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CHANGES IN FOREST HABITAT CLASSES UNDER ALTERNATIVE CLIMATE AND LAND-USE CHANGE SCENARIOS IN THE NORTHEAST AND MIDWEST, USA

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Abstract. Large-scale and long-term habitat management plans are needed to maintain the diversity of habitat classes required by wildlife species. Planning efforts would benefit from assessments of potential climate and land-use change effects on habitats. We assessed climate and land-use driven changes in areas of closed- and open-canopy forest across the Northeast and Midwest by 2060. Our assessments were made using projections based on A1B and A2 future scenarios developed by the Intergovernmental Panel on Climate Change. Presently, forest land covers 70.2 million ha and is evenly divided between closed- and open-canopy habitats. Projections indicated that total forest land would decrease by 3.8 or 4.5 million ha for A2 and A1B, respectively. Within persisting forest land, the balance between closed and open-canopy habitats depended on assumed harvest rates of woody biomass. Standard harvest rates led to closed-canopy habitat attaining a slight majority of total forest land area. Intensive harvest rates resulted in the majority of forest land being in open-canopy habitat for A1B or maintained the even split between closed- and open-canopy habitats for A2. Ultimately, managers need to identify benchmark habitat conditions informed by historical conditions and wildlife population dynamics and plan to meet these benchmarks in dynamic forest landscapes.

Keywords: Wildlife Habitat; Bioenergy; Biomass Harvest; Climate Change; Young Forest; Early Successional Habitat; FIA; Forest Projections.

1 Introduction

A significant challenge in natural resources management is providing sufficient habitat for wildlife species that have diverse and sometimes conflicting habitat needs (Magules and Pressey, 2000; Noon et al., 2009). Suites of species are associated with particular forest habitat classes characterized by different compositions, ages, and structures (Hagan et al., 1997; Patton, 2011). For example, some species (e.g., Cerulean warbler, *Setophaga cerulea*) are associated with mature, deciduous forests whereas others (e.g., Kirtland's warbler, *Setophaga kirtlandii*) are found in disturbance-dependent early successional, coniferous habitat. Successful conservation and management of species with different habitat associations requires management plans that are large-scale and long-term in scope; such plans are necessary to ensure that diverse habitat needs are simultaneously met and maintained through time (Hamel et al., 2005).

Changing climate and land-use conditions are expected to drive, in part, the large-scale dynamics of forest habitat and wildlife distributions over the coming decades and centuries (Iverson and Prasad, 1998; Matthews et al., 2004; Schwartz et al., 2006). Ecologists have long recognized large-scale associations between distributional limits of forest types and wildlife and climate conditions (Booth, 1990; Newton, 2003; Prentice et al., 1992). As climate changes, some forest ecosystems and forest-associated species might shift their distributions to track hospitable climate conditions, and others might adapt to new climates (Iverson and Prasad, 1998; Matthews et al., 2004; Parmesan, 2006). When forest ecosystems and forest-associated species are unable to move or adapt, their geographic ranges may shrink, or they may become extirpated from portions of their former range (Parmesan, 2006; Thomas et al., 2004). To aid long-term conservation and management planning, researchers often model the potential distributions of forest types and wildlife species under alternative climate change scenarios (Iverson and Prasad, 1998; Matthews et al., 2004; Schwartz et al., 2006). The direction and magnitude of modeled distributional changes are often scenario-specific, and accounting for uncertainty caused by scenario selection is a significant challenge (Beaumont et al., 2008).

Within large-scale patterns established by climate, land-use decisions further modify the extent and configuration of forest types and wildlife species diversity (Opdam and Wascher, 2004; Pearson et al., 2004). Forest conversion (e.g., to urban lands) reduces forest area and potentially fragments forest ecosystems; this can result in small, extirpation-prone wildlife populations that are too isolated to be rescued or re-established by immigrants from the surrounding landscape (Robinson and Wilcove, 1994; Verboom et al., 1991). Land-use conversion may also place remaining forest habitat in close proximity to anthropogenic land-uses, including agricultural and urban areas. This proximity can alter food availability, ecological processes, and biotic interactions in ways that hasten the decline of wildlife populations (e.g., via increased nest predation pressure, Donovan et al., 1995). Researchers and managers recognize the importance of simultaneously assessing climate and land-use change effects (Pearson et al., 2004). Despite this, few such assessments exist due, in part, to a lack of climate and land-use projections that share common assumptions about future demographic, economic, and technological conditions (Bierwagen et al., 2010).

Quantity and quality of habitat can be affected by increases in woody biomass utilization for bioenergy. Woody biomass currently accounts for the greatest share of bio-energy generation in the U.S. at about 53% (U.S. Energy Information Administration, 2012). Annual woody biomass consumption for electricity generation is projected to increase over the next 20 years (U.S. Energy Information Administration, 2012). Energy markets for woody biomass may lead to the harvesting of stands previously viewed as being non-commercial (e.g., due to poor wood quality) and to shorter rotation times between harvests (Janowiak and Webster, 2010). Forest harvest can increase the area of early successional or young forest habitat, benefiting wildlife dependent on this habitat (e.g., Annand and Thompson, 1997). Plantations of short-rotation woody crop species (e.g., *Salix, Populus*) might serve as a source of biomass, and plantation establishment may have negative or positive effects on wildlife, depending on the land-use type that is converted. Studies have generally found that plantations support fewer species of wildlife than unmanaged forest (Moore and Allen, 1999), but the conversion of non-forest land-use types (e.g., agricultural fields) to plantations might benefit forest wildlife by increasing total area and connectivity of habitat (Cook and Beyea, 2000). Ultimately, the value of plantations to wildlife species depends on how they are managed (Hartley, 2002).

Energy markets for woody biomass may also provide sufficient incentives to remove small-diameter woody material in addition to processing logging residues. Integrated harvest, which includes removal of both logging residues and small diameter trees, is one of several biomass procurement regimes that may be used to supply woody biomass for co-firing in electrical plants (Aguilar et al., 2012). The removal of woody residue previously left behind might negatively affect the abundance or quality of important microhabitat features, including downed woody material and snags, and the wildlife that depend on them, although more research is needed (Riffell et al., 2011). These concerns have led to several states adopting Best Management Practices (BMPs) specifically designed to minimize impacts of woody biomass removal on water quality, soils, biodiversity and wildlife habitat (Shepard, 2006; Skog and Stantkurf, 2011).

