Conservation of Biological Diversity

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Introduction

PEOPLE ENJOY A variety of ecosystem services, or benefits, from forests, including water purification, recreation, income from tourism, timber products, and the cultural and economic benefits from hunting, fishing, and gathering (Shvidenko et al. 2005). Across the Northern United States, growing human populations will place increased service demands on forests for the foreseeable future. The type, magnitude, and stability of future services from northern forests will depend in part on the level of biological diversity in those forests.

Biodiversity and Climate

Biological diversity, or biodiversity, is broadly defined as the variety of life, which encompasses diversity within species, diversity among species, and diversity among organisms. As defined by the Convention on Biological Diversity (2010), forest biodiversity consists of the multiplicity and genetic diversity of plants, animals, and micro-organisms that inhabit forests. Biodiversity can be an important asset by itself, for instance when an infusion of genetic material from wild stocks increases crop productivity (Myers 1997). A high level of biodiversity in forests increases resiliency and helps ensure an uninterrupted flow of services, even when resources are subjected to natural or human-caused disturbances that inevitably occur (Bengtsson et al. 2000, Tilman et al. 2006). Shifley et al. (2012) describe current contributions of northern forests to biological diversity. Because of the links among biodiversity, ecosystem health and function, and ecosystem services, assessments of the potential effects of alternative futures on forest biodiversity are a useful source of information for policy makers and natural resource managers.

Because climate affects the spatial distribution and diversity of forest ecosystems and species across the landscape, changing climate and land-use patterns can be expected to affect levels of forest biodiversity during the coming decades and centuries (Iverson and Prasad 1998, Schwartz et al. 2006). As climate changes, some forest ecosystems and forest-associated species can shift their distributions to track the conditions that favor them, and other species are able to adapt to their new environment (Parmesan 2006). But if a species is unable to move or adapt, its geographic range may shrink or it may become extirpated from portions of its former range (Parmesan 2006).

Key Findings

Present Day

- Forests cover 174 million acres in the North, or 41 percent of the total land area.
- Current tree species richness by State ranges from 49 species sampled in Rhode Island to 99 species sampled in West Virginia, with 160 tree species recorded regionwide.
- Northern forests provide habitat for more than 363 terrestrial vertebrate species, with richness varying among habitat classes and wildlife taxa; the highest level of richness is associated with birds, and with forests of open-canopy structure and hardwood dominance.
- With one exception—amphibians in closed-canopy hardwoods—most forest wildlife species also frequent one or more nonforest habitats.

Projected 2010 to 2060

- Forest area is projected to decrease slightly across the North, with the rate ranging from 3.5 to 6.4 percent and with losses concentrated around existing urban and suburban areas.
- Under all scenarios, area is projected to decrease for four forest-type groups (aspen-birch, elm-ash-cottonwood, oak-hickory, and spruce-fir) and increase for one group (maple-beech-birch); white-red-jack pine, which is projected to increase under current rates of biomass utilization, would decrease under high biomass-utilization scenarios.
- Under all scenarios, the forest area in the large diameter size class is projected to increase by 3.5 to 5.7 percent and the medium diameter size class is projected to decrease; the small diameter size class is projected to decrease under scenarios that assume no increase in biomass harvesting for energy, but increase or remain stable under high biomass utilization scenarios.
- Closed-canopy habitat classes are projected to gain acreage whereas open-canopy habitat classes are projected to lose acreage; intensive biomass harvesting rates for bioenergy are projected to result in losses for closed-canopy classes and to a lesser degree for open-canopy classes.
- Increasing fragmentation and parcellation are expected to reduce high-density core forests within large patches and the average size of family forest landholdings in every State; these changes would decrease the ecological services and socioeconomic benefits of core forests and diminish the ability to manage large patches of forest.
- Overall tree species richness is projected to decline through 2060; projected reductions in early successional habitats such as oak-hickory could have a large effect on tree species richness in such ecosystems.

Human Impacts on Forests

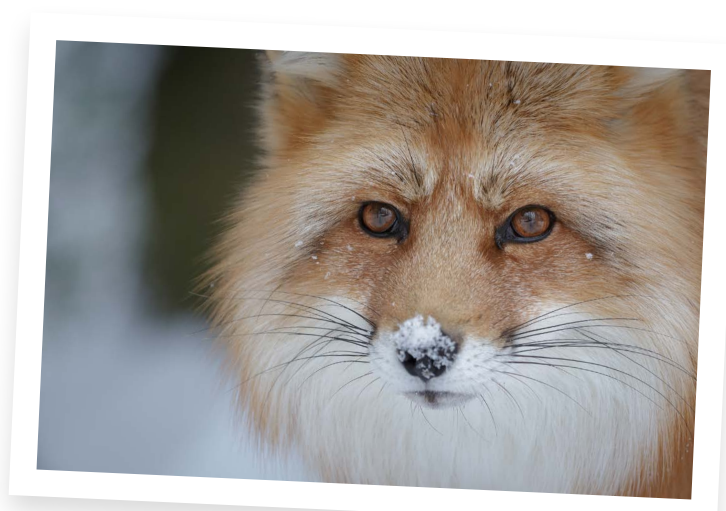
Within the large-scale patterns established by climate, land-use decisions further modify the extent and configuration of forests ecosystems, ecosystem functioning, species diversity, and genetic diversity (Opdam and Wascher 2004).

Landscape patterns, landowner objectives, and the area, composition, and structure of forests all affect the amount and quality of habitat for wildlife. Conversion to urban or other uses reduces forest area and potentially fragments forest ecosystems; this increases the proximity and exposure of remaining forests to human influences. Public and private land management decisions affect forest structure and composition. Species and genetic diversity can be reduced by forest loss, fragmentation, and urbanization, or can be increased by forest management strategies (Battles et al. 2001, Pimm and Askins 1995).

Assessments of forest biodiversity under a range of plausible climate and land-use change scenarios can help answer questions about current actions and future outcomes: How will the areas of forest ecosystem types change in the future? What are the potential consequences of climate and land-use changes for species in protected areas? What changes will occur in the area of forest habitats and how will changes affect at-risk wildlife? Answers to these questions will be valuable input for informing policy and management decisions that influence the trajectories of climate change and land-use patterns.

Modeling Change

The projections for changes in forest condition used three greenhouse gas emissions storylines developed by the Intergovernmental Panel on Climate Change (IPCC 2007, Chapter 2): A1B assumes moderate greenhouse gas emissions associated with moderate gains in population and large gains in income and energy consumption—but with a balanced renewable/fossil fuel portfolio; A2 assumes high greenhouse gas emissions associated with large gains in population, high energy consumption, and moderate gains in income; and B2 assumes low greenhouse gas emissions associated with moderate gains in population, income, and energy consumption. General circulation models (climate models) provide estimates of temperature and precipitation changes by geographic area. The scenarios discussed in this chapter refer to combinations of storylines with general circulation models (climate change models) and assumptions about future forest harvest rate—either a continuation of harvest rates observed in the recent past or greatly increased biomass harvesting for energy. See Chapter 2 for detailed descriptions of storylines, climate models, and scenarios.



PREDICTED CHANGES IN FOREST CONDITIONS AND OWNERSHIP

In 2010, forest area in the North was estimated at 174 million acres, accounting for 41 percent of total land cover. Projections of future conditions suggest a decrease from 2010 to 2060 under all three storylines (Fig. 3.1), with the largest decrease projected for A1B (-6.4 percent), the smallest decrease for B2 (-3.5 percent), and an intermediate decrease for A2 (-5.4 percent) (Wear 2011).

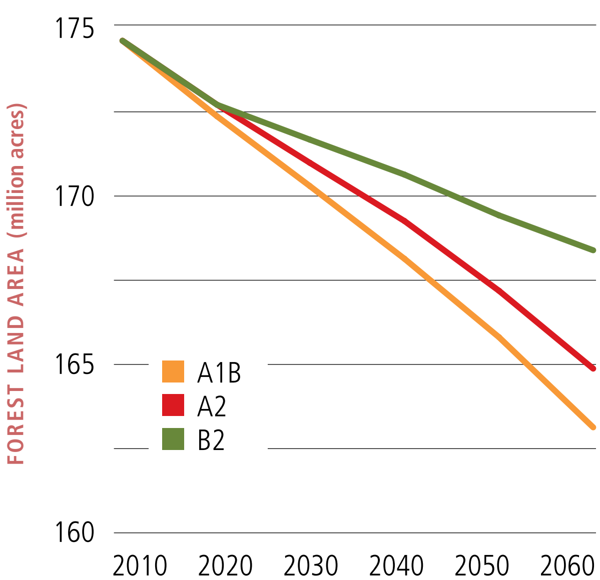


FIGURE 3.1

Forest land area estimates under baseline conditions (2010) and projections (2020 to 2060) under greenhouse gas emission storylines A1B, A2, and B2 (IPCC 2007) for the North (Wear 2011). Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income; and B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption.

Forest Composition

Under the three scenarios that predict a continuation of current harvesting levels, two major groups—maple-beech-birch (*Acer* spp.–*Fagus* spp.–*Betula* spp.) and white-red-jack pine (*Pinus strobus – P. resinosa – P. banksiana*)—are projected to increase from 2010 to 2060 and four others[1] are projected to decrease (Fig. 3.2). Under scenarios of increased harvesting to support high biomass utilization, similar trends are expected with the exception of white-red-jack pine, which is expected to decrease, albeit slightly (Fig. 3.2). The white-red-jack pine group is projected to experience a large increase in harvesting rates under the high biomass utilization scenarios, which suggests a potential explanation for its switch from increasing to decreasing forest area among the alternative scenarios, although other processes and model uncertainty may also be at play.

[1] *spruce-fir (Picea spp.—Abies spp.); oak-hickory (Quercus spp.—Carya spp.); elm-ash-cottonwood (Ulmus spp.–Fraxinus spp.–Populus spp.); and aspen-birch (Populus spp.—Betula spp.).*

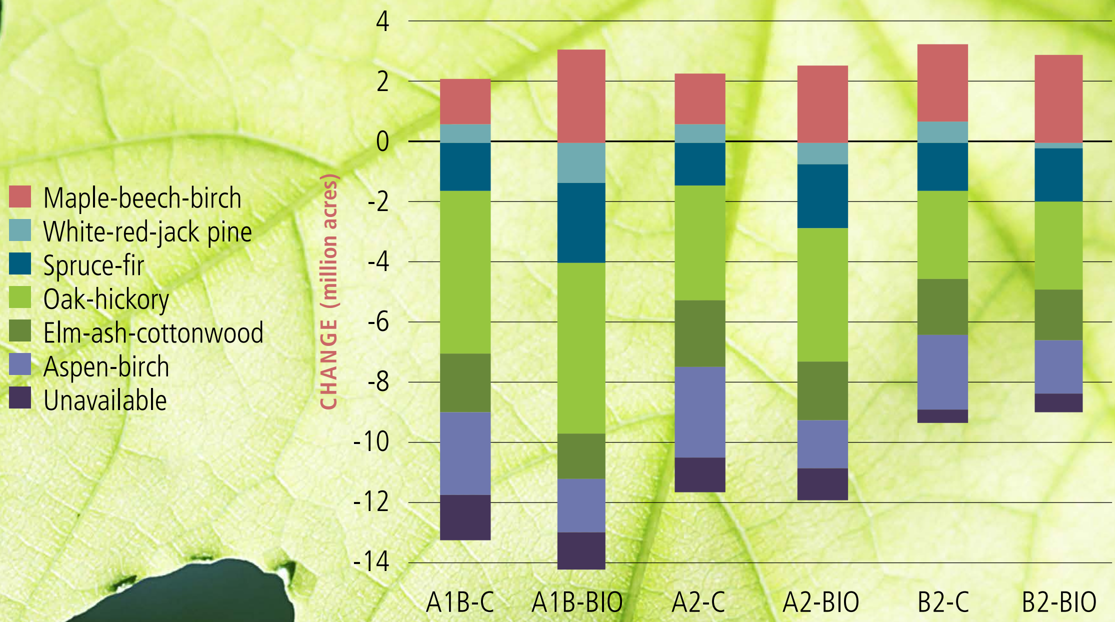


FIGURE 3.2

Projected change in northern forest area by forest-type group, 2010 to 2060, under six scenarios, each representing a global greenhouse storyline (IPCC 2007) paired with a harvest regime. Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income; and B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption. Scenario projections assume harvest will continue at recently observed levels (labeled –C) or increase to reflect increased harvest for bioenergy production (labeled –BIO).

