EVALUATING THE EFFECTS OF HOUSING GROWTH ON ASPECTS OF FOREST ECOLOGY AND MANAGEMENT

By

Alexia Anastasia Sabor

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Executive Summary

Humans have affected forested landscapes in North America as long as people have existed there (Stearns 1997, Denevan 1992). In the latter half of the 20th century, however, forest landscapes have been affected by human-caused changes to a greater extent and more rapidly than ever before. While the total proportion of urban land area doubled between 1950 and 2000, the area of rural, low-density housing experienced a five-fold increase over the same time period (Brown et al. 2005). Increases in housing density are especially prominent in areas with attractive recreational and aesthetic amenities, such as forests and water bodies (Stynes et. al. 1997, Hammer et. al. 2004, Radeloff et. al. 2005).

Impacts on forest ecosystems due to housing growth are expected to continue at a high rate in the coming decades. By 2030, additional residential development is expected to occur on over 44 million acres of privately owned rural forest land in the conterminous U.S. (Stein et al. 2005). Unlike conversion to uses such as agriculture or disturbances such as fire, conversion to developed purposes generally represents a permanent change in land use status.

Housing growth is associated with a host of negative environmental impacts, including habitat loss and fragmentation (Gonzalez-Abraham et al. 2007), the spread of invasive species (Gavier-Pizzaro et al. 2010), and wildlife populations (Theobald et al. 2007). Residential development in and around forested areas may also reduce timber harvest and forest management activities (Wear et al. 1999, Kline et al. 2004).

The upper Midwest — Michigan, Minnesota, and Wisconsin — is one region of the U.S. that has experienced significant landscape change due to increases in housing density (Gobster et al. 2000). Although there has been some population growth in this region since the 1970s (Johnson 1998), housing density has increased far more rapidly (Gustafson et al. 2003). For example, in 1940, over 70% of the land area in Wisconsin's North Woods had fewer than five housing units/mi^2. By 1990, however, this had declined to just over 40%, while the land area with 10-20 housing units/mi^2 was nearly 4.5 times greater in size than it had been in 1940 (Hammer et. al. 2004). Most forests in the region exhibit at least low level housing density or are located less than 25 km from human settlements (Radeloff et al. 2005).

The objective of my dissertation was to analyze the effects of housing growth on different forest types throughout the United States and to examine the relationships among housing density, standing and down dead wood, and timber harvest in forests of the upper Midwest.

To achieve these objectives, I utilized data collected by the Forest Inventory and Analysis (FIA) program of the USDA Forest Service. FIA collects georeferenced data on vegetation and other natural features on approximately 128,000 forested plots throughout the U.S and is the only comprehensive source of inventory information on private and public forest land in the United States. FIA data are potentially of great value to land managers, consultants, researchers, and others interested in the scale and pattern of forest change over time and space. However, since FIA is required to alter plot locations before they are released to the public in order to prevent the direct or indirect disclosure of personal information pertaining to plot ownership, there have been many unanswered questions about the utility of using web-available FIA plot location data for ecological research. Therefore, my objective in chapter one was to quantify the amount of error introduced by using FIA data with altered plot locations in conjunction with other datasets so that researchers can evaluate whether perturbed FIA data are suitable for certain kinds of ecological research or answering management questions.

To achieve this objective, I associated FIA plot locations using both altered and true plot locations with three other geospatial datasets that a) represented a range of map unit sizes, b) are widely available, and c) are likely to be useful in answering a broad range of research questions. Although misclassification rates associated with the use of altered plot locations exhibited a strong inverse relationship to the mean map unit size of the other geospatial datasets used in the analyses, in most instances the altered plot locations did not seriously compromise the quality of the information conveyed in the results. In addition to this main finding, my approach provides others with an analytic framework to evaluate the sensitivity of their own geospatial data to errors introduced by altered FIA plot location data.

Having established that FIA data with altered plot locations could produce accurate results when used in conjunction with housing density data, in chapter two I combined these two datasets to examine which forest types in the conterminous U.S. are most likely to be affected by residential development now and in the future. I also explored how these impacts vary depending on land ownership and the extent to which broad forest categories approximate the housing densities of the individual forest types that comprise them.

I found that nearly one-fifth of all forest plots in the United States are located in areas where housing density exceeds the threshold beyond which land is no longer classified as rural, as are plots of many rare forest types. Housing densities associated with forest plots in state and local government ownership represent an intermediate level between those of federally owned public lands and those in private ownership, suggesting that use of a simple dichotomy between public and private ownership is not entirely sound. Broad forest types were not a reliable proxy for housing density of the individual forest types that comprised them, particularly in the case of rare forest types. These results are the first to examine the relative impact of housing growth on different forest types and ownerships nationwide, and provide an important step towards understanding how further housing growth may alter the conservation functions and values of forest lands throughout the U.S.

In chapters three and four, I focused more closely on the ecological and economic impacts of housing growth in one region, the upper Midwest. In chapter three, I analyzed the relationships between standing and down dead wood and residential development in Michigan, Minnesota, and Wisconsin. My results indicated that housing density, along with ecological section, ownership, stand age, live basal area and forest type are all important in predicting the abundance of snags, maximum DBH of snags, fine woody debris, and both the length and the diameter of coarse woody debris. However, these relationships are complex and patterns were not consistent among states, highlighting the need to exercise caution when generalizing even statewide studies to the regional scale.

In chapter four, I examined the relationship of housing density to the likelihood and volume of timber harvest in the upper Midwest, hypothesizing that both of these responses would be negatively correlated with increases in housing density. My results provided some evidence that increases in housing density may affect timber harvesting, but they strongly suggest that this activity is influenced by many other factors as well, particularly land ownership and stand age. In addition, my analysis indicated that ownership was the factor most likely to exert a strong influence on the decision to initiate timber harvest but did not play a significant role in determining how much timber was removed. Once the choice to engage in silvicultural management has been made, housing density, stand age, and type of timber (i.e., hardwood or softwood) were the factors that most strongly influenced how much timber was harvested.

Taken together, the results presented in this dissertation can provide forest managers and ecologists across the U.S. with a better understanding of the human dimension of forest management and land use change. The methods developed and tested in this study also offer the possibility that the data collected by the FIA program will be more widely utilized by scientists and managers. As the only consistent, nationwide source of data on U.S. forests using repeated measures on plots over time, this valuable dataset could be used to great advantage in many research applications and scenario building.

My results highlight the broad extent to which forests in the conterminous United States have already been impacted by housing growth and the degree to which they are likely to be further affected in the next twenty years. From a management perspective, our results indicate that rare forest types need to be protected from further encroachment by housing growth, but they also point to the need to ensure that more common forest types are adequately represented on public lands. In addition, given that forest lands owned by state and local governments are much more likely be located in areas where the housing density is very similar to privately owned land, non-governmental organizations involved in forest conservation should start thinking about the impacts of housing density beyond the realm of a simple public-private dichotomy.

Forest lands in federal ownership were more likely to have more standing dead trees, larger mean sizes of coarse woody debris, and greater quantities of large-fraction fine woody debris, and they were more likely to experience timber harvest than either privately owned forest plots or those owned by state and local governments. This is consistent with federal mandates to manage forests for multiple uses, a goal to which other forest owners do not necessarily subscribe. While housing density was an important in factor in predicting aspects of timber harvest and dead wood size and abundance, the relationships were often not as consistent as they were with ownership.

It has been suggested that changes in the ways forests are managed and utilized are not inherently problematic, but may simply be the result of shifts in social values and reflect efficient markets at work (Klein and Azuma 2007). While both of these may be true, negative impacts of housing growth on timber harvest and forest ecology are still cause for concern because people want to use timber products, have wildlife, and enjoy scenic beauty. Since housing densities in many areas of the country, including the upper Midwest, will likely continue to rise in coming decades, scientists, forest managers and policy makers will need to monitor the potential economic and ecological impacts of housing growth on the nation's forests.

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Chapter 1: Adding uncertainty to Forest Inventory and Analysis (FIA) plot locations: Effects on analyses using geospatial data

Abstract

The Forest Inventory and Analysis (FIA) program of the USDA Forest Service alters plot locations before releasing data to the public to ensure landowner confidentiality and sample integrity, but using data with altered plot locations in conjunction with other spatially explicit data layers produces analytical results with unknown amounts of error. We calculated the potential error from using altered location data in combination with other data layers that varied in mean map unit size. The incidence of errors associated with the use of altered plot locations exhibited a strong inverse relationship to the average map unit size of the other datasets used in the analyses. For a 30 m x 30 m resolution land cover map, plot misclassification rates ranged from 32-66%, while for ecological subsection data (mean polygon size of 9,067 $km^{2)}$ only 1-10% of plots were misclassified. Housing density data derived from the U.S. Decennial Census (mean polygon size = 5.7 km^2) represented an intermediate condition, with 5-70% of data points misclassified when altered plot locations were used. These analyses demonstrate the impacts of altering FIA plot locations and represent an important step toward making the FIA database more helpful to a broad variety of end users.

Introduction

As contemporary organizations gather, analyze, and share large quantities of personal and household data, maintaining the privacy of their subject groups is becoming increasingly important (Muralidhar and Sarathy 2005). In particular, the confidentiality of personal information collected electronically by businesses, the medical industry, and the government has become a topic of great concern (Chen and Rea 2004; O'Herrin et al. 2004). Both private and public institutions increasingly struggle to balance their obligation to protect the privacy of the individuals who are the source of these data against users' needs for accurate information (Domingo-Ferrer et al. 2004). To minimize the possibility of disclosing personal information, data-collecting institutions and agencies may apply a variety of masking procedures (Brand 2002; Domingo-Ferrer et al. 2004; Lechner and Pohlmeier 2004). Data masking necessarily involves some information loss, but the magnitude of such losses and their potential effects on the accuracy of data analyses are unknown.

The FIA database of the USDA Forest Service provides an excellent case study of the conflicts that arise as agencies attempt to balance users' needs while maintaining data privacy and sample integrity. FIA collects georeferenced data on vegetation and other natural features on approximately 128,000 forested plots throughout the U.S, and is the only comprehensive source of inventory information on private and public forest land in the U.S. Data collected on each plot from the 1970s to the present are available in tabular format on the Internet (http://www.fia.fs.fed.us/tools-data/data/). This database is of great potential value to land managers, consultants, researchers, and others interested in the scale and pattern of forest change over time and space. For example, FIA data have been used to examine rates of timber harvest (Munn et al. 2002), monitor the effects of climate change (Iverson and Prasad 1998; Stolte 2001), predict tree species distribution (Schwartz et al. 2001), and assess damage caused by natural disasters (Faust et al. 1994).

While the geographic coordinates and landowner information included in the FIA database allow spatially explicit analyses, the program has long been concerned that disclosing precise plot locations could compromise sample integrity. First, such disclosure may attract other activities that either intentionally or unintentionally affect plot composition (e.g., damaged trees, trampled vegetation, and compacted soils), thereby altering inventory results. Second, disclosures of exact plot locations may make public certain proprietary information on growth and yield or management practices, resulting in landowners' refusing to allow repeated FIA assessments on their property (McRoberts et al. 2005). In 2000, this concern was formalized when the U.S. Congress, as part of the Interior and Related Agencies Appropriations Act (H.R.3423), mandated that FIA plot location and ownership data receive confidential treatment. This language prevents FIA from disclosing sample locations to individuals outside the program in such a way that individual land ownership and other proprietary information could be determined with certainty. Moreover, the risk of revealing landowners' personal information has grown as the FIA program increasingly works with state agencies, universities, other federal agencies, and contractors to implement fieldwork, analysis, reporting, and monitoring.

Due to the new legislation and the variety of program partners, a new policy regarding the direct or indirect release or disclosure of personal information pertaining to plot ownership had to be developed. Thus, the Web-available plot locations released by FIA are now altered in two ways. The majority of FIA plots undergo perturbation, in which the plot coordinate data are altered but still are located within a 1.6 km radius of the true plot location. A much smaller subset of privately owned plots also undergoes swapping, in which the plot location data are first perturbed and then exchanged with data from other plots similar in both ownership and ecological condition (Lister et al. 2005). Users do not have any way of discerning either the extent to which plot locations have been perturbed within the 1.6 km radius or exactly which plot locations may have undergone swapping.

The FIA program's intent is to maintain the ecological validity of its data while decoupling plot-landowner information by adding uncertainty to plot locations. Prior studies concluded that perturbing and swapping have minimal effects on analyses of variables included in the FIA database if the area of interest (AOI) is large enough. For example, a study by Lister et al. (2005) showed that adding uncertainty to FIA plot locations had steadily decreasing effects on multiplot estimates of board-foot volume as circular AOIs increased from 5 km to 20 km in radius. McRoberts et al. (2005) similarly found that perturbing and swapping had negligible effects for design-based estimation of forest attributes included in FIA when the radii of circular AOIs exceeded 30 km. Many users, however, are interested in examining the relationships between FIA data and other spatially explicit data, either in raster format or containing irregularly shaped polygons, on finer scales that reflect typical private ownerships or correspond to community interests in political or economic activities. Errors incurred when conducting such analyses using perturbed FIA plot location data may or may not be similar to those incurred when using circular AOIs or when examining only data included in the FIA database. For example, Coulston et al. (2006) found that the extent to which perturbed FIA plot locations influence the development and accuracy of linear regression models is significantly affected by the cell size and spatial autocorrelation among cells of the raster datasets containing the independent variables. Thus, there are many unanswered questions about the utility of altered FIA plot location data for ecological research.

