficult to predict. Similarly, the area of land devoted to bioenergy production in the future can be estimated based on various scenarios of technological advances and market forces. For example, many states have enacted mandates requiring utilities to generate a proportion of their electricity using renewable energy, and burning wood is often the easiest source to bring online quickly (Drabick 2003). However, the effects of increased exploitation of forests for bioenergy on forest composition, age class distribution, and the spatial pattern and ecological functioning of the forest mosaic are not intuitively obvious. Short-rotation, intensively cultured tree crops may be favored for bioenergy in the long term, but immediate demand will likely be met by increasing harvest rates of natural forests using conventional silvicultural practices that maximize yield. Spatial models that simulate the landscape consequences of strategic forest management decisions provide a useful tool to investigate such questions (Gustafson and Crow 1999, Larson et al. 2004, Azevedo et al. 2005). Virtual experiments can be conducted to relate levels of divestiture and bioenergy production to landscape pattern and ecological sustainability.

This article describes an investigation of the potential consequences of divestiture and bioenergy production on measures of landscape pattern that are related to ecological function and sustainability. The objectives of our study were to (1) accurately simulate the forest management practices of multiple owners in a real, working forest landscape, (2) conduct virtual experiments by varying levels of divestiture in this landscape, (3) conduct virtual experiments by varying the amount of land managed to grow feedstocks for bioenergy, and (4) evaluate the effect of these experimental treatments on measures of landscape pattern. Our study adopts the assumption of the Montreal Process that trends in indicators of sustainability will reflect trends in actual ecological conditions and function (Montreal Process Working group 1999). The Montreal Process includes indicators of all aspects of sustainability, including economics, recreation, and biodiversity. This study focuses on a subset of these indicators that is specifically related to the conservation of biodiversity and/or landscape composition and pattern.

Methods

Study Area. The study area is a 170,380-ac landscape in Menominee County, Michigan (Figure 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, with cedar and other native conifers dominant in the lowlands.

Table 1. Effect of cutting by HARVEST in various forest types for the two experimental treatments.

NIPF prescription	Divestiture experiment	Bioenergy experiment
Aspen clearcut	Aspen age = 1 yr	Aspen age = 1 yr
Aspen removal	Convert to upland softwood, age unchanged	Aspen age = 1 yr
Northern hardwood (periodic harvest every 15 yr)	Reduce age by 15 yr, forest type unchanged	Reduce age by 15 yr, forest type unchanged
Upland softwood	Upland softwood age = 20 yr	Upland softwood age = 1 yr
Lowland softwood	Lowland softwood age = 20 yr	Lowland softwood age = 1 yr
Lowland conifer	Reduce age by 10 yr	Lowland conifer age = 1 yr

Table entries describe the resulting cell age and forest type when a cell (on NIPF only) was harvested. The effects on investment land (i.e., REIT and TIMO) and state land did not vary between the two experiments.

Two investment groups own large, relatively contiguous holdings that together dominate the study area (Figure 1). Each investment owner has specific management objectives that are determined by the products manufactured by the paper company from whom they acquired the land. One is a real estate investment trust (REIT) that owns 55,005 ac that have historically been managed primarily for softwoods. The other is a timber investment and management organization (TIMO) that owns 19,570 ac that are managed primarily for hardwoods. The Michigan Department of Natural Resources (MDNR) owns 11,065 ac that are managed for both timber and wildlife. Nonindustrial private forestland (NIPF) tracts total 84,740 ac and are scattered throughout the study area. Based on the consensus of foresters working in the study area and the US Forest Service Woodland Owner Survey (Brett Butler, unpublished data, Aug. 18, 2005), we estimated that 40% of NIPFs is not managed for timber (i.e., no timber is cut), and that the remainder is managed for generic timber objectives as described later. Based on hunter numbers and hunting party size from MDNR deer hunter surveys and assuming that 60% of hunters hunt from hunting camps (Craig Albright, personal communication, Mar. 13, 2007), we estimated that NIPF currently have one hunting camp per 711 ac.