Tree diameter provides a reliable measure of forest stand structural stage, which in turn is indicative of successional stage. Under the three scenarios that predict a continuation of current harvesting levels (A1B-C, A2-C, and B2-C), the forest area in the large diameter size class (≥11 inches d.b.h. for hardwoods and ≥9 inches d.b.h. for softwoods) is projected to increase by 3.5 to 5.7 percent from 2010 to 2060.

The forest area is projected to decrease for both the medium diameter class (≥5 inches d.b.h. and smaller than the large diameter class) and the small diameter class (<5 inches d.b.h.) (Fig. 3.3). However, under scenarios of high biomass utilization, forest area in the small-diameter size classes is projected to increase under A1B-BIO and A2-BIO, and to remain unchanged under B2-BIO.

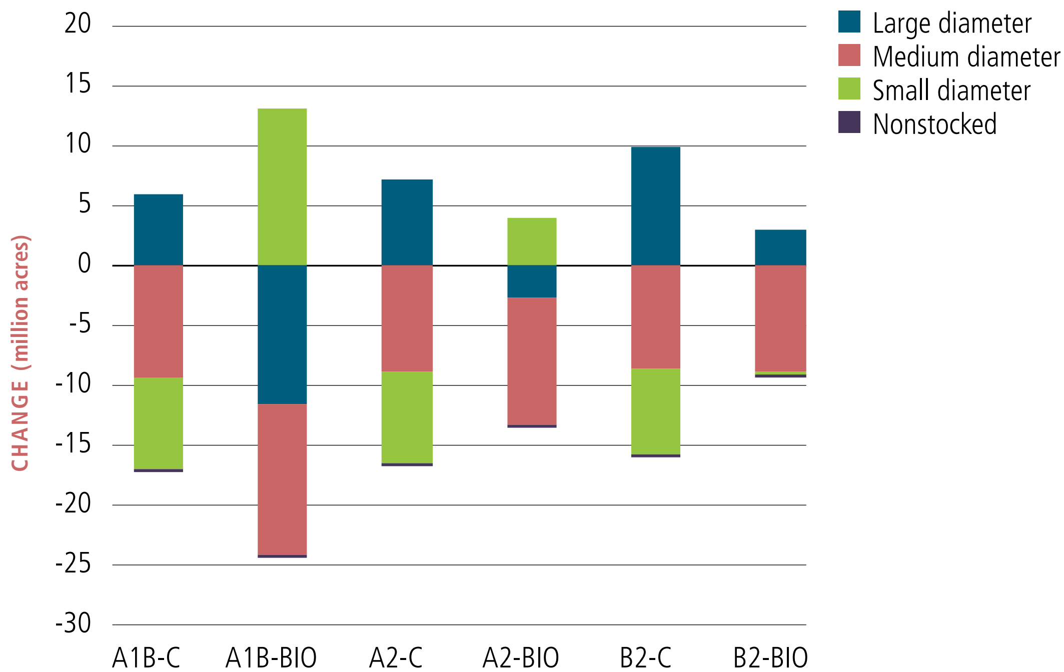


FIGURE 3.3

Projected change in northern forest area by stand-size class, 2010 to 2060, for six scenarios, each representing a global greenhouse storyline (IPCC 2007) paired with a harvest regime. Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income; and B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption. Scenario projections assume harvest will continue at recently observed levels (labeled –C) or increase to reflect increased harvest for bioenergy production (labeled –BIO). Small diameter class <5 inches d.b.h.; medium diameter class ≥5 inches d.b.h. and smaller than large diameter class; large diameter class ≥11 inches d.b.h. (hardwoods) or ≥9 inches d.b.h. (softwoods).

Although trends in forest tree species composition for the entire North are likely to be relatively consistent across scenarios, the direction and magnitude of change would vary dramatically among States, with projected increases in some States offset by decreases in others. For example, the elm-ash-cottonwood (*Ulmus* spp.–*Fraxinus* spp.–*Populus* spp.) group is projected to decrease by 12 to 18 percent from 2010 to 2060. On a per-State basis, however, this group would decrease by ≥25 percent in a number of States and actually increase by ≥25 percent in several other States within the same scenario, suggesting subregional variations (Fig. 3.4). These trends do not address additional threats from Dutch elm disease (*Ophiostoma ulmi*) nor threats from the emerald ash borer (*Agrilus planipennis*), a nonnative insect pest discussed in detail in Chapter 5.

Forest Structure

Unlike forest composition, forest structure projections vary widely among States, both in response to the greenhouse gas storylines and in response to levels of harvesting. Some States would experience increases in large or small diameter classes under continued harvesting levels, but trends would reverse under high biomass utilization scenarios. For instance, a continuation of current forest harvesting rates (scenarios A1B-C, A2-C, and B2-C) would result in modest or major reductions in forest acreage in the small diameter class, for the vast majority of States. However under high biomass utilization scenarios (A1B-BIO, A2-BIO, and B2-BIO), many of those States would experience moderate or major increases in small diameter forest area (Fig. 3.5).

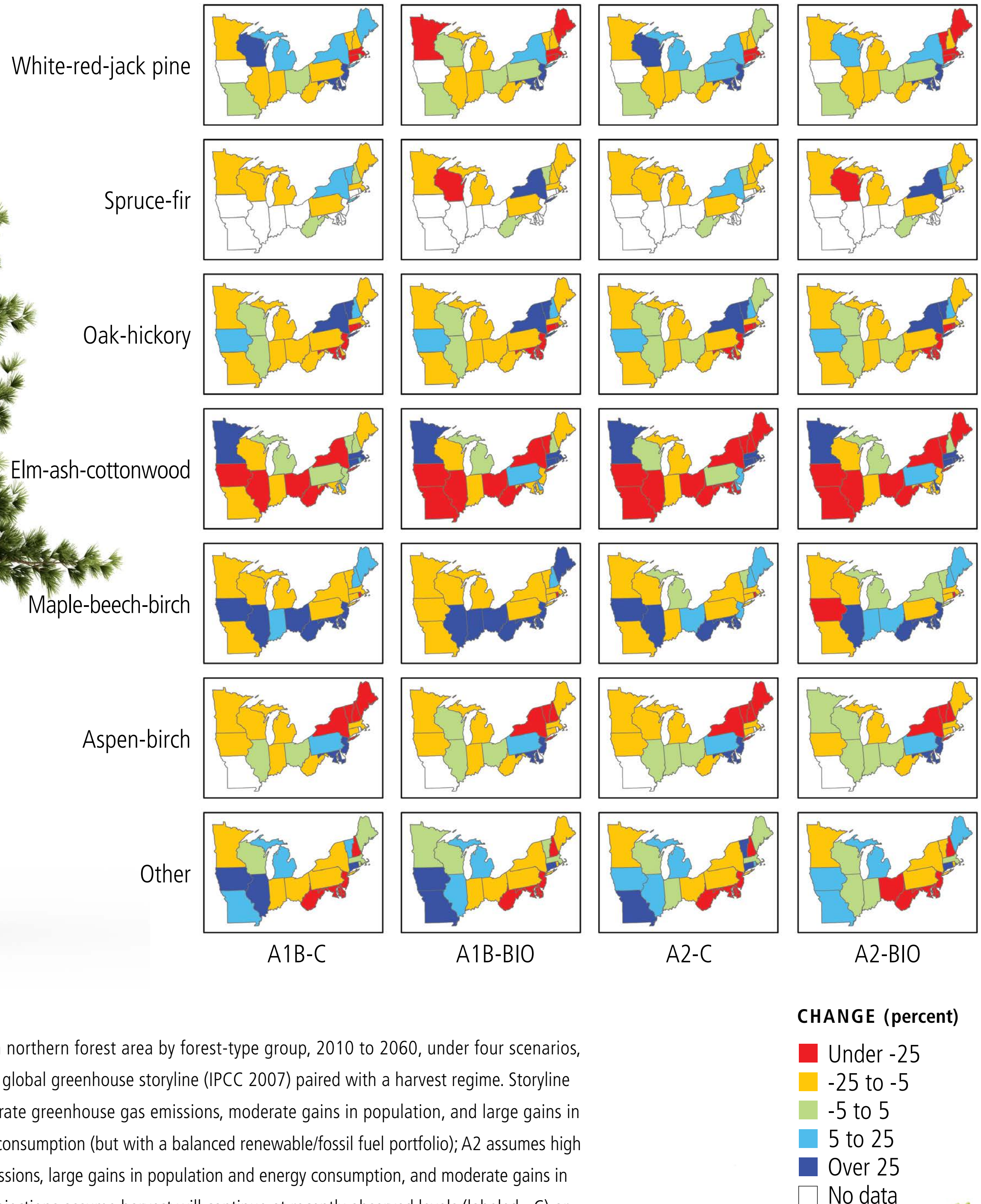


FIGURE 3.4

Projected change in northern forest area by forest-type group, 2010 to 2060, under four scenarios, each representing a global greenhouse storyline (IPCC 2007) paired with a harvest regime. Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income. Scenario projections assume harvest will continue at recently observed levels (labeled –C) or increase to reflect increased harvest for bioenergy production (labeled –BIO).

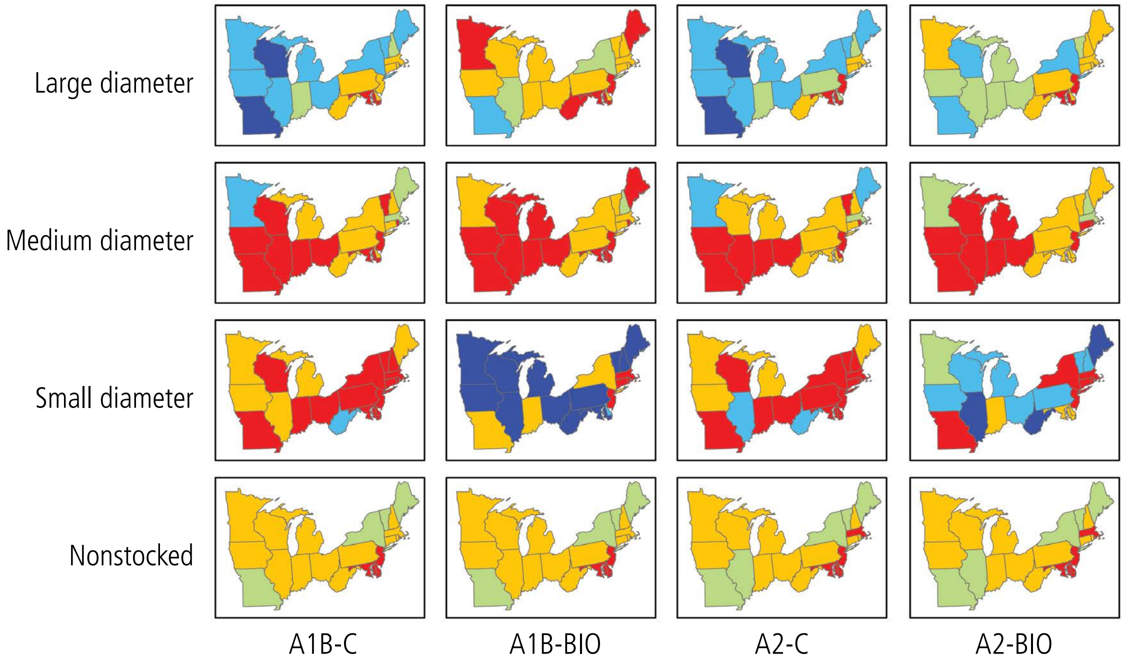


FIGURE 3.5

Projected change in northern forest area by stand-size class, 2010 to 2060, under four scenarios, each representing a global greenhouse storyline (IPCC 2007) paired with a harvest regime. Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income. Scenario projections assume harvest will continue at recently observed levels (labeled –C) or increase to reflect increased harvest for bioenergy production (labeled –BIO). Small diameter class <5 inches d.b.h.; medium diameter class ≥5 inches d.b.h. and smaller than the large diameter class; large diameter class ≥11 inches d.b.h. (hardwoods) or ≥9 inches d.b.h. (softwoods).