FIA Spatial Data Services (SDS) was created to facilitate the connection between user-generated geospatial data to FIA's true geospatial information to generate derived products that comply with the confidentiality law (USDA Forest Service 2004). Although SDS Centers play a valuable role in meeting the needs of those who wish to use FIA data, many users will find SDS Centers too geographically distant to visit themselves and SDS will face limitations in their ability to address all users' requests within a reasonable time frame. Therefore, our objective was to quantify the amount of error introduced by using FIA data with altered plot locations in conjunction with other datasets so that researchers can evaluate whether perturbed FIA data are suitable for conducting certain kinds of ecological research or answering management questions. To do so we chose three datasets that represent a range of map unit sizes, are widely available, and are likely to be useful in answering a broad variety of research questions: 1) a 30 m x 30 m land cover classification, 2) census partial block group data with polygon sizes ranging from <0.01 km^2 to 1,640 km^2, and 3) ecological subsection data with polygon sizes ranging from 469 km^2 to 80,600 km^2.

Methods

Study area

The study region includes Michigan, Minnesota, and Wisconsin, an area covering 494,014 km^2. This area is characterized by cold, snowy winters and warm, humid summers with precipitation evenly distributed throughout the year. A gradual transition zone, defined by temperature, frontal movement, and vegetation extends from north-central Minnesota to southeastern Wisconsin and then across the Lower Peninsula of Michigan (Stearns 1997).

Within the study area, approximately 210,000 km^2 are designated as forest land, defined by the USDA Forest Service as a minimum land area of 0.41 ha in size that is at least 10% stocked by forest trees of any size or that formerly had such tree cover and that is not currently developed for a nonforest use (Bechtold and Patterson 2005). Predominant forest types in these three states include maple/beech/birch, aspen/birch, spruce/fir, and oak/hickory (Shifley and Sullivan 2002). This region encompasses a wide range of land cover types, varies greatly in housing density, and includes many ecological subsections, making it particularly suitable for a study of this type.

Data sources

Forest Attributes

Our analyses were conducted using FIA plot location data collected from Michigan, Minnesota, and Wisconsin during 2000-2003. FIA field survey plots occur at an intensity of approximately one 0.41 ha plot per 1,200 ha in Minnesota and Wisconsin and one plot per

800 ha in Michigan (R. McRoberts 2005, personal communication). Field survey personnel collect quantitative and qualitative data on stand condition, land use, ownership, timber volume, tree species, and tree condition (Miles et al. 2001). We used FIADB version 1.7 downloadable data files from the National FIA Data Base Retrieval website (*http://ncrs2.fs.fed.us/4801/fiadb/index.htm*) to obtain the publicly available perturbed and swapped plot coordinates. Exact plot coordinates were obtained and analyzed at the USDA Forest Service Spatial Data Services Center in St. Paul, Minnesota.

Land Cover

We used the National Land Cover Data (NLCD) derived from Landsat Thematic Mapper (TM) satellite imagery circa 1992. The TM images, combined with supporting information such as topography, census, agricultural statistics, soil characteristics, and other land cover maps, have been classified into a hierarchical, 21-class land cover scheme applied consistently over the U.S. at a 30 m x 30 m resolution (Vogelmann et al. 2001). Eighteen of the 21 cover classes occur within the study region, with major land cover classifications in the study region including herbaceous cultivated (36.3%), forested upland (27.0%), water (20.8%), and wetlands (12.9%). Developed area accounts for 1.7% of land cover, while all other cover types account for less than 2.0% of the total surface area of this region.

We also aggregated NLCD data into eight broader aggregations of these 18 classes (e.g., coniferous, deciduous, and mixed forest all became simply ‘forest’) and calculated the average area of all patches of contiguous pixels formed by grouping pixels of like classes into homogeneous landscape units using an 8-neighbor rule. When we did this, the average patch size across all categories was 0.41 km^2 (stdev = 96.35 km^2).

Housing density

Housing density for the year 2000 was estimated using U.S. Decennial Census data at the partial block group (PBG) level via methods developed by Hammer et al. (2004). Due to concerns about privacy and sampling error, certain data are released only for aggregations of census blocks (block groups). However, block groups are divided by a variety of political boundaries, such as congressional districts and minor civil divisions, which permit division into multiple partial block groups. PBGs have a mean size one-tenth that of block groups and therefore provide a much better spatial resolution while including the complete array of population and housing attribute information available at the block group level. Sizes of PBGs in the study region range from <0.01 to 1,640 km^2 (mean = 5.7 km^2), while housing densities range from 0.0 to 16,945 units/ km^2 (mean = 71.45 units/km^2).

Ecological subsections

The National Hierarchical Framework of Ecological Units divides the country into progressively smaller areas of land and water based on physical and biological characteristics and ecological processes (Cleland et al. 1997). Ecological subsection boundaries are typically delineated by discrete changes in surficial geology (Great Lakes Ecological Assessment 2004). For these analyses we used ecological subsections as delineated by USDA Forest

Service ECOMAP (McNab and Avers 1994), which included 90 ecological subsections ranging in size from 469 to 80,600 km^2 (mean = 9,067 km^2).

Data analyses

Individual FIA plots were classified as perturbed or swapped based on the linear distance between true and altered plot locations. Those plots having a linear distance of $\leq$ 1.6 km between true and altered plot coordinates were categorized as perturbed while those with linear distances > 1.6 km were considered swapped. This threshold was chosen based on the maximum extent to which plot coordinates are perturbed and was intended to ensure that the subset of data categorized as swapped did not include any data points that were merely perturbed. Information on land cover type, housing density, and ecological subsection was associated with each true FIA plot location and its perturbed or swapped counterpart in a Geographic Information System (GIS). Information on land cover, housing density, or ecological subsection was missing from some plot locations because perturbation or swapping moved the plot location outside the study region or into a water body. These plot records were eliminated, as were duplicate plot records, yielding 21,498 records for perturbed plots and 491 records for swapped plots. In general, perturbed and swapped data were analyzed separately. In some instances, however, all 21,989 plots were analyzed together to determine whether there were any differences in the results because users will not be able to differentiate between perturbed and swapped plot location data when using FIA data available on the Internet. In those instances, both the results from individual and combined analyses are reported.

We graphed housing densities derived using true plot location data against those derived using perturbed and swapped FIA locations on a log-log scale to examine whether there was a linear relationship between these two sets of results. Data points that did not fall on a straight line were examined on a map to determine whether the locations of these plots exhibited any distinctive spatial pattern, such as clustering around public lands or water bodies. In addition, we created residual plots comparing housing densities at true and altered plot locations to assess whether there was any bias to estimates derived using altered plot locations. We used Pearson correlation coefficients and their p-values to evaluate the linearity and strength of the relationship between true, perturbed, and swapped locations both by state and over the entire study area and performed paired t-tests to assess differences in mean housing density values among these data. Correlation coefficients and paired t-tests for housing densities derived using perturbed plot coordinates were analyzed at the county and ecological subsection level but such analyses were not possible using swapped plot locations due to insufficient sample sizes.

Since NLCD and ecological subsection data are categorical rather than continuous, we could not perform the same types of analyses on these data as on housing density. Instead, we calculated the percentage of changes that occurred in our results when using either true or perturbed FIA plot locations. In addition, we computed simple Kappa (κ) coefficients for NLCD and ecological subsection data. Kappa is a measure of agreement between two categorical datasets that assumes a value between 0 and 1, with 1 being complete agreement between datasets. Kappa is positive whenever the observed agreement exceeds chance agreement, with its magnitude reflecting the strength of agreement (Cohen 1960). NLCD data were analyzed using both the full suite of 18 land cover classes in the study region and the eight broader aggregations of these 18 classes.

Results

In general, the magnitude of the effects of perturbing and swapping FIA plot coordinates strongly depended on the mean map unit size of the comparative dataset used in the analyses. Here, we review our results in order from finest- to coarsest-scale data layers.

National land cover data

When we used the full set of 18 land cover types, 51.5% of perturbed plots and 66.8% of swapped plots exhibited differences in land cover when compared to those derived using true FIA plot coordinates (Tables 1 and 2). When compared to land cover types derived using true plot locations, Kappa coefficients for perturbed data ($\kappa = 0.36$) and for swapped data ($\kappa = 0.16$) indicated considerable lack of agreement between these datasets. Although many of these land cover type changes occurred between closely related cover types, such as coniferous and deciduous forest, the use of aggregated land cover categories still resulted in the misclassification of 32.7% of perturbed plots ($\kappa = 0.50$) and 51.7% ($\kappa = 0.36$) of swapped plots (Tables 3 and 4), again indicating a strong lack of agreement between these datasets. Upon examining a map, we saw no spatial patterning among plots that changed NLCD categories due to the added uncertainty in the FIA plot coordinate data.

Data analyses using the combined set of perturbed/swapped plot location data yielded results similar to those for perturbed data alone, with land cover type changes occurring

51.7% of the time when we used all 18 land cover types and 33.9% of the time when we used aggregated land cover categories.

Housing density

Graphs of log transformed housing density for true versus perturbed or swapped coordinates demonstrated a distinctly linear relationship but included considerable scatter (Figs. 1 and 2). When we mapped plot locations for those points that fell along the axes (i.e., exhibited a housing density of zero for either the true or perturbed/swapped locations but not both), it was apparent that many of these plots occurred in areas exhibiting high spatial heterogeneity in housing density, such as PBGs with no houses intermixed with PBGs containing 5 to 64 houses per km^2.

Residual plots for both perturbed and swapped plot locations (Figs. 3 and 4) exhibited lines of points: 1) down the y axis due to instances in which actual housing densities equaled 0 and the value derived from altered plot locations was greater, and 2) along the diagonal due to instances where actual housing densities were greater than 0 but the value derived from altered plot coordinates was 0. There was no apparent bias in estimates based on altered plot locations in cases where neither the actual nor the estimated housing density was greater than zero.

Housing densities derived using true FIA plot locations were highly correlated (Pearson $R = 0.68$, $P < 0.0001$) with those derived using perturbed locations. Approximately 85% of all plots exhibited a housing density difference of ≤ 0.5 units/km^2 between true and perturbed coordinates. In about 5% of cases, however, differences of > 10 housing units/km^2 resulted from using perturbed plot coordinates (Fig. 5). Similarly, housing densities derived using swapped plot locations were significantly correlated with those derived using true coordinates (Pearson R = 0.41, P < 0.0001), although less strongly than those derived using perturbed plot location data. Swapping resulted in nearly 41% of all plots exhibiting a housing density difference of $\leq$ 0.5 units/km^2 and approximately 17% exhibiting differences of > 10 housing units/km^2 as compared to the housing densities derived using true plot coordinates (Fig. 6).

Housing density data derived using perturbed or swapped coordinates exhibited very similar distributions in relation to true coordinates (Figs. 7 and 8). Paired t-tests confirmed there were no significant differences between the means for these datasets either when all plots in the study area were included in the analysis or when tests were performed on data for individual states. When aggregated at the county level, housing densities derived using perturbed FIA plot locations were significantly different at the $\alpha = 0.05$ level from those derived using true plot locations only 3% of the time. Likewise, when housing density was aggregated to the level of ecological subsection, values derived using true plot locations differed significantly only 7% of the time from those derived from perturbed plot locations. When mapped, there was no clear relationship between the amount or spatial patterning of housing development that could explain the significant differences in housing densities at the county level.

Data analyses using the combined set of perturbed/swapped data indicated that approximately 84% of all plots exhibited housing density differences of $\leq$ 0.5 units/km^2, a figure comparable to that for perturbed data alone. Similarly, paired t-tests showed no significant differences between mean housing densities for perturbed/swapped versus true plot location data when examined across the study area or by state. Mean housing densities for counties differed significantly at approximately the same rate (3%) as when perturbed data were analyzed alone; however, mean housing density was significantly different from that derived using true plot locations approximately 13% of the time.

Ecological subsections

When we used perturbed coordinates, FIA plots changed ecological subsection only 0.5% (n=107) of the time when compared to true plot locations. Among those plots that changed subsection, 27% (n=29) were moved to a different section within the same province and about 6% (n=6) were moved to a different province. Changes in ecological subsection also occurred relatively infrequently when swapped coordinates were used, affecting only 10.3% (n=43) of plots in this sample. Of this 10.3% of affected plots, 14% (n=6) changed sections within the same province and 21% (n=9) switched to a different province. Simple Kappa statistics also indicated strong levels of agreement between ecological subsections derived using true, perturbed, and swapped plot locations, with $\kappa = 0.99$ for perturbed data and $\kappa = 0.91$ for swapped data. Analysis of the combined perturbed/swapped data provided results similar to those obtained using only perturbed data, with ecological subsection changes occurring at a rate of less than 1%.

Discussion

FIA perturbs and swaps plot location data in order to comply with the law and maintain the ecological integrity of their sample plots while still providing useful data to outside users. Our results suggest that perturbed and swapped FIA plot locations may be used in correlative studies with other spatially explicit data layers having a wide range of polygon shapes and sizes without seriously compromising the quality of the information conveyed in the results. However, the misclassification rates associated with the use of altered plot locations exhibits a strong inverse relationship to the mean map unit size of the other geospatial datasets used in the analyses. Thus, we suggest that users carefully evaluate the appropriateness of using perturbed and swapped plot locations for any other geospatial dataset they wish to use in combination with FIA data.

When using coarse-scale ecological subsection data, we found that relatively few data points were misclassified due to the inclusion of uncertainty in FIA plot location data. Of those plots that were assigned an incorrect ecological subsection, the majority (65-67%) were assigned to a different subsection within the same ecological section as the correct subsection and usually immediately adjacent to it. Since ecological subsections represent very broad ecotones rather than discrete ecological boundaries (Rowe 1996), the analytical consequences of occasionally misclassifying the subsections in which FIA plots occur are probably negligible in most cases.

