

# Simulating Landscape-Level Effects of Constraints to Public Forest Regeneration Harvests due to Adjacent Residential Development in Northern Wisconsin

Brendan C. Ward, David J. Mladenoff, and Robert M. Scheller

**Abstract:** Residential development in rural forests is currently one of the most prominent land use changes in the United States. Such development can have important effects on ecosystem processes and landscape patterns. It can also constrain forest management on public lands, influencing tree species composition and age structure at the landscape scale. We used the forest landscape simulation model LANDIS-II to analyze changes to the landscape patterns of tree species composition, age structure, windthrow disturbance, and aboveground live biomass across a range of constraints to public forest timber harvests on a northern Wisconsin landscape. Our results demonstrate that decreasing harvest area through the use of buffers reduces the dominance of early-successional species, increases the dominance of mid to later-successional species, and shifts the stand age distribution within the landscape toward older forests. These buffers increase the spatial heterogeneity of dominant species and age structure at the landscape scale, and lead to more windthrow disturbance. We believe that our study can inform policymakers, forest managers, and scientists about potential long-term landscape-level effects of interaction between residential development and forest management. *FOR. SCI.* 51(6):616–632.

**Key Words:** Landscape ecology, landscape modeling, landscape composition, landscape structure.

RESIDENTIAL DEVELOPMENT in previously rural forests has become one of the most prominent land use changes in the United States (Turner et al. 1998, Dale et al. 2000). According to Sampson and DeCoster (2000), roughly 2.26 million acres of forestland within the United States were converted to developed land uses each year during the 1990s. Another assessment estimates that approximately 17 million acres of forests in the United States will be converted to developed land uses over the next 50 years (Haynes 2002), emphasizing the key role that development will play in forested landscape dynamics in the coming years (Marcin 1993). The increase in rural residential development is especially prominent in landscapes with high natural amenity values, such as the lakes district of northern Wisconsin (Klase and Guries 1999, Hammer et al. 2004). In particular, this area had considerable increases in housing unit density during the 1970s and 1990s, with indications for high growth potential in the future (Hammer et al. 2004). Because rural residential development in the upper Midwest is related to aesthetic and recreational qualities in the environment (McGranahan 1999, Rissman and Daniels 2002), it is likely to occur in ecologically sensitive areas, such as riparian forests (Schnaiberg et al. 2002, Brown 2003, Walsh et al. 2003).

Direct effects of development include the creation of

semi-permanent forest openings and replacement of the natural understory and midstory with human modified vegetation communities such as lawns, hedges, and non-native trees. In contrast, many other natural forest processes (e.g., fire) and anthropogenic processes (e.g., timber harvest) create canopy openings that experience regrowth of the native vegetation within a few years. Residential development can also result in fragmentation and loss of habitat, which can affect biodiversity, species persistence, dispersal, and population size (Friesen et al. 1995, Theobald et al. 1997, Mancke and Gaven 2000). Fragmentation and small-scale loss of habitat, when integrated over ecosystems and landscapes, can make an area unsuitable for habitation by regionally important species such as the gray wolf (*Canis lupus*) (Mladenoff et al. 1995, Mladenoff and Sickley 1998) and neotropical migrant songbirds (Keyser 2002). Indirect effects of rural residential development include changes in forest composition and structure that arise from altered fire, windthrow, and timber harvest regimes.

Several studies have analyzed the relationships among rural residential development, land use, and land cover in forested landscapes. For example, Kline et al. (2003) found that timber-producing forest area was likely to decline with increasing dispersed residential development, potentially leading to increased forest cover because areas are not

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harvested (Radeloff et al. 2001, Brown 2003). In contrast, Schneider and Pontius (2001) found that development resulted in deforestation detectable at the watershed scale. Interaction between development and landscape processes may also result in long-term changes in landscape species composition and age structure. In particular, active forest management for resource production is a key component of the regional economy of northern Wisconsin (Marcouiller and Mace 1999, Watkins et al. 1999, Rissman and Daniels 2002) and increased residential development may lead to a decrease in forest production (Barlow et al. 1998, Wear et al. 1999, Munn et al. 2002) with important consequences for future landscape patterns. Studies have noted an increase in the intensity and frequency of conflicts between developed land uses and forest management (Sirmon et al. 1993, Edwards and Bliss 2003). Some have suggested that new residents in these landscapes lack knowledge of the role of forest management in rural areas (Martus et al. 1995, Egan and Luloff 2000), view forest management as a threat to the aesthetic quality of their surroundings (Cubbage 1995, Green et al. 1996), and may be more capable of restricting forest management through legal and political channels (Malmsheimer and Floyd 1998).

Residential development and the increase of new residents in rural forests can influence adjacent public forest management through several interrelated mechanisms. First, landscape context becomes increasingly important in determining the viability of forest management decisions. Development frequently occurs concurrently with ownership fragmentation and decreasing private forest parcel size (Sampson and DeCoster 2000; Mehmood and Zhang 2001; Gobster and Rickenbach 2004). Increasingly within northern Wisconsin, corporate industrial forest owners are subdividing and selling forests (Gobster and Rickenbach 2004), perhaps as the economic return of forest resource production is surpassed by conversion to developed land uses (A. Rissman, Wisconsin Department of Natural Resources, personal communication Nov. 4, 2003). Large private nonindustrial forests are also sold to generate revenue or following the death or retirement of aging landowners (Birch et al. 1998, Mehmood and Zhang 2001). Due to these trends, public forest managers must cope with the increasing number and diversity of adjacent landowners who often have disparate views on how neighboring forests should be managed (Marcin 1993, Martus et al. 1995, Shelby et al. 2004). Indeed, factors such as negative public attitudes toward timber harvest and increasing demand for recreation opportunities can restrict the range of management options available to public forest managers (Marcin 1993, Barlow et al. 1998, Radeloff et al. 2001). Thus forest managers may be required to adapt the spatio-temporal dynamics of timber harvest to account for the influence of nearby landowners.

Second, adjacent residential development can influence timber harvest on public lands through political and regulatory processes. Public forests frequently incorporate public involvement during the development of management plans, this involvement can be politically influential if sufficient neighboring landowners are invested in the process

(Sirmon et al. 1993, Shelby et al. 2004). For example, local community associations can exert political pressure on public forest managers to limit or alter harvests in areas perceived as aesthetically sensitive. The introduction of new regulations to forest management can incorporate provisions for neighboring development, such as stipulations that timber harvest be permitted only beyond specified distances from developed areas (Cubbage 1995, Martus et al. 1995, Malmsheimer and Floyd 1998). These restrictions on harvest behavior can result in important adjacency effects at the landscape scale, as forest area is removed from production due to the land use status of neighboring parcels (Munn et al. 2002).

Studies have demonstrated that variation in management regimes can generate markedly different landscape patterns over time (Franklin and Forman 1987, Li et al. 1993, Kittredge et al. 2003). These patterns include variation in tree species composition, age structure, patch shape, and patch size across the landscape. In simulation experiments, landscape patterns of tree age structure were found to be persistent over long periods of time and proved to be quite difficult to alter to achieve different management objectives at some later date (Wallin et al. 1994). This persistence is due in part to long rotation periods (e.g., 120 years) and the negative financial consequences resulting from major changes to management objectives before the end of a given rotation. Therefore, constraints on forest management due to adjacent residential development could produce important long-term changes to landscape patterns and structure (Barlow et al. 1998, Munn et al. 2002).

In regions where timber harvest is a key disturbance agent, constraints on harvest could lead to changes in tree age structure and species composition within the landscape. As forest management occurs or fails to occur on a given site, the successional pathway will be altered as a function of the species and age cohorts present. Over time, this altered pathway could result in the replacement of certain species by others, such as the transition from early-successional species on intensely managed sites to late-successional species in the absence of intensive management practices (Pastor and Mladenoff 1994). Furthermore, the tree species diversity and age structure complexity of these stands could increase following the decline of the early-successional dominants as subsequent forests differentiate according to other ecological constraints, such as soil nutrient and water characteristics (Pastor and Mladenoff 1994). Changes to forest management could also lead to important changes in the diversity and patterns of understory plant species (Metzger and Schultz 1984, Scheller and Mladenoff 2002) and changes in habitat quality for vertebrate species including wolves (Mladenoff et al. 1995), deer (Johnson et al. 1995), and birds (Keyser 2002). Furthermore, these changes may favor certain species and disfavor others. Because of these interacting processes, the alteration of timber harvest dynamics within intensively managed landscapes could result in important long-term landscape-level effects.

## Objective

Our primary goal was to analyze the long-term ecological effects of hypothetical constraints to public forest management due to adjacent residential development in northern Wisconsin. In particular, we wished to explore trends in landscape patterns of forest tree species composition, age structure, and disturbance processes that arise from different levels of spatial constraints to regeneration harvests on public forests. We used a forest landscape simulation model, LANDIS-II (Scheller and Mladenoff 2004), to simulate forest processes based on scenarios with varying constraints to regeneration harvests. Simulation modeling is a particularly useful tool for analyzing the future relationships between human land use and environmental dynamics at the ecosystem and landscape levels (Turner et al. 1998, Pijanowski et al. 2000). Furthermore, scenario modeling holds great potential for analyzing a range of possible effects from a system of interacting processes when the future outcomes are unknown (Peterson et al. 2003).