CHANGE (percent)

- Under -25
- -25 to -5
- -5 to 5
- 5 to 25
- Over 25
- No data

THREATS TO BIODIVERSITY IN PROTECTED AREAS

The Convention on Biological Diversity defines a protected area as "…a geographically defined area which is designated or regulated and managed to achieve specific conservation objectives" (UNEP 1992). Protected areas are classified under a system used by the International Union for Conservation of Nature (2012) and include (from most to least protected): strict nature reserves, wilderness areas, national parks, natural monuments or features, habitat and species management areas, protected landscapes, and managed resource protection areas.

Protected areas differ in their permanence and level of protection. For example, federally established wilderness areas are unlikely to experience high levels of resource extraction, but lands voluntarily enrolled in conservation programs (such as the Conservation Reserve Program) may only be temporarily protected. Forest land that is withdrawn from harvesting through statute, administrative regulation, or designation without regard to productive status is labeled "reserved forest," which is usually publicly owned (Bechtold and Patterson 2005). Of the 174 million acres of northern forests, <6.4 million acres (3.7 percent) are designated as reserved forest (Fig. 3.6); they align with four of the protected-area classes described above: strict nature preserves, wilderness areas, national parks, and natural monuments or features.

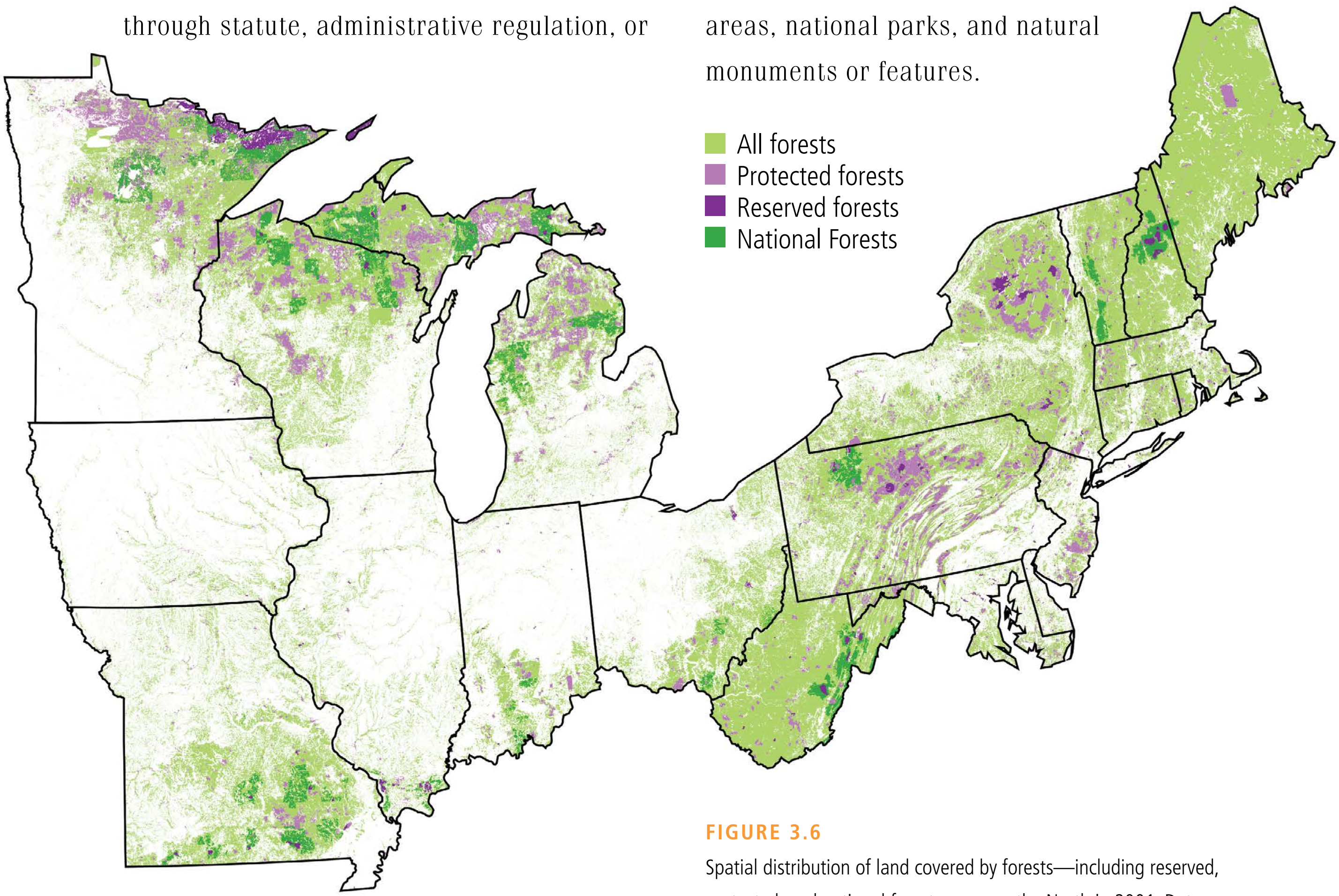


FIGURE 3.6

Spatial distribution of land covered by forests—including reserved, protected, and national forests—across the North in 2001. Data sources: ForestOwn_v1 geospatial database (Nelson et al. 2010); and Protected Area Database (CBI 2012).

Much of the public land base (including most national forests) does not meet the strict definition for protected areas. But these lands do provide protection through long-term stability in both ownership and land use (Chapter 6). In this report, projections of land-use change excluded Federal land because it is assumed that forested Federal land (such as national forests and national parks) will remain forested and in Federal ownership (Chapter 2).

Protected areas can act as important refuges for biodiversity by excluding or controlling factors (such as habitat loss or invasive species) that cause loss of biodiversity outside their boundaries. Forest biodiversity in protected areas can be an important source of ecosystem services, or benefits, especially in landscapes where land cover is predominantly nonforest. These benefits include erosion control, carbon sequestration, recreation, and economic returns from tourism and other rural enterprises.

Changing climate and land-use patterns may threaten the ability of protected areas to effectively conserve biodiversity (Hannah 2008, Radeloff et al. 2010, Stein et al. 2007). Although many protected areas are geographically fixed, distributions of forest ecosystems and species are dynamic and depend partly on climate conditions and the conditions in surrounding landscapes. For example, some species could disappear from protected areas that lose suitable conditions, but these losses could be offset if species are able to take advantage of an increase in suitable climate conditions in other protected areas (Hannah 2008). For other species, distributions could expand if the overall area with suitable climate and land-use conditions increases. The ability of forest-associated plants and animals to move with changing climate conditions would be reduced in landscapes with isolated protected forests that border nonforested areas, lack connectivity to other forested acres, or are subject to interspersed urbanization (Hannah 2008, Opdam and Wascher 2004). This is not likely to be an issue in landscapes with relatively contiguous forest cover, such as northern Minnesota. If changing climate conditions cause one species to be an overall gainer or loser of territory, the result could be a change in species representation, which is the proportion of range in protected areas that each species commands (Hannah 2008). Some proposed management strategies to maintain species representation include creating additional protected areas, increasing connectivity of protected areas, assisting migration by relocating species to new areas (Hannah 2008), and reducing human/urban pressures on remaining areas to increase their resiliency and ability to recover from disturbances (Giordano and Boccone 2010, Zipperer 2002).

Across the North, housing densities in and around protected areas are expected to increase at the expense of forest and other land uses (Radeloff et al. 2012, Radeloff et al. 2010, Stein et al. 2007, Theobald 2010).

The natural amenities offered by protected areas attract people who want to live in relatively undeveloped areas and are willing to commute long distances or can take advantage of telecommuting (Radeloff et al. 2010). Purchasers of seasonal homes and an increasing number of retirees are also contributing to housing growth in and around protected areas. In 2000, 35 million housing units were located within 31 miles of national forests, and this number is projected to increase by 16 million units by 2030. And within national forests, the number of housing units on private inholdings was projected to increase from 1.8 million in 2000 to 1.9 million by 2030 (Radeloff et al. 2010).

FOREST FRAGMENTATION

The value of forest habitats depends not only on attributes of composition and structure but also on the size and arrangement of forest fragments within the larger landscape. Growth of residential development and other land-use changes can threaten forest biodiversity (Radeloff et al. 2010, Stein et al. 2007) by reducing habitat abundance, increasing fragmentation, changing ecological processes (e.g., suppressing wildfires), and altering biotic interactions (e.g., introducing nonnative species or predatory pets). Such changes can also isolate forested areas, reducing the ability of species to respond to changing climate. Policies such as promotion of clustered developments and financial mechanisms such as tax benefits offered by land trusts, could help reduce the impact of housing growth on the biodiversity of protected areas (Radeloff et al. 2010).

Various metrics of forest fragmentation offer useful indicators of habitat suitability for species that require large fragments, forest interiors, ecotones (forest edges), proximity to nonforested land covers, or corridors for movement between fragments. Landscape metrics typically are derived from maps of current and projected forest cover that delineate individual fragments of land cover. These maps are needed to assess potential future changes in landscape patterns.

A raster map showing baseline forest cover (distribution and density) of the North in 2009 (Fig. 3.7A) was developed following the methodology of Wilson et al. (2012), having 250-m spatial resolution. The forest cover data set was combined with a model of future nighttime illuminated areas, based on a satellite-image time series recorded over the past decade. For future projections, it was assumed that nighttime illumination is associated with developed areas and that recent trends in the growth of illuminated areas will continue through the projection period. The 2009 forest cover map was adjusted using a map of projected illuminated areas, with the appropriate adjustment factor supplied by projections of forest cover in 2060 under storylines A1B, A2, and B2 from an econometric model developed by Wear (2011). This analysis lacked sufficient resolution to include reversion to forest from pastureland, cropland, or rangeland. Map results for storyline A1B were visually similar to A2, but differed from B2; for simplicity, map results are only portrayed for A1B (Fig. 3.7B), but quantitative results are portrayed for both A1B and B2 (see subsequent figures).

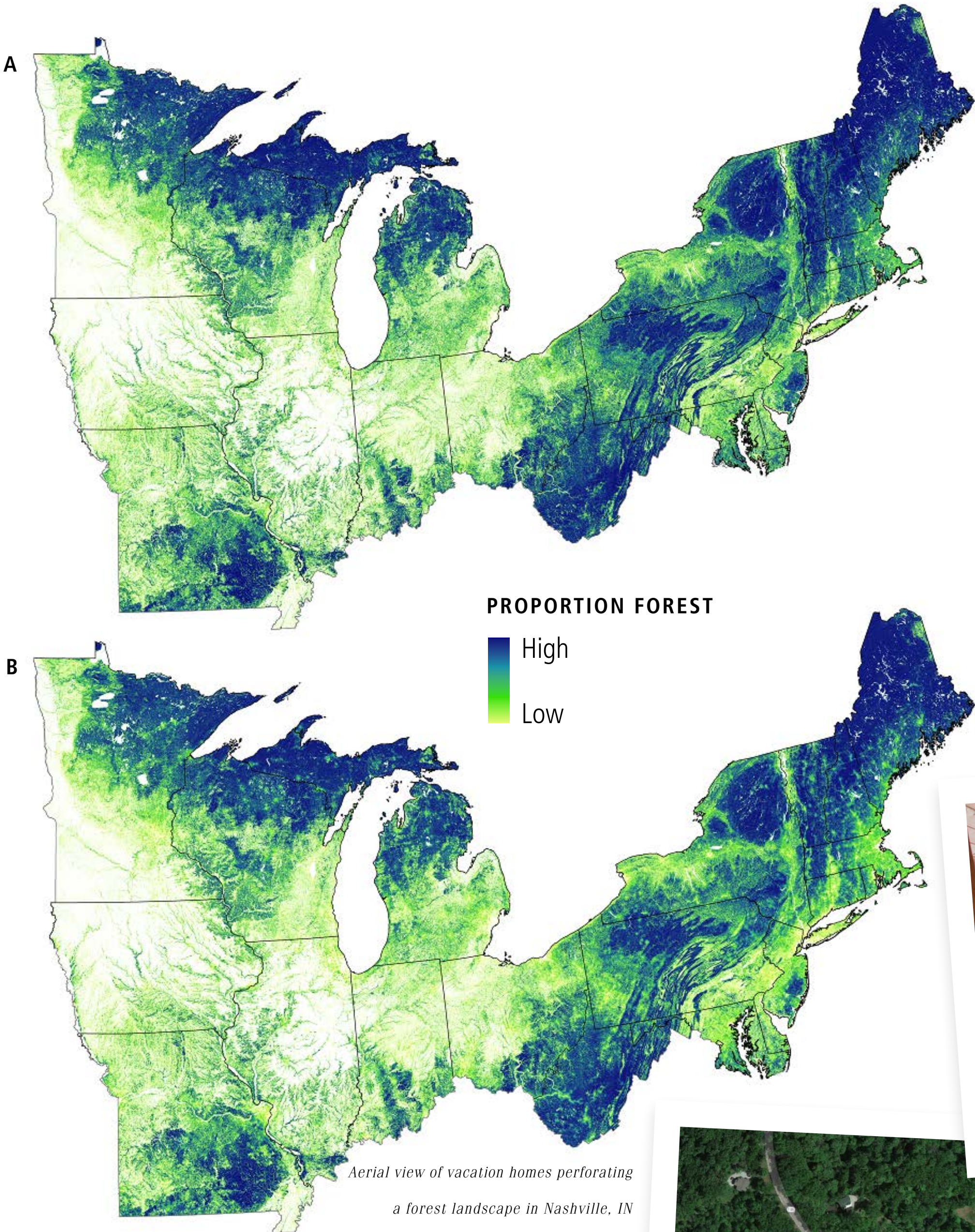
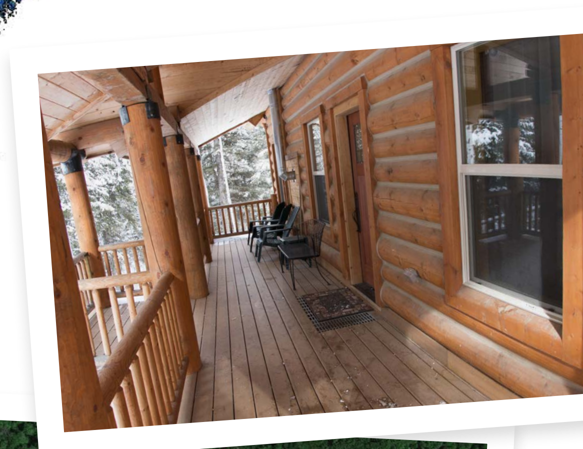