Efforts to conserve diverse wildlife communities would benefit from assessments of current habitat conditions and from projections of climate and land-use change effects on a suite of forest habitat classes. The Northern Forests Futures Project (NFFP), a joint effort by the USDA Forest Service and several partners, is projecting and assessing the potential impacts of climate and land-use changes on forest extent, composition, and structure across 20 U.S. states in the Northeast and Midwest. These projections are being made under common sets of assumptions about future demographic, economic, and technological conditions. For NFFP projections, the USDA Forest Service is capitalizing on forest composition and structure data provided by its Forest Inventory and Analysis (FIA) program (Woudenberg et al., 2010). Past studies have used FIA data to estimate status and trends of coarse-scale habitat characteristics, like young hardwood forest area, or area of old softwood forest (Schmidt et al., 1996; Trani et al., 2001). Finer scale habitat information for many forest-associated vertebrate species can be obtained from more detailed FIA data on tree species, size, and condition, for both live and dead trees (Nelson et al., 2011). One challenge in using FIA data for habitat assessments is relating FIA data to habitat classes contained in wildlife species-habitat matrices. For example, tree canopy cover thresholds are used to characterize NatureServe (2011) forest habitat domains (hereafter, "classes"), but historical and current FIA data do not include estimates of tree canopy cover.

Multi-species management planning is often based on coarse-filter assessments of the structure, function, and composition of habitat mosaics (Noon et al., 2009; Schulte et al., 2006). The types and areas of coarse habitat classes in a region can be used to broadly define the amount of habitat potentially available to broad suites of species (Beaudry et al., 2010). As part of the NFFP effort, we used projections of forest conditions in 2060 and ancillary data sets to assess potential changes in areas of forest habitat classes. The primary objective of this study was to assess potential changes to wildlife habitat classes over time under a suite of future scenarios assuming different trajectories for climate, land-use, and biomass utilization. We defined our habitat classes using thresholds in canopy cover, providing us with the ability to crosswalk our habitat classes to a wildlife-habitat matrix created by NatureServe and to report species richness by taxonomic group and conservation status for each class.

2 Methods

2.1 Region of Interest Our study area encompasses the USDA Forest Service's Eastern Region (Fig. 1). The Eastern Region is heavily forested relative to the whole U.S. (42% vs. 33%, respectively) and contains 32% of the nation's timberland (Shifley et al., 2010). Approximately 5 million private forest owners hold the majority (55%) of the region's forest land and mostly adopt a low intensity management approach to their lands. The region supports 124 million people (41% of U.S. population) who depend on forests to supply a wide variety of ecosystem services (Shifley et al., 2010). Among a variety of forest resource issues, stakeholders in the region are concerned about the ability of forest habitat to support diverse wildlife communities (Dietzman et al., 2011).

Well-informed forest and policy management decisions depend on assessments of the potential effects of alternative decisions on a suite of forest resources and services. As part of the 2010 Forest and Rangeland Renewable Resources Planning Act (RPA) Assessment, the USDA Forest Service used alternative future scenarios of climate change, land-use change, and human population growth (described below) to project forest and rangeland conditions in 2060 for the Eastern Region (USDA Forest Service, 2012a). We used FIA data (described below) to estimate current conditions and to project potential future forest conditions. We applied alternative RPA scenarios to project potential changes in forest habitat classes needed to sustain regionally diverse wildlife communities.

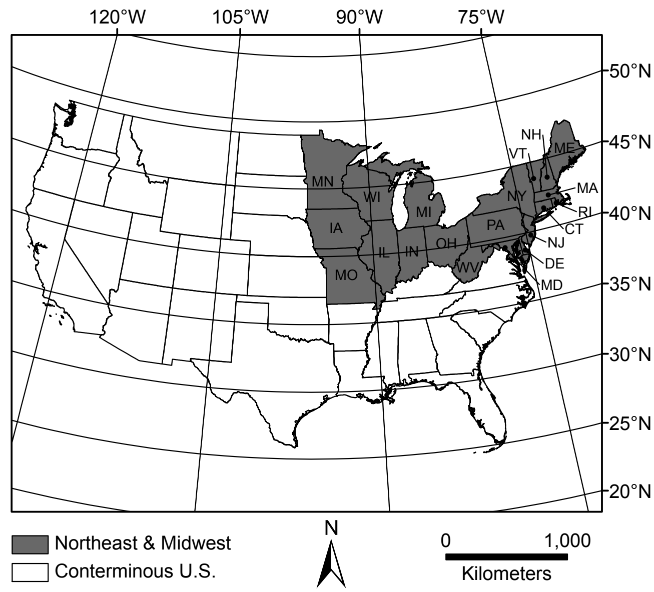

Figure 1: Map showing the location of states across the Northeast and Midwest, USA, an area corresponding to the USDA Forest Service's Eastern Region. States include: Connecticut (CT), Delaware (DE), Illinois (IL), Indiana (IN), Iowa (IA), Maine (ME), Maryland (MD), Massachusetts (MA), Michigan (MI), Minnesota (MN), Missouri (MO), New Hampshire (NH), New Jersey (NJ), New York (NY), Ohio (OH), Pennsylvania (PA), Rhode Island (RI), Vermont (VT), West Virginia (WV), and Wisconsin (WI).

2.2 FIA Data FIA's definition of forest land includes components of both land cover and land use. FIA forest land is defined as having "...at least 10 percent cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated" (Woudenberg et al., 2010, p. 47). Forest land is not developed for a non-forest use such as agriculture, residential, or industrial use, and includes commercial timberland, some pastured land with trees, forest plantations, unproductive forested land, and reserved, noncommercial forested land. FIA forest land requires a minimum area of 0.405 ha and minimum continuous canopy width of 36.58 m (Woudenberg et al., 2010). FIA sample plots follow a nationally consistent configuration comprised of a cluster of four fixed-radius circular subplots, on which land use (e.g., proportion forest cover), tree (e.g., species, height, and diameter at breast height: DBH, 1.37 m) and other site variables are collected. At least one FIA plot is selected for each 2400-ha hexagon from a nationally consistent hexagonal sampling frame. Field crews install, monument, and measure ground plots if any portion of a plot contains a forest land use (Bechtold and Scott, 2005; Reams et al., 2005). FIA began collecting tree canopy cover data only recently; such data are absent from historical FIA inventories.

FIA data from 2004-2008 were used to produce estimates of current conditions, assigned the decadal label of '2010' and referred to as 'baseline'. These same FIA data were also used to model future projections, by decade, as described below. Estimates of baseline and future conditions were produced using estimators within PC-EVALIDator tools in the Northern Forest Futures Database (NFFDB) (Miles et al., 2013).