Spatial Timber Harvest Simulation. We simulated the divestiture and bioenergy scenarios using the timber harvest simulator HARVEST Ver. 6.1 (Gustafson and Rasmussen 2005). 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 age classes 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 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 within complex ownership patterns.

Simulation Parameters. The current management strategies of all the major owners on this landscape reflect both best management practices (Peterson et al. 1998) and the specific objectives of each owner for forest and game commodities and forest habitat conditions. The REIT owner manages mainly for softwoods in even-aged stands and plantations, the TIMO owner manages primarily for hardwoods in uneven-aged stands, the MDNR 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 the NIPF landowners who harvest timber.

We simulated the divesture of investment owner land by (1) identifying 40-ac investment owner parcels that are candidates for divestiture (i.e., not containing desired tree species) and (2) simulating hunting camp development on a proportion of those parcels. Working with representatives of the investment owners, parcels with the potential to be divested were identified as 40-ac parcels that were both more than 50% lowland types and currently owned by the REIT or more than 50% softwood types and currently owned by the TIMO. We simulated three levels of divestiture that encompass the range expected in the study area (0, 2, and 5% of investment holdings), with divestiture and subsequent hunting camp development being completed within 20 years. Each divestiture treatment assigned a random subset of the potentially divested parcels to a unique "divested" category in the ownership map. New hunting camp development was simulated by creating a single 3- to 10-ac opening (hunting camp) within each divested parcel (Craig Albright, personal communication, Mar. 13, 2007). Because the locations of existing hunting camps are not known, we also used HARVEST (prior to conducting the experiments) to disperse 119 hunting camps on NIPF initial conditions map, which is a rate of one hunting camp per 711 ac. The undeveloped land on "hunting" parcels was managed like other NIPF, so approximately 40% of hunting parcels was unmanaged.

We simulated increased bioenergy production in two concurrent ways: (1) by manipulating the percentage of "managed" NIPF using three treatment levels (50, 70, and 90%) and (2) increasing the intensity of timber cutting in selected forest types on managed NIPF (Table 1). We assumed that management intensity on investment and public lands would be unchanged. We used the initial conditions map containing existing hunting camps, but divestiture and the establishment of new hunting camps was not included to avoid confounding this experiment.

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. In the divestiture experiment, the percentage of "managed" NIPF was 60%, as in Gustafson et al. (2007). The specific stands chosen for harvest were randomly selected by the model at runtime using the "dispersed" dispersion method of HARVEST. Timber harvest was not allowed in unmanaged NIPF. Two de-