Conversely, combining 30 m x 30 m-pixel NLCD data resulted in frequent misclassification of land cover type when perturbed or swapped data were used. Even when we combined land cover data into broader categories, many points were still misclassified using perturbed and swapped data. Although the average patch size increased to 0.41 km^2 when we aggregated contiguous cells with the same broad land cover classification, a size more than 450 times greater in area than a 30 m x 30 m pixel, these data still represent a much finer spatial scale than the other data layers used in this analysis. In addition, even when pixels were aggregated the incidence of single pixels averaged 44.2% across all land cover categories. Although smoothing these data by merging isolated pixels with their neighbors or by specifying a minimum patch size would most likely result in fewer differences between true and perturbed or swapped locations, the extent to which these practices would affect analytical outcomes is beyond the scope of the present study. Based on our results and the likelihood of relatively high levels of misclassification in fine-resolution data such as NLCD, particularly in highly heterogeneous landscapes (Smith et al. 2003), we do not advise using data with such a small mean map unit size in conjunction with perturbed FIA plot locations.

Housing density represents an intermediate situation, in which a small to moderate number of data points are affected by the inclusion of uncertainty in FIA plot location data. The majority of all FIA plots showed differences of $\leq$ 0.5 housing units/ km^2 regardless of whether true or altered plot locations were used. These differences were not statistically significant for the overall study region or individual states and only infrequently (3-13% of the time) significant when plots were aggregated at the ecological subsection or county level. Nonetheless, there are a small number (5-17%) of instances in which using altered locations resulted in differences of > 10 housing units/ km^2 per plot. These differences are most likely to occur in regions with high levels of heterogeneity in housing densities, and thus researchers should use particular caution when associating FIA and PBG data in such areas.

Our results both concur with and differ from those of prior studies of the impacts of altering plot locations on data analyses. Overall, our findings agree with those of Lister et al. (2005), McRoberts et al. (2005), and Coulston et al. in concluding that analyses using fine-scale data are more likely to reflect the effects of perturbation and swapping in FIA plot location data (Fig. 9). Coulston et al. (2006) indicated that perturbed plot locations should only be used with fine (30m -500m) resolution raster data with a high degree of contagion (i.e., exhibiting generally clumped patterns of landscape categories, cf. Li and Reynolds 1993, Riitters et. al. 1996). Our results reinforce their conclusions, as we found extremely high rates of misclassification when using altered plot locations in conjunction with NLCD data. The small mean patch size and high incidence of single cells even after we aggregated pixels using eight categories indicate these data do not exhibit high degrees of contagion, thus explaining the minimal improvement in misclassification rates we observed. Conversely, there were few statistically significant differences between housing densities calculated using altered and true plot locations despite the relatively small average size (5.67 km^2) of housing PBGs. Since the mean PBG size is much smaller than the minimum circular AOI size prior studies (Lister et al. 2005, McRoberts et al. 2005) have found necessary to mitigate the effects of altering plot locations, our results suggest that it may be possible to use perturbed and swapped FIA data in conjunction with finer scale data than was previously supposed.

Although perturbed and swapped data exhibit similar overall patterns of error, the magnitude of the error generated using swapped data is generally greater than that from perturbed data. Thus, to obtain a particularly conservative error estimate it would be desirable to treat all privately owned plots as if they had been swapped. On the other hand, analyses in which we combined both perturbed and swapped data generally produced results most comparable to those derived using perturbed data alone. We surmise this is the case because only a small percentage of plots are swapped, and thus perturbed plots are more important in influencing the outcomes of data analyses. Thus, we suggest that in most instances it will not be necessary for users to treat privately owned plots as a separate case. We also note that while the threshold distance value used in this study ensured accurate categorization of all plots in the subset of swapped data, it is likely that the subset of perturbed data include some plots that are swapped since FIA has no criterion specifying a minimum distance beyond which plots must be swapped. Given that only a small proportion of plots are ever swapped, however, we feel confident that the number of such plots analyzed as perturbed is small and the effects of their inclusion are negligible given the large sample size of perturbed data.

Our results, along with those of Coulston et al. (2006), highlight the importance of landscape configuration and contagion as well as map unit size. While not a factor explicitly addressed in this study, we found that analyses using NLCD data were not much improved when like pixels were aggregated and that housing density was most likely to be assigned incorrectly in areas where this attribute was very heterogeneous. Landscape pattern is widely recognized as an important component of ecological study (Turner et al. 2001) and thus we suggest users pay particular attention not only to the size of their map units but to the configuration of those units when deciding whether using altered FIA data is appropriate for their purposes.

Our results can provide researchers with an analytic framework to evaluate the sensitivity of their own geospatial data to errors introduced by altering FIA plot location data.

For those who wish to use land cover, census, or ecoregion data or other geospatial datasets with similar mean map unit sizes, the results of this study may be directly applicable to their analyses. In other cases, users can perform simple analyses to determine how much their results would change due to perturbation of plot locations. For example, users could re-perturb web-available FIA plot locations randomly within a 0.8 km radius circle, perform analyses such as we have done, and use these results as a proxy for the difference between true and perturbed plot locations. Alternatively, users could evaluate the likelihood that altered plot locations would affect their analyses by placing a 1.6 km buffer around the plot and calculating the percentage of the buffer area that falls in a different polygon. Plots with a high probability of changing to a dissimilar polygon could then be removed from the dataset provided this did not bias the sample.

Finally, we suggest that further research along the lines of the present study and those conducted by Lister et al. (2005), McRoberts et al. (2005), and Coulston et al. (2006) are needed to help maintain confidentiality while continuing to making this valuable database more broadly available to those who require spatially explicit information to answer questions about forest ecology, management, and policy.

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Table 1. Confusion matrix for 18 land cover categories using perturbed FIA plot locations (n=21,498).

Overall accuracy = 0.49

* indicates insufficient cell count to ensure confidentially if reported

OW = open water, LIR = low intensity residential, HIR = high intensity residential, CIT = commercial/industrial, RSC = rock/sand/clay, QMG = quarries/mines/gravel pits, TB = transitional barren, DF = deciduous forest, EF = evergreen forest, MF = mixed forest, SL = shrubland, GH = grassland/herbaceous, PH = pasture/hay, RC = row crops, SG = small grains, URG = urban/recreational grasses, WW = woody wetlands, EHW = emergent herbaceous wetlands . (Next page).

NLCD category for true plot locations

NLCD category for perturbed plot locations

	OW	LIR	HIR	CIT	RSC	QMG	TB	DF	EF	MF	SL	GH	PH	RC	SG	URG	WW	EHW	User's accuracy
OW	0.06	*				*		0.01	0.01	0.01			<0.01	<0.01			0.01	0.02	0.21
LIR	0.02	0.34	0.26	0.11		*		<0.01	*	*		*	<0.01	<0.01		0.24	*	*	0.32
HIR	*	0.14	0.26	0.10				<0.01		*		*	*	<0.01		0.09		*	0.27
CIT	0.01	0.07	0.13	0.22		*		<0.01	*	*			0.01	<0.01	*	0.06	<0.01	*	0.22
RSC					*			*	*			*							0.40
QMG		*	*			0.39		*	*	*			*	*				*	0.52
TB							0.14	0.01	0.01	0.01	*	*	*	*			<0.01	*	0.13
DF	0.30	0.11	0.14	0.15	*	0.23	0.28	0.49	0.24	0.37	0.34	0.37	0.18	0.09	0.10	0.13	0.18	0.16	0.45
EF	0.09	*		*		*	0.09	0.05	0.31	0.13	*	0.06	0.02	0.01	*	*	0.05	0.02	0.32
MF	0.07	*	*	*		*	0.14	0.07	0.13	0.18	*	0.06	0.01	<0.01	*		0.05	0.02	0.18
SL	*							<0.01		*	*			*			0.01	*	0.08
GH	*	*					*	0.02	0.02	0.01		0.20	0.01	<0.01			0.01	0.01	0.14
PH	0.13	0.08	*	0.09		*	*	0.09	0.03	0.04	*	0.08	0.35	0.15	0.13	0.11	0.05	0.13	0.35
RC	0.10	0.12	0.07	0.18		*	*	0.12	0.06	0.04		0.10	0.34	0.67	0.25	0.18	0.05	0.17	0.69
SG	*	*						<0.01	*				0.01	0.01	0.38	*	*	0.03	0.44
URG	*	0.03	0.07	0.06				<0.01	*	*			<0.01	<0.01	*	0.15	*	*	0.16
WW	0.14	0.04	*	0.03		*	14.00	0.11	0.15	0.17	0.23	0.07	0.04	0.02	0.04	*	0.53	0.17	0.51
EHW	0.05	*	*	0.03		*	*	0.02	0.03	0.02	*	*	0.03	0.02	0.07		0.06	0.25	0.27
Producer's accuracy	0.06	0.06	0.34	0.26	0.22	0.67	0.39	0.14	0.49	0.31	0.18	0.09	0.20	0.35	0.67	0.38	0.15	0.53	

Table 2. Confusion matrix for 18 land cover categories using swapped FIA plot locations (n=491).

Overall accuracy = 0.33

* indicates insufficient cell count to ensure confidentially if reported

OW = open water, LIR = low intensity residential, HIR = high intensity residential, CIT = commercial/industrial, RSC = rock/sand/clay, QMG = quarries/mines/gravel pits, TB = transitional barren, DF = deciduous forest, EF = evergreen forest, MF = mixed forest, SL = shrubland, GH = grassland/herbaceous, PH = pasture/hay, RC = row crops, SG = small grains, URG = urban/recreational grasses, WW = woody wetlands, EHW = emergent herbaceous wetlands . (Next page).

NLCD category for true plot locations

NLCD category for swapped plot locations

	OW	LIR	HIR	CIT	RSC	QMG	TB	DF	EF	MF	SL	GH	PH	RC	SG	URG	WW	EHW	User's accuracy
OW																		*	0.00
LIR								*		*									0.00
HIR								*						*					0.00
CIT																			n/a
RSC														0.38					0.00
QMG														*					0.00
TB																	*		0.00
DF				*				0.44	0.48	0.46	*	*	0.21		*	*	0.24	*	0.57
EF								0.03	*	*							0.08		0.05
MF							*	0.05	*	*			*				*		0.17
SL								*									*		0.00
GH								*	*			*	*						0.13
PH								0.14		*			0.28	0.16			0.13	*	0.12
RC								0.15	*	0.14	*	*	0.31	0.32		*	0.13	*	0.19
SG								*											0.00
URG								*											0.00
WW								0.11	*	*		*	*	*			0.30	*	0.38
EHW	*							0.04	*	*			*	*			*	*	0.04
Producer's accuracy	0.00	n/a	n/a	0.00	n/a	n/a	0.00	0.44	0.04	0.11	0.00	0.14	0.28	0.32	0.00	0.00	0.30	0.11	

Table 3. Confusion matrix for 8 aggregated land cover categories using perturbed FIA plot locations (n=21,498).

Overall accuracy = 0.67

* indicates insufficient cell count to ensure confidentially if reported (Next page)

NLCD category for true plot locations

NLCD category for perturbed plot locations

	Water	Developed	Barren	Forested	Shrubland	Herbaceous natural	Herbaceous planted	Wetland	User's accuracy
Water	0.06	*	*	0.01			<0.01	0.01	0.21
Developed	0.04	0.53	*	0.01		*	0.01	0.01	0.52
Barren		*	0.23	0.01	*	*	<0.01	<0.01	0.24
Forested	0.46	0.14	0.45	0.63	0.54	0.49	0.13	0.26	0.59
Shrubland	*			<0.01	*		*	<0.01	0.08
Herbaceous natural	*	*	*	0.02		0.20	<0.01	0.01	0.14
Herbaceous planted	0.23	0.21	*	0.18	*	0.19	0.79	0.16	0.81
Wetland	0.20	0.10	0.21	0.15	0.26	0.09	0.06	0.54	0.53
Producer's accuracy	0.06	0.53	0.24	0.63	0.09	0.20	0.80	0.54	

Table 4. Confusion matrix for 8 aggregated land cover categories using swapped FIA plot locations (n=491).

Overall accuracy = 0.48

* indicates insufficient data to ensure confidentially restrictions if released (Next page).

NLCD category for true plot locations

NLCD category for swapped plot locations	Water	Developed	Barren	Forested	Shrubland	Herbaceous natural	Herbaceous planted	Wetland	User's accuracy
Water								*	0.00
Developed				*			*		0.00
Barren								*	0.00
Forested		*	*	0.54	*	*	0.36	0.37	0.71
Shrubland				*				*	0.00
Herbaceous natural			0.02		*	*		0.13	
Herbaceous planted			0.28	*	*	0.51	0.27	0.47	
Wetland	*			0.15		*	0.11	0.32	0.33
Producer's accuracy	0.00	0.00	0.00	0.54	0.00	0.14	0.51	0.32	

Figure 1. Housing density for true vs. perturbed FIA plot locations (log-log scale, n=21,498).