2.3 Future Scenarios The RPA Assessment used climate (Coulson and Joyce, 2010; Coulson et al. 2010), land-use (Wear, 2011), and population (Zarnoch et al., 2010) projections consistent with greenhouse gas emission scenarios developed by the Intergovernmental Panel on Climate Change (IPCC) (USDA Forest Service, 2012a). The analyses by RPA represented an adaptation of the broad IPCC scenarios to a regional-scale through downscaling of climate change, economical, and population projections (USDA Forest Service, 2012a; USDA Forest Service, 2012b; Wear et al., 2013). Emissions scenarios were consistent with IPCC storylines that assumed different trajectories of change for global populations and gross domestic product (Tab. 1). For the RPA Assessment, the USDA Forest Service elected to use IPCC's A1B, A2, and B2 storylines because these captured a range of potential futures likely to drive variation in natural resources (USDA Forest Service, 2012a). These storylines also had marker emission scenarios that used common assumptions about driving forces in storylines, were intended to illustrate their respective storylines, and were subjected to greater scrutiny (USDA Forest Service, 2012a). While capturing a range of potential futures, these storylines are not tied to specific policy or management actions. We chose to use emission scenarios from the A1B and A2 storylines for our forest habitat assessments. We eliminated the B2 storyline because recent observations of greenhouse gas emissions (Raupach et al., 2007) suggest that projected emissions under this storyline may underestimate actual emissions.

There are many sources of uncertainty when assessing future changes in natural resources conditions (Beaumont et al., 2008). The A1B and A2 storylines capture some uncertainty by representing a range of likely future climate, land-use, and population conditions. Projected changes in climate for each storyline's emission scenario depend on the general circulation model (GCM) used to simulate future climate conditions. For the RPA, the USDA Forest Service projected future climate change using projections from three GCMs: CGCM 3.1 MR (T47) developed by the Canadian Centre for Climate Modeling, and Analysis; CSIRO MK 3.5 (T63) developed by Australia's Commonwealth Scientific and Industrial Research Organization; and MIROC 3.2 MR (T42) developed jointly by Japan's National Institute for Environmental Studies, Center for Climate System Research, University of Tokyo, and Frontier Research Center for Global Change. These GCMs had average or above average sensitivity to greenhouse gas emissions (Randall et al., 2007), showed a reasonable degree of accuracy when simulating present-day mean climate conditions (Reichler and Kim, 2008), and produced a range of future climate conditions. To address uncertainty resulting from choice of IPCC storylines and GCMs, we assessed potential changes of forest habitat conditions in 2060 under six scenarios representing unique combinations of A1B and A2 IPCC storylines and CGCM, CSIRO, and MIROC GCMs. Table 2 summarizes projected changes in climate, land-use, and population under each scenario. Maps of current and projected changes in climate conditions can be found in Tavernia et al. (2013).

Table 1: Projections of global population and global gross domestic product (GDP) associated with Intergovernmental Panel on Climate Change storylines. Source: USDA Forest Service (2012a).

Storyline	2010	2020	2040	2060
Global Population (millions)				
A1	6,805	7,493	8,439	8,538
A2	7,188	8,206	10,715	12,139
B2	6,891	7,672	8,930	9,704
Global GDP (2006 trillion USD)				
A1	54.2	80.6	181.8	336.2
A2	45.6	57.9	103.4	145.7
B2	67.1	72.5	133.3	195.6

2.4 Forest Projections Estimation of future forest conditions relied on the Forest Dynamics Model (FDM) developed by Wear et al. (2013) (Fig. 2). The FDM is a set of interlinked submodels which take an existing forest inventory and produce predictions of future inventories, given assumptions about climate, timber market conditions, and land use change. Climate, market, and land use assumptions link the forest forecasts to the IPCC storylines (Wear et al., 2013). The FIA Database (FIADB) provided the foundational data for the FDM. FIA inventories for each analysis unit were summarized at the plot level, and only plots classified as forest were maintained so that the Forest Dynamics Database of beginning inventories reflects the forest land base for 2010. For each plot for each inventory i) major forest type group was assigned based on forest type, ii) variables for physical characteristics (slope, aspect, etc.) were retained, and iii) biophysical attributes (basal area, growing stock volume, number of trees, etc.) at the population (expanded) and per-acre scale were calculated. Transition, partitioning, and imputation submodels were used to predict change in forest plot conditions through time.

The transition submodel projects changes in forest type, forest age, and harvesting and the exogenous climate models project changes in key climate variables. The transition submodel selects a specific outcome (condition) from among all possible outcomes through combinations of forest type, forest age, harvest history and climate variables represented in a probability matrix. The partitioning submodel groups plots from the 2010 baseline record (we call these donor plots) based upon a set of biophysical attributes. The plot characteristics that defined the partitioned groups included forest attributes such as stand age, slope, and ownership, as well as climate variables such as average temperature and precipitation. Given the conditions for each future plot, the imputation submodel draws a random donor plot (with replacement) from the plot's appropriate group as defined by the partitioning submodel. For example, if the transition probabilities state that a 50-year-old oak-hickory plot will become a 55-year-old oak-hickory plot (instead of an elm-ash-cottonwood plot or some other forest type group), then the model imputes what this 50-year-old plot will look like in five years by randomly picking an oak-hickory plot from the group of existing 55-year-old plots that have similar ecological characteristics. The time step for imputation was set as the span of years between re-measurements for individual FIA plots. Hence, the time step for imputation in the Northcentral and Northeast states was five years (Wear et al., 2013). FDM projections were validated by applying the model to past FIA inventories for select states and comparing the projections to present-day FIA inventories. Additionally, calibrations were made to the FDM at the state-level following reviews by USDA Forest Service FIA and state foresters and planners to account for past trends in forest change (Moser and Shifley, 2012; Wear et al., 2013).

The transition and imputation submodels used probabilistic (Monte Carlo) methods to simulate variance associated with different model components. Note that the algorithm for the imputation submodel was run 26 times with random selection for donor plots in each time step resulting in 26 "inventories". The stochastic nature of the imputation model implies that no two inventories were exactly alike. Aggregate validation was performed by comparing 95% confidence intervals for trees per acre, total biomass, and sawtimber biomass for both hardwoods and softwoods using all 26 inventories (Wear et al., 2013). Ultimately, only one inventory was used to estimate forest attributes for each future scenario in the NFFDB. The selected inventory was the one with the greatest "central tendency," defined as minimum proportional distance of total growing stock volume, trees per acre and sawtimber volume for softwoods and hardwoods from their means over the 50-year projection period (Wear et al., 2013). The completed database summarizes the results of the FDM for future decades by summarizing plot conditions for a projected future date in the same way that one would summarize current or past forest conditions (Miles, 2013).

Projections of forest conditions for combinations of IPCC storylines and CGCM included variations which accounted for higher assumed increases in woody biomass utilization in the future. While not tied to specific forest management policies, these alternative scenarios (designated as "BIO" scenarios) show substantial growth in harvesting reflecting the expansion in demands for forest biomass in bioenergy production associated with each scenario. These expansions were applied to the scenarios by adjusting harvest probabilities to reflect the harvests predicted by the U.S. Forest Products Model (Ince et al., 2011). Probability of harvest was adjusted for these scenarios in the FDM by increasing the probability of harvest by a pre-determined percentage via a scale parameter. Consequently, for each BIO scenario, the probability of harvest was increased by the same proportion across the region. Note that global assumptions regarding plantation woody biomass were included in the RPA projections; however, plantation biomass does not play a significant role in the projections from the FDM (Ince et al., 2011; USDA Forest Service, 2012a). Harvest projections for the BIO scenarios deviated from the original scenarios starting in 2020 for the A1B and A2 storylines. For the A1B-BIO storyline, harvest levels were projected to reach approximately 3 times the 2010 level by 2060. Similarly, for the A2-BIO storyline, harvest levels were projected to be roughly 2.5 times the 2010 level by 2060 (Wear et al., 2013).