## Study Assumptions

We used a portion of northern Wisconsin as our study landscape, including its public land boundaries, forest cover types, and prevailing harvesting techniques for various forest types. We made several assumptions regarding constraints on management within our study that represent plausible outcomes of residential development in forested regions. We first assumed that even-aged regeneration harvests (e.g., clearcuts) will be constrained on public forests by adjacent residential development because of the highly visible nature of such methods compared to uneven-aged, selection, and partial-cutting management techniques. In many cases, residents perceive these harvests as a destruction of the nontimber amenity values of the forest, such as aesthetics, presence of mature trees, or wildlife species associated with closed-canopy forests (Cubbage 1995, Edwards and Bliss 2003). Residents are thus more likely to oppose these techniques on nearby forests through legal or political channels (Barlow et al. 1998; Radeloff et al. 2001).

We assumed that regeneration harvests are constrained by no-harvest buffers within stand boundaries. This assumption relies on the premise that the spatial extent of timber harvest on public land can be restricted by adjacent development, such as accommodations for individual landowners or through more generalized requirements in forest management plans and regulations. Buffers are frequently used by managers when planning the spatial characteristics of timber harvests in aesthetically sensitive areas (Wisconsin Department of Natural Resources 2004a) or when required by legislation (Martus et al. 1995). While forest managers may implement harvests on such buffers to maintain aesthetic quality and desired species composition (Wisconsin Department of Natural Resources 2004a), we assumed that no harvesting occurs within these aesthetic buffers to isolate the potential effects of their use at the landscape scale.

We assumed that current dominant tree species data derived from stand-level forest inventories would drive the harvest activities over the course of the simulations. In particular, we assumed that a primary goal of forest management is to maintain these dominant species in perpetuity using simulated harvest types and re-entry periods that reflect actual harvest practices. We did not account for changes in management priorities and the spatial locations of target vegetation communities that have been recommended (Rissman 2003) but not yet implemented on adjacent public forests. We further assumed that other major changes to policy objectives or regulations would not occur over the course of the simulations.

We also assumed that the spatial influence of residential development within the landscape can be approximated using housing unit density. Previous research suggests that the likelihood of timber harvest on a given stand can be expressed as a negative function of residential development (Barlow et al. 1998, Wear et al. 1999, Munn et al. 2002). For landscapes with large proportions of seasonal home ownership, housing unit density can serve as a more complete indicator of human settlement than population estimates (Hammer et al. 2004). We further assumed that the spatial location and extent of constraints in the landscape will not vary over time. Although it is clearly true that housing unit density and human influence on public forest management will evolve over time in the real world, it is outside of our scope to be able to predict such changes within our simulations.

Our assumptions signify that we do not presume that our simulations predict real-world outcomes. However, they do allow us to isolate and focus attention on the effects of otherwise highly coupled landscape interactions. They also allow us to explore landscape-level trends when the precise relationships between rural residential development and public forest management are unknown.

## Methods

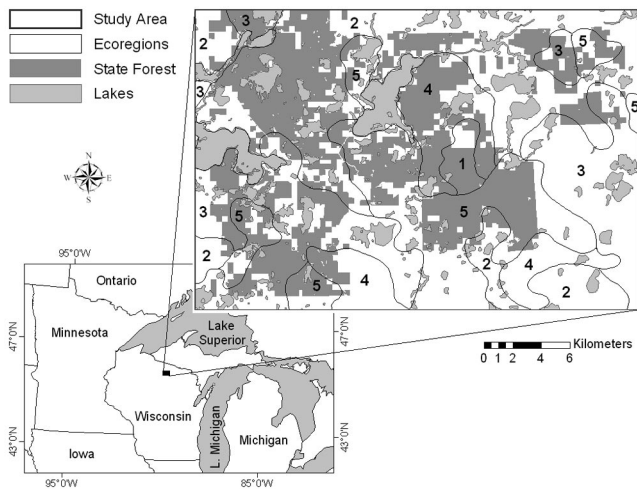
### Study Area

Our study landscape was derived from land ownership and forest cover data in north-central Wisconsin (Figure 1) and defined using a 1-km buffer around the southern unit of the Northern Highland–American Legion State Forest in Oneida County. The study area is approximately 651 km<sup>2</sup> in size, with roughly 64% of the area in upland forest types, 13% in lakes, and 23% in lowland vegetation types. This landscape was chosen for our study because of a high level of interface between public and private ownerships, relatively high rates of residential development in recent years, and predominant use of even-aged regeneration harvests on the state forest. The boundary of the public forest simulated within our study is derived from the forest stands that are owned and managed by the state forest.

### Ecological Context

The climate of northern Wisconsin can be characterized as humid-continental, with large ranges of temperature and





**Figure 1.** Location of northern Wisconsin study area. Ecoregions were defined by homogenous soil types (percentage of study area in parentheses): Ecoregion 1 has very poorly drained organic soils (<1%), Ecoregion 2 has excessively drained sands (24%), Ecoregion 3 has hilly topography with well drained loamy sands (18%), Ecoregion 4 has level topography with well drained loamy sands (7%), and Ecoregion 5 has hilly topography with well drained sandy loams (15%).

moderate levels of precipitation. Average temperatures for Jan. and July are  $-10^{\circ}\text{C}$  and  $21^{\circ}\text{C}$ , respectively (National Oceanic and Atmospheric Administration. Comparative Climatic Data Publication: Climatological Normals 1971–2000, La Crosse, WI. <ftp://ftp1.ncdc.noaa.gov/pub/data/ccd-data/NRMAVG.DAT>. accessed July 7, 2003). The region receives roughly 80 cm of precipitation per year, much of which falls during the summer months. The surficial geology was largely shaped by the glacial advances and retreats of the recent Wisconsin glacialiation, ending about 10,000 years before present (Hole 1976). The study area falls within Subsection 212Xb of the USFS National Hierarchical Framework of Ecological Units (Cleland et al. 1997, Wisconsin Department of Natural Resources 1999), otherwise known as the Northern Highlands Pitted Outwash. The outwash plains are predominantly sandy and contain numerous lakes and ponds, nearly all of which formed in depressions in the outwash following the melting of buried ice (Eckstein et al. 2001).

In northern Wisconsin, the forest species composition and age structure of a given site are determined predominantly by disturbance history, soil texture, nutrient availability, topography, and hydrologic characteristics (Curtis 1959). Common upland tree species include aspen (*Populus tremuloides* Michx.), *P. grandidentata* Michx.), paper birch (*Betula papyrifera* Marsh.), red maple (*Acer rubrum* L.), and red oak (*Quercus rubra* L.). Jack pine (*Pinus banksiana* Lamb.), red pine (*P. resinosa* Ait.), white pine (*P. strobus* L.), and scrub oak (*Q. ellipsoidalis* E.J. Hill) are important on outwash sands within the study landscape. Tree species more important on the moraines in the study landscape include sugar maple (*A. saccharum* Marsh.), eastern hemlock (*Tsuga canadensis* Carr.), yellow birch (*B. alleghaniensis* Britton), balsam fir (*Abies balsamea* Mill.), white spruce (*Picea glauca* Voss), northern white cedar (*Thuja*

*occidentalis* L.), green ash (*Fraxinus pennsylvanica* Marsh.), white ash (*F. americana* L.), basswood (*Tilia americana* L.), and black cherry (*Prunus serotina* Ehrh.).

During extensive harvests of old pine and hemlock–northern hardwood forest communities of the late nineteenth to the early twentieth century, most of northern Wisconsin was cleared and more was burned (Stearns and Likens 2002). Harvest and subsequent fire are responsible for the increased proportion of aspen relative to historical levels, aspen acreage peaked in the late 20th century with corresponding dramatic decreases in proportions of red and white pine on the outwash sands (Mladenoff and Pastor 1993). Contemporary forests are predominantly second or third growth, established following the early logging. Today, timber harvest is the predominant forest disturbance, with low levels of disturbance by windthrow and wildfire.