FIGURE 3.7

Spatial distribution and density of northern forests (A) in 2009, and (B) projected for 2060 under storyline A1B that assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio) (IPCC 2007).



Aerial view of vacation homes perforating a forest landscape in Nashville, IN

Imagery ©2015 Google

Projected reductions in forest extent and density are portrayed as differences between the 2009 and 2060 maps (Fig. 3.8). Not surprisingly, given that area of nighttime illumination was used as a proxy for development, forest losses are projected to be concentrated around existing urban and suburban areas and major transportation corridors (Fig. 3.8).

To quantify current and projected future forest fragmentation, a spatial integrity index was adapted that integrates forest patch size, local forest density (which picks up likelihood of edge conditions), and patch connectivity to core forest areas (Kapos et al. 2000). Because values for the index are calculated from pixels that are labeled or classified as forest, pixels with high forest canopy density were reclassified as forest, and those with low density as nonforest, for current and future conditions, and then spatial analysis was performed to determine spatial integrity values. A value of 10 is defined as core forest, and the lowest values (approaching zero) are defined as highly fragmented forest, with intermediate values representing varying degrees of opportunity for rehabilitating landscapes to connect core forest areas.

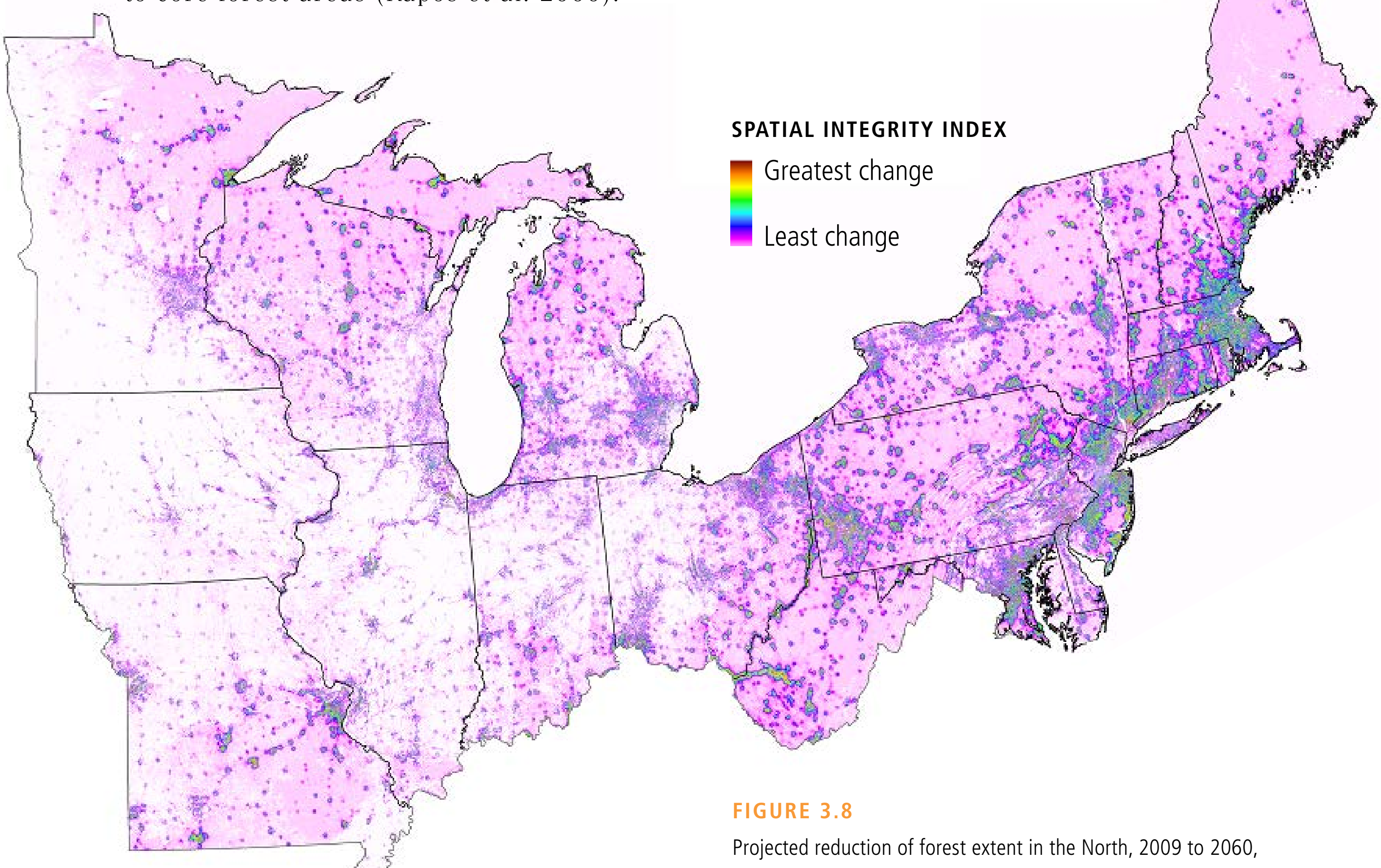


FIGURE 3.8

Projected reduction of forest extent in the North, 2009 to 2060, under a greenhouse gas emissions storyline A1B (IPCC 2007) that assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio).

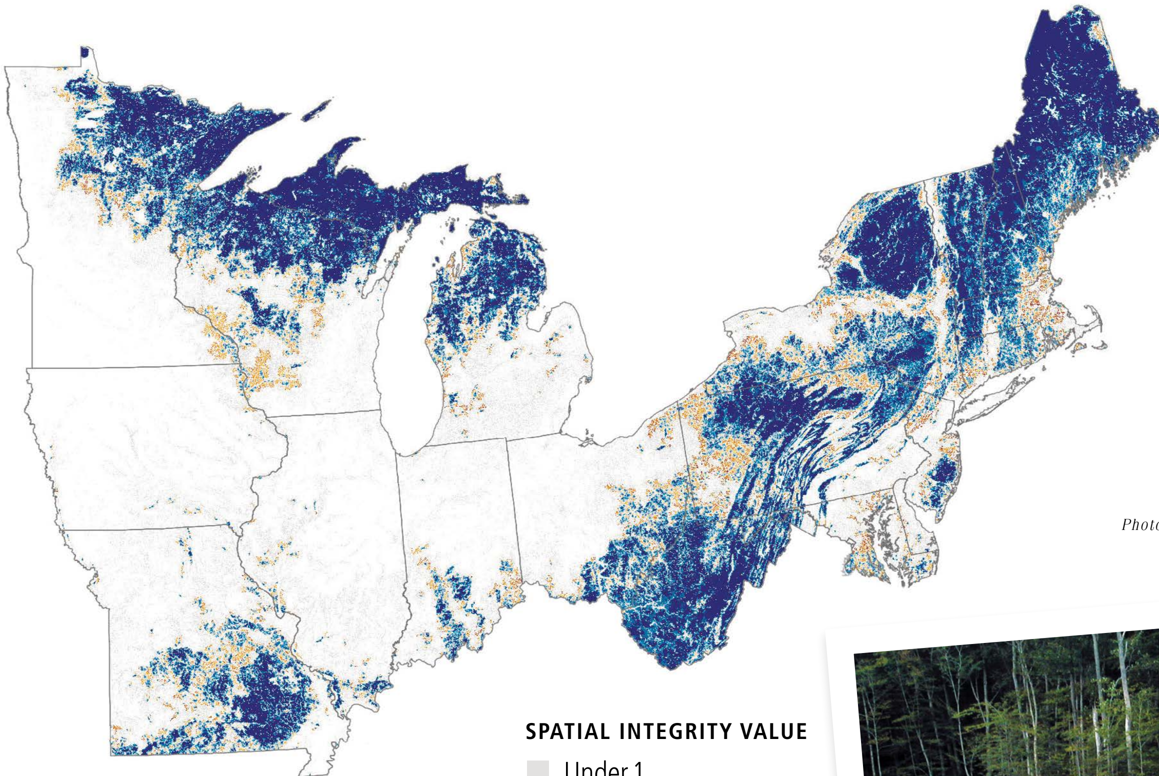


FIGURE 3.9

Forest spatial integrity index values, adjusted for housing densities associated with the wildland-urban interface or urbanized areas, 2009, across the North. Index values range from less than 1 (highly fragmented forest) to 10 (core forest with large patches and high local forest density).

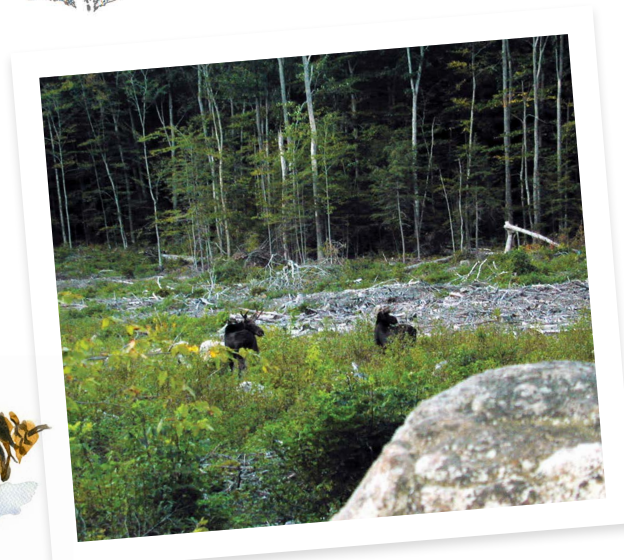


Photo by Mariko Yamasaki, U.S. Forest Service

Pixels having high tree canopy cover (inferred as representing high forest density) can occur in areas that also contain residential urban development. To determine the effects of such development on ecological processes and biotic interactions, two housing density classes were overlaid on the spatial integrity map: (1) forest pixels with a local housing density >15 houses per square mile, which is considered the threshold for the wildland-urban interface; and (2) pixels with >41 houses per square mile, which represent more urbanized areas. Assuming that this level of residential development contributes some characteristics of fragmented land, such as edge conditions and barriers to biotic movement, the spatial integrity value was reduced by two classes for the wildland-urban interface and one class for urbanized areas in the following summary (Fig. 3.9). Following procedures similar to those used for the 2009 spatial integrity index map, but excluding wildland-urban interface data which are not available for future projections, spatial integrity index values were estimated for 2060 and used to estimate the State-level changes in the spatial integrity index discussed later in this chapter.