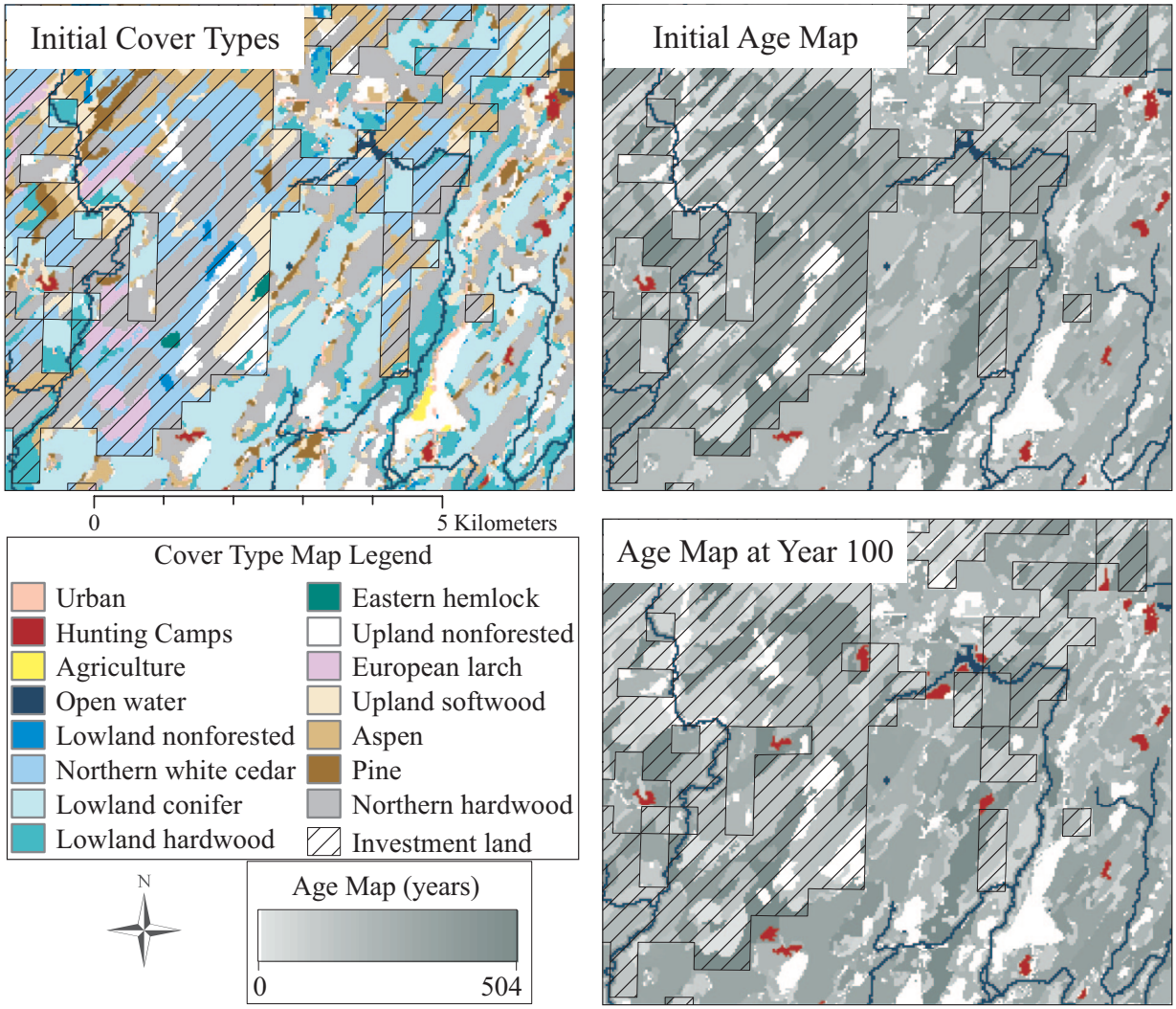

Figure 2. Forest age maps for a subset of the study area at year 0 and year 100 showing the spatial pattern of hunting camp development, simulated harvest activity, and the aging of undisturbed forest.

terministic succession processes (aspen senescence and upland softwood senescence) were simulated on all ownerships at each time step (Gustafson et al. 2007).

Initial conditions maps (stand boundaries, stand age, and forest type) were created from stand maps maintained by the investment and state owners. Because stand maps were not available for the NIPF owners, we estimated maps for these lands using a combination of a Landsat Thematic Mapper image classification created by MDNR (MDNR 2001; for forest type) and US Forest Service Forest Inventory and Analysis (FIA) data (for forest age, by forest type). To create the forest type map for NIPF, 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 eight-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 stand data from the three major landowners to create the final input maps.

Calculation and Analysis of Indicators. Our study focused on three Montreal Process indicators (Montreal Process Working group 1999) 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 examined two indicators under criterion 6 (maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of societies), namely, (6.1.a) value and volume of wood and wood products production and (6.2.a) the area of forestland managed for general recreation and tourism. Spatial 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 years, 16–30

Table 2. Regression parameters for Montreal Process indicators of sustainability regressed on the percent of investment land divested.