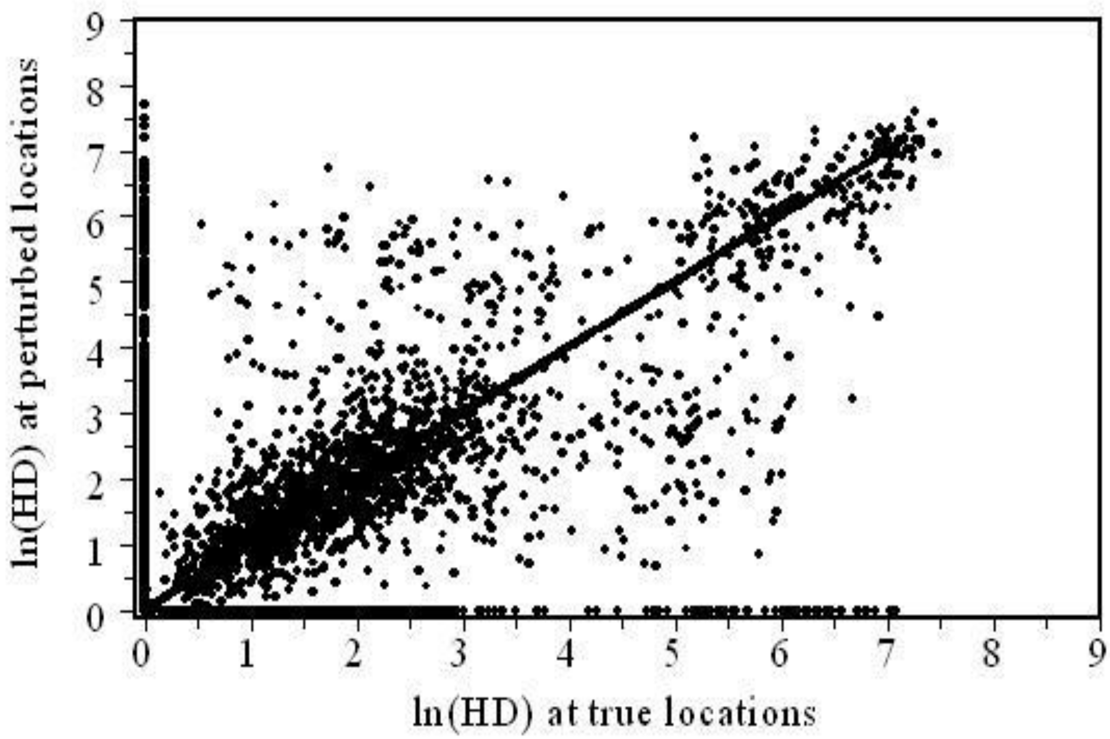


Figure 2. Housing density for true vs. swapped FIA plot locations (log-log scale, n=491).

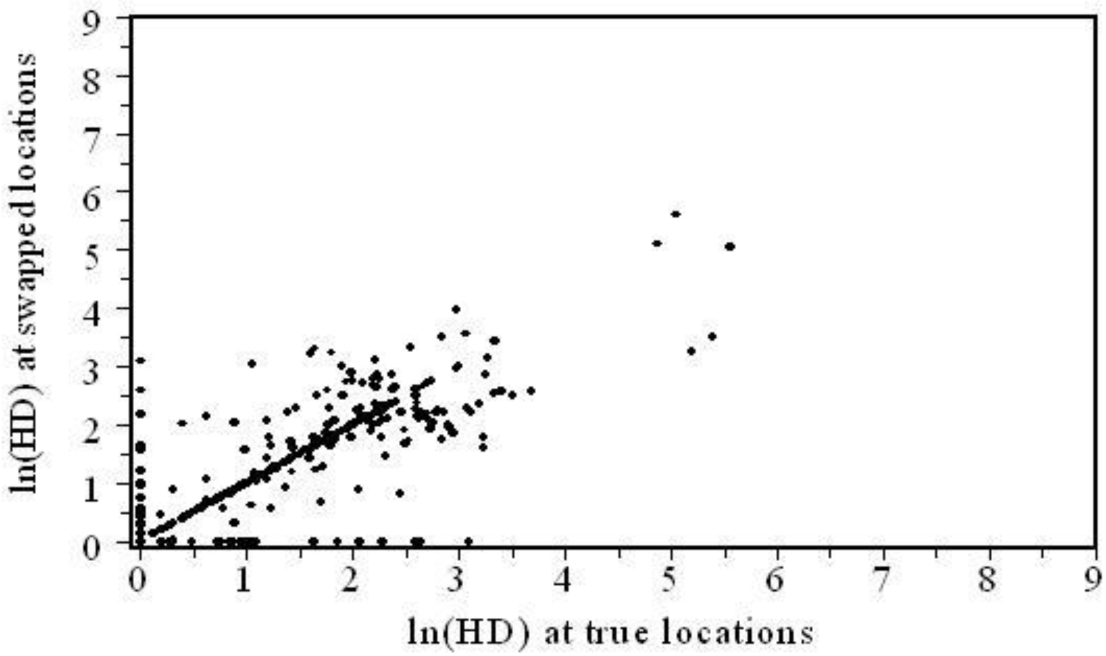


Figure 3. Residual plot for housing density using true vs. perturbed FIA plot coordinates (n=21,498).

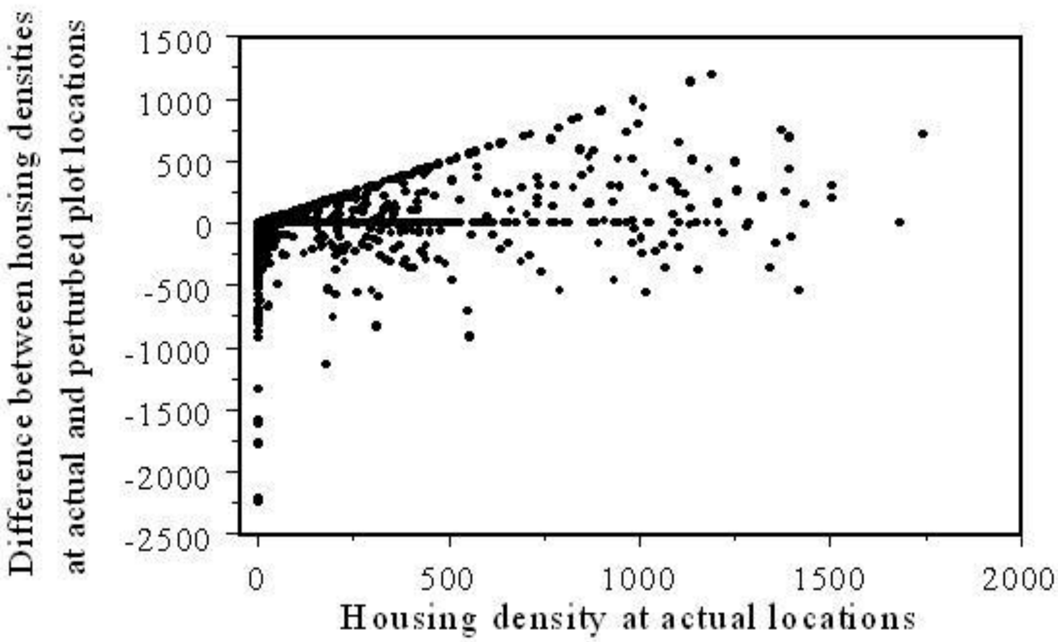


Figure 4. Residual plot for housing density using true vs. swapped FIA plot coordinates (n=491).

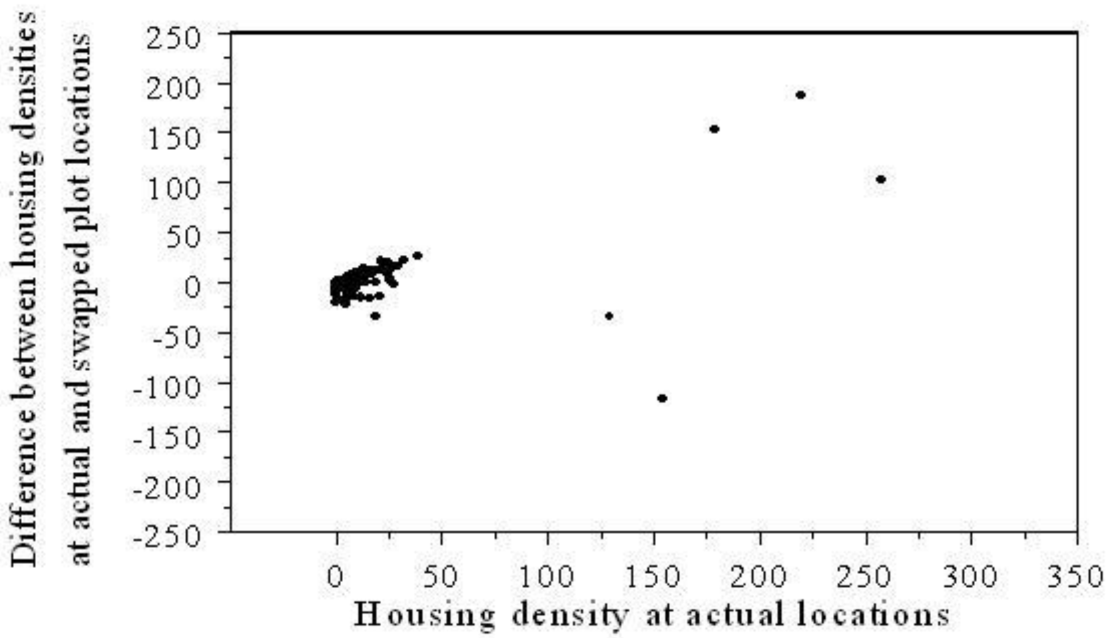


Figure 5. Cumulative distribution of differences in housing density corresponding to true vs. perturbed FIA plot coordinates (n=21,498).

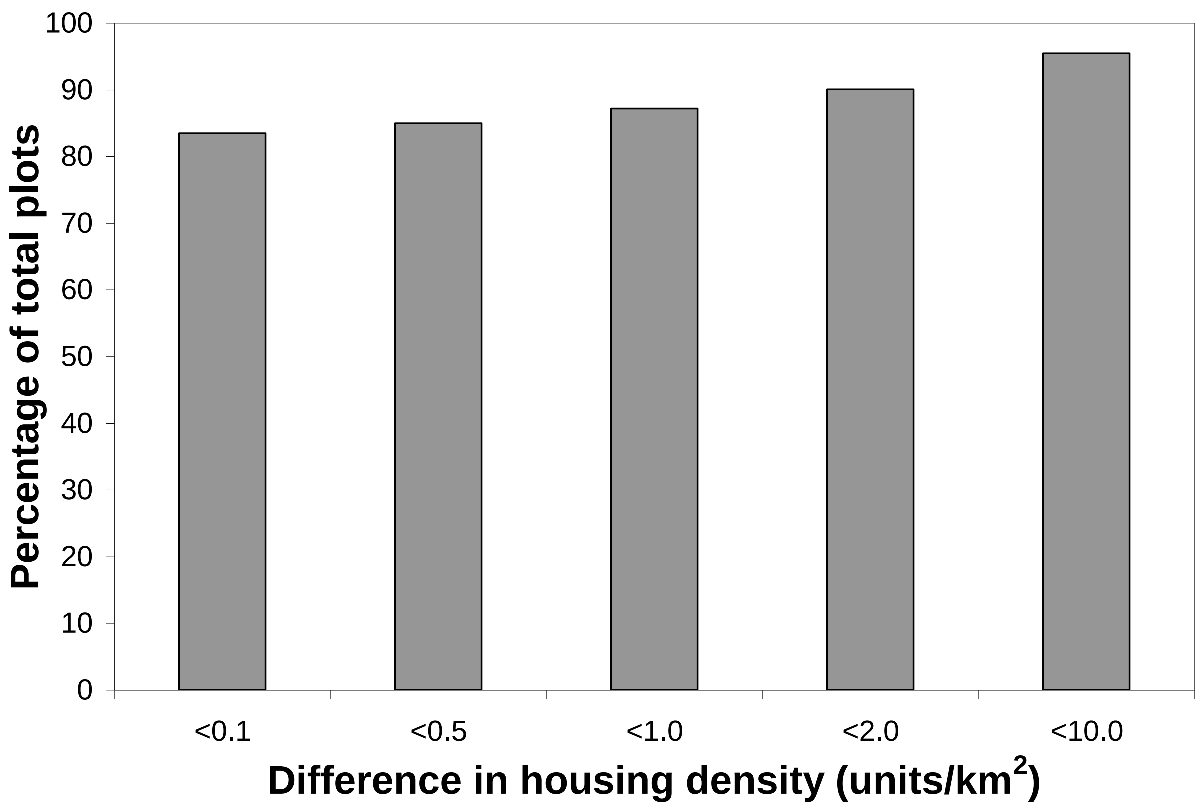


Figure 6. Cumulative distribution of differences in housing density corresponding to true vs. swapped FIA plot coordinates (n=491).

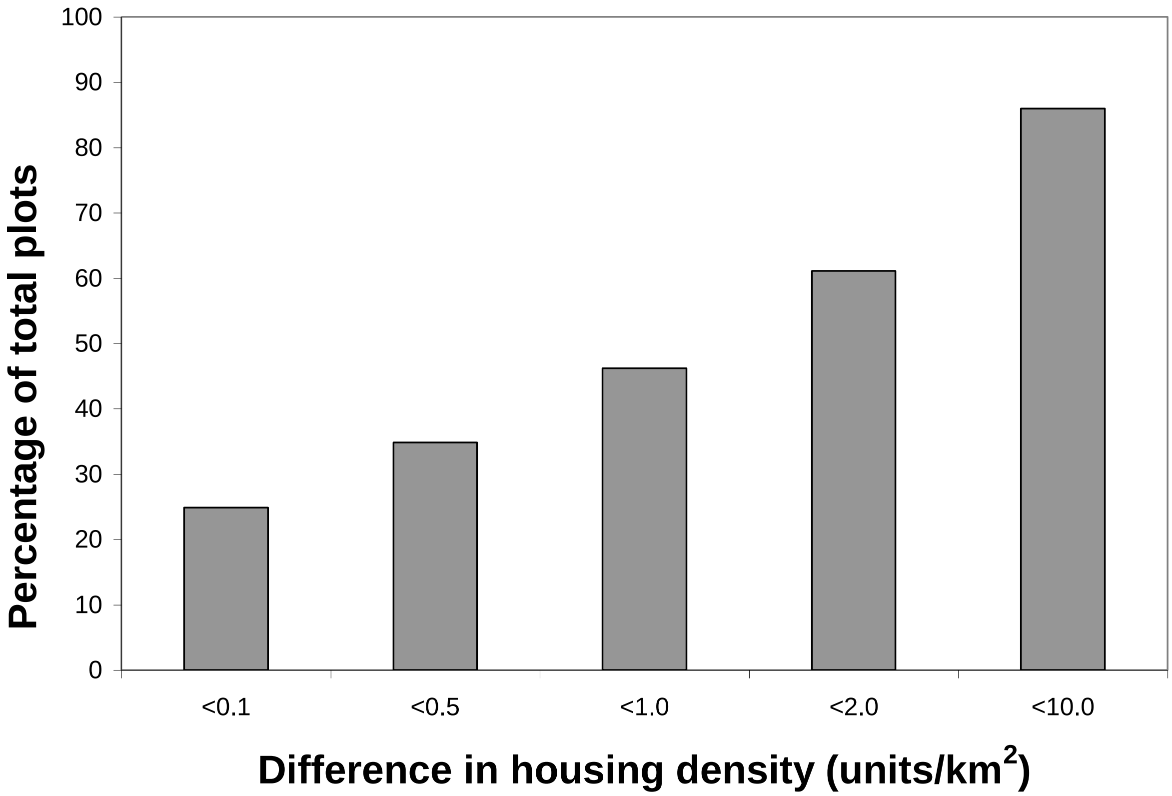


Figure 7. Boxplots of housing density distributions derived using true and perturbed FIA plot coordinates (n=21,498). Box indicates 25th percentile, median, and 75th percentile, star indicates sample mean.