## Forest Landscape Simulation Model Model Background

We chose LANDIS-II (Scheller and Mladenoff 2004) as the spatially explicit forest landscape model for this study. This model was designed to operate on large spatial extents ( $10^4$ – $10^6$  ha) and relatively long temporal scales (10– $10^2$  years), using a 10-year time step. Each grid cell tracks the presence or absence of 10-year cohorts of each species, and is embedded within an ecoregion that is assumed to have homogenous soils and climate. LANDIS-II simulates cohort establishment, growth, and mortality within the landscape through spatially dynamic forest processes such as seed dispersal and disturbance (Mladenoff and He 1999, Scheller and Mladenoff 2004). LANDIS, and the biomass-driven version, LANDIS-II (Scheller and Mladenoff 2004), are flexible forest landscape models that have been used to study the effects of timber harvest (Gustafson et al. 2000, Radeloff, V.C., et al. Modeling forest harvesting effects on landscape pattern in the Northwest Wisconsin Pine Barrens. Unpublished.), fire regimes (Mladenoff and He 1999, Scheller et al. 2005, Yang et al. 2004), and climate change (Scheller, R.M., and D.J. Mladenoff. A spatially dynamic simulation of the effects of climate change, harvesting, and wind on tree species migration, forest composition, and biomass in northern Wisconsin, manuscript in preparation). Scheller and Mladenoff (2004) present sensitivity tests of the LANDIS-II biomass calculations. We used the Harvest module to simulate timber harvests within the study landscape (Gustafson et al. 2000). We made refinements to the seed dispersal probability distribution used by LANDIS-II to more accurately represent long-distance dispersal. In particular, we implemented dispersal probability with two negative exponential probability distributions parameterized by the effective and maximum dispersal distances of a species (see Ward (2004) for rationale and description).

## Input Data Preparation

LANDIS-II requires a suite of input maps and parameters to perform simulations. All input maps were prepared

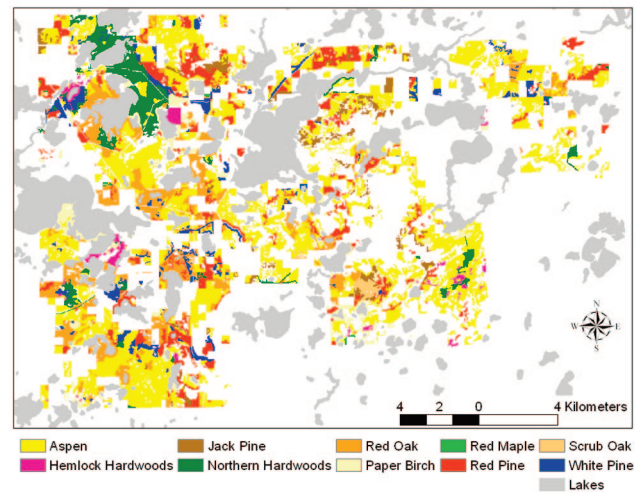
using a resolution of 25 m. Tree species life history attributes including longevity, age at sexual maturity, effective and maximum seed dispersal distances, shade tolerance, fire tolerance, and ability to sprout were derived from previous applications of LANDIS and LANDIS-II in northern Wisconsin (Mladenoff and He 1999, Scheller and Mladenoff 2004) and the literature (Burns and Honkala 1990, Iverson et al. 1999, Wisconsin Department of Natural Resources 2004a).

## Ecoregions

LANDIS-II requires delineation of fairly homogenous ecoregions within the landscape to model differences in seedling establishment probabilities and growth rates. Ecoregion boundaries were delineated using State Soil Geographic (STATSGO) database for Wisconsin soil polygons (US Department of Agriculture, Natural Resources Conservation Service (USDA-NCRS) 1994). (Figure 1). Tree species seedling establishment probabilities were determined for each ecoregion using the LINKAGES gap model (Post and Pastor 1996) as modified by Scheller et al. (2005), STATSGO soil data, and 30-year mean climatic data (1960–1990, ZedX 1995) using established methods (Scheller et al. 2005). We estimated maximum annual aboveground net primary productivity for each species in each ecoregion using PnET-II (Aber et al. 1995) and species-specific parameters obtained from the literature (Reich et al. 1995, Bolster et al. 1996, Walters and Reich 1996, Reich et al. 1999, Smith and Martin 2001, Green et al. 2003) using existing procedures (see above Scheller and Mladenoff, 2004). We identified lakes, wetlands, lowland forests, and urban areas using a combination of classified Landsat satellite imagery (Wisconsin Department of Natural Resources 1998, Dymond et al. 2002) and forest management reconnaissance data (Wisconsin Department of Natural Resources 2004b). These areas were excluded from model processing because LANDIS-II does not simulate ecological dynamics on these land types.

## Management Areas

Management areas are used to define specific harvest actions within LANDIS-II (Gustafson et al. 2000). Within our study area, we defined management areas by forest



**Figure 2.** Management areas on simulated public forest in northern Wisconsin study area.

cover type (e.g., aspen, northern hardwoods, etc.) from stand dominance data contained within the state forest stand reconnaissance database (Wisconsin Department of Natural Resources 2004b) (Figure 2). Due to the lack of clear species management goals on a stand-by-stand basis within the database and the dynamic nature of long-term management goals on public forests, we chose to define our harvest actions for the continued dominance of species groups on their respective management areas. Although we simulated forest dynamics on all upland forest sites in the study area, we only simulated harvest activity on the 162 square kilometers contained within our management areas, which constitute roughly 25% of the study area. No harvest activity was specified outside the public forest.

We defined harvest prescriptions for each management area by harvest type, re-entry period, stand-ranking algorithm, and a cohort removal mask for each forest cover type (Gustafson et al. 2000) (Table 1). A cohort removal mask defines the age cohorts of each species that are to be removed in a given harvest event. We derived stand re-entry periods from the average of those used on the state forest (P. Schultz, Northern Highland–American Legion State Forest, personal communication Apr. 14, 2004, Wisconsin Department of Natural Resources 2004b). Harvest types were derived from those commonly used in management of the

**Table 1.** Harvest type and frequency by forest cover type

Management area	Forest cover type	Harvest type	Re-entry period	Percentage of managed areas
1	Aspen	Clearcut	50	54
2	Hemlock hardwoods	Selection	20	2
3	Jack pine	Clearcut	60	3
4	Northern hardwoods	Selection	20	6
5	Red oak	Shelterwood	90	12
6	Paper birch	Shelterwood	70	6
7	Red maple	Clearcut	60	<1
8	Red pine	Clearcut	120	12
9	Scrub oak	Clearcut	50	<1
10	White pine	Shelterwood	120	5

species groups, such as clearcutting for aspen and selection cuts for northern hardwoods. Age ranking was used for aspen and jack pine stands to harvest the oldest stands first, assuming that harvesting all stands as close to their rotation length will be a management priority for these species. An economic ranking function (Gustafson et al. 2000) was used for the remaining management areas to define the order in which stands were harvested, with the intent of holding species dominance constant over time rather than accurately representing the economic priorities that are set by forest managers in reality. Each harvest type uses a different cohort removal mask; clearcutting removes all cohorts of all species on a stand, and selection harvests remove certain age cohorts of certain species. Shelterwood harvests were conducted for red oak, paper birch, and white pine management areas by simulating two harvests 10 years apart. In the first harvest, select old and young cohorts of the target species were retained while all cohorts of other species were removed. In the second harvest, some or all of the retained cohorts were removed as appropriate for the target species. The cohort removal mask provides only a rough approximation of selection and shelterwood techniques used in actual forest management.

We modeled tree-planting for red and jack pine on their respective management areas because this is a common silvicultural practice for ensuring successful regeneration in northern Wisconsin (Wisconsin Department of Natural Resources 2004a). The planting algorithm restricted natural regeneration of competing species for 10 years after planting to approximate the combined effects of site preparation, herbicide application, and postplanting competition control. We also specified follow-up treatments for planted areas to remove competing species; one at 20 years after the initial harvest, then every 30 years until the next complete harvest of the stand.

### ***Forest Cover Map***

For initialization, LANDIS-II requires a list of the age cohorts of each species present on each site in the landscape. We combined a recent, multidecade phenological classification of Landsat imagery (Dymond et al. 2002) with Forest Inventory and Analysis (FIA) data from 1996 and State Forest Stand Reconnaissance (RECON) data from 2004 using a probabilistic assignment process similar to He et al. (1998). The FIA database is a widely used landscape-level measurement of forest conditions (Miles et al. 2001) that provided us with a sample of forest parameters (stand age, tree species, dbh, and stem density) based on inventory plots located in each Land Type Association (Wisconsin Department of Natural Resources 1999). Land Type Associations were used as the spatial unit for deriving our forest parameters from the FIA database because the location of inventory plots with respect to our ecoregions was unknown. The RECON database is a periodically updated listing of management-oriented forest conditions on most stands in the state forest (Wisconsin Department of Natural Resources

2004b). It provided us with information on each stand's tree species composition, age, and planned management actions.

The mapping of tree species age cohorts involved the formation of tree species composition clusters and probabilistic assignment of these clusters to grid cells in the classified Landsat imagery. We derived tree species composition clusters by statistically clustering species' importance values (Schulte et al. 2002) at the FIA plot level, using the FASTCLUS procedure (SAS Institute, Inc., 1990) with Ward's minimum variance method and 64 cluster seeds. Importance value is the average of relative basal area and relative number of trees per hectare. Dominant species and their 10-year age classes were determined for each FIA plot within each cluster primarily using maximum importance value. Ages were approximated for secondary species within each FIA plot in each cluster using a linear scaling of their dbh against the dbh and age of the dominant species. These ages were limited to 100 years based on the assumption that all cohorts originated following the extensive harvests and fires during the early 20th century.