Land-use change was a major consideration when developing models to project forest conditions for the selected IPCC storylines and GCMs. Land-use change was projected using econometric models developed by Wear (2011). These models were linked to historical land-use data to ensure that land-use change estimates are fairly consistent with trends in urbanization intensity and urban land-use change. The most important components used in the econometric models pertained to urbanization and allocation of rural land. Urbanization projections were driven by population and personal income projections for the IPCC storylines (Wear, 2011). The

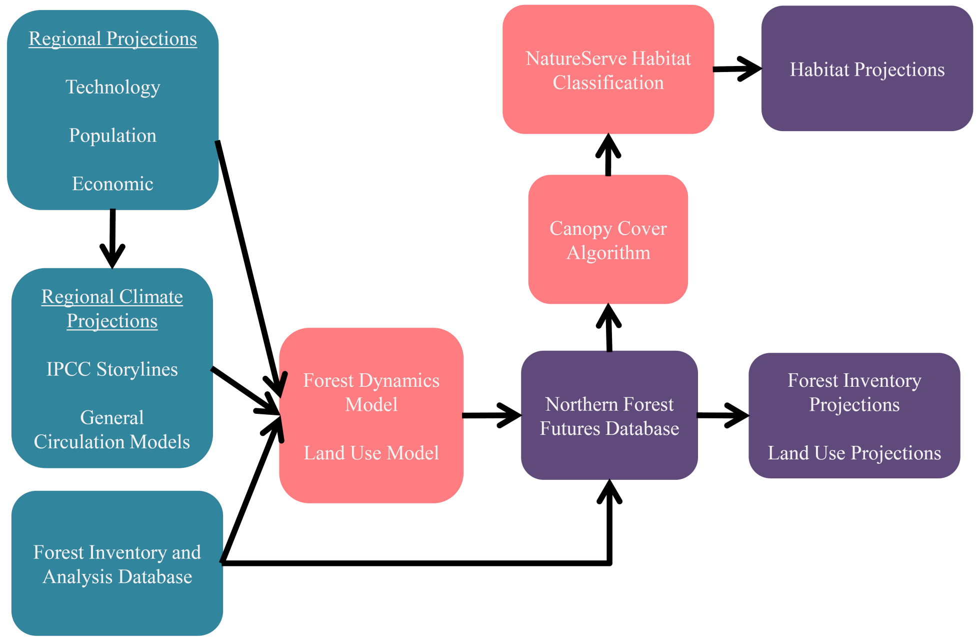

Figure 2: Modeling process used to project changes in the areas of forest habitat classes by 2060. Population, technological, economic, and climate projections served as input to Forest Dynamics and Land Use Models and drove changes in the extent and composition of forests from 2010 to 2060 at 5-year increments. Input projections represented future scenarios resulting from the combination of Intergovernmental Panel on Climate Change (IPCC) storylines and General Circulation Models. Data and trends from the USDA Forest Service's Forest Inventory and Analysis Database established 2010 forest conditions and informed projections of forest change out to 2060. Projected land-use conditions and forest inventories were summarized in the Northern Forest Futures Database (NFFDB). A canopy cover algorithm assigned canopy cover estimates to forested conditions in the NFFDB, enabling forest projections to be translated into forest habitat classes found within a NatureServe habitat classification system. Green nodes represent input data, pink nodes indicate modeling steps, and purple nodes are output products. Flowchart adapted from Wear et al. (2013).

allocation of rural land (land not converted to urban) was greatly based on the existing distribution of different non-urban land classifications from historical data. All federal land, water area, enrolled Conservation Reserve Program lands, and utility corridors were held constant for all projections (Wear, 2011).

Output from the forest projection process described above was combined with data from the FIADB (Woudenberg et al., 2010) to produce the NFFDB (Fig. 2) (Miles, 2013; Miles et al., 2013).

2.5 Habitat and Species Richness Assessments

Within the NFFDB, we assigned FIA forested conditions to six different habitat classes defined to match classes in a wildlife-habitat matrix created by NatureServe (2011) and purchased by the USDA Forest Service for the NFFP (Tab. 3). Using canopy cover thresholds, we arrayed habitat classes along two dimensions addressing differences in structure and composition. With respect to structure, we identified classes as being either closed- ($\geq$ 66% total canopy cover) or open-canopy (10 to 66%). Our closed-canopy definition is consistent with NatureServe's (2011) definition of 'forest' habitat class whereas our open-canopy definition encompasses both NatureServe's 'woodland' (40 to 66% canopy cover) and 'savanna' (10 to 40% canopy cover) habitat classes.

Following consultation with NatureServe staff (J. McNees, pers. comm., NatureServe, December 19, 2011), we included NatureServe 'savanna' in our open-canopy class because regenerating forest with sparse canopy is not synonymous with a savanna ecosystem, and because actual savanna habitat is very rare in our study area. To avoid confusion with FIA's definition of forest land, which encompasses all three NatureServe (2011) habitat classes – 'forest', 'woodland', and 'savanna', we refer to closed- and open-canopy habitat classes. Closed- and open-canopy classes were further refined based on differences in composition. We labeled areas as hardwood or conifer when > 66% of the canopy consisted of hardwood or conifer tree species, respectively. Habitats were labeled as mixed when neither hardwood nor conifer tree

Table 2: Mean monthly temperature (T) and precipitation (PPT), land-use conditions (%), and population under baseline conditions and six future scenarios for the Northeast and Midwest, USA. Future scenarios were defined using unique combinations of Intergovernmental Panel on Climate Change storylines and General Circulation Models (GCM). Mean climate conditions were calculated using area-weighted means of county-level values. Baseline and future climate data from Coulson and Joyce (2010) and Coulson et al. (2010), land-use from Wear (2011), and population projections from Zarnoch et al. (2010).

Storyline	GCM	T (°C)	PPT (mm)	Urban	Forest	Crop	Pasture	Population (mill.)
Baseline	Historical	9.1	80.5	9.4	41.4	39.3	9.9	124.1
A1B	CGCM	11.4	84.4	15.5	38.7	36.5	9.3	157.6
	CSIRO	11.5	79.8	15.5	38.7	36.5	9.3	157.6
	MIROC	13.1	72.6	15.5	38.7	36.5	9.3	157.6
A2	CGCM	11.8	83.1	14.2	39.2	37.2	9.4	178.0
	CSIRO	11.2	86.2	14.2	39.2	37.2	9.4	178.0
	MIROC	12.4	75.0	14.2	39.2	37.2	9.4	178.0

cover exceeds 66% of the total canopy cover, consistent with NatureServe's definitions.