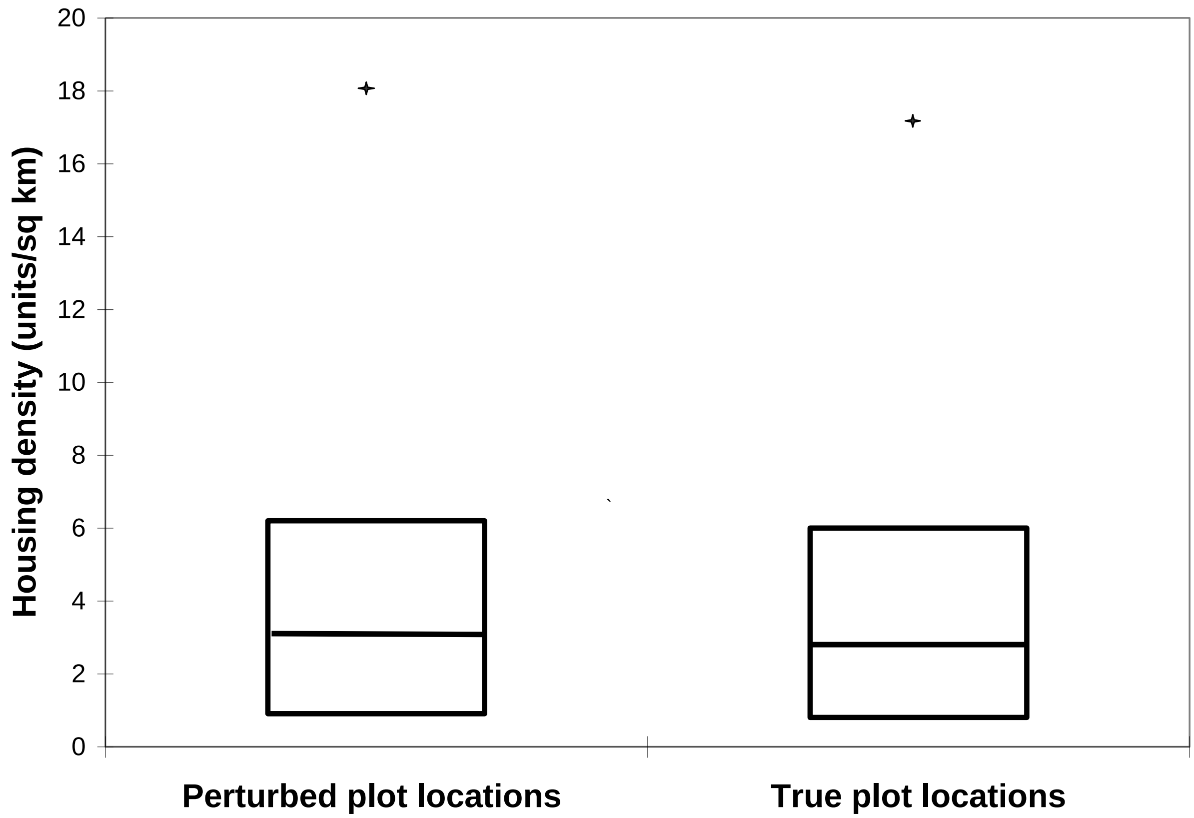


Figure 8. Boxplots of housing density distributions derived using true and swapped FIA plot coordinates (n=491). Box indicates 25th percentile, median, and 75th percentile, star indicates sample mean.

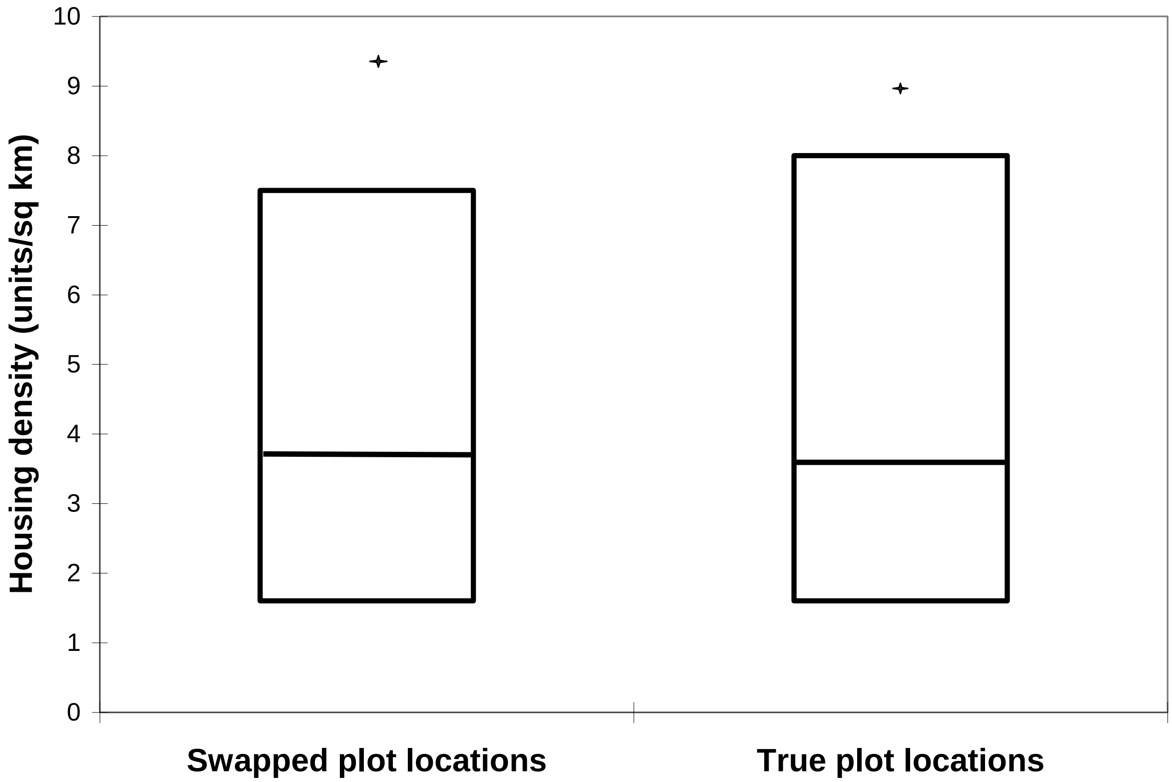
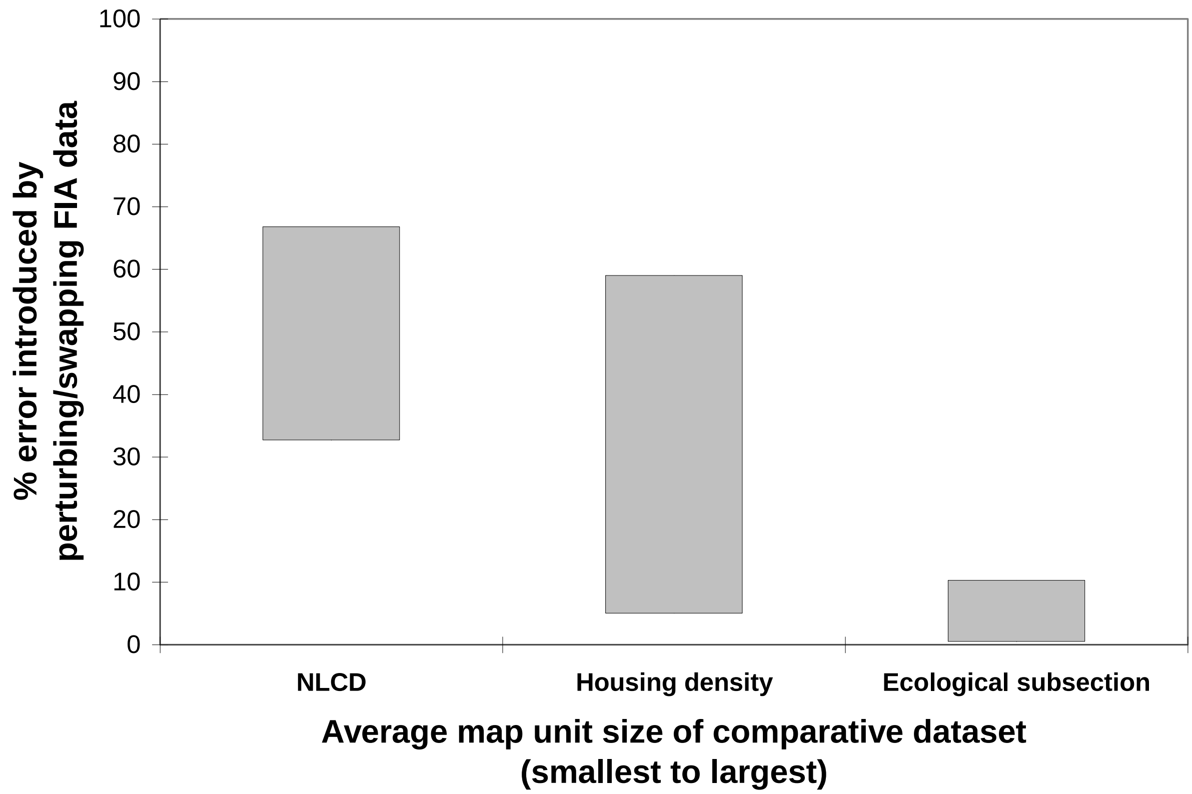


Figure 9. Summary of error distributions when comparing true and perturbed/swapped FIA plot locations to data layers at multiple spatial scales.



Chapter 2: Effects of residential development and land ownership on different forest types in the conterminous U.S. in 2000 with projections to 2030

Abstract

Despite the widespread impact of housing growth on forest ecosystems, little is known about the extent to which any of the more than 100 individual forest types in the United States are affected. Our goals in this study were to determine which forest types in the conterminous U.S. were most likely to be affected by residential development now and in the future and the extent to which those effects vary between different ownership categories. We also wanted to examine the extent to which broad forest categories reflect the housing densities of the individual forest types that comprise them. To address these issues, we associated data collected by the USDA Forest Service's Forest Inventory and Analysis database from over 91,000 forested plots with housing density estimates produced using the 2000 U.S. Decennial Census. We found that nearly 20% of all forest plots were located in areas where housing density was at least 6.5 housing units/ km^2. Approximately 8% of all forest types had at least half of their plots located in areas with housing densities of 6.5 housing units/ km^2 or more and we project the number of forest types exhibiting that level of residential development will more than double by 2030. Over one-third of the rarest forest types had at least 25% of their plots located in areas with housing densities of at least 6.5 housing units/ km^2, and nearly half of these rare forest types are projected to be at that level of housing density by 2030. The forest types that are currently experiencing the highest levels of residential development are

located primarily in the eastern portion of the U.S., but several rare Western forest types are expected to undergo the greatest increases in the percentage of plots located in areas where housing density $\geq$ 6.5 housing units/ km^2 by 2030. Housing densities associated with forest plots owned by state and local governments represented an intermediate condition between federally owned public lands and those in private ownership, suggesting that use of a simple dichotomy between public and private ownership is not entirely sound. Broad forest types are not a reliable proxy for housing density associated with the individual forest types that comprise them, particularly in the case of rare forest types. These results can be used to help refine the focus of future studies, policy decisions, and conservation efforts.

Introduction

The area of developed land in the U.S. increased nearly 57% between 1982 and 2007, during which time over 650,000 ha of rural land was converted to developed uses each year (USDA 2009b). Forest land was the largest source of rural land conversion during this time, with over 275,000 ha converted annually (USDA 2009b). Estimates project that an additional 10 million ha of forested land will have converted to developed uses between 1997 and 2030 (Alig and Plantinga 2004). Land use conversion between rural uses (e.g., from forest to agriculture and back again) has been common historically, but conversion to developed uses is generally a permanent alteration of rural lands (White et al. 2009). While the conversion of some forested lands to developed uses is inevitable, knowing which forests are most likely to be affected by land use changes can help scientists, resource managers, and communities plan more effective resource conservation strategies.

The increasing conversion of forested land to developed uses in recent decades has been fueled in large part by high rates of housing growth. While the U.S. population more than doubled between 1940 and 2000, the number of housing units more than tripled during the same time period (Hammer et al. 2009b). These population and housing growth trends are expected to continue over the next 20 years, with population estimated to reach 360 million and the number of housing units 158 million by 2030 (U.S. Census Bureau 2008).