Response variable	d.f.	Slope	t	Prob $>\|t\|$	R^2
0- to 15-yr age class (% abundance)	17	0.022	2.70	0.0159	0.27
16- to 30-yr age class (% abundance)	17	0.08	4.84	0.0002	0.57
31- to 55-yr age class (% abundance)	17	0.22	5.05	0.0001	0.59
56- to 70-yr age class (% abundance)	17	0.30	4.80	0.0002	0.56
>70-yr age class (% abundance)	17	−0.20	−5.72	<0.0001	0.65
Uneven-aged age class (% abundance)	17	−0.50	−5.31	<0.0001	0.62
Mean age class patch size (ac)	17	−0.045	−7.66	<0.0001	0.77
Age class edge density (ft/ac)	17	0.068	9.75	<0.0001	0.85
Age class contagion (unitless)	17	−0.0006	−4.09	0.0009	0.48
Forest interior (ac)	17	−350.18	−9.72	<0.0001	0.85
Mean forest type patch size (ac)	17	−0.02	−9.84	<0.0001	0.85
Forest type edge density (ft/ac)	17	0.036	11.97	<0.0001	0.89
Forest type contagion (unitless)	17	−0.0024	−2.15	0.047	0.18
Area of recreation land (ac)	12	−748.82	−363.05	<0.0001	0.99
Patch size of recreation land (ac)	12	−94.12	−4.54	0.0008	0.62
Wood volume (ft^3/yr)	17	−10,921.5	−5.44	<0.0001	0.63

d.f., degrees of freedom.

years, 31–55 years, 56–70 years, and more than 70 years [independent of type]) and an uneven-aged class consisting of all northern hardwood, aspen, or hemlock cells with an age of more than 70 years, and all upland softwood cells more than 60 years of age. These types tend to develop an uneven age structure by the age of 70 years, 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 [relative tendency of pixels to be adjacent to a pixel of the same class], area of forest interior habitat [forest more than 150 m from an opening {cut within 20 years} or nonforest edge]), and forest edge habitat (all noninterior forest). Volume of wood produced under each treatment was estimated by combining total number of acres harvested by each owner and forest type (over 100 years) with yield information, using the methods of Gustafson and Loehle (2006). The yield tables contained the cubic foot volume of all merchantable trees (by forest type) in 10-year age classes based on the best available inventory data (from adjacent Wisconsin). Recreation indicators were total area and mean patch size of land open to public recreational use (i.e., investment and MDNR owners). For each indicator, we regressed the mean (across 100 years of simulated time) of the indicator against the treatment variable (percent of investment land divested or percent NIPF managed for bioenergy). The significance of effects was determined conservatively using $\alpha = 0.01$.

Results

Divestiture. A total of 39 and 94 hunting camps were established under the % and 5% divestiture rates, respectively (e.g., Figure 2). Divestiture of investment land tended to increase the amount of intermediate seral stages at the expense of late seral and uneven-aged forests (Table 2). Although this result is statistically significant, it may not be biologically significant because the slopes of the relationships are quite small (Figure 3a). Divestiture also tended to fragment the forest when mapped by age classes. As the percentage of divested investment land increased, mean age class patch size, contagion, and the amount of forest interior decreased, while edge density increased (Table 2). For forest type classes, divestiture did not have a significant effect on the relative abundance of forest types, excepting a slight decrease (slope more than −0.01%) in pine and northern hardwood (not shown). These common upland types were the most likely types to be converted to hunting camps. Similar to the results for age classes, divestiture increased edge density and decreased patch size of forest types (Table 2). Divestiture reduced the wood volume extracted from the landscape (Figure 3b). Increasing the percent of investment land divested directly reduced both the amount of land open to public access and the average size of such tracts (Table 2; Figure 3c).

Bioenergy. Increasing the percentage of the land base dedicated to production of bioenergy feedstocks increased the amount of younger seral stages at the expense of late seral and uneven-aged forests (Table 3; Figure 4a). Bioenergy production increased fragmentation of age classes by all measures (patch size, edge density, contagion, and interior; Table 3; Figure 4b). Conversely, fragmentation of forest types tended to decrease with increasing bioenergy production (Table 3). Again, although most of these results are statistically significant, they may not be biologically significant because the slopes of the relationships are small. Increased bioenergy production was assumed to have no effect on public access, because ownership did not change in this experiment. Wood volume extracted from the landscape increased as the percentage of the land base dedicated to production of bioenergy increased (Table 3; Figure 4b).