Tree species composition clusters were assigned to the Dymond et al. (2002) land cover classes using dominant species importance value as a proxy for percentage canopy coverage within each class. Each grid cell in each forest land cover class of the classified Landsat imagery was assigned a tree species composition cluster by probabilistically sampling from the clusters assigned to that class at a rate proportional to the number of their constituent FIA plots within each Land Type Association (Wisconsin Department of Natural Resources 1999). Thus the classification approximated the frequency of inventoried forest cover types within each Land Type Association. For grid cells that occurred on management areas, we used the RECON data to increase the spatial precision of the assignment process by preferentially sampling from a subset of the composition clusters that were similar in composition and age to that measured on a given stand. The RECON data were also used instead of the Dymond et al. (2002) classification for land cover classes that were ambiguous with respect to forest types (e.g., clearings) and that fell within the boundaries of young forested stands (e.g., recently harvested aspen stands). When no RECON data were available for these ambiguous land cover classes, pixels were assumed to be regenerating aspen clearcuts due to their size and the prevalence of this species on upland sites within our study area. Using this classification process, species composition and age structure were mapped on every upland forest grid cell on the landscape, with greater compositional and age precision on stands managed by the state forest.

### ***Landscape Disturbance Processes***

We simulated windthrow disturbance to provide insight about possible interactions between spatial constraints to timber harvest and landscape disturbance processes. We derived minimum, maximum, and mean windthrow patch size estimates from historical analysis of northern Wisconsin disturbance regimes (Schulte and Mladenoff 2005).



Windthrow frequency was calibrated so that a severe windthrow disturbance on a given site was repeated approximately once every 1,200 years in the absence of timber harvest, which was equivalent to historical estimates for our study area (Schulte and Mladenoff 2005). Windthrow parameters were held constant across each scenario in our study so that observed windthrow disturbance rates would vary only in response to different spatial patterns of species age cohorts. We chose to exclude fire from our analysis based on the assumption that fire suppression is almost completely effective within this landscape.

**Residential Development**

We estimated the spatial distribution and intensity of residential development using housing unit densities derived from the US Census of Population and Housing in 2000 (US Census Bureau. 2002. Census 2000 TIGER/Line files for Wisconsin) at the census block level. We corrected these densities by removing state forest lands from census blocks and recalculated densities over the remaining areas.

**Experimental Design**

We simulated forest landscape dynamics in our study area based on 11 different scenarios. All scenarios were simulated for 300 years, which was chosen to permit stabilization of most major trends in our study. Nine constraint scenarios were based on a treatment matrix of three housing density levels and three harvest buffer sizes (Table 2); these constraints to timber harvest were held constant within each scenario for the duration of the simulation. We also simulated one control scenario with no constraints to harvest activity and one scenario without harvests so that the relative effects of the constraint scenarios could be measured against these extremes. Each constraint scenario was defined by a discrete buffer size around census blocks that exceeded a threshold of housing unit density. Census housing unit density thresholds were generated using an equal-area classification of census blocks that fell within 1 kilometer of the public forest. Buffer sizes were derived from the range of aesthetic buffers used by the Department of Natural Resources when planning harvests in aesthetically or ecologically sensitive areas (Wisconsin Department of Natural Resources 2004a). Regeneration harvests were not allowed where these buffers intersected public forest stands. Selection harvests were not altered in any constraint scenario. In combination, housing unit density and buffer size

generated scenarios where harvesting was constrained on 6–26% of the area within our management areas (Table 2).

This experimental design, encompassing a range of housing density thresholds and buffer sizes, was chosen because the precise relationship between residential development and constraint to adjacent harvests was unknown. Our lowest density threshold could represent the situation where housing growth is high across large areas of the landscape, which is a trend suggested by recent research (Hammer et al. 2004). Such a situation could arise in a landscape rich in natural amenities as new residents seek to maximize the undeveloped characteristics in their surroundings and minimize the presence of neighbors, thus leading to dispersed development across large areas. In contrast, the highest density threshold could represent the situation where housing growth is low in areas that are minimally developed and is high on areas that are already more developed. This situation could occur where new residents seek proximity to developed areas, such as lakes (Schnaiberg et al. 2002).

**Data Analysis**

We analyzed quantitative and spatial trends in tree species dominance, age structure, and aboveground live biomass to assess the effect of constraint to regeneration harvests on landscape composition and structure. In particular, we analyzed tree species dominance using proportional landscape dominance and angular second moment (Musick and Grover 1991). Proportional species dominance is a quantitative measure of the relative area dominated by a given species over time. The species with the maximum aboveground live biomass on a cell is considered to be the dominant species of that cell. Dominant species were aggregated into cover types (Figure 3) for the quantitative analysis, but data were retained at the species level for analysis of landscape structure. Angular second moment is a spatial measure of landscape heterogeneity among categorical variables, and ranges between 0 for maximum heterogeneity and 1 for a single dominant species (Musick and Grover 1991).

We aggregated grid cell ages into 40-year age classes for the quantitative analysis and age map. We analyzed grid cell age patterns using inverse difference moment among the maximum 10-year age cohort on each cell. Inverse difference moment is a measure of landscape heterogeneity among interval variables that ranges between 0 for maximum age heterogeneity and 1 for landscapes dominated by a single age cohort (Musick and Grover 1991). We used APACK landscape pattern analysis software (Mladenoff, D.J., and B. DeZonia. 2001. APACK: User’s guide. <http://landscape.forest.wisc.edu/projects/apack>. accessed February, 2004) with an eight pixel neighborhood to calculate all landscape spatial metrics. We averaged aboveground live biomass across all upland forest cells. We also analyzed aboveground live biomass trends for species that showed large responses in proportional dominance or were of special interest within the study area.

**Table 2. Percentage of management areas restricted from regeneration harvests in each constraint scenario**

Buffer size (m)	Census 2000 housing unit density threshold (housing units per km <sup>2</sup> )		
	2	5	20
50	9	8	6
100	17	16	12
150	26	24	19