Much of the housing growth in the past seven decades has been decentralized, occurring at relatively low and moderate densities in areas rich in natural amenities that are beyond the urban fringe (Mc Granahan 1999, Radeloff et al. 2005a, 2005b, Theobald 2005, Hammer et al. 2009a). Although much of the land experiencing housing growth in rural and exurban areas may still have substantial tree cover, the ecological conditions in these forests may have changed substantially (Alig et al. 2003, Stein et al 2005). Multiple negative ecological impacts caused by housing growth and associated development have been well documented and include habitat loss and fragmentation (Theobald et al. 1997, Hawbaker et al. 2005), decreased biodiversity (Pidgeon et al. 2007), altered hydrology and reduced water quality (Paul 2001, Atasoy et al. 2006), increases in the number and/or range of exotic species (Parendes and Jones 2000, Meekins and McKarthy 2002, Yates et al. 2004), and increased probability of wildfire (Cardille et al. 2001, Syphard et al. 2007).

Housing density increases may have significant impacts on both public and privately owned forests. Private forests constitute nearly 60% of the total forest land in the U.S. (Smith et al. 2009), and over 17 million hectares of private forests are expected to experience increases in housing density by 2030 (Stein et al. 2005). Many habitat types are found disproportionately on privately owned properties (Scott et al 2001), making these properties of particular concern for conservation planning (Robles et al. 2008). Although publicly owned forests may be less directly affected by housing density increases, residential development in and around protected areas is often significant and may diminish the conservation value of many public lands (Radeloff et al. 2010, Wade and Theobald 2010).

Despite the widespread impact of housing growth on forest ecosystems, little is known about the extent to which any of the more than 100 individual forest types recognized in the United States are affected (Smith et al. 2007). Forests are not uniformly significant in their conservation value (Robles et al. 2008), nor are all forest types equally likely to undergo housing development (Sabor et al. 2003). Prior studies have utilized either simple forest/non-forest categories (Radeloff et al. 2005b, Robles et al. 2008, Stein et al. 2010) or a limited number of broad forest categories (Theobald and Romme 2007) to identify areas across the United States where housing density increases may have significant impacts on forest ecosystems. While this research provides valuable information about the geographic regions where housing growth may more heavily impact forests in general, it does not permit analysis of the extent to which any particular forest types may be threatened by housing growth now or in the future.

Our goal in this study was to determine which forest types in the conterminous U.S. currently have the highest housing densities and which are most likely to be affected by future housing density increases. In addition, we wanted to determine how the impacts of residential development varied between public and private landowners and the extent to which broad forest categories reflected the housing densities associated with the individual forest types that comprise them.

Methods

Data sources

We conducted our analyses using Forest Inventory and Analysis (FIA) data collected in the conterminous U.S. downloaded from the National FIA Database (FIADB 4.0) DataMart (*http://199.128.173.17/fiadb4-downloads/datamart.html*). These data are collected by the USDA Forest Service using a nationally standardized plot design and common data collection procedures. The standard plot consists of four ~7.3 meter radius subplots on which all trees ≥ 12.7 cm in diameter are measured. Boundaries between distinct conditions on the plot, as defined by changes in type of ownership, forest type, stand age, reserve status, and other factors, are also mapped. Each condition on the plot is assigned a condition proportion, and the sum of all plot conditions should always equal 1.0 (USDA 2009a).

We used FIA data collected from forested plots sampled between 1998 and 2008 from 47 states (no data from Oklahoma were available in this time period). We excluded plots or portions of plots where the forest type was exotic hardwoods, exotic softwoods, or nonstocked. We also excluded plots where the sum of all condition proportions for that plot exceeded 1.0. Our final sample consisted of 91,213.05 plots or portions of plots (Fig. 1). These data represented 130 forest types from 27 forest type groups.

We used housing density for the year 2000 calculated using US Decennial Census data at the partial block group (PBG) level and projected housing density estimates for the year

2030 based on these calculations. Because of concerns about privacy and sampling error, certain data are released only for aggregations of census blocks (block groups). However, block groups are divided by a variety of political boundaries, such as congressional districts, that permit division into multiple partial block groups. The size of PBGs varies (mean =2.45 km^2, SD = 27.09 km^2) and is larger in rural areas than urban areas. PBGs have an average size one-tenth that of block groups and therefore provide a finer spatial resolution while including the complete array of population and housing attribute information available at the block group level (for a more complete description of PBG delineation and housing projection methods, see Hammer et al. 2004 and Radeloff et al. 2010).

Housing density and FIA data were compiled using a geographic information system (GIS). Although the web-available FIADB includes geographic coordinates for every plot location in the database, these are not the precise locations of the plot centers. The Forest Service is legally required to perturb the plot coordinates in the web-available FIA data to ensure that individual landowner and other proprietary information cannot be determined with certainty by those outside the program (USDA 2009a). However, analyses conducted by Sabor et al. (2007) found there were rarely significant differences in housing densities at the PBG level whether true or altered FIA plots locations were used.

Data analyses

We assigned each FIA plot a housing density category derived from the housing density of the PBG with which the plot was associated (Table 1). We then calculated the number of plots in each housing density category in each of three ownership categories:

Federal, state and local (S&L) government, and private. Due to privacy restrictions in the web-available FIA data, we were unable to distinguish between private corporate and private noncorporate forest owners in our analyses.

Housing density of 6.17 units/km^2 (approximately one housing unit[1] per 40 acres) is commonly regarded as the threshold beyond which land is no longer classified as rural (Stein et al. 2005, Theobald 2005). This threshold was approximated by housing density categories 5-9 in this study, which included plots with housing densities $\geq$ 6.5 units/km^2. For each ownership category, we calculated the percentage of plots that were located in PBGs where housing density $\geq$ 6.5 units/km^2 in 2000, the percentage of plots that were projected to be in PBGs in those housing density categories in 2030, and the difference in percentage between those years. For comparative purposes, we also calculated the percentage of plots in each ownership category that were located in PBGs where housing density was 0-1.49 housing units/km^2 (categories 1 and 2) in 2000 and 2030 and the difference between them.

For each forest type group and forest type, we calculated the percentage of plots of that type group or type in each ownership category. For each combination of ownership and forest type or forest type group, we also calculated the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 in the year 2000 and/or projected to have that level of housing density in 2030, and the difference in percentage between those years.

We ranked forest types based on the number of plots of that type included in the sample and examined housing density in 2000 and 2030 for the rarest 15% and 30% of forest

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[1] A housing unit is defined as a house, an apartment, a mobile home, a group of rooms, or a single room that is occupied (or if vacant, is intended for occupancy) as separate living quarters (U.S. Census Bureau 2010).

types. We also ranked forest type groups and forest types by percentage of plots located in PBGs where housing density ≥ 6.5 units/km^2 in 2000, percentage of plots projected to be located in PBGs with that level of housing density in 2030, and expected difference in percentage between those years. Finally, we examined differences in these three measures between forest type groups and the forest types that comprise them.

Results

Housing density and forest ownership

Approximately 63% of plots included in the study sample were privately owned and 37% were publicly owned. In 2000, 50.7% of all plots were located in PBGs where housing density was 0-1.49 housing units/km^2. Among Federally owned plots, 75.8% were located in PBGs in these categories, compared with only 48.7 % of those in S&L ownership and 27.6% of those owned privately. Just 18.5% of all plots were located in PBGs where housing density ≥ 6.5 units/km^2 in 2000, but there were considerable differences between ownerships. Only 5.8% of plots in Federal ownership were located in PBGs with that level of housing density, compared to 19.5% of S&L and 30.4% of privately owned plots (Fig. 2).

By 2030, only 47.1% of plots are projected to be located in PBGs where housing density is 0-1.49 housing units/km^2, a 3.6% decrease from 2000. A decrease is projected for all ownerships, with 71.3% (-4.5%) of Federal, 44.4 % (-3.9%) of S&L, and 25.2% (-2.4%) of privately owned plots in PBGs where housing density is 0-1.49 housing units/km^2. Conversely, the percentage of plots located in PBGs where housing density ≥ 6.5 units/km^2 is expected to increase to 24.5% (+6.0%) overall by 2030, with concomitant increases in the percentage of plots under Federal (+3.0%), S&L (+5.8%) and private (+8.9%) ownership in these categories (Table 2).

Rare forest types

The rarest 30% of forest types included 38 forest types, comprising 2.1% of all forest plots in the study, while the rarest 15% included 20 forest types and comprised 0.32% of all forest plots sampled (Table 3). Of the rarest 15% of forest types, 46.7% (n=7) had over three-quarters of their plots in private and/or S&L ownership. In particular, the majority of three rare western forest types -- Sitka spruce (*Picea sitchensis*, 69.7%), Bishop pine (*Pinus muricata*, 74.7%), Oregon ash (*Fraxinus latifolia*, 79.9%) -- were privately owned, as were 100% of Spruce pine (*Pinus glabra*) plots.

In 2000, 25% (n=5) of the rarest 15% of forest types had at least one-quarter of their plots located in PBGs where housing density $\geq$ 6.5 units/km$^{2.}$ This number is projected to increase to 35% by 2030, with 15% (n=3) of these rare forest types projected to experience double-digit changes in the percentage of plots in these categories. Two of these forest types are primarily privately owned (Bishop pine, with an expected 25.3% increase in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, and Oregon ash, with an expected increase of 11.6%), and one is primarily Federally owned (Table mountain pine, *Pinus pungens*, with an expected 13.2% increase in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2).

Of the rarest 30% of forest types, 55.2% (n==21) had over three-quarters of their plots in private and/or S&L ownership. In 2000, 60.5% (n=23) had at least one-tenth and 34.2%

(n=13) had at least one-quarter of their plots located in PBGs where housing density $\geq$ 6.5 units/km^2, and this number is projected to increase to 47.4% (n=18) by 2030. In addition, 13.2% (n=5) were expected to have double-digit changes in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 between 2000 and 2030.

Housing density changes by forest type and owner

In 2000, 7.7% (n=10) of forest types had at least half of their plots located in PBGs where housing density $\geq$ 6.5 units/km^2 (Table 4). The primary landowner for each of these forest types was private. When we examined each combination of forest type and owner, however, we found considerable variation among owners. For Federally owned plots, only two forest types (1.5%) had at least half of their plots located in PBGs where housing density $\geq$ 6.5 units/km^2, compared to 13 forest types (10%) with plots in S&L ownership and 16 forest types (12.3%) of those with privately owned plots (Table 5 and Table 8, column b).

By 2030, 16.9% (n=22) of forest types are projected to have at least half of their plots located in PBGs where housing density $\geq$ 6.5 units/km^2. Primary ownership for these forest types was private. The projected number of forest types with at least half of their plots located in PBGs where housing density $\geq$ 6.5 units/km^2 was unchanged for Federal ownership, but expected to increase to 19 (+ 4.6% from 2000) for S&L ownership and 31 (+ 11.5% from 2000) for private ownership (Table 6 and Table 8, column c).

Across all owners and forest types, there was an expected mean increase of 5.9% in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 between 2000 and 2030. The percentage of forest types anticipated to experience changes greater than the mean was 51.5% across all owners, 23.1% for those with plots in Federal ownership, 33.1%

for those with plots in S&L ownership, and 56.2% for those with plots in private ownership (Table 8, column d). Forest types projected to have increases of $\geq$10% in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 was 17.7% across all ownerships, 13.1% for Federal, 16.2% for S&L, and 23.1% for private owners (Table 8, column e). Only 0.7% of forest types are expected to have increases $\geq$ 25% in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 across all ownership and for Federal ownership. S&L ownerships are expected to have a 2.3% increase in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, while private ownerships are not expected to have any increases of this magnitude (Table 7 and Table 8, column f).

The same 10 forest types exhibited the highest percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 in 2000 and 2030 (Table 9). These 10 types include one rare Western forest type (Oregon ash) and one rare Eastern type (Eastern white pine/eastern hemlock, *Pinus strobus/Tsuga canadensis*). Both of these forest types are primarily privately owned. Two of these forest types (Oregon ash and Virginia pine/Southern red oak, *Pinus virginiana/Quercus falcata)* were also among those that had the greatest increase in the percentage of plots projected to be located in PBGs where housing density $\geq$ 6.5 units/km^2 between 2000 and 2030 (Table 10). Of those 10 forest types, three were Western forest types (Bishop pine, Gray pine (*Pinus sabiniana)*, Oregon ash). Three were among the rarest 15% of forest types (Bishop pine, Oregon ash, Table Mountain pine), with a fourth in the rarest 30% of forest types (Gray pine). Nine of these forest types are primarily in private ownership, while one (Table Mountain pine) is primarily in federally ownership.

Housing density changes by forest type group and comparisons with forest types

In 2000, there were six forest type groups for which at least one-quarter of plots were located in PBGs where housing density $\geq$ 6.5 units/km^2 (Fig. 3; see also Table 4). All of these were Eastern forest types, and all were primarily privately owned. By 2030, there are projected to be 11 forest type groups for which at least one-quarter of plots are located in PBGs where housing density $\geq$ 6.5 units/km^2. Of these, 90.1% are primarily privately owned and 0.9% is primarily in S&L ownership.

In the majority of cases, the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 in 2000 was not greater for individual forest types than the average for the forest type group in which they are included. However, 43.1% (n=56) of forest types did exceed the mean for their forest type group, with 27.7% (n=36) differing by at least 5% and 13.8% (n=18) differing by 10% or more.

There were several instances where percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 was relatively low for the forest type group but high for a single forest type within that type group (Fig. 3 and Table 4). For example, the Hemlock-sitka spruce type group had just 6.3% of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, but 29.6% of plots in the Sitka spruce forest type across all owners and 66.3% of Federally owned plots exceeded this threshold. Similarly, the Other Western softwoods forest type group had only 1.4% of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, while Bishop pine had 29.8% of plots at that level, and the Pinyon-juniper forest type group had 4.1% of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, with the Eastern redcedar (*Juniperus virginiana*) forest type having 36.2% of plots at that level.