Discussion

Based on similar ownership and management patterns, Gustafson et al. (2007) concluded that the Menominee County landscape is probably being managed sustainably in terms of indicators of biodiversity. This conclusion was based on the assumption that native species are adapted to presettlement disturbance regimes (Swanson et al. 1994, Moore et al. 1999) and that sustainability in heavily modified landscapes is more likely when trends in Montreal Process indicators are moving toward presettlement conditions rather than away from them, as is the case in Menominee County. Gustafson et al. (2007) used General Land Office notes collected by surveyors in the early to mid 1800s (Comer et al. 1995) and assumed that these data are a valid representation of the environment to which native species are adapted. Using these assumptions, the results of the current study suggest that divestiture of investment land will have negative effects on indicators of ecological sustainability (i.e., moving them further away from presettlement conditions). For example, although divestiture tends to produce a more even age class distribution, which is desirable from a silvicultural perspective, this represents a greater departure from the distributions found in the presettlement landscapes in this region (Comer et al. 1995.) Although it is not clear how much departure will trigger a negative ecosystem response that results in ecosystem dysfunction, it is clear that divestiture moves ecological indicators away from sustainability rather than toward it. Similarly, increased roundwood harvest for bioenergy will move indicators away from presettlement values.

Although our results are statistically sig-

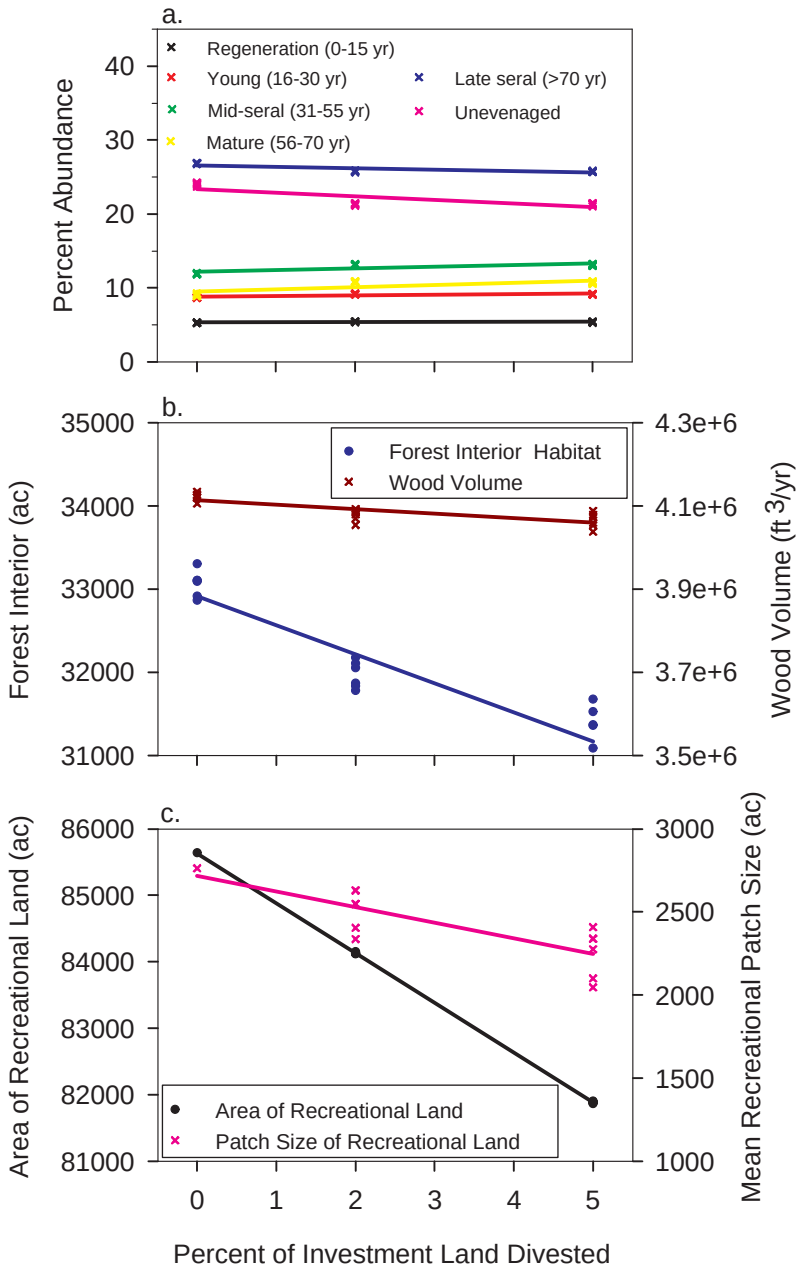

Figure 3. Relationship between the percent of investment land divested and (a) the percent abundance of age classes, (b) the area of interior forest and volume of wood harvested, and (c) land open to public recreational use.