Using the NatureServe matrix, we tabulated numbers of terrestrial vertebrate species within the study area, by major taxon (amphibians, birds, mammals, reptiles) associated with each of six habitat classes, and by global rank (NatureServe, 2011). Habitat associations reflected species' entire annual cycles, i.e., a species could be associated with a habitat type during any season. Rank is defined as follows: 1 = critically imperiled; 2 = imperiled; 3 = vulnerable to extirpation or extinction; 4 = apparently secure; 5 = demonstrably widespread, abundant, and secure. A small number of records had "T" ranks (infraspecific taxon: subspecies or varieties); these were combined with "G" ranks (global ranks), and all results were labeled as ranks (G1-G5). For the purposes of our coarse-filter assessment, we summarized numbers and global ranks to characterize wildlife communities that might be affected by projected changes in habitat classes. Projections of changing habitat associations or global ranks for individual species fell outside the scope of our study. Such species-specific assessments might be important and appropriate if the objective is to inform species-level conservation objectives, but our assessment focused on changes in coarse habitat classes.

FIA does not provide estimates of canopy cover, so we used a computer algorithm to derive estimates of canopy cover from FIA data, enabling us to crosswalk NFFDB area projections to habitat classes (Fig. 2). A canopy cover modeling approach (Toney et al., 2009) was used to estimate canopy cover for trees ($\geq$ 5 in. d.b.h., on subplots), if present, or saplings (1-4.9 in. d.b.h., on microplots) on forested FIA conditions within 20 states of USDA Forest Service's Eastern Region, during the inventory period 2004-2008. Canopy cover estimation was based on tree species-specific predicted crown dimensions, and tree stem location coordinates recorded by field crews within FIA subplots and microplots. Tree and sapling crown width predictions are based on Bechtold (2003) and Bragg (2001). An optional spatial statistic (Ripley's K) included as a predictor in Toney et al. (2009) was not utilized for canopy cover modeling in the present study. Because FIA plots may contain multiple conditions, tree and sapling canopy cover estimates were weighted based on condition proportion and appended to the CONDITION table in the NFFDB.

A small number of forested FIA conditions contained no trees or saplings. Thus, no canopy cover estimates were available for these conditions, and canopy cover could not be used to assign habitat classes to those conditions. During a plot visit, a field crew can look beyond subplot boundaries to determine some condition attributes via visual interpretation, including those conditions containing no trees at the time of field data collection. For conditions with no trees or saplings (e.g., prior to regeneration, or with only small seedlings; i.e., estimated canopy cover = 0), habitat classes were recoded to valid classes using other FIA condition attributes, described in Nelson et al. (2012). Of 52,860 forested conditions within the database, 51,398 (97.2%) contained trees and/or saplings, from which canopy cover was predicted and subsequent habitat classes were assigned. For the remaining 2.8% of forested conditions, 1.6% were assigned to one of the six habitat classes based on other condition attributes and the remaining 1.2% of conditions were labeled as 'no data' because they had neither canopy cover, nor ancillary condition data. Plots associated with the 'no data' conditions were excluded from further analyses.

Table 3: Forest habitat classes (adapted from NatureServe Habitat Classes, 2011).

Class Name	NatureServe Habitat Class	Description
Closed-canopy forest	Forest	Woody vegetation at least 6 m tall (usually much taller) with a fairly continuous and complete (two-thirds or greater) canopy closure.
Closed-canopy hardwood	Forest-hardwood	Angiosperms comprise over two-thirds of the canopy.
Closed-canopy conifer	Forest-conifer	Gymnosperms comprise over two-thirds of the canopy.
Closed-canopy mixed	Forest-mixed	Composed of both hardwood and conifer trees, neither dominating as much as two-thirds of the canopy.
Open-canopy forest	Woodland & Savanna	Crowns often not interlocking; tree canopy discontinuous (often clumped), averaging between 40 and 66 percent overall cover (Nature Serve Woodland), or, mosaic of trees or shrubs and grassland; between 10 and 40 percent cover by trees and shrubs (Nature Serve Savanna).
Open-canopy hardwood	Woodland-hardwood	Angiosperms comprise over two-thirds of the canopy.
Open-canopy conifer	Woodland-conifer	Gymnosperms comprise over two-thirds of the canopy.
Open-canopy mixed	Woodland-mixed	Stand composed of both hardwood and conifer trees, neither dominating as much as two-thirds of the canopy.

3 Results

Birds were the most numerous terrestrial vertebrate species associated with closed- or open-canopy forest habitat classes within the study area at 189 species, followed by mammals (85), reptiles (52), and amphibians (50). For every one of the six individual habitat classes, birds and mammals had most species (Fig. 3). Amphibian species outnumbered reptiles in all three closed-canopy classes; reptile species outnumbered amphibians in all three open-canopy classes (Fig. 3).

Overall, 25 of 376 species (6.6%) were listed within one of the three most at-risk ranks (G1-G3), ranging from a low of 1.1% for birds, to a high of 14.1 % for mammals. Amphibians and reptiles were intermediate, with 12.0% and 9.6%, respectively. Figure 4 presents numbers of species by global rank within each habitat class. The habitat classes with highest and lowest percentages, respectively, of at-risk species (G1-G3) were closed-canopy hardwood (7.3%), and open-canopy mixed (1.9%). Note that many species were associated with multiple habitat classes, so it is not valid to sum species counts across habitat classes in Figure 3 or 4.

Per-plot estimates of canopy cover were used to assign habitat classes. Because almost all FIA forested conditions were assigned habitat labels, total area of habitat classes was essentially equivalent to FIA forest land area for the study area (Nelson et al., 2012). Mean canopy cover of forest land across the study area was 60.4%, ranging from lows of 41.6 – 55.3% in Minnesota, Maine,

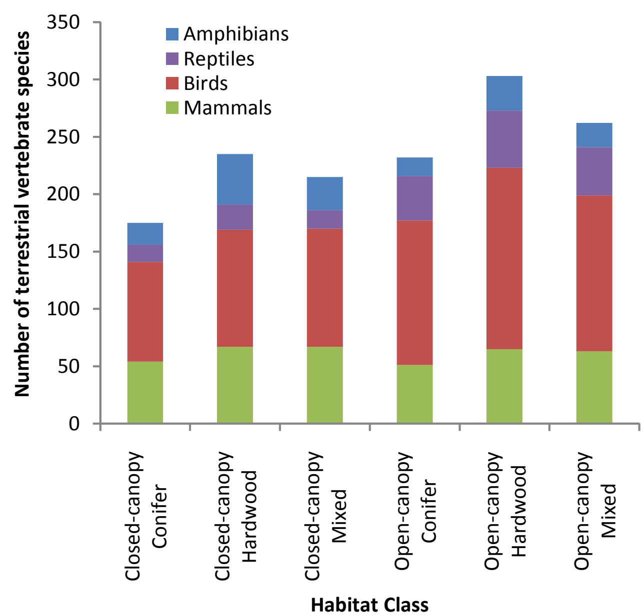

Figure 3: Number of terrestrial vertebrate species associated with closed- and open-canopy conifer, hardwood, and mixed forest, Midwest and Northeast USA, by major taxon. (Adapted from NatureServe, 2011)

Wisconsin and Michigan, to highs of 74.2 – 75.8% in Rhode Island, Massachusetts, and Connecticut.