Areas with the highest level of spatial integrity are characterized by forests with (1) a patch size >1,544 acres; (2) a local forest density of >90 percent in the surrounding 1,213-acre neighborhood; and (3) a housing density of <15 houses per square mile. These forests are thus considered to be core for this scale of analysis, with an index value of 10. An index value of 1 translates into a forest that is highly fragmented, with low or no connectivity to the core, small patch sizes, and ≤10 percent forest density. Values between 1 and 10 correspond to increasing proximity to core forest areas, increasing patch size, and increasing local forest density (Fig. 3.9), with local arrangement and interspersion of spatial integrity values for the wildland-urban interface more easily seen when the map focus is zoomed to a finer scale, as is shown for a subset of the North (Fig. 3.10).

This analysis combined fragmentation estimates from 2009 and housing densities from 2000 to demonstrate future challenges to maintaining biodiversity. This was done for two reasons. First, because fragmentation and urbanization have a delayed impact on forest ecosystems, a phenomenon known as extinction-debt trajectory, their existence portends areas where we can expect future changes in biodiversity that are not yet apparent (Malanson 2002, Tilman et al. 1994). Second, predicting the extent, intensity, location, and very importantly, the spatial distribution of future development depends on many factors, and thus comes with many uncertainties. This, combined with the high sensitivity of most measures of fragmentation to spatial variations in the data sources used, prompted the use of current information.

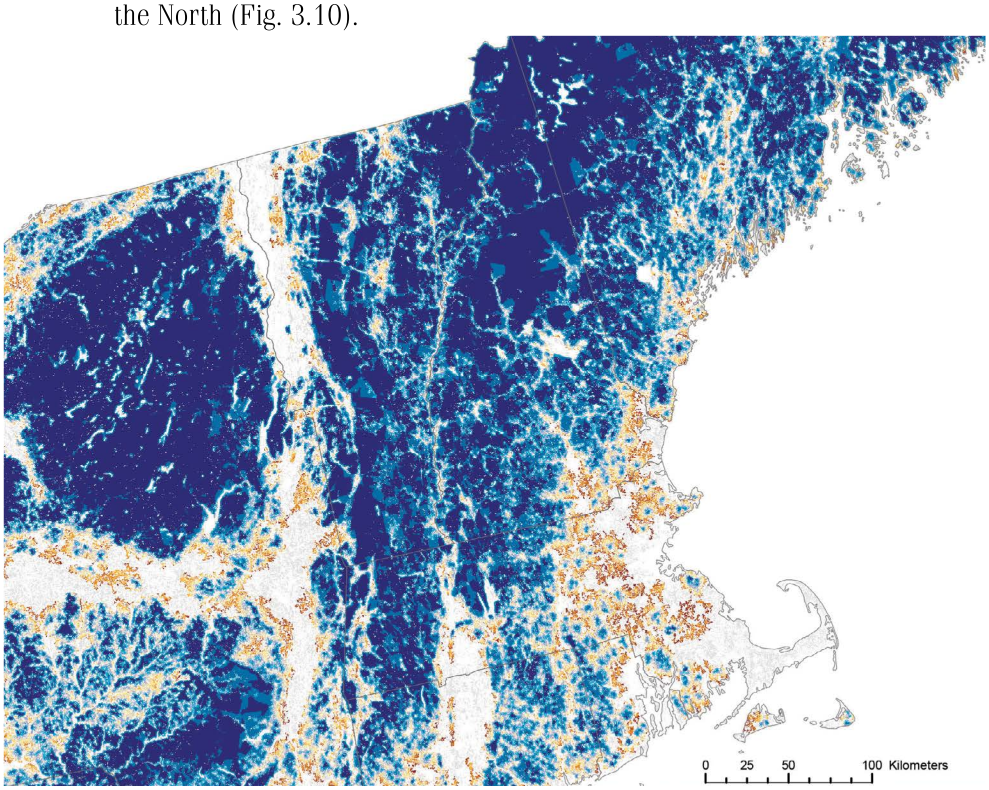


FIGURE 3.10

Forest spatial integrity index values, adjusted for housing densities associated with the wildland-urban interface or urbanized areas, 2009, in one area of the North. Index values range from less than 1 (highly fragmented forest) to 10 (core forest with large patches and high local forest density).

SPATIAL INTEGRITY VALUE

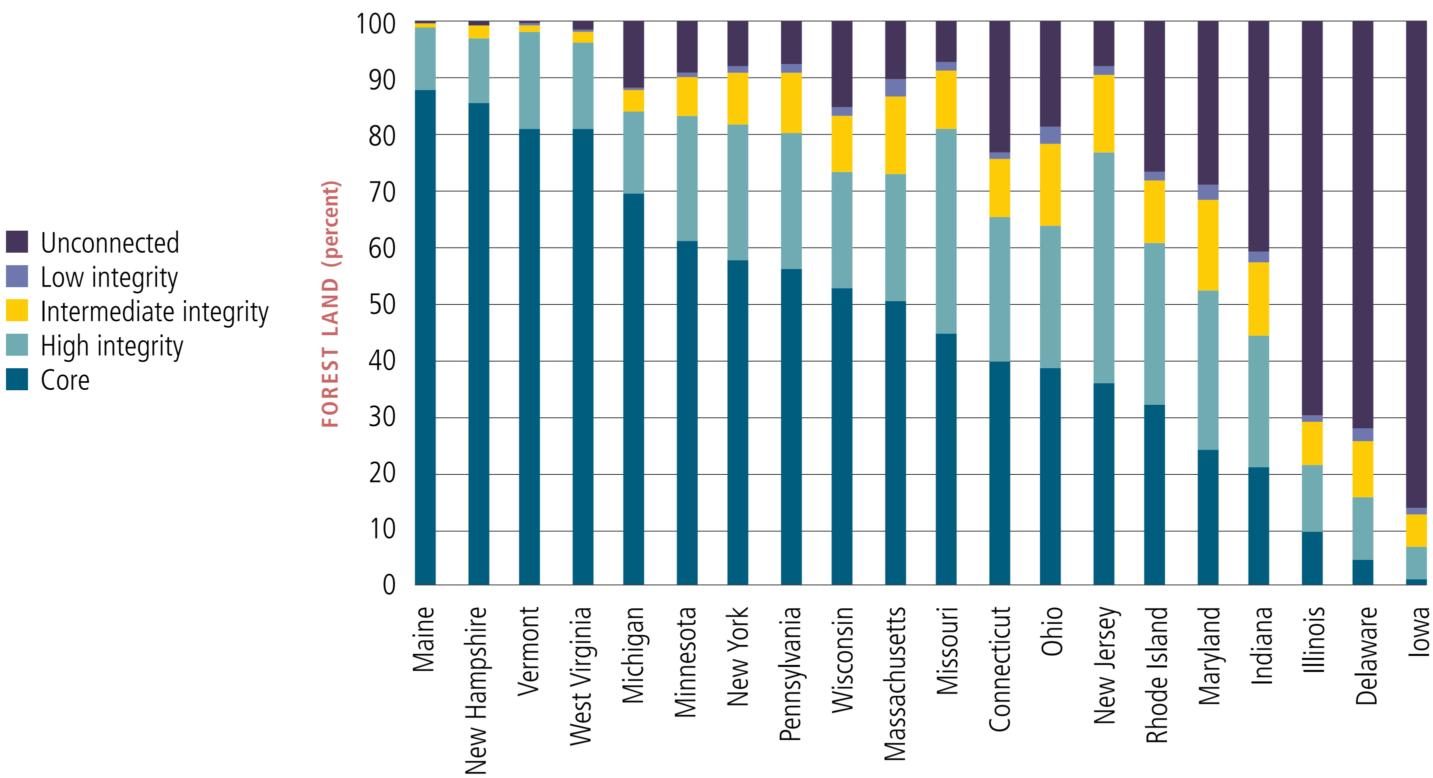
- Under 1
- 1
- 2
- 3
- 4
- 5
- 6
- 7
- 8
- 9
- 10

Figure 3.11 shows the current distribution of forest spatial integrity categories, adjusted for housing densities associated with the wildland-urban interface or urbanized areas. Maine supports the greatest percentage of core forest (88 percent), and Iowa has the least (1 percent). The pattern is reversed for unconnected forest, with 87 percent for Iowa and less than 1 percent for Maine. The majority of forests (56 percent) in Rhode Island are in neither of these two extreme categories, but are characterized by intermediate categories of low, medium, or high integrity forest.

All States are projected to lose some forest acreage by 2060. Losses outweigh gains, but losses vary by State and are generally larger under storyline A1B than B2 (Fig. 3.12). Under both storylines, some States and spatial integrity class combinations would increase, most of which would occur in unconnected, low integrity, and intermediate integrity forests. Such increases typically are linked to fragmentation of core and high integrity forests, resulting in losses to those classes. In contrast, decreases in unconnected, low integrity, and intermediate integrity forests typically are not balanced by increases in core and high integrity forest, suggesting that acreage would convert to another use. No State is projected to gain forest in the core forest class.

FIGURE 3.11

Forest area by spatial integrity index class in the North, adjusted for housing densities associated with the wildland-urban interface or urbanized areas, 2009.



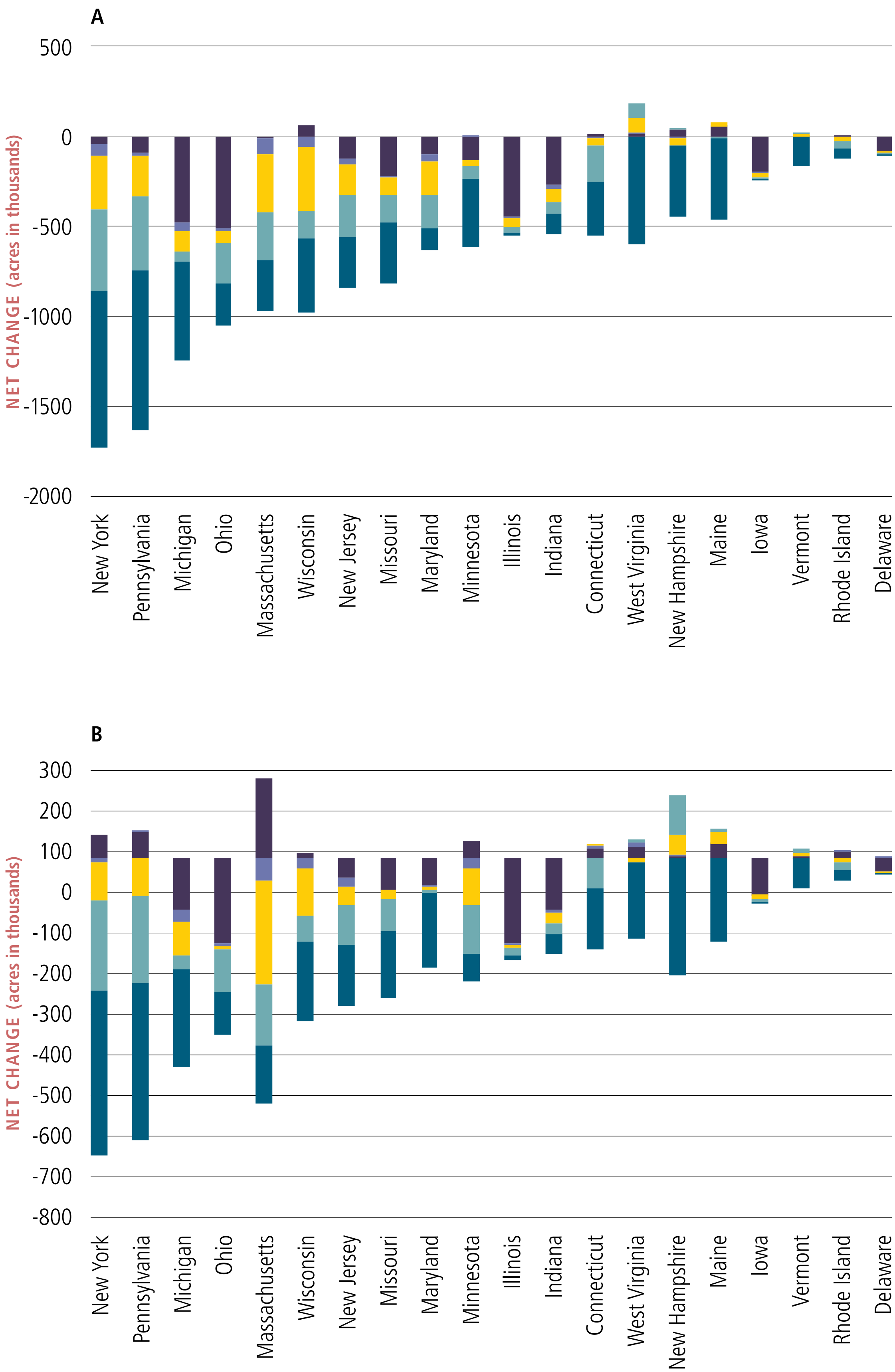


FIGURE 3.12

Projected change in forest area and spatial integrity index class in the North, 2009 to 2060, adjusted for housing densities associated with the wildland-urban interface or urbanized areas, under two greenhouse gas emissions storylines (IPCC 2007): (A) storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); and (B) storyline B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption.