nificant (i.e., trends are not zero), are they ecologically significant? For example, increasing the proportion of NIPF dedicated to bioenergy production from 50 to 90% on this landscape would be expected to reduce the abundance of late seral forests by only 1.9%, the average size of age class patches by 1.7 ac and the amount of forest interior habitat by 1,968 ac on this 170,380-ac landscape. This modest reduction is related to an abundance of lowland forests that are not expected to be heavily harvested even for bioenergy. Similarly, the divestiture of 5% of investment land for hunting camps would be expected to reduce the abundance of late seral forests by 1.0%, the average size of age class patches by 0.25 ac, and the amount of forest interior habitat by 1,750 ac. Would these reductions in habitat be sufficient to jeopardize the viability of populations of species that depend on them? Would ecosystem integrity and function be compromised? Our results can not provide these answers without detailed population and metapopulation viability analyses, but they do suggest that the impacts of divestiture and bioenergy production should not be dismissed lightly. The cumulative erosion of specific habitats across the working landscapes of the country may be quite significant.

Divestiture may reduce public access to forested land for recreation, which is an important indicator under criterion 6 of the Montreal Process. Our methods assumed that divested land will be purchased exclusively by hunters and therefore be closed to public access. Although some divested lands in other regions may be purchased by owners who will maintain public access, this is believed to be unlikely in this landscape. Recreational activities such as hunting require relatively large contiguous blocks of land. Our results show that not only does the amount of accessible land decrease, but the average size of blocks of such land also declines markedly (Table 2). These results also have social justice implications in that divestiture diminishes the hunting experience of those without the means to own hunting land. The less affluent hunter has less land open for hunting and must conduct hunts within more confined areas.

Divestiture and bioenergy production have opposite effects on wood supply. For example, divesting 5% of investment land reduces wood volume extracted from the landscape by 54,608 ft^3/year, while increasing the NIPF devoted to bioenergy production to 90% increases wood volume by 579,280 ft^3/year. However, wood harvested for bioenergy is not available for wood products. Our simulations assumed that wood for bioenergy production will come exclusively from wood harvest above and beyond that used by the forest products industry. It is not unreasonable to believe that geopolitical and market forces may soon dramatically increase the demand for cellulose to produce liquid fuel (Smeets and Faaij 2007), which will result in an increased price for wood. This would almost certainly reduce the wood available to the traditional forest products industry and increase its price. Such a possibility may change the economics behind divestiture decisions.

Given the negative landscape ecological effects of divestiture and bioenergy production, and assuming these effects are biologically significant, do our results suggest mitigation strategies? The effects on age class distribution were directly related to the amount of land on which landownership changed. Therefore, mitigation would involve reducing the amount of land divested either through market or

Table 3. Regression parameters for Montreal Process indicators of sustainability regressed on the percent of NIPF where timber cutting occurs (bioenergy).

Response variable	Slope	t	Prob $>\|t\|$	R^2
0- to 15-yr age class (% abundance)	0.031	33.47	<0.0001	0.98
16- to 30-yr age class (% abundance)	0.026	34.34	<0.0001	0.99
31- to 55-yr age class (% abundance)	0.016	14.77	<0.0001	0.93
56- to 70-yr age class (% abundance)	0.007	2.50	0.024	0.24
>70-yr age class (% abundance)	−0.048	−27.23	<0.0001	0.98
Uneven-aged age class (% abundance)	−0.031	−12.33	<0.0001	0.90
Mean age class patch size (ac)	−0.042	−33.94	<0.0001	0.98
Age class edge density (ft/ac)	0.021	49.06	<0.0001	0.99
Age class contagion (unitless)	−0.0002	−26.60	<0.0001	0.98
Forest interior (ac)	−49.19	−12.44	<0.0001	0.90
Mean forest type patch size (ac)	0.0012	9.05	<0.0001	0.83
Forest type edge density (ft/ac)	−0.0099	−4.94	<0.0001	0.58
Forest type contagion (unitless)	0.000008	0.43	0.675	0.00
Wood volume (ft^3/yr)	14,482.0	26.82	<0.0001	0.98