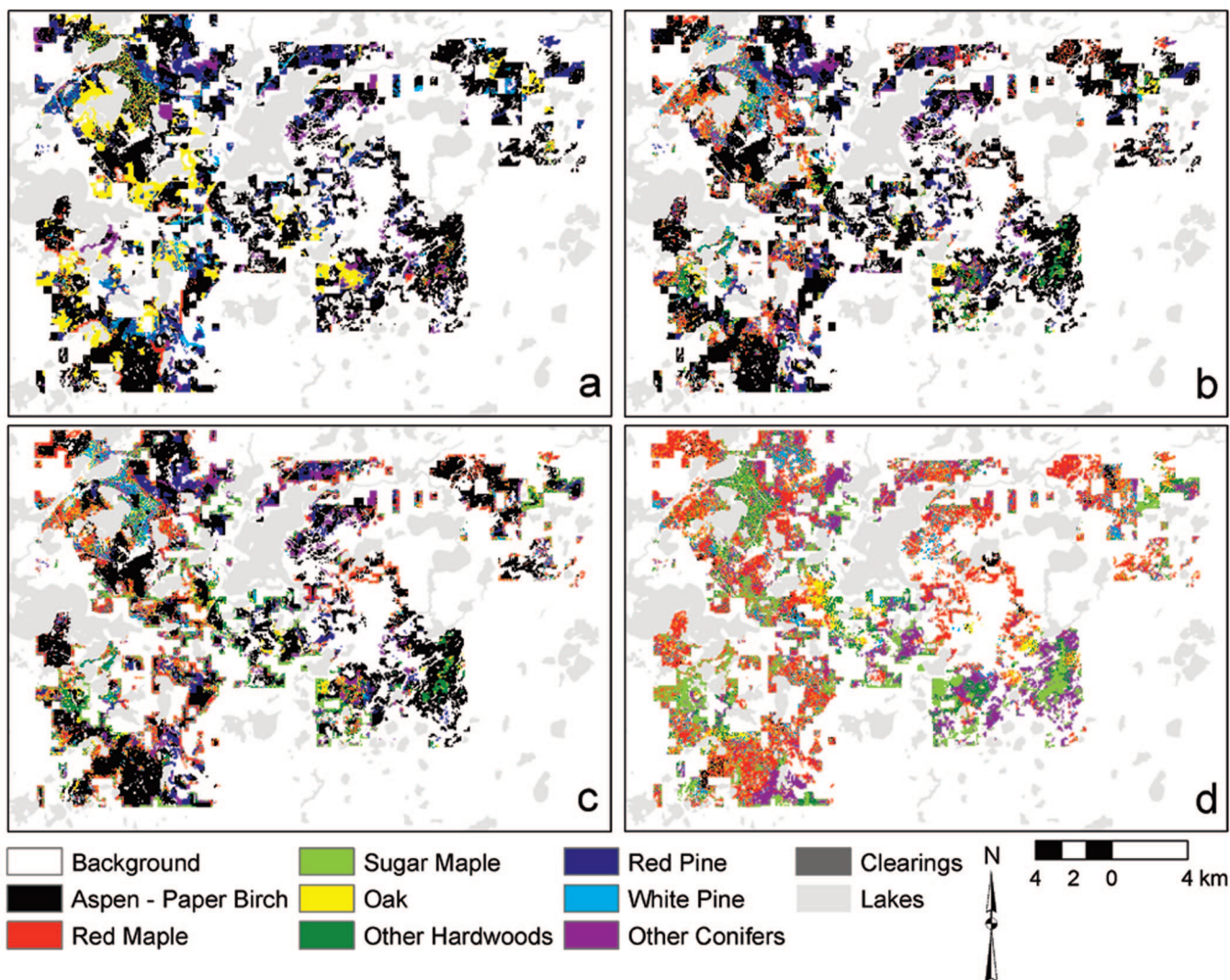


Figure 3. Landscape cover types of the simulated public forest at (a) year 0, and (b) at year 300 for control scenario without any restrictions to regeneration harvest; (c) regeneration harvest constrained within 150 m buffers around census blocks with more than two houses per km<sup>2</sup>, and (d) no harvest during simulation. Species were aggregated into cover types as follows: aspen–paper birch (*Populus tremuloides*, *P. grandidentata*, *Betula papyrifera*), oak (*Quercus rubra*, *Q. ellipsoidalis*), other hardwoods (*B. alleghaniensis*, *Fraxinus americana*, *F. pennsylvanica*, *Prunus serotina*, *Tilia americana*), and other conifers (*Abies balsamea*, *Picea glauca*, *Pinus banksiana*, *Thuja occidentalis*).

## Results

### Landscape Composition

#### Tree Species Dominance

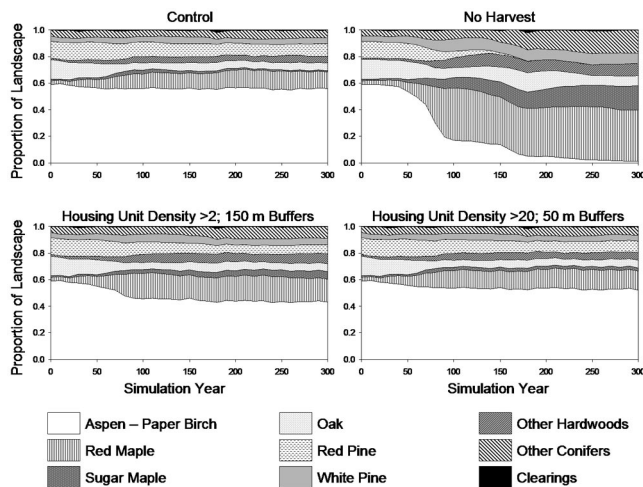
Temporal variability in tree species dominance is evident in maps of selected scenarios (Figure 3) and in trends of proportional species dominance (Figure 4). In the control scenario, red maple dominance increases until year 100, after which it remains steady at roughly 12% of the landscape. The oaks decline in dominance over time, reflecting the invasion of more shade-tolerant species including red maple, sugar maple, and white pine. There are also moderate increases in the dominance of basswood, black cherry, yellow birch, white spruce, and balsam fir. The aspen–paper birch cover type declines slightly until year 100 as paper birch is replaced by red maple and other hardwoods.

Among scenarios, mid- to late-successional species such as red maple, sugar maple, basswood, black cherry, white

spruce, and balsam fir increase proportionally in dominance with decreasing area managed using regeneration harvests (Figure 4). In contrast, early-successional species such as aspen, paper birch, jack pine, and red pine decrease with decreasing levels of regeneration harvest. Much of the increase of red maple between scenarios is within and immediately adjacent to the buffers, especially those that occurred on aspen and red pine stands (Figure 3). The most dramatic changes in species dominance occur within the first 80 years of the simulations, predominantly reflecting the replacement of short-lived species such as aspen and paper birch in the buffers (Figure 4). In contrast, the replacement of red pine occurs more gradually due to its longer lifespan.

Without timber harvest, red maple essentially replaces aspen and dominates the landscape by year 100 (Figure 4). Red maple, red oak, white pine, and balsam fir predominantly replace red and jack pine. Sugar maple dramatically increases in dominance, especially on stands previously





**Figure 4.** Trends in proportional species dominance among four scenarios of varying constraint to regeneration harvest with specified housing unit density thresholds and buffer sizes. Dominance is assigned to the species with the maximum biomass on each upland forest pixel in the public forest. Species are aggregated into cover types according to Figure 3.

dominated by hemlock, yellow birch, red maple, and red oak. The oaks exhibit a somewhat cyclic trend in dominance, showing declines between years 50 and 120 and after year 250, but dramatic increases between year 120 and 250. From maps of landscape dominance, it appears that red oak, and to a much smaller degree scrub oak, invade older aspen and jack pine stands, especially following a windthrow event (Figure 3).

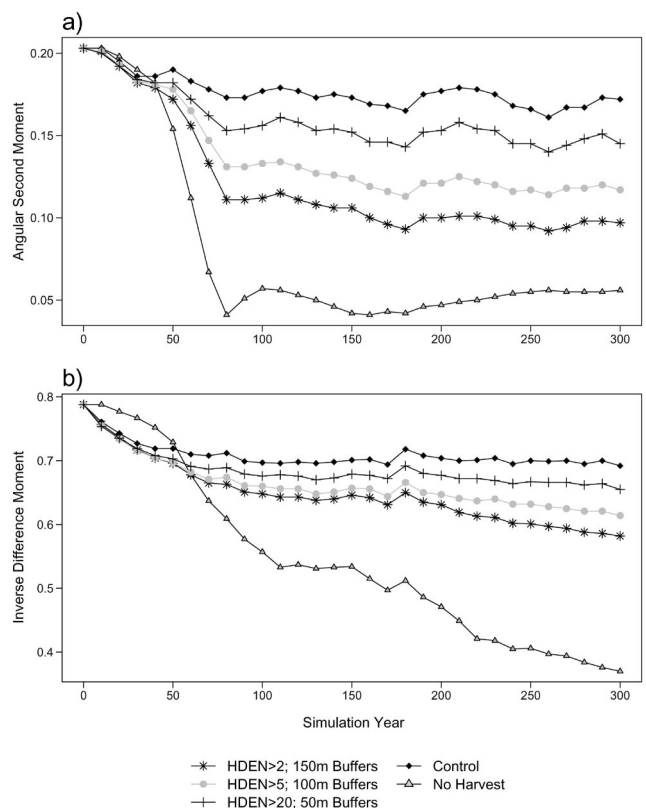
### *Dominant Species Spatial Pattern*

The angular second moment metric shows an increase in landscape heterogeneity of dominant tree species over time (Figure 5a). In all scenarios, heterogeneity increases over time in response to windthrow, timber harvest, seed dispersal, and succession. After year 50, the control scenario exhibits the least heterogeneity over the remainder of the simulation, whereas the no harvest scenario shows a dramatic increase in heterogeneity before year 80, after which the angular second moment metric stabilizes. The heterogeneity of dominant species increases with increasing constrained area, resulting in an angular second moment that is up to 44% less than the control scenario at year 300. The angular second moment of most constraint scenarios appears to stabilize after year 100, indicating that the period of greatest divergence between scenarios is between years 50 and 80. This divergence results predominantly from the different establishment opportunities for shade-tolerant species as a function of buffer size.

### *Age Structure*

#### *Tree Species Age*

The age of each upland forest grid cell shows strong spatial variability over time due to disturbance and establishment processes (Figure 6). By year 300, the control scenario is largely dominated by cohorts less than 80 years



**Figure 5.** Spatial metrics of landscape heterogeneity using (a) angular second moment of species dominance, and (b) inverse difference moment of grid cell age, among five scenarios of varying constraint to regeneration harvest with specified housing unit density (HDEN) thresholds and buffer sizes.

in age (Figure 7). However, old cohorts of long-lived species (e.g., hemlock) are rare in the landscape due to harvest, dispersal limitations due to the spatial isolation of long-lived species, and age-related mortality.

The scenarios with harvest buffers show increasing proportions of cohorts older than 80 years at the end of the simulations compared to the control scenario due to the lack of cohort removals in the buffers, as exemplified by the scenario with the greatest constraint to harvests (Figure 7). With constraint to timber harvest, cohorts greater than 160 years in age show up to a 250% increase over the control scenario, demonstrating the presence of long-lived shade-tolerant species, such as sugar maple, white spruce, and white pine in the buffers even if they do not dominate the sites (Figure 3c). Old cohorts in the buffers are also evident in landscape-level maps of grid cell age (Figure 6c).