- Unconnected
- Low integrity
- Intermediate integrity
- High integrity
- Core

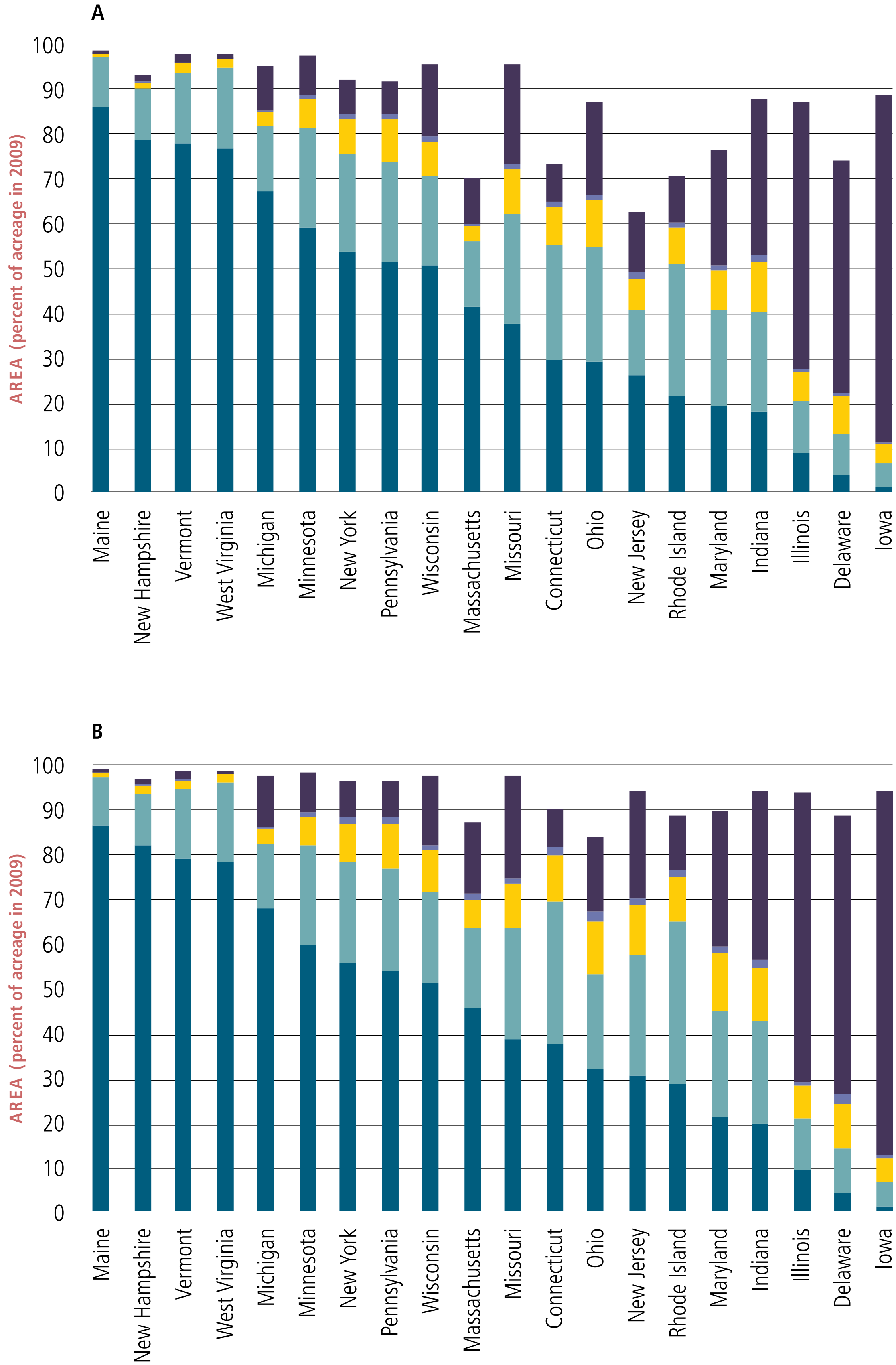


FIGURE 3.13

Projected forest area in the North, 2060, and the spatial integrity index class, adjusted for housing densities associated with the wildland-urban interface or urbanized areas, under two greenhouse gas emissions storylines (IPCC 2007): (A) storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); and (B) storyline B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption.

- Unconnected
- Low integrity
- Intermediate integrity
- High integrity
- Core

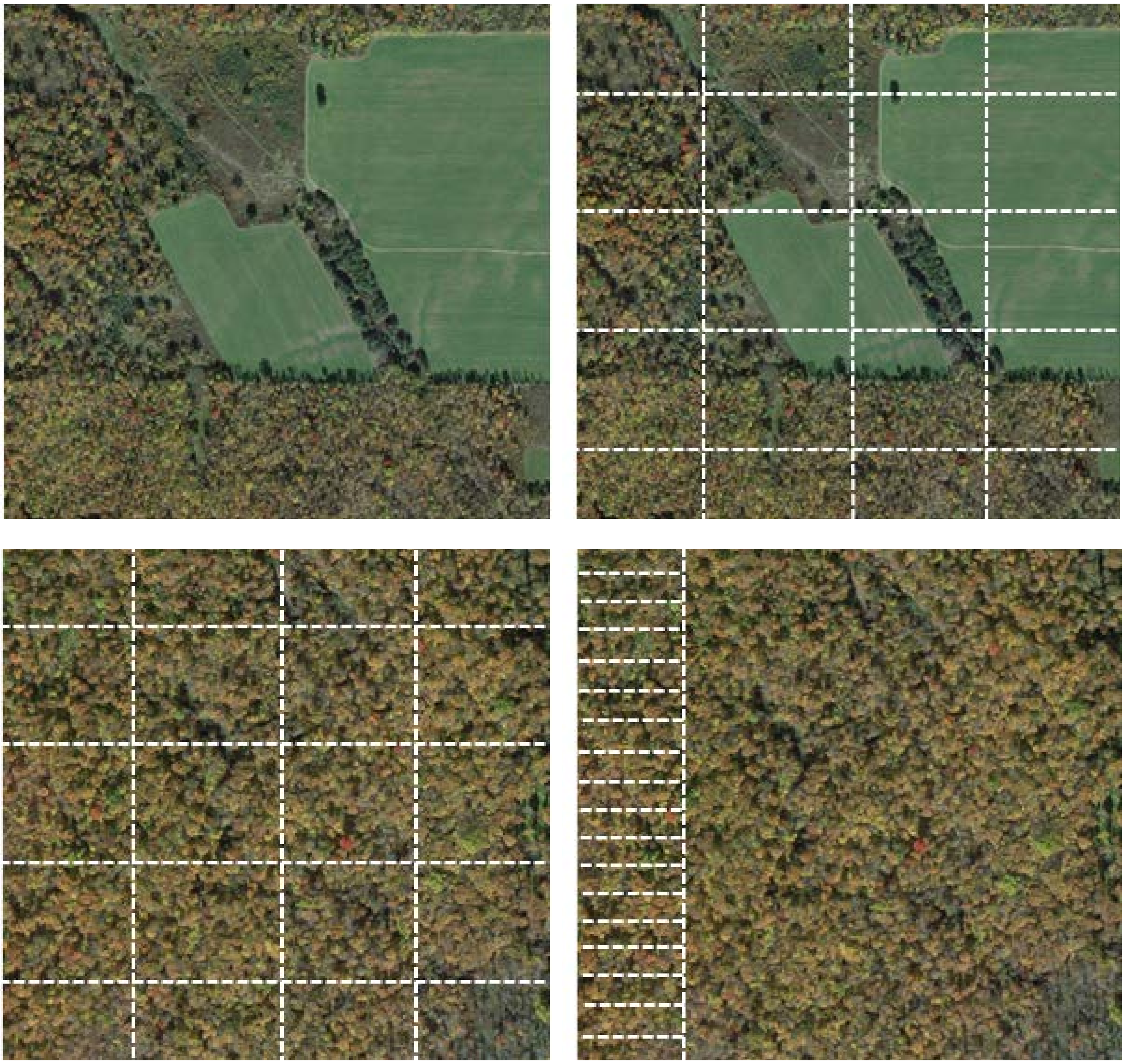
Figure 3.13 shows two projected distributions of future spatial integrity values, calculated as the percentage of the 2009 forest acreage in each spatial integrity category, sorted from highest to lowest percentage of core forest. New Jersey, Massachusetts, Rhode Island, Connecticut, and Delaware are all projected to lose ≥25 percent of forest acreage by 2060 under storyline A1B, but under B2 these five States are projected to lose much less acreage (10 to 16 percent).

FOREST PARCELLATION

One factor affecting forest fragmentation and management is forest parcellation (sometimes called parcelization). Haines et al. (2011) defines parcellation as the division of large landholdings into smaller ones. Parcellation does not necessarily imply that ecological services from affected forests will be lost, but Haines et al. (2011) have shown evidence of a possible relationship between parcellation and fragmentation. Although this relationship has not yet been studied further, it is plausible enough to be of concern, especially because of the growing trend in forest parcellation over the past several decades (DeCoster 1998, Mehmood and Zhang 2001, Zhang et al. 2009).

The distinction between fragmentation and parcellation is shown in Figure 3.14. Two of the landscapes are fragmented—one with all fragments under one ownership and the other divided in multiple ownerships. A third forest landscape does not appear to be fragmented at all; it is a contiguous forest, but one that is divided into 20 different ownerships and so is parcelized. Although a small portion of the fourth forest landscape is divided into many parcels, the biggest portion is not parcelized at all; 80 percent of it is under one ownership. The distinction between the two parcelized forest landscapes demonstrates the potential weakness of using average parcel size as a parcellation measurement. Both landscapes have the same average ownership size (same area and same number of ownerships), but these two forests will most likely face very different sets of development pressures. Nonetheless average ownership size was used as the best proxy for detecting parcellation because it is the only measurement available to use at the scale required by the data set.

Parcellation was correlated with estimated average family forest ownership size using National Woodland Owner Survey data from 2006 (Butler 2008) at the State and county levels. Estimates were based on the average number of forested acres in the State for families that had some holdings in the county of interest. Because respondents indicate how many acres they own, not in a county, but in the State, an ownership may be more than one parcel, with multiple parcels across different counties. We regressed the natural log of those estimates on the log of current (2010) U.S. housing density data as well as several significant State categorical variables (true/false categories). The county-level model has very low explanatory power, with an R^2 of 0.08, but county estimates were retained to illustrate



spatial pattern in cartographic outputs (Fig. 3.15). In contrast, the state-level model has an R^2 of 0.77. These models projected average parcel size for 2060 using U.S. housing density predictions. In all, eight sets of projections were created: the two described above and those based on the county and state-level housing density prediction sets for storylines A1B, A2, and B2.

FIGURE 3.14

Schematic illustrating hypothetical combinations of parcellation and fragmentation: (A) no parcellation on a fragmented landscape; (B) parcellation into uniform ownerships on a fragmented landscape; (C) parcellation into uniform ownerships on a contiguous landscape; and (D) parcellation into ownerships of varying sizes on a contiguous landscape. Hypothetical ownership parcel boundaries are illustrated by dashed lines. Imagery: ©Google 2015.