Degrees of freedom for all analyses = 17.

policy incentives. Current divestiture trends are the result of interacting domestic regulatory and tax policies, international trade balances, and corporate investment decisions (Yin et al. 2000). Turning the divestiture tide would require significant political and social will to change the policies that have driven paper companies to divest their forested land holdings and put many forest ecosystems at risk. Fragmentation effects can be mitigated by applying the results of Gustafson (2007), which in this case would be primarily by clustering intensive uses. The recreational losses associated with divestiture could be mitigated by clustering divested parcels. This would help mitigate the decrease in accessible tract size. Our results do not suggest a way to mitigate the negative effects of bioenergy production other than setting aside some currently harvested land to provide older age class forests. The most likely option for this is on publicly owned land, where the political process may determine that old forests are a more important societal benefit than wood products when the private sector is providing an adequate supply of wood. Other options that do not require setting aside land might include the purchase of "working forest" conservation easements by nongovernmental organizations that could moderate the harvest intensity on land managed for bioenergy or adopting sustainable woody biomass harvest guidelines (e.g., Minnesota Forest Resources Council 2007).

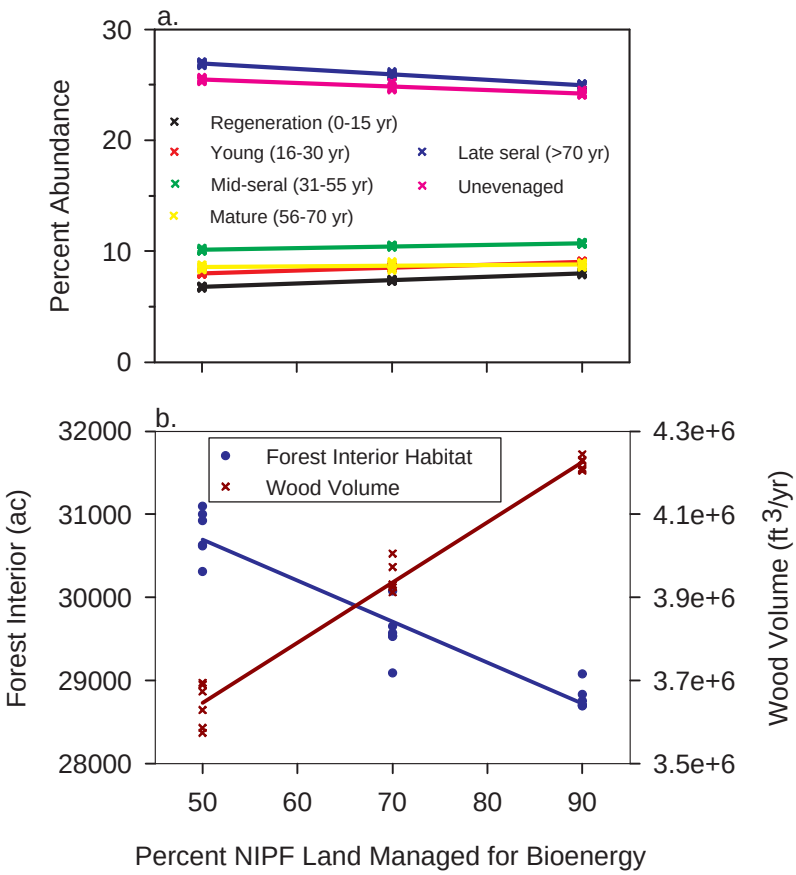

Figure 4. Relationship between the percent of NIPF managed for bioenergy and (a) the percent abundance of age classes, (b) the area of interior forest, and the volume of wood harvested.

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