Across the Northeastern and Midwestern U.S., total area of all forest land currently stands at 70.5 million ha.

Table 4: Area (millions of ha) and percent change of six closed- (CC) and open-canopy (OC) forest habitat classes across the Northeast and Midwest. Estimates are provided for 2010 baseline conditions and for six 2060 scenarios representing unique combinations of two Intergovernmental Panel on Climate Change storylines (IPCC) and three General Circulation Models (GCM). Two 2060 scenarios assuming intensive biomass utilization for bioenergy (A1B-BIO, A2-BIO) are also included. Changes in habitat classes between 2010 and 2060 were driven by projected climate and land-use changes, forest succession, and forest harvest. See Table 3 for explicit definitions of forest habitat classes.

IPCC	**GCM**	**Total Habitat**	**CC Hardwood**	**CC Conifer**	**CC Mixed**	**OC Hardwood**	**OC Conifer**	**OC Mixed**
Baseline	Historical	70.2	28.5	1.7	4.4	24.1	6.7	4.7
A1B-BIO	CGCM	65.7 (-6.4%)	22.8 (-20.0%)	1.4 (-17.6 %)	3.2 (-27.3%)	27.6 (14.5%)	6.4 (-4.5%)	4.4 (-6.4 %)
A1B	CGCM	65.7 (-6.4%)	29.6 (3.9%)	2.1 (23.5%)	4.6 (4.5%)	19.6 (-18.7%)	5.8 (-13.4%)	4.0 (-14.9%)
	CSIRO	65.7 (-6.4 %)	29.5 (3.5%)	2.0 (17.6%)	4.7 (6.8%)	19.6 (-18.7 %)	5.8 (-13.4%)	4.2 (-10.6%)
	MIROC	65.7 (-6.4%)	29.4 (3.2%)	2.1 (23.5%)	4.5 (2.3 %)	19.7 (-18.3%)	5.8 (-13.4%)	4.3 (-8.5%)
A2-BIO	CGCM	66.4 (-5.4%)	26.4 (-7.4%)	1.6 (-5.8%)	3.9 (-11.4%)	24.0 (-0.4%)	6.2 (-7.5%)	4.4 (-6.4%)
A2	CGCM	66.4 (-5.4%)	30.1 (5.6%)	2.0 (17.6%)	4.7 (6.8 %)	19.7 (-18.3%)	5.9 (-11.9%)	4.1 (-12.8%)
	CSIRO	66.4 (-5.4%)	29.9 (4.9%)	2.0 (17.6%)	4.6 (4.5 %)	19.8 (-17.8%)	5.8 (-13.4%)	4.3 (-8.5%)
	MIROC	66.4 (-5.4 %)	29.9 (4.9%)	2.0 (17.6%)	4.7 (6.8%)	19.9 (-17.4 %)	5.9 (-11.9%)	4.0 (-14.9%)

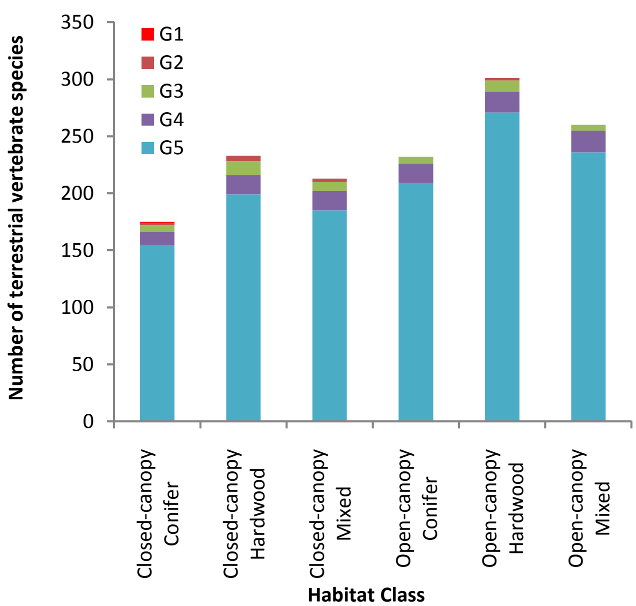

Figure 4: Number of terrestrial vertebrate species associated with closed- and open-canopy conifer, hardwood, and mixed forest, Midwest and Northeast USA, by global rank. (Adapted from NatureServe, 2011)

Of 0.6 million ha in nonstocked conditions, 0.3 million ha were assigned to habitat classes and 0.3 million ha were omitted from the habitat classification (0.4% of total forest land area), resulting in 70.2 million ha assigned to six habitat classes (Tab. 4). The region is dominated by the closed-canopy hardwood (40.6% of forest habitat) and open-canopy hardwood (34.3%) habitat classes with no other class exceeding 10% of forest habitat. Forest land is approximately evenly split between the groups of closed- and open-canopy habitat classes (49.3% and 50.7%, respectively).

Sampling errors associated with baseline per-state estimates of 2010 forest land area ranged from 0.4% (Michigan) to 4.2% (Delaware), with a median value of 1.1% across 20 states. Per-county estimates of total forest land area resulted in considerably larger sampling errors than for per-state estimates. Delaware sampling errors for per-county baseline estimates of total forest land area ranged from 6.0 – 21.1% (median = 9.8%), and per-county sampling errors for Michigan baseline estimates ranged from 1.4 – 20.1% (median = 5.6%). Due to the smaller sampling errors for per-state estimates of baseline conditions, and the additional (but unknown) uncertainty introduced in the projections modeling process, all subsequent results are reported only for per-state or region-wide scales.

Assuming standard forest harvest levels, loss of habitat area was projected under both IPCC storylines with the magnitude of loss ranging from 3.8 million ha (5.4%) under A2 to 4.5 million ha (6.4%) under A1B (Tab. 4). While projected losses for total habitat area did not differ among GCMs for either storyline, choice of GCM did affect projected changes for individual habitat classes, but these effects were relatively minor and varied across habitat classes. For example, areas for the open-canopy mixed habitat class ranged from 4.0 (CGCM) to 4.3 million ha (MIROC) whereas areas for open-canopy conifer did not vary under the A1B storyline. Patterns of change for habitat classes were consistent across both IPCC storylines (Tab. 4). All three closed-canopy forest habitat classes gained area; percent gains were greatest for closed-canopy conifer and least for either closed-canopy hardwood or mixed, depending on the GCM. Conversely, all three open-canopy habitat classes lost area; percent losses were greatest for open-canopy hardwood and least for open-canopy conifer or mixed, depending on the GCM. Closed-canopy habitat classes (54.7 to 55.3%) were projected to increase relative to open-canopy habitat classes (44.7 to 45.3%) as a percent of total habitat regardless of the scenario considered.