Without harvests, the landscape undergoes an increase in grid cell age (Figure 6); the relative proportions of older age cohorts increase dramatically by year 300 due to an increase in long-lived shade-tolerant species (Figure 7). However, the spatial influence of windthrow is evident in the age map at year 300 (Figure 6d) and in the maintenance of the younger age cohorts over time (Figure 8). The simulated 150-year lifespan of red maple drives a moderate expansion of the cohorts between 80 and 140 relative to the control

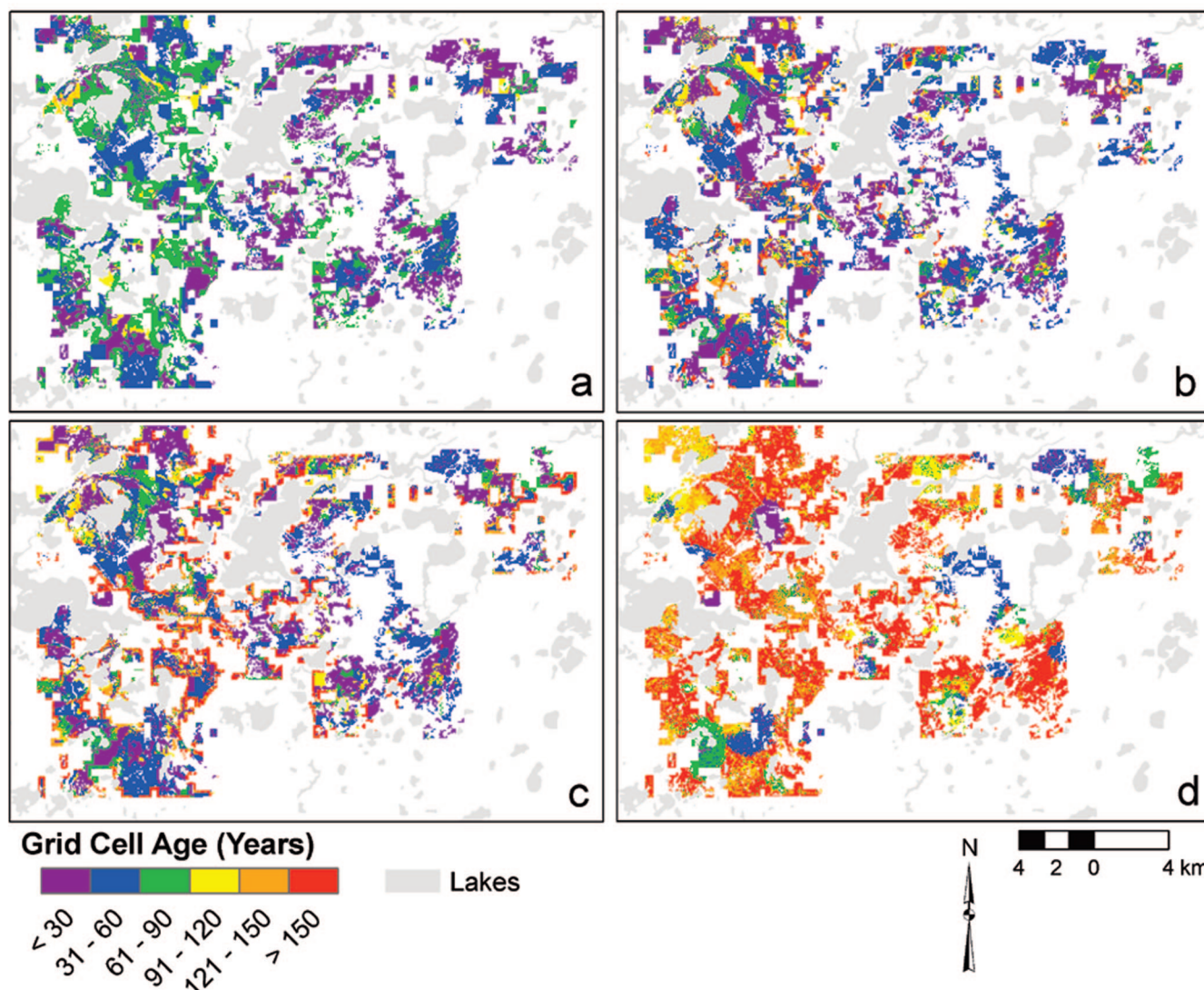


Figure 6. Landscape grid cell age of the simulated public forest at (a) year 0, and (b) at year 300 for control scenario without any restrictions to regeneration harvest, (c) regeneration harvest constrained within 150 m buffers around census blocks with more than 2 houses per km<sup>2</sup>, and (d) no harvest during the simulation.

scenario, because cohorts of this species that establish during the simulation succumb to age-related mortality near the end of this age range.

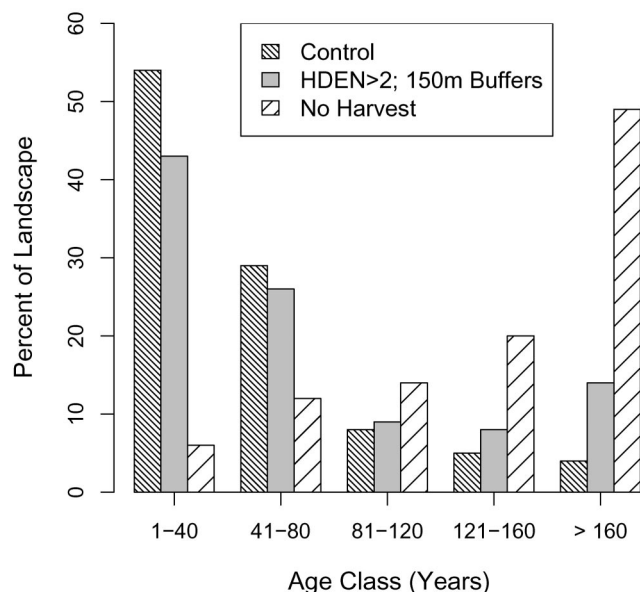
### Age Spatial Structure

Similar to species dominance, landscape age heterogeneity increases over time (Figure 5b). The heterogeneity of stand age remains fairly stable in the control scenario, as measured by the inverse difference moment metric, because harvesting and mortality restrict the range and spatial distribution of age cohorts present. The scenarios with harvest buffers show increasing heterogeneity with increasing reduction in harvest area, reflecting an increasing range of age cohorts present over time. The inverse difference moment decreases up to 16% relative to the control scenario by year 300. Without management, landscape heterogeneity of stand age increases dramatically as the landscape ages, reflecting a greatly expanding range of cohorts present due

to spatially variable processes of disturbance, mortality, and establishment.

### Aboveground Live Biomass

Spatial heterogeneity due to timber harvest and windthrow is evident in maps of aboveground live biomass (Figure 8). The scenarios with harvest buffers generally show increases in biomass within the buffers relative to the control scenario by year 300, as exemplified by the scenario with the largest constraint to harvest. Among these scenarios, landscape average aboveground live biomass diverges most strongly from the control between years 90 and 120 as older cohorts of aspen succumb to mortality, resulting in up to 15% less biomass than the control scenario. The scenarios with the smallest buffers generally show the lowest levels of average aboveground live biomass relative to the control, suggesting that increases in red maple biomass fails to offset decreases in aspen biomass in the small buffers.



**Figure 7.** Distribution of percent of landscape area by age class for control, regeneration harvest constrained within 150 m buffers around census blocks with greater than two houses per km<sup>2</sup> (HDEN), and no harvest scenarios.

The trends in aboveground biomass for selected species show interesting responses to various levels of harvest constraint. For aspen, the control scenario maintains the highest overall levels of biomass, whereas the scenarios with the largest buffers show the largest reductions in biomass (Figure 9a). Without timber harvest, aspen experiences a sharp peak at year 30 followed by an apparently exponential decline in biomass to very low levels. For red maple, the control scenario maintains the lowest levels of biomass, whereas increasing constraints on harvesting leads to increasing biomass (Figure 9b). The differences among scenarios with buffers are most evident between year 90 and year 150 because varying constraints to harvest give red maple varying levels of opportunity to invade other stands. After year 150, red maple is widely dispersed and variation in constraints to harvest does not produce markedly different opportunities for red maple establishment. In the absence of harvest, red maple biomass increases exponentially until year 100, after which it remains fairly stable at roughly 56 Mg/ha.

Red pine experiences a slight increase in biomass until year 40, probably because the initial age distribution of red pine is 30–40 years younger than the specified rotation age (Figure 10a). After this point, red pine biomass varies among scenarios. The control scenario maintains moderate overall levels of red pine biomass, whereas the least constrained harvest leads to greater biomass. Without harvest, red pine biomass decreases more slowly than aspen due to its greater longevity. White pine biomass increases with reductions in harvest area similar to red maple, but at lower average levels of biomass (Figure 10b). However, white pine increases at a much slower rate than red maple due to more restricted dispersal and establishment throughout the landscape. In the absence of harvest, white pine biomass

increases exponentially until year 120, after which it appears to increase slowly for the remainder of the simulation.