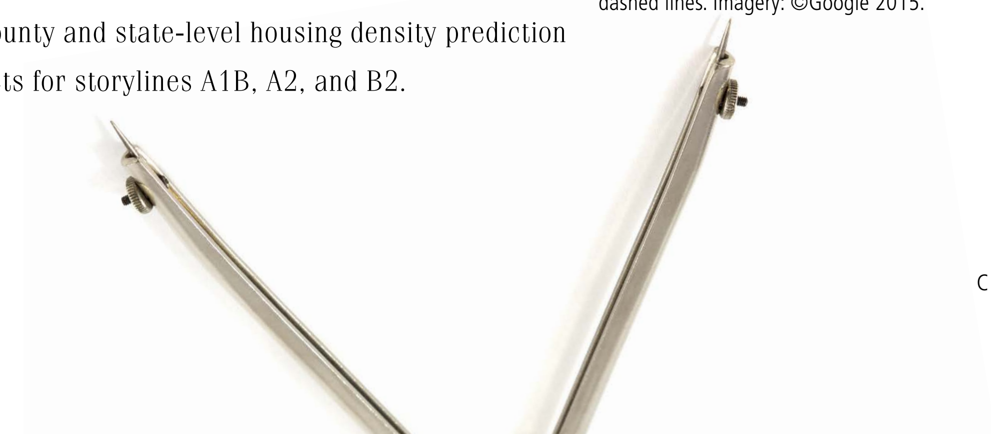


Figure 3.15 shows projected changes in average family forest ownership size at the county level for storyline A1B. Differences in projections across the scenarios are not visually discernible, so the other scenarios are not displayed. Overall, each county and each State is expected to see a reduction in average ownership size, which indicates an increase in parcellation.

The projected changes represent the difference between using fitted or actual current ownership size in projections. The reason the differences shown do not use actual current estimated average ownership size is that the map would be misleadingly noisy. Indeed, some of the projected future sizes are larger than actual current sizes. This is because some actual current sizes are much smaller than the fitted current values. To indicate gains in average ownership size would suggest that the model somehow predicts reverse parcellation in those counties, which is certainly not what is occurring. The map is designed simply to give an understanding of how parcellation is expected to follow increasing housing density predictions.

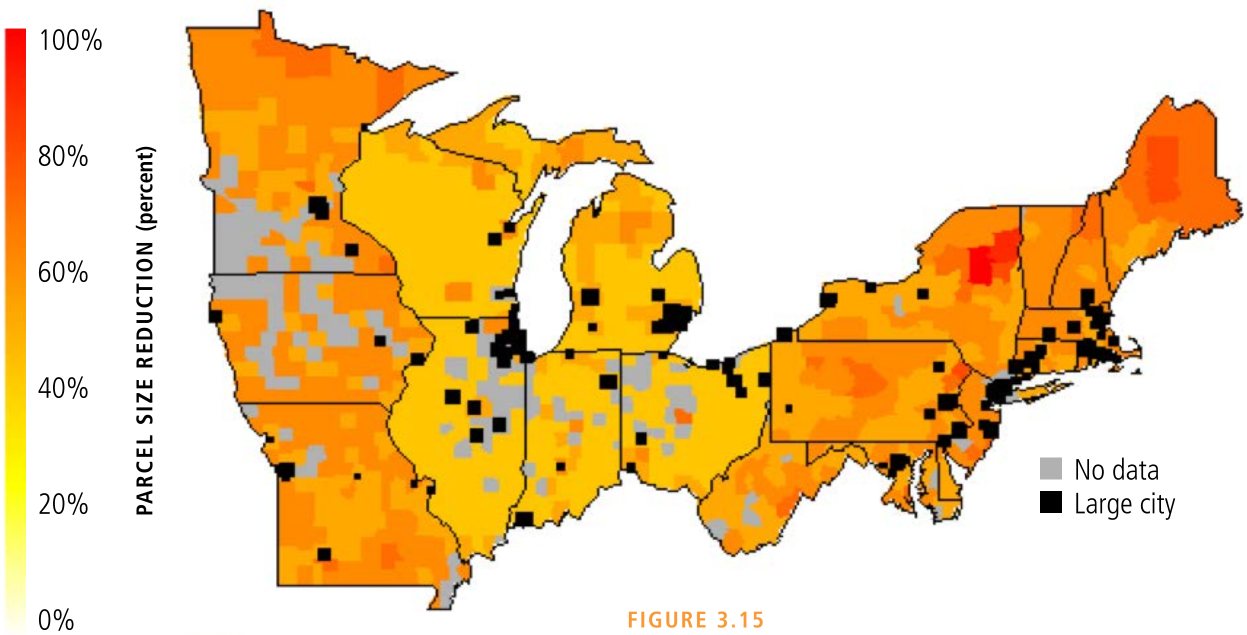


FIGURE 3.15

County-level projected reductions in average family forest parcel size, 2006 to 2060, under greenhouse gas emissions storyline A1 that assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy (Mondal et al. 2013).

Much of the extreme parcellation (>70-percent reduction in average size) is projected to occur far from city centers, most notably in northern Maine, New York, and Minnesota. However, a relatively low proportion of northeastern Minnesota forest acreage is in family forest ownership; in two counties <10 percent of forest acreage is estimated to be held by families. Thus, although parcellation of family forests is projected to be quite severe in that area, it may be less of a concern than in the Adirondacks of New York or in northern Maine.

These projections are simplistic and should be interpreted with caution. They are meant only to offer an idea of the direction of change that can be expected by 2060. With this caveat in mind, we anticipate a much more parcelized family forest landscape, regardless of which of the anticipated scenarios play out. Insofar as forest fragmentation follows this projected parcellation, the forest landscape would be much more fragmented by 2060 as well. With that level of fragmentation, losses in ecological services and socioeconomic benefits can be expected.

TREE SPECIES AND GENETIC DIVERSITY

One of the most basic measures of biological diversity is the total number of species within a given geographic area. This is referred to as species richness. In general, large areas are likely to contain more species than smaller areas. Different measures of diversity are used to account for these differences. Alpha diversity is the number of species observed within a small, homogeneous area; and gamma and epsilon diversity estimate species richness over increasingly larger, heterogeneous areas. Average species richness is reported for trees on survey plots (alpha diversity) established by Forest Inventory and Analysis (FIA), within States (gamma diversity), and across the entire region (epsilon diversity). Numbers of future species are not projected because this report addresses only potential habitat abundance within the region; species distributions also are affected by many nonhabitat factors, including invasive pests, disease, and competition.

Across the 20 Northern States, 160 tree species were recorded on FIA plots during the baseline period (2010). An average of 76 tree species was recorded in each State, ranging from 47 species in Rhode Island to 99 species in West Virginia. The number of species per FIA plot also varied among States and among major species groups (Fig. 3.16).

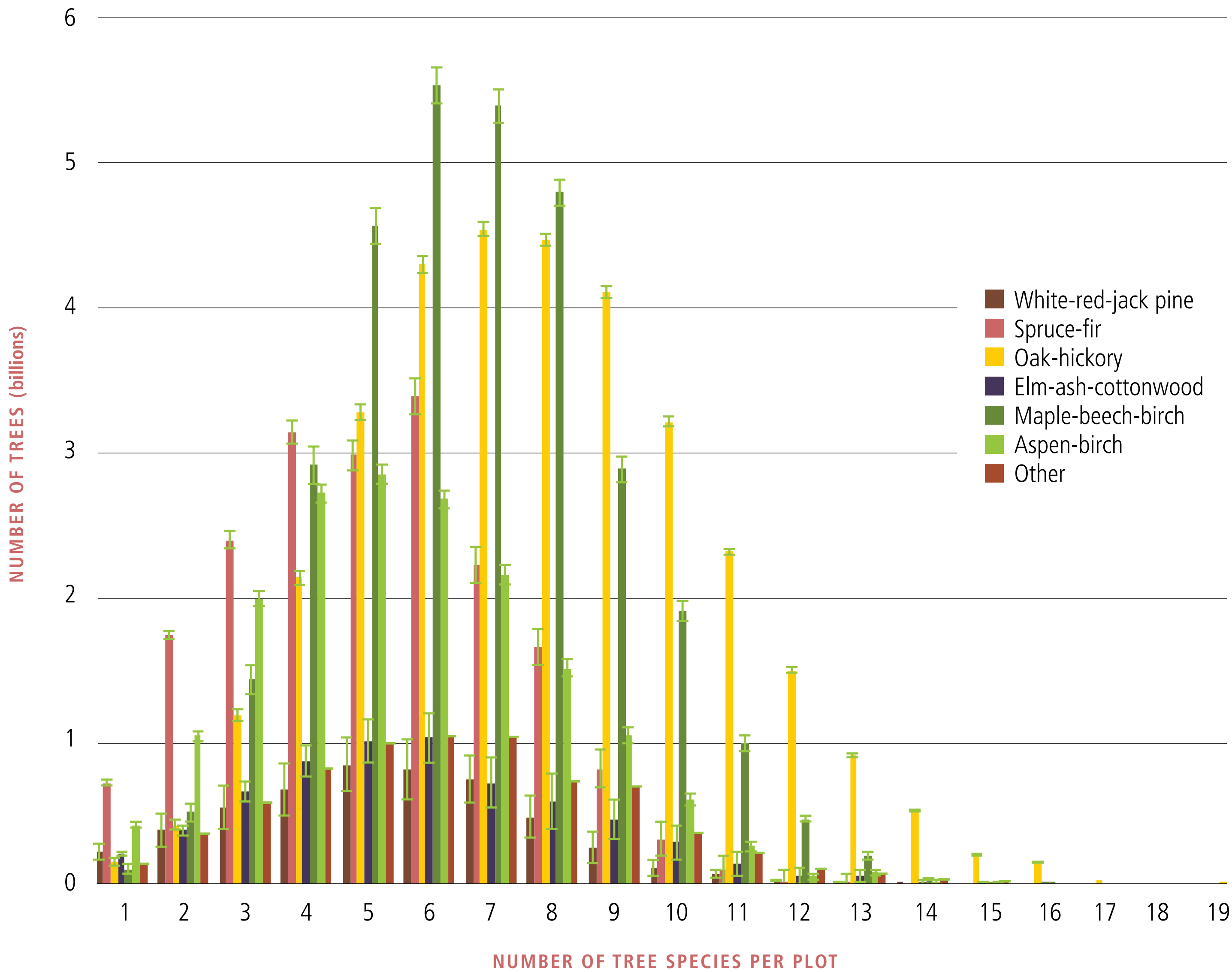


FIGURE 3.16

Number of trees (≥5 inches d.b.h), by forest-type group and tree frequency, 2010, per survey plot in the North; data from FIA. Error bars indicate 68-percent sampling error associated with the estimated number of trees for each column.

Given such a large geographic extent and the high number of FIA plots, it is not surprising that the number of tree species per plot presents such a classically normal distribution. What is notable, however, is the prominent shift to the right (more species per plot) in the maple-beech-birch (the northern hardwoods) and the oak-hickory (*Quercus* spp.–*Carya* spp.) groups. These two groups demonstrate different underlying processes. Maple-beech-birch is primarily composed of shade-tolerant species and regenerates largely by gap-phase processes. Oak-hickory forests have a more disturbance-mediated origin and so their regeneration and development dynamic reflects a more even-age initiation and release; they can be characterized as the climax forests where disturbance has "reset the clock."

Collecting ash tree seeds

Photo by Tom Arbour, Ohio Department of Natural Resources, used with permission

Genetic Diversity

Biological diversity refers not only to species diversity but also to genetic diversity. To illustrate, consider the variation in characteristics associated within cultivated plants of a single species, like apples (*Malus* spp.). Similarly, genetic diversity within a single tree species results in a variety of traits, resulting in different physical and functional characteristics that help trees adapt to local conditions. A species under threat of rapid, widespread mortality is at risk of losing genetic diversity, even if individual trees persist in some locations.