Under the high biomass utilization scenarios, loss of habitat area was projected under both IPCC storylines, with one exception: A1B-BIO-CGCM open-canopy hardwood class gained 3.5 million ha (Tab. 4). Across the other eleven classes, A1B-BIO-CGCM closed-canopy mixed displayed the greatest percent loss, and A2-BIO-CGCM open-canopy hardwood displayed the least (Tab. 4). Thus, patterns of change were mostly consistent, but not in magnitude across IPCC storylines (Tab. 4). Under the A1B-BIO-CGCM scenario, closed-canopy habitat classes were in the minority (41.6% versus 58.4% for open-canopy classes) whereas, under the A2-BIO scenario, the closed- and open-canopy classes remained relatively balanced (48.0% versus 52.0%, respectively).

The greatest spatial contrasts seen in Figure 5 pertain to states with well-established timber industries, such as Minnesota, Wisconsin, and Maine. These states showed general increases in forest land area for all closed canopy classes for the original scenarios, but displayed some of the greatest losses in closed canopy classes for the BIO scenarios.

4 Discussion

Adopting a coarse-filter approach, we used climate and land-use projections sharing a common set of assumptions to assess potential changes in forest habitat classes across the Northeast and Midwest from 2010 to 2060. For all scenarios considered, our assessments suggest that the total area of forest habitat classes will decrease, and this loss in total habitat area has the potential to negatively affect wildlife populations. For an individual species, the degree of these effects may depend, in part, on the spatial pattern of habitat loss. Although we do portray regional variation in habitat trends among states, we did not directly assess spatial patterns of habitat loss at a fine scale. Overall reduction in habitat area can lead to smaller and more isolated forest patches. These patches support fewer individuals and are less likely to receive immigrants from other areas, increasing the likelihood of local extirpation and decreasing likelihood of recolonization or population rescue (Hanski, 1999). Habitat in smaller forest patches in this region of North America is also more exposed to negative ecological influences (e.g., nest predators, Donovan et al., 1995) from surrounding non-forest land-uses, contributing to local population declines. Land-use and climate changes may have synergistic effects on species. For example, reduced connectivity among forest patches might influence the ability of a species to locate and occupy climatically suitable environments as these shift in response to changing climate conditions (Hannah, 2008; Opdam and Wascher, 2004). If habitat loss is widespread, regional declines and extirpations may result.

Our assessments suggest that uncertainty about future demographic, economic, technological, and climate conditions (as represented by different IPCC-GCM scenarios) contributes to uncertainty about the extent of habitat loss. While we did not quantify it, additional uncertainty arises from the unknowable possibility that future forest and land-use management actions might greatly depart from historically observed actions. Policy (e.g., promoting growth near existing urban centers) and financial mechanisms (e.g., tax deductions resulting from conservation easements) might be used to limit negative effects of land-use change on forest wildlife.

The number of terrestrial vertebrate species varied among major taxa and among closed- and open-canopy hardwood, conifer, and mixed forest habitat classes. Birds and mammals dominated species richness. The number of species at-risk rank was relatively low (6.6%), with the largest percentages observed for mammals and amphibians. While numbers of species were not projected for future conditions, consideration for at-risk species may be needed for habitat classes projected to decline in future decades.

Researchers have reported decades-long declines in the area of early successional forest habitat across the Northeast and Midwest (Trani et al., 2001). These declines

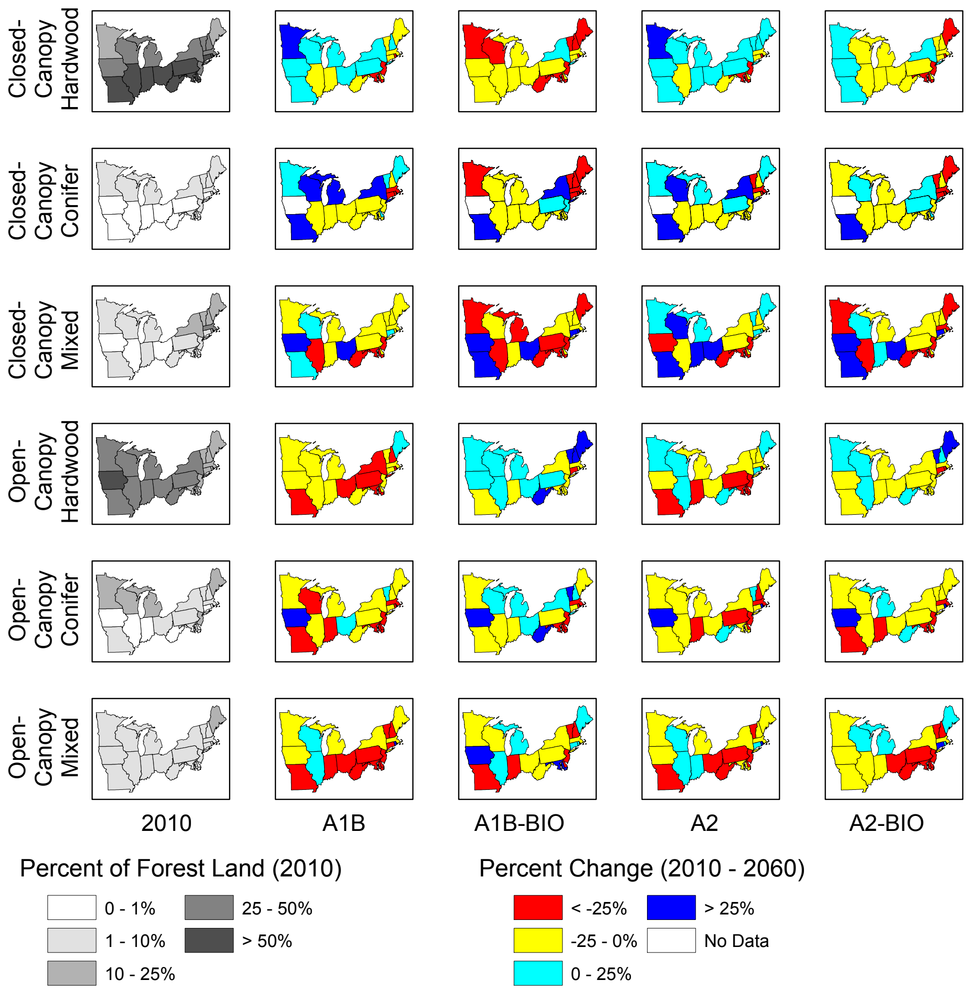

Figure 5: Percent change in area of closed- and open-canopy habitat classes, 2010-2060, by state and future scenarios. Percent of 2010 forest land area within each habitat class is shown for reference (left column). Future scenarios involved two Intergovernmental Panel on Climate Change storylines (A1B, A2) and the CGCM 3.1 MR (T47) general circulation model. Scenarios assumed either standard harvest rates (A1B, A2) or intensive harvest rates for high biomass utilization (A1B-BIO, A2-BIO).