## Windthrow

LANDIS-II simulated a total of 190 km<sup>2</sup> of windthrow by year 300 in the control scenario, affecting an average of roughly 4% of the managed areas within the landscape each decade. The total amount of windthrow area increases proportionally with decreasing timber harvest in the landscape, resulting in 12–50% more windthrow over the duration of the simulation in the constraint scenarios compared to the control. Furthermore, the total amount of windthrow contained within the buffers increases with increasing constraint to regeneration harvests, ranging from 18% for the least reduction in harvest area up to 57% for the largest reduction in harvest area. While the spatial location of a windthrow event is modeled randomly, simulated windthrow damage to a given species increases with species age. Thus older forests are more likely to experience windthrow mortality. This is evident in differences in the amount of windthrow among scenarios.

## Discussion

Our results show a number of changes in landscape pattern and structure in response to decreasing harvest area, including shifts in tree species dominance toward late-successional species, increasing prevalence of older cohorts, increasing landscape heterogeneity of dominant species and age, and increasing windthrow. One key effect of unharvested buffers is a decrease in the landscape dominance and aboveground live biomass of early-successional tree species, such as aspen, and an increase in the dominance of mid and later-successional species, such as red maple. The increase in red maple dominance is consistent with its ability to act as both an early- and late-successional species across a wide range of site conditions (Tift and Fajvan 1999), whereas aspen is highly intolerant of competition (Huffman et al. 1999). Stearns and Likens (2002) noted a similar shift in dominance from aspen and paper birch to red and sugar maple over 100 years following disturbance in a forest very near the study area. However, unlike other areas in northern Wisconsin that show a strong compositional trend toward sugar maple (Eckstein et al. 2001), sugar maple is relatively restricted within our simulations due to the prevalence of poor soil types and frequent harvest disturbance, with the exception of the no harvest scenario.

Our results also demonstrate that red maple increases in dominance across the landscape in the absence of harvest buffers, suggesting that it could become increasingly important in the real landscape at current levels of management. In particular, red maple shows increasing dominance in red oak, paper birch, and white pine stands within our control scenario. This trend is consistent with its higher shade tolerance relative to other midtolerant species, such as white pine and red oak (Curtis 1959), its ability to compete well on sites with relatively poor soil quality, its capacity to



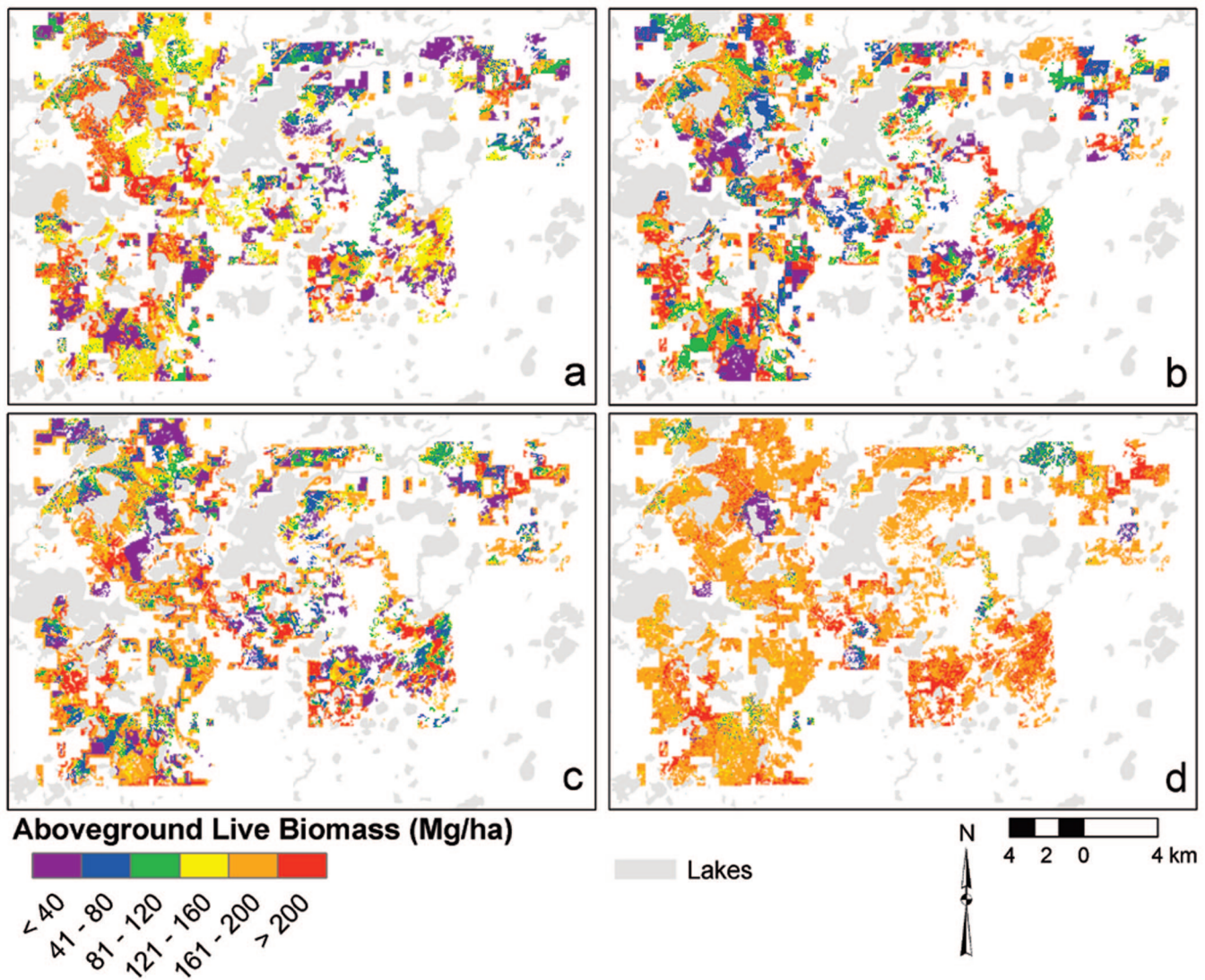


Figure 8. Average aboveground live biomass on simulated public forest at (a) year 0, and (b) at year 300 for control scenario without any restrictions to regeneration harvest, (c) regeneration harvest constrained within 150 m buffers around census blocks with more than two houses per km<sup>2</sup>, and (d) no harvest during simulation.

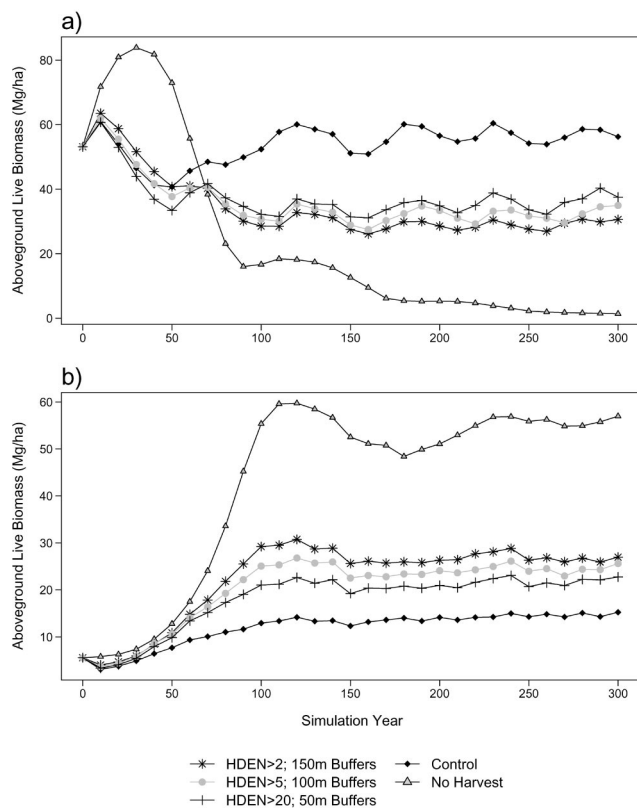
disperse over greater distances than other shade tolerant species, and its ability to sprout prolifically following disturbance (Burns and Honkala 1990). However, this trend could also reflect limitations in our simulated harvests on red oak, paper birch, and white pine management areas, particularly for complex treatments such as shelterwood.

### Ecological Implications

Unlike buffers designated primarily for ecological purposes, such as riparian buffers, the harvest buffers analyzed within our study may be limited in ecological functionality due to landscape context, species composition, and age structure. For example, the buffers create linear areas of older, shade-tolerant species around early-successional species, which could serve as dispersal corridors for vertebrate species. However, given the proximity of these buffers to developed and harvested areas, it is likely that they will be relatively low in habitat quality for many forest-dwelling

species (Mladenoff et al. 1995, Keyser 2002). Narrow buffers in proximity to human-disturbed areas can inflate populations of edge and human-associated native and exotic species, with important implications for native, forest-dependent species (Mancke and Gaven 2000). Buffers of older forest interspersed with even-aged forests could also increase populations of white-tailed deer (*Odocoileus virginianus*) with potentially detrimental effects on regeneration of historically important tree species (Alverson et al. 1988, Mladenoff and Stearns 1993, Eckstein et al. 2001).