Discussion

Land use conversion due to housing density increases is known to be a primary determinant of environmental change affecting forested ecosystems. Our results reveal details about the specific forest types and ownerships categories that are currently most affected by residential development and those most likely to experience future increases that can be used to help refine the focus of future studies, policy decisions, and conservation efforts.

Typically, studies have focused on the dichotomy between publicly and privately owned lands. However, our results indicate that this distinction is always useful due to variation in housing densities between classes of publicly owned land. In examining the percentage of plots in the lowest housing density categories, the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2, and the expected degree of change in those two categories between 2000 and 2030, we found that S& L ownership represents an intermediate condition between federally owned public lands and those in private ownership. Thus, although they are technically protected, lands under S&L ownership are likely affected by landscape change to a greater extent than federally protected areas.

Our results indicated that many forest types already have 50% or more of their plots located in PBGs where housing density exceeds the commonly accepted definition of ‘rural’ and that there could be a 220% increase in the number of forest types at that level of housing density by 2030. The extent of these increases is projected to be greatest on and around plots in private ownership, but important changes are also expected to occur in PBGs that include public lands. The greatest decreases in the number of plots in the lowest housing density categories is expected to occur in PBGs where public lands occur, and only the publicly owned portions of any forest type are expected to experience increases of 25% or more in the number of located in PBGs where housing density $\geq$ 6.5 units/km^2. These results support those of other recent studies that reported housing growth in and around protected areas has increased at a rate faster than that for the nation as a whole (Radeloff et al. 2010, Wade and Theobald 2010).

Of the rarest 30% of forest types included in this study, more than 10% of these forest types had at least a quarter of their plots located in PBGs where housing density $\geq$ 6.5 housing units/km^2 in 2000, and nearly half of these rare forest types are expected to have housing densities at this level by 2030. In addition, many of these rare forest types are expected to experience double-digit increases in the percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 in the coming decades. Thus, housing density increases may represent a serious threat to rare forest types remaining on the landscape and by extension to rare or threatened species that depend on these habitat types. Several of the rarest forest types, including Bishop pine, Sand pine (*Pinus clausa)*, and Redwood (*Sequoia sempervirens*), are based upon tree species that are considered threatened or near threatened (IUCN 2010). Interestingly, among rare forest types there is generally an inverse correlation between elevation and the number of plots in PBGs with housing densities $\geq$ 6.5 units/km^2 (Burns and Honkala 1990, R.Guries pers. comm..), highlighting the need to account for landscape position as well as prevalence in conservation planning.

Our results also indicated that the forest types currently experiencing the highest levels of housing density now and those expected to experience the highest levels by 2030 are all

Eastern forest types. However, among those forest types expected to experience the greatest increase in the percentage of plots located in PBGs where housing density $\geq$ 6.5 housing units/km^2 there are several Western forest types, including multiple rare types. These results are consistent with other studies indicating that although the East has higher average population and housing densities than the West, areas of the West may experience a greater increase in the number of forested plots that are located in PBGs that exceed the $\geq$ 6.17 units/km^2 threshold since they have experienced a faster rate of population increase than the East since the 1940s (Hammer et al. 2009b).

Broad forest categories, such as forest type group, are not a reliable proxy for the housing densities associated with the individual forest types that comprise them, with nearly one-half of all types having a higher percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 than the mean for the forest type group in which they are included. In several instances, rare forest types with a high percentage of plots located in PBGs where housing density $\geq$ 6.5 units/km^2 are included in forest type groups where the mean number of plots with that level of residential development is quite low. Therefore, we recommend that specific forest types are preferable for use in situations where anything more than a distinction between forest and nonforest is desirable.

The projections provided here are based on past trends, and long-term economic conditions in the upcoming decades that differ markedly from those upon which the housing density projections are based may result in greater or lesser impacts on forested habitats. However, housing growth has been substantial in every decade since the 1940s, even in times of economic recession, and population is expected to continue growing over the next 20 years

(Hammer et al. 2009b, Radeloff et al 2010). The Census Bureau intercensal housing estimates suggested that 12 million new housing units had been built between 2000-2007 (http://www.census.gov/popest/housing), which predates the start of the current economic recession. In addition, the housing densities used in this study should be considered conservative estimates, as they do not exclude the area in public land in the density calculations (Theobald 2005, R.Hammer, pers. comm). Thus, although recent changes in the U.S. economic situation may somewhat slow housing growth in the immediate future, the longer-term impacts of housing development are still likely to intensify.

Future housing density increases are also likely to act synergistically with other environmental stressors, including global climate change, on forest ecosystems. While the exact effects of future climate change on forests remains unclear, even moderate-change scenarios predict increases in the range and population size of pests and pathogens, as well as changes in tree species distribution (Scheller and Mladenoff 2008, CBO 2009). If climate shifts reduce forest habitat within protected areas, residential development outside these areas may limit tree species migration and the amount of new habitat available for colonization.

While the results of this study do not provide precise predictions about the future in all parts of the U.S., they do provide an important step toward understanding those factors that could alter the conservation functions and values of forestlands. They also provide a basis for conservation planning and highlight the need for integrating housing growth into those plans. Using the results from this and other similar studies, government agencies and conservation organizations can plan and target efforts to prevent or reduce conversion of some of their most valuable forest lands, such as those that are rare nationally or locally and those with few holdings in Federal ownership.

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Table 1. Housing density categories used in analysis

hu = housing unit(s)

Category	# hu/km2	Equates to:
0	0	0 hu/acre
1	>0-1.49	up to approx. 1 hu/240 acres
2	1.5- 2.49	up to approx. 1 hu/120 acres
3	2.5-4.49	up to approx. 1 hu/60 acres
4	4.5-6.49	up to approx. 1 hu/40 acres
5	6.5-12.49	up to approx. 1 hu/20 acres
6	12.5-25.49	up to approx. 1 hu/10 acres
7	25.5-50.49	up to approx. 1 hu/5 acres
8	50.5-249.49	up to approx. 1 hu/acre
9	$\geq$ 249.5	more than 1 hu/acre

Table 2. Differences in housing densities 2000-2030, by ownership

	% plots with HD ≤ 1.5 units/km2		
	2010	2030	Difference
All ownerships	50.7	47.1	-3.6
Federal	75.8	71.3	-4.5
State & local	48.7	44.8	-3.9
Private	27.6	25.2	-2.4

	% plots with HD ≥6.5 units/km2		
	2010	2030	Difference
All ownerships	18.5	24.5	6.0
Federal	5.8	8.8	3.0
State & local	19.5	25.3	5.8
Private	30.4	39.3	8.9

Table 3. Percent of plots with housing densities of 6.5 units/km^2 or more for rarest 30% of forest types.

Bold line indicates division between rarest 15% and rarest 30% of forest types.

[1] Highlighted cells indicate forest types where $\geq$ 10% of plots have housing densities $\geq$ 6.5 units/km^2.

[2] Highlighted cells indicate forest types where change in the percentage of plots with housing densities $\geq$ 6.5 units/km^2 is $\geq$ 10%.

[3] Highlighted cells indicate forest types where primary ownership is private and/or state & local. (Next page).

Forest type	**%HD 00 ≥6.5[1]**	**%HD 30 ≥6.5[1]**	**CH % 00-30[2]**	**% Fed owned**	**% Priv/S&L owned[3]**
Giant sequoia (S.giganteum*)*	0.0	0.0	0.0	100.0	0.0
Southwestern white pine *(P.strobiformus)*	0.0	0.0	0.0	63.6	36.4
Bishop pine (*P muricata)*	29.8	55.0	25.3	0.0	74.7
Oregon white oak (*Q.garryana)*	0.0	0.0	0.0	20.4	56.9
Alaska-yellow-cedar *(C.nootkatensis)*	0.0	0.0	0.0	100.0	0.0
Port-Orford cedar *(C.lawsoniana)*	4.4	4.4	0.0	90.0	8.3
Sugar pine *(P.lambertiana)*	0.0	0.0	0.0	100.0	0.0
Incense-cedar *(C.decurrens)*	3.6	3.6	0.0	100.0	0.0
Oregon ash (*F .latifolia)*	56.1	67.7	11.6	10.2	79.9
Knobcone pine *(P. attenuata)*	0.0	0.0	0.0	65.1	34.9
Spruce pine (*P glabra*)	28.8	28.8	0.0	0.0	100.0
Mangrove	39.0	39.0	0.0	9.7	17.5
Atlantic white-cedar (*C.thyoides)*	23.6	30.7	7.1	30.6	21.3
Table Mountain pine (*P.pungens)*	16.2	29.4	13.2	90.8	7.8
Sitka spruce (*P.sitchensis)*	29.6	34.8	5.3	15.9	69.7
Giant chinkapin (*C.chrysophylla)*	0.0	0.0	0.0	65.6	34.4
Noble fir (*A.procera)*	3.4	3.4	0.0	70.5	25.3
Foxtail/bristlecone pine (*P.balfouriana/P.aristata)*	0.0	0.0	0.0	91.9	8.1
Blue spruce *(P.pungens)*	11.0	17.1	6.1	53.7	44.2
Western white pine (*P.monticola)*	2.9	2.9	0.0	92.6	5.1
Black locust (*R.pseudoacacia)*	43.3	50.2	6.9	9.1	84.3
California laurel (*U.californica*)	12.7	19.8	7.1	40.1	45.3
Gray pine (*P.sabiniana*)	15.7	27.7	12.0	43.0	51.8
Intermountain maple woodland	0.6	9.6	9.0	55.2	38.9
Pacific madrone (*A. menziesii)*	16.2	20.0	3.9	36.6	54.9
Bigleaf maple *(A.macrophyllum)*	36.1	42.8	6.7	21.3	65.2

Forest type	**%HD 00 ≥6.5[1]**	**%HD 30 ≥6.5[1]**	**CH % 00-30[2]**	**% Fed owned**	**% Privately owned[3]**
Cottonwood/willow (*Populus* spp./*Salix* spp.)	32.9	37.1	4.3	3.2	79.0
Redwood (*S.sempervirens*)	24.1	33.4	9.3	8.3	76.9
Palms	19.8	23.9	4.1	14.4	45.6
Coast live oak (*Q.agrifolia)*	20.9	23.6	2.7	28.2	58.1
Pitch pine (*P.rigida*)	36.7	44.5	7.7	21.7	41.0
Black walnut (*J.nigra*)	43.5	50.7	7.2	3.1	87.9
East.white pine/east. hemlock (*P.strobus/T.canadensis*)	52.2	60.0	7.8	14.9	68.4
Sand pine (*P.clausa*)	24.2	28.8	4.6	45.4	47.0
Pond pine (*P.serotina*)	28.1	38.0	9.9	32.6	56.3
Scarlet oak (*Q.coccinea*)	41.3	52.4	11.1	24.3	55.8
Western redcedar (*T.plicata*)	9.1	12.4	3.3	56.4	33.9
Limber pine (*P.flexilis*)	0.7	1.6	0.9	72.8	19.8

Table 4. Percent of plots with housing densities of 6.5 units/km2 or more in 2000 & 2030 and percent change 2000-2030

[1] F = Federal, P = Private, S = State and local government

Forest type group/forest type	%HD 00 ≥6.5	%HD 30 ≥6.5	CH % 00-30	Primary owner[1]
Alder/maple	**24.2**	**28.4**	**4.2**	**P**
Bigleaf maple *(A.macrophyllum)*	36.1	42.8	6.7	P
Red alder (*A.rubra*)	21.2	24.7	3.5	P
Aspen/birch	**15.0**	**20.8**	**5.8**	**P**
Aspen (*Populus* spp.)	14.3	20.2	5.9	P
Balsam poplar (*P.balsamifera*)	12.6	19.3	6.8	P
Paper birch (*B.papyrifera*)	19.2	24.3	5.1	P
California mixed conifer				
California mixed conifer	3.6	5.0	1.4	F
Douglas-fir	**5.0**	**7.7**	**2.7**	**F**
Douglas-fir (*P.menziesii*)	5.0	7.7	2.7	F
Port-Orford cedar *(C.lawsoniana)*	4.4	4.4	0.0	F
Elm/ash/cottonwood	**28.7**	**35.5**	**6.7**	**P**
Black ash/American elm/red maple (*F.nigra/U.americana/A.rubrum*)	22.7	31.5	8.8	P
Cottonwood (*Populus* spp.)	19.4	21.0	1.5	P
Cottonwood/willow (*Populus* spp./*Salix* spp.)	32.9	37.1	4.3	P
Oregon ash (*F .latifolia)*	56.1	67.7	11.6	P
Red maple/lowland	43.0	50.0	7.0	P
River birch/sycamore (*B.nigra/P.occidentalis*)	39.0	44.2	5.2	P
Silver maple/American elm (*A.saccarinium/U.americana*)	30.0	38.5	8.5	P
Sugarberry/hackberry/elm/green ash	29.3	35.8	6.5	P