have been attributed to a number of different causes including forest maturation of abandoned farmland, altered forest management practices, forest ownership patterns that discourage harvest, disrupted natural disturbance regimes (e.g., fire suppression), and land-use conversion (Askins, 2001; Lorimer and White, 2003; Trani et al., 2001). Assuming that early successional forests can be characterized as having more open canopies, projections of open-canopy habitat classes in our assessment suggested that declines of this habitat type may continue into the near future. With the exception of intensive biomass utilization scenarios, we found that all open-canopy habitat classes declined and that regional habitat became dominated by closed-canopy habitat classes. These projected declines may negatively affect not only open-canopy associated species but also species typically associated with closed-canopy habitats that depend upon open-canopy areas during certain times of the year (e.g., Streby et al., 2011; Vitz and Rodewald, 2006). Ultimately, the future status of wildlife species dependent on young forests or open-canopy habitat will depend on the scale, type, and frequency of anthropogenic and natural disturbances occurring in landscapes across the Northeast and Midwest.

The harvest of woody biomass for bioenergy is perceived as having the potential to mitigate climate change by alleviating, to a degree, dependence on traditional fossil fuel sources (White, 2010). Climate change mitigation policies promoting biomass harvest might increase the profitability of harvesting in stands previously seen as being non-commercial (e.g., due to poor wood quality) and lead to shorter rotation times (Janowiak and Webster, 2010). Harvest of woody biomass has the potential to open up forest canopies and turn back succession, influencing the balance between closed- and open-canopy habitat classes. Our intensive biomass utilization scenarios led to smaller decreases for open-canopy habitat classes relative to the other scenarios considered; one class (open-canopy hardwood) even displayed an increase under the A1B-BIO-CGCM scenario. Under the intensive biomass utilization scenarios, the percent cover of forest land in the open-canopy habitat classes remained stable or increased relative to current conditions. This contrasts with our other scenarios in which percent cover of open-canopy habitat classes declined and the closed-canopy habitat classes attained a slight majority. Policies and tactics associated with woody biomass harvest will partly determine the degree to which wildlife species dependent on open-canopy habitat classes might benefit. Biomass harvest for bioenergy might incentivize the removal of woody residue, or woody materials typically left behind after harvest (e.g., tops, dead wood). These materials contribute to important microhabitat conditions that can influence the habitat quality of an area. Several states have adopted BMPs specifically designed to minimize impacts of woody biomass removal on water quality, soils, biodiversity and wildlife habitat (Shepard, 2006; Skog and Stantkurf, 2011). We did not examine changes in microhabitat features as a result of intensive biomass utilization due to limitations of available FIA data and the projection technique.

Recall that some of the stark contrasts between the original scenarios and the BIO scenarios regarding canopy cover classes occurred in northern states with relatively high current levels of forest products utilization, such as Minnesota, Wisconsin, and Maine. This was greatly due to the high current probabilities of harvest within these states resulting in greater increases in harvest probability within the FDM for the BIO scenarios. This is logical, as the probability of harvest was increased by the same proportion across the region when the FDM was adjusted for higher biomass utilization (Wear et al., 2013). Consequently, many of these same states also show lower decreases (and sometimes increases) in forest area for the open canopy classes when compared to the original scenarios A1B and A2. While the variability in direction and magnitude of change among scenarios cautions against over-interpretation, these results suggest that future trends in forest habitat conditions will vary across states presenting unique challenges to wildlife managers in different areas.

Interpreting the significance of projected shifts in the representation of closed- and open-canopy habitat classes is difficult without appropriate ecological context. One viewpoint is that the historical balance between closed- and open-canopy habitat classes should be the standard because these are the conditions under which organisms evolved (Askins, 2001; Litvaitis, 2003; Lorimer, 2001; Thompson and DeGraaf, 2001). Estimating the frequency and extent of historical disturbance events and open-canopy habitat is difficult for a variety of reasons, including difficulty differentiating natural from anthropogenic disturbances and spatiotemporal variation in disturbance rates (Lorimer, 2001). To cope with temporal variability in disturbance rates, researchers have suggested managing habitat classes to maintain a balance that falls within the range of historical variability (Thompson and DeGraaf, 2001). With respect to wildlife management, it is important to consider the minimal amount of open-canopy (or other habitat class) required to support viable populations (Askins, 2001; Lorimer, 2001). Studies have indicated that species dependent on open-canopy habitat might respond to decreasing habitat areas in non-linear, threshold fashions although these thresholds might occur at relatively low levels of habitat cover (e.g., Betts et al., 2010). Identifying appropriate benchmarks for habitat management remains an active field of research.

It can be difficult to associate FIA data with habitat classes in established wildlife-habitat matrices. The method presented here provides an operational approach to predicting per-condition tree canopy cover from FIA tree data, with resulting classifications used to assign FIA conditions to closed- and open-canopy habitat classes, for which population estimates were produced. Although FIA's forest land definition requires a minimum of 10 percent canopy cover, a small area of FIA forest land was characterized by canopy cover below this threshold. Such conditions likely occur shortly after full canopy removal (e.g., harvest, wildfire, etc.), but before regenerating seedlings have established significant canopy. Tree canopy cover predictions allowed FIA data to be used with NatureServe's (2011) wildlife-habitat matrices to summarize species distribution across habitat classes. Because choice of habitat classification systems can affect resulting estimates of habitat abundance, work continues to link FIA data with a variety of habitat classifications systems, including the National Vegetation Classification System (Federal Geographic Data Committee, 2008).

A significant challenge for wildlife managers is developing and implementing large-scale and long-term plans aimed at maintaining a suite of habitat conditions suitable for diverse wildlife communities. To maintain wildlife communities in the future, wildlife managers will need to cope with landscape dynamics driven by changes in climate and land-use. Our assessments suggest that the overall area of forest habitat might decline and the balance between different habitat classes might shift in the future. The influence of assumptions about biomass utilization on the balance between closed- and open-canopy habitat classes highlights the importance of policy and management decisions in determining habitat conditions in the future. Ultimately, managers will need to identify benchmark habitat conditions informed by historical conditions and wildlife population dynamics and to develop plans to meet these benchmarks in dynamic forest landscapes.

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