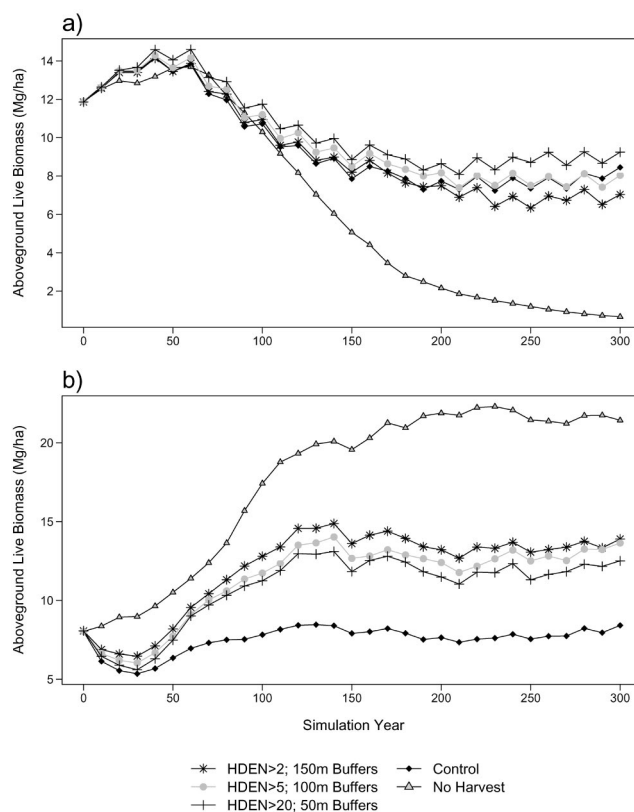
The buffers could also increase the relative proportion of old-growth forest in the landscape by allowing trees to reach biological maturity. However, the buffers may fail to achieve old-growth characteristics typical of this landscape, such as high compositional and structural heterogeneity, dominance by old sugar maple and hemlock on mesic sites, and dominance by old red and white pine on dry and dry-mesic sites (Eckstein et al. 2001). The small size and



**Figure 9.** Average aboveground live biomass for (a) aspen (*Populus tremuloides*, *P. grandidentata*), and (b) red maple among five scenarios of varying constraint to regeneration harvest with specified housing unit density (HDEN) thresholds and buffer sizes.

disturbance history of the buffers can have strong effects on the species composition and structure of both the overstory and the understory vegetation components (Metzger and Schultz 1984, Miller et al. 2002, Scheller and Mladenoff 2002), such as the dominance by relatively short-lived, shade-tolerant tree species observed in our study. The proximity of these buffers to residential development and managed areas could also inhibit the development of old-growth characteristics. The limited landscape-level dispersal and establishment of long-lived shade-tolerant tree species, such as hemlock (Mladenoff and Stearns 1993, Eckstein et al. 2001) and sugar maple, would further limit the development of old-growth forest characteristics on mesic sites within the landscape. Tree species that were more prevalent in the landscape before Euro-American settlement (Mladenoff and Pastor 1993, Eckstein et al. 2001, Schulte et al. 2002), such as white pine, could fail to achieve increased dominance in the landscape due to competition with shorter-lived, fire-intolerant, and shade-tolerant species (Stearns 1949, Stearns and Likens 2002), as indicated by the very slight expansion of white pine dominance in our constraint scenarios. Thus the buffers may make only minimal contributions toward the establishment of high-quality habitat for forest-dependent species and the restoration of old-growth characteristics in the landscape.

Our results emphasize the key role of windthrow disturbance on forest composition and structure in the absence of



**Figure 10.** Average aboveground live biomass for (a) red pine and (b) white pine among five scenarios of varying constraint to regeneration harvest with specified housing unit density (HDEN) thresholds and buffer sizes.

fire and timber harvest. In particular, windthrow increases the mortality of aging early-successional tree species, decreases aboveground live biomass, creates additional opportunities for establishment of shade-tolerant species, and increases landscape heterogeneity. These effects can have ecological ramifications, such as accelerated succession (Abrams and Scott 1989) and increased structural and spatial variability in forest composition and age (Zhang et al. 1999; Stearns and Likens 2002; Schulte and Mladenoff 2005). Furthermore, windthrow in proximity to developed areas can have important consequences for human habitations and municipal infrastructure, with potentially escalating damage costs with increasing levels of development (Changnon 2001).

### Assumptions and Possible Effects on Results

Our harvest buffers may limit ecological and aesthetic functionality to a greater degree than buffers used in the real landscape because we did not allow any harvesting within them. We did not harvest them to isolate their cumulative effects at the landscape scale, but real harvest buffers could undergo moderate levels of harvest to enhance aesthetic quality and ecological function (Wisconsin Department of Natural Resources 2004a). Harvest actions could be used to create establishment opportunities for regionally important species, such as white pine and red oak (Eckstein et al.

2001), and remove trees that are highly susceptible to windthrow. Although not addressed within this study, forest managers may also need to account for fire risk and available fuels in these buffers due to their proximity to developed areas (Sturtevant et al. 2004).

Although our study focuses on a single public forest in northern Wisconsin, the interaction between rural residential development and adjacent public land management is not a phenomenon confined to our study landscape. Higher property values and population growth in proximity to public lands are nationwide indicators of the role that public lands play in the spatial patterns of development (Klase and Guries 1999, Radeloff et al. 2001, Rasker and Hansen 2000). However, the influence of residential development on public lands and subsequent long-term landscape-level effects likely varies with respect to the type of public land, ecological and management dynamics, and socioeconomic context. For example, residential development will likely exert a different type of influence on management of US Forest Service lands in the Western United States (Graham et al. 2004) than on small, highly fragmented county forests in the Pine Barrens of northwestern Wisconsin (Radeloff et al. 2001).

Our study demonstrates that landscape-scale simulation modeling can be used to explore key issues related to the management of public forests in close proximity to development. However, more research needs to be conducted to assess the range of influences that forest residents and residential development can exert on the management of nearby public forests, perhaps involving surveys, case-studies, and analyses of legislation trends. Simulation studies could capitalize on finer-scale residential development data to yield additional insights about possible long-term landscape-level effects of constraint to timber harvests due to residential development. Additional research should also focus on the long-term impacts of private forest parcelization and residential development on landscape patterns and processes because private forest dynamics could become increasingly important in the near future (Gobster et al. 2000, Gobster and Rickenbach 2004).

## Conclusions

The primary objective of our study was to analyze the long-term, landscape-level ecological effects of constraints to public forest management that arise from adjacent residential development. While many of these constraints do not currently exist on public lands within the study landscape, we simulated a series of plausible theoretical scenarios to gain insights into potential long-term effects. Our results demonstrate that such constraints of rural residential development on public forest management are likely to influence landscape-level patterns of forest composition, age structure, aboveground live biomass, and windthrow disturbance. If this constraint is defined by no-harvest buffers around areas of a given level of housing unit density, landscape structure and processes appear most sensitive to total area restricted from harvest. We found that even small

reductions in harvest area (e.g., 6%) can increase landscape-level heterogeneity of dominant species, stand age, and alter the allocation of aboveground live biomass among tree species. However, the magnitude of such changes in the real landscape could vary as a function of landscape context, site- and species-specific growth and competition rates, timber harvest rates, and types of management conducted on harvest buffers.

Unharvested buffers around developed areas can change landscape patterns of tree species dominance by creating habitat for mid-to-later successional species interspersed with early seral species. This increases the landscape heterogeneity of dominant tree species and age structure. Restricting areas from harvest can lead to greater amounts of windthrow, especially in buffers where early successional species are approaching the limit of their lifespan. The buffers can also accumulate greater biomass for shade-tolerant species at the exclusion of shade-intolerant species. Increasing the area within buffers can increase the complexity of forest management in the landscape by reducing of the area available for harvest and therefore influence the resulting forest composition and structure.

As policymakers, forest managers, and scientists begin to address the influence of adjacent residential development on forest management, we believe careful thought should be given to the long-term effects of management decisions on landscape patterns. Specifically, we believe that management plans and forest policy should explore alternatives that are sensitive to the long-term social, economic, and ecological constraints of forested landscapes undergoing increasing levels of rural residential development. Our results suggest that the use of buffers can have important impacts on the landscape scale over long periods of time while failing to ensure increased aesthetic and ecological qualities. We therefore recommend that the spatial distribution of buffers within the landscape be carefully evaluated within the landscape to balance the concurrent goals of forest resource production, aesthetics, and the creation of old-growth and high-quality habitat for native species. The landscape modeling tools and analytical techniques used in this study can contribute important insights about trade-offs and long-term, large-scale cumulative effects.

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