Such is the concern for ash trees, which are under serious threat from emerald ash borer, a nonnative insect pest that causes rapid mortality in several ash species (Chapter 5). In response, work is underway to preserve germplasm by collecting ash seeds for long-term storage under a combined effort by three U.S. Department of Agriculture agencies (Agricultural Research Service, Forest Service, and Natural Resources Conservation Service) the U.S. Department of the Interior's Bureau of Land Management, and more than 50 cooperators.

The goal of this effort is to preserve seeds from many individual trees of each ash species, evenly spaced across the range. To date, samples have been collected from more than 3,000 individual ash trees, including white (*Fraxinus americana*), green (*Fraxinus pennsylvanica*), black (*Fraxinus nigra*), blue (*Fraxinus quadrangulata*), and pumpkin (*Fraxinus profunda*) ash species. Collection priority is given to portions of the range that are under current infestation. Working collections of seeds (samples made available to researchers and breeders) are held at national seed laboratories with duplicate samples held as a security backup at alternate locations (Karrfalt and Carstens 2013).

The implications for biodiversity are several. The disturbance patterns that initiated many oak-hickory forests—such as land clearing (and subsequent reforestation), large fires, and even-age forest management—are declining in extent and intensity. Furthermore, oak-hickory forests, being primarily located in the southern reaches of the region, are more likely to adjoin major metropolitan areas and thus more likely to lose acreage to urban and suburban encroachment. Oak-hickory is the largest group by area (Chapter 4, Appendix 4) and is projected to lose the most acreage; on a relative basis, elm-ash-cottonwood is projected to be the biggest loser (Chapter 4). Maple-beech-birch and white-red-jack pine are projected to gain acreage. The implication of this projected trend is that overall, tree species diversity would decline through 2060, primarily because of the decrease in acreage of oak-hickory, a group that benefits from some forms of canopy disturbance.

FORECASTS OF CHANGES IN HABITAT AND AT-RISK WILDLIFE SPECIES

Some species require forest habitats that can have unique characteristics such as the species composition, age, and size of trees (Patton 2011). For example, Canada warblers (*Cardellina canadensis*) breed in young forests (6 to 20 years) whereas Acadian flycatchers (*Empidonax virescens*) breed in areas dominated by mature forests. Conservation and management of species with divergent habitat requirements cannot be successful unless they are based on a landscape approach that focuses on simultaneously providing different types of habitats and successional stages in separate areas (Hamel et al. 2005) and managing changes in habitat as forests grow and are subjected to disturbances. Efforts to conserve diverse groups of wildlife can benefit from assessments of habitat responses to changes in climate and land-use change. This report considered species richness and projected changes in area for six forest habitat classes—closed-canopy and open-canopy hardwoods, conifers, and mixed forests—using a NatureServe (2011) data set to project habitat classes, species richness, and species-habitat class associations.

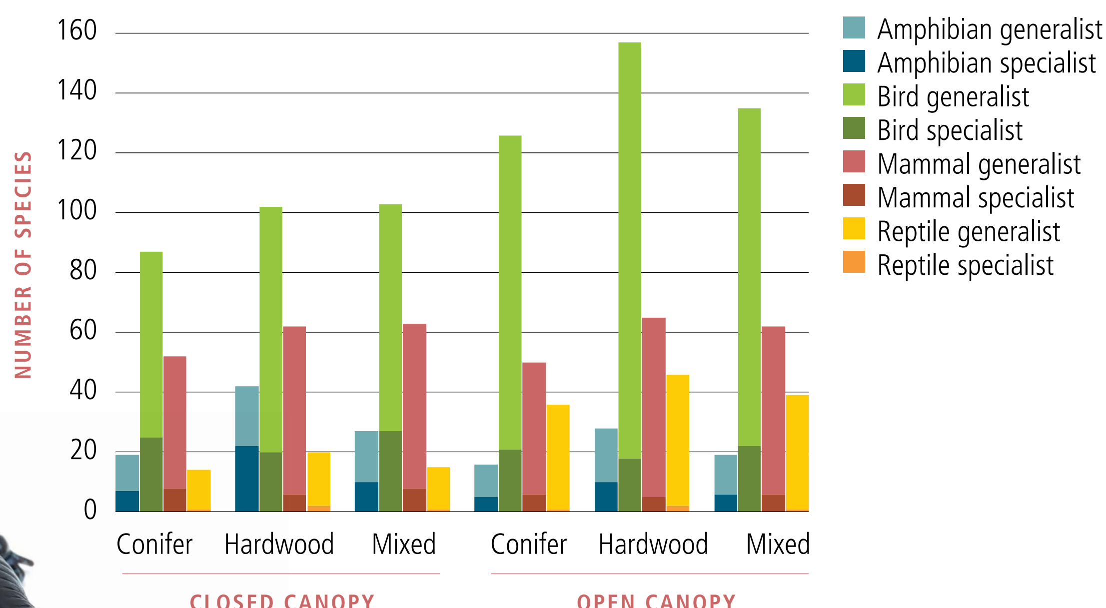
Northern forests provide habitat for 363 terrestrial vertebrate wildlife species, including mammals, reptiles, amphibians, and birds. Many of these species—such as the coyote (*Canis latrans*), American crow (*Corvus brachyrhynchos*), common garter snake (*Thamnophis sirtalis*), and American toad (*Anaxyrus americanus*)—are common generalists, meaning that they inhabit a wide variety of habitats (not necessarily forested), do not depend on any one type of habitat for their survival, and are mostly plentiful across their geographic range. Many other species are more specialized and spend their entire lives within forests; they include the northern hawk owl (*Surnia ulula*), hoary bat (*Lasiurus cinereus*), fisher (*Martes pennanti*), wood frog (*Lithobates sylvaticus*), and coal skink (*Plestiodon anthracinus*).

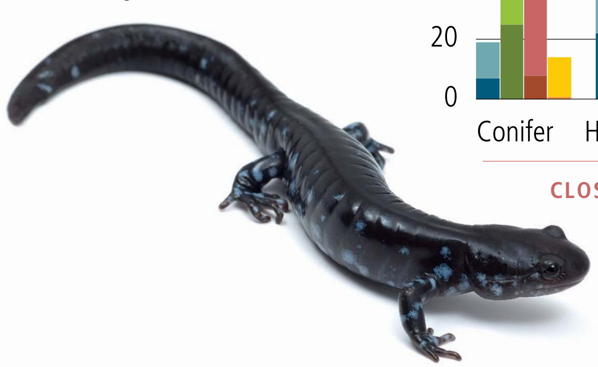
A few species are even more specialized and depend on a specific forest habitat for their survival, such as the open canopies of mixed pine and oak-pine barrens required by the treefrog (*Hyla andersonii*) or the closed-canopy hardwood forests required by the streamside salamander (*Ambystoma barbouri*). Other species may frequent numerous nonforest habitat types but rely on northern forests for critical life stages. An example is the wood duck (*Aix sponsa*), which spends most of its life on water but selects tree cavities for nesting sites. For other species, primarily migratory birds, northern forests make up only part of their geographic range, either as a seasonal destination as with Kirtland's warbler (*Dendroica kirtlandii*) and the northern shrike (*Lanius excubitor*), or as a stop along a migratory route as with the gray-cheeked thrush (*Catharus minimus*). Northern forests are also home to two known endemic species of salamander, the Cheat Mountain salamander (*Plethodon nettingi*), and the blue-spotted salamander (*Ambystoma laterale*), which can be found nowhere else on earth.

Abundance of terrestrial vertebrate wildlife species varied among habitat classes and among major taxa (Fig. 3.17). Species richness is highest in open-canopy forests and hardwood habitats. Bird species accounted for the highest richness, with mammals second in all forests. Bird species numbered >150 species in open-canopy hardwoods, the greatest richness for any taxon-habitat combination. Amphibians were the least rich taxon in open-canopy habitats, and reptiles were the least rich in closed-canopy habitats. Species that limit their habitats to forests (defined as specialists) were in the minority across all habitats except closed-canopy hardwoods, where amphibian specialists were more abundant than amphibian generalists.

FIGURE 3.17

Distribution of terrestrial vertebrate species among closed- and open-canopy conifer, hardwood, and mixed forest habitats across the North in 2010; note that species associated only with forest habitats are termed "specialists" and species associated with both forest and nonforest habitats are termed "generalists."





Forest projection models were used to assess climate and land-use driven changes in forest habitat classes under the assumption that current harvesting levels will continue without the influence of increased bioenergy demands (scenarios A1B-C, A2-C, and B2-C). Results showed that total forest habitat area is expected to decrease from a 2010 value of 173 million acres to 162 million acres for A1B-C, 164 million acres for A2-C, and 168 million acres for B2-C. Across all scenarios, closed-canopy habitat classes would gain acreage but open-canopy classes would lose (Fig. 3.18, Table 3.1). We also projected the potential influence of intensive biomass harvesting rates for bioenergy (scenarios A1B-BIO, A2-BIO, and B2-BIO) (Fig. 3.18); most open-canopy classes would lose some acreage and closed-canopy classes would lose even more (Tavernia et al. 2013).

FIGURE 3.18

Projected change in closed- and open-canopy conifer, hardwood, and mixed forest habitats of the North, 2010 to 2060, for six scenarios, each representing a global greenhouse storyline (IPCC 2007) paired with a harvest regime. Storyline A1B assumes moderate greenhouse gas emissions, moderate gains in population, and large gains in income and energy consumption (but with a balanced renewable/ fossil fuel portfolio); A2 assumes high greenhouse gas emissions, large gains in population and energy consumption, and moderate gains in income; and B2 assumes low greenhouse gas emissions with moderate gains in population, income, and energy consumption. Panel A scenario projections assume harvesting will continue at recently observed levels (scenarios labeled –C); panel B scenario projections assume increased harvest for bioenergy production (scenarios labeled –BIO).

Table 3.1—Projected area, 2060, and percent change from the 2010 baseline of closed-canopy and open-canopy forest habitat classes (NatureServe 2011, Tavernia et al. 2013) across the Northern United States under three scenarios; note that changes were driven by projected climate and land-use changes, forest succession, and forest harvesting.

Scenerio	Total habitat	Closed canopy forests			Open canopy forests		
		Hardwood	Conifer	Mixed	Hardwood	Conifer	Mixed
	Million acres (percent change in acreage)						
2010	173.4[a]	70.4	4.3	10.8	59.5	16.7	11.7
A1B-C[b]	162.4 (-6)	73.1 (4)	5.2 (21)	11.4 (6)	48.5 (-19)	14.3 (-14)	9.9 (-15)
A2-C	164.1 (-5)	74.3 (6)	5.0 (16)	11.6 (7)	48.7 (-18)	14.5 (-13)	10.1 (-14)
B2-C	167.5 (-3)	74.9 (6)	5.0 (16)	12.1 (12)	50.6 (-15)	14.6 (-13)	10.2 (-13)

[a]*0.6 million acres of the total 174 million acres of forest area could not be associated with habitat classes (Tavernia et al. 2013).*

[b]*Under the three scenarios, A1B-C assumes moderate levels of greenhouse gas emissions associated with moderate gains in population growth and large gains in income and energy consumption (but with a balanced renewable/fossil fuel portfolio); A2-C assumes high levels of greenhouse gas emissions associated with large gains in population growth and energy consumption with moderate gains in income; and B2-C assumes low levels of greenhouse gas emissions associated moderate gains in population growth, income, and energy consumption.*

Early successional forest habitats in the North have experienced decades-long decreases in acreage (Trani et al. 2001). Researchers attribute these decreases to maturation of forests on abandoned farmland, altered forest management practices, forest ownership patterns that discourage harvesting, fire suppression and other disruptions to natural disturbance regimes, and land-use conversion (Askins 2001, Lorimer and White 2003, Trani et al. 2001). Given that young forests characteristically have small diameter trees and relatively open canopies, decreases can be expected to continue under all scenarios.

If so, a broad segment of the wildlife community would be affected. For example, many species that typically inhabit mature, closed-canopy forests relocate to early successional habitat at certain times of the year (Vitz and Rodewald 2006). Wildlife managers can use information about historical habitat conditions and wildlife population dynamics to identify a mix of closed- and open-canopy habitats capable of supporting diverse wildlife communities (Askins 2001, Litvaitis 2003, Lorimer 2001, Thompson and DeGraaf 2001). As illustrated by the intensive harvesting scenarios, policy and management decisions can alter forest habitat conditions and, consequently, help achieve goals for habitat improvement.

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