(*C.laevigata/C.occidentalis/Ulmus* spp./*F.pennsylvanica*)				
Sycamore/pecan/American elm (*P.occidentalis/ C.illinoensis/U.americana*)	28.7	34.9	6.2	P
Willow (*Salix* spp.)	22.0	29.9	7.9	P
Fir/spruce/mountain hemlock	**1.1**	**2.6**	**1.5**	**F**
Alaska-yellow-cedar *(C.nootkatensis)*	0.0	0.0	0.0	F
Blue spruce *(P.pungens)*	11.0	17.1	6.1	F
Engelmann spruce/subalpine fir (*P.engelmannii/A.lasiocarpa*)	0.9	1.7	0.8	F
Engelmann spruce (*P.engelmannii*)	1.1	3.3	2.2	F
Grand fir (*A.grandis*)	1.4	2.4	1.0	F
Mountain hemlock (*T.mertensiana*)	0.0	1.3	1.3	F
Noble fir (*A.procera)*	3.4	3.4	0.0	F
Pacific silver fir (*A.amabilis*)	0.0	3.3	3.3	F
Red fir (*A.magnifica*)	0.9	3.6	2.7	F
Subalpine fir (*A.lasiocarpa*)	0.3	1.3	0.9	F
White fir (*A.concolor*)	1.9	3.1	1.2	F
Hemlock/Sitka spruce	**6.3**	**8.9**	**2.5**	**F**
Sitka spruce (*P.sitchensis)*	29.6	34.8	5.3	P
Western hemlock (*T.heterophylla*)	4.1	6.3	2.1	F
Western redcedar (*T.plicata*)	9.1	12.4	3.3	F
Loblolly/shortleaf pine	**27.7**	**37.1**	**9.4**	**P**
Loblolly pine (*P.taeda*)	26.4	36.0	9.6	P
Pitch pine (*P.rigida*)	36.7	44.5	7.7	P
Pond pine (*P.serotina*)	28.1	38.0	9.9	P
Sand pine (*P.clausa*)	24.2	28.8	4.6	P
Shortleaf pine (*P.echinata*)	23.3	30.9	7.6	P
Spruce pine (*Pinus glabra*)	28.8	28.8	0.0	P
Table Mountain pine (*P.pungens)*	16.2	29.4	13.2	F
Virginia pine (*P.virginiana*)	61.4	72.1	10.7	P

Lodgepole pine				
Lodgepole pine (*P.contorta*)	1.5	3.1	1.6	F
Longleaf/slash pine	**21.9**	**32.8**	**11.0**	**P**
Longleaf pine (*P.palustrus*)	26.3	36.7	10.5	P
Slash pine (*P.elliottii*)	20.5	31.6	11.1	P
Maple/Beech/Birch	**31.3**	**37.1**	**9.4**	**P**
Black cherry (*P.serotina*)	51.0	58.4	7.4	P
Cherry/ash/yellow-poplar (*Prunus* spp./*Fraxinus* spp./*L.tulipifera*)	57.1	67.0	9.9	P
Elm/ash/locust (*Ulmus* spp./*Fraxinus* spp./*R.pseudoacacia)*	39.6	45.6	6.0	P
Hard maple/basswood (*Acer* spp./*Tilia* spp.)	23.9	31.9	8.0	P
Red maple (*Acer rubrum)*/upland	30.6	38.3	7.6	P
Sugar maple/beech/yellow birch (*A.saccharum/F.grandifolium/B.allaghaniensis*)	29.8	37.1	7.3	P
Oak/gum/cypress	**24.9**	**34.3**	**9.4**	**P**
Atlantic white-cedar (*C.thyoides*)	23.6	30.7	7.1	S
Baldcypress/water tupelo (*T.distichum/N.aquatica*)	20.2	27.7	7.5	P
Overcup oak/water hickory (*Q.lyrata/C.aquatica*)	9.3	15.7	6.3	P
Swamp chestnut/cherrybark oak (*Q.michauxii/Q.falcata*)	21.6	27.8	6.3	P
Sweetbay/swamp tupelo/red maple (*M.virginiana/N.sylvatica/A.rubrum*)	31.5	43.7	12.2	P
Sweetgum/Nuttall/willow oak (*L.styraciflua/Q.nuttallii/Q.phellos*)	23.6	32.3	8.6	P
Oak/hickory	**38.8**	**48.9**	**10.0**	**P**
Black locust (*R.pseudoacacia)*	43.3	50.2	6.9	P
Black walnut (*J.nigra*)	43.5	50.7	7.2	P
Bur oak (*Q.marcorcarpa)*	13.2	16.3	3.1	P
Chestnut oak (*Q.prinus*)	45.0	55.7	10.7	P
Chestnut/black/scarlet oak (*Q.prinus/Q.velutina/Q.coccinea*)	45.3	54.6	9.3	P
Mixed upland hardwoods	37.7	47.7	10.0	P

Northern red oak (*Q.rubra*)	37.7	48.2	10.4	P
Post oak/blackjack oak (*Q.stellata/Q.incana*)	28.7	40.7	12.0	P
Red maple/oak (*A.rubrum/Quercus* spp.)	54.8	61.9	7.1	P
Southern scrub oak (*Quercus* spp.)	49.3	57.4	8.0	P
Sassafras/persimmon (*S.albidum/D.virginiana*)	40.2	50.2	10.0	P
Scarlet oak (*Q.coccinea*)	41.3	52.4	11.1	P
Sweetgum/yellow-poplar (*L.styraciflua/L.tulipifera*)	45.5	53.9	8.4	P
White oak (*Q.alba*)	29.7	39.2	9.6	P
White/red oak/hickory (*Q.alba/Q.rubra/Carya* spp.)	36.7	47.2	10.5	P
Yellow-poplar (*L.tulipifera*)	50.6	62.2	11.6	P
Yellow-poplar/white/Northern red oak (*L.tulipifera/Q.alba/Q.rubra*)	54.0	64.0	10.0	P
Oak/pine	**37.3**	**47.4**	**10.1**	**P**
Eastern redcedar (*J.virginiana*)/hardwood	34.2	47.4	13.2	P
Eastern white pine/Northern red oak/white ash (*P.strobus/Q.rubra/F.americana*)	55.9	64.6	8.7	P
Loblolly pine (*P.taeda*)/hardwood	35.2	44.8	9.6	P
Longleaf pine/oak (*P.palustris/Quercus* spp.)	37.6	48.7	11.1	P
Other pine (*Pinus* spp.)/hardwood	34.6	42.0	7.4	P
Shortleaf pine/oak (*P.echinata/Quercus* spp.)	26.0	34.0	8.0	P
Slash pine/hardwood	28.1	42.2	14.1	P
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	57.4	72.5	15.1	P
Other hardwoods				
Pacific madrone (*A. menziesii)*	16.2	20.0	3.9	P
Other Western softwoods	**1.4**	**2.9**	**1.5**	**F**
Bishop pine (*P. muricata)*	29.8	55.0	25.3	P
Foxtail/bristlecone pine (*P.balfouriana/P.aristata)*	0.0	0.0	0.0	F
Knobcone pine *(P. attenuata)*	0.0	0.0	0.0	F
Limber pine (*P.flexilis*)	0.7	1.6	0.9	F
Misc. Western softwoods	4.6	9.9	5.3	F

Southwestern white pine *(P.strobiformus)*	0.0	0.0	0.0	F
Western juniper (*J.occidentalis*)	1.8	3.8	2.0	F
Whitebark pine (*P.albicaulis*)	0.0	0.0	0.0	F
Pinyon/juniper	**4.1**	**6.6**	**2.4**	**F**
Eastern redcedar (*J.virginiana*)	36.2	50.5	14.3	P
Juniper woodland (*Juniperus* spp.)	1.6	3.1	1.5	F
Pinyon/juniper woodland (*P.edulis/Juniperus* spp.)	1.0	2.3	1.3	F
Rocky Mountain juniper (*J.scopulorum*)	1.5	2.5	1.0	F
Ponderosa pine	**3.3**	**5.6**	**2.4**	**F**
Incense-cedar *(C.decurrens)*	3.6	3.6	0.0	P
Jeffrey/Coulter pine/bigcone Douglas fir (*P.jeffreyi/P.coulteri/P.macrocarpa*)	2.4	3.9	1.5	F
Ponderosa pine (*P.ponderosa*)	3.3	5.8	2.4	F
Sugar pine *(P.lambertiana)*	0.0	0.0	0.0	F
Redwood	**23.8**	**33.0**	**9.2**	**P**
Giant sequoia *(S.giganteum)*	0.0	0.0	0.0	F
Redwood (*S.sempervirens*)	24.1	33.4	9.3	P
Spruce/fir	**10.9**	**14.6**	**3.7**	**P**
Balsam fir (*A.balsamea*)	12.1	16.6	4.5	P
Black spruce (*P.mariana*)	6.2	8.0	1.8	S
Northern white-cedar (*T.occidentalis*)	13.3	18.5	5.2	P
Red spruce (*P.rubens*)	11.3	14.7	3.4	P
Red spruce/balsam fir (*P.rubens/A.balsamea*)	11.8	14.7	2.9	P
Tamarack (*L.laricina*)	9.6	12.0	2.4	S
White spruce (*P.glauca*)	15.8	20.4	4.6	P
Tanoak/laurel	**6.2**	**10.8**	**4.6**	**P**
California laurel (*U.californica*)	12.7	19.8	7.1	P
Giant chinkapin (*C.chrysophylla)*	0.0	0.0	0.0	F
Tanoak (*L.densiflorus*)	5.3	9.8	4.5	P

Tropical hardwoods	**22.0**	**25.6**	**3.6**	**S**
Mangrove	39.0	39.0	0.0	S
Palms	19.8	23.9	4.1	P
Western larch				
Western larch (*L.occidentalis*)	2.6	3.3	0.7	F
Western oak group	**13.2**	**18.5**	**5.3**	**F**
Blue oak (*Q.douglasii*)	12.8	21.9	9.1	P
California black oak (*Q.kelloggii*)	12.9	16.5	3.6	F
Canyon live/interior live oak (*Q.chrysolepsis/Q.wislizeni*)	11.7	15.8	4.1	F
Coast live oak (*Q.agrifolia)*	20.9	23.6	2.7	P
Gray pine (*P.sabiniana*)	15.7	27.7	12.0	P
Oregon white oak (*Q.garryana)*	0.0	0.0	0.0	P
Western white pine				
Western white pine (*P.monticola)*	2.9	2.9	0.0	F
White/red/jack pine	**33.0**	**41.6**	**8.6**	**P**
Eatern hemlock (*T.canadensis*)	33.9	40.9	7.0	P
Eastern white pine (*P.strobus*)	46.4	55.6	9.2	P
Eastern white pine/eastern hemlock (*P.strobus/T.canadensis*)	52.2	60.0	7.8	P
Jack pine (*P.banksiana*)	18.9	25.6	6.7	S
Red pine *(P.resinosa*)	29.6	40.3	10.7	P
Woodland hardwoods	**4.3**	**8.3**	**4.0**	**F**
Cercocarpus woodland (*Cercocarpus* spp.)	0.4	1.3	0.9	F
Deciduous oak woodland (*Quercus* spp.)	5.1	9.3	4.2	F
Evergreen oak woodland (*Quercus* spp.)	4.1	7.5	3.4	F
Intermountain maple woodland (*Acer* spp.)	0.6	9.6	9.0	F
Mesquite woodland (*Prosopis* spp.)	3.9	7.9	4.0	F
Misc. woodland hardwoods	12.6	19.7	7.1	P

Table 5. Forest types where one or more ownerships had ≥ 50% of plots located in PBGs where housing density ≥ 6 units/km2 in 2000

[1] F = Federal, P = Private, S = State and local government.

Forest type	**All owners**	**Federal**	**State & local**	**Private**	**Primary owner[1]**	**Secondary owner[1]**
Port-Orford cedar *(C.lawsoniana)*	4.4	0.0	100.0	33.5	P	S
Misc. Western softwoods	4.6	0.0	85.4	12.8	F	P
Table Mountain pine (*P.pungens)*	16.2	11.0	0.0	78.9	F	P
Sand pine (*P.clausa*)	24.2	21.1	68.6	20.0	F	P
Sycamore/pecan/American elm (*P.occidentalis/ C.illinoensis/U.americana*)	28.7	11.9	53.1	27.7	P	S
Sitka spruce (*P.sitchensis)*	29.6	66.3	23.3	22.4	P	S
Eastern redcedar (*J.virginiana*)/hardwood	34.2	11.8	54.6	36.0	P	S
Eastern redcedar (*J.virginiana*)	36.2	19.4	50.8	37.1	P	
Mangrove	39.0	0.0	41.5	50.0	P	F
Sassafras/persimmon (*S.albidum/D.virginiana*)	40.2	25.6	57.8	40.1	P	F
Scarlet oak (*Q.coccinea*)	41.3	12.2	60.9	47.0	P	S
Black locust (*R.pseudoacacia)*	43.3	5.4	67.3	45.5	S	P
Black walnut (*J.nigra*)	43.5	57.7	56.7	41.6	P	F
Chestnut oak (*Q.prinus*)	45.0	25.7	41.2	63.4	P	F
Chestnut/black/scarlet oak (*Q.prinus/Q.velutina/Q.coccinea*)	45.3	23.4	37.3	55.7	P	F
Sweetgum/yellow-poplar (*L.styraciflua/L.tulipifera*)	45.5	20.1	57.2	46.8	F	P
Eastern white pine (*P.strobus*)	46.4	17.6	33.7	56.3	F	P
Southern scrub oak (*Quercus* spp.)	49.3	7.1	45.5	57.8	F	P
Yellow-poplar (*L.tulipifera*)	50.6	29.1	45.7	54.6	F	P
Black cherry (*P.serotina*)	51.0	26.3	31.9	59.9	F	P
East. white pine/east. hemlock (*P.strobus/T.canadensis*)	52.2	6.4	37.3	61.7	F	P

Yellow-poplar/white/Northern red oak (*L.tulipifera/Q.alba/Q.rubra*)	54.0	24.7	54.4	62.2	P	F
Red maple/oak (*A.rubrum/Quercus* spp.)	54.8	27.6	35.2	61.7	P	F
Eastern white pine/Northern red oak/white ash (*P.strobus/Q.rubra/F.americana*)	55.9	25.0	41.4	66.1	P	F
Oregon ash (*F .latifolia)*	56.1	0.0	100.0	57.8	F	P
Cherry/ash/yellow-poplar (*Prunus* spp./*Fraxinus* spp./*L.tulipifera*)	57.1	23.3	40.8	60.9	P	S
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	57.4	24.0	41.8	66.3	P	F
Virginia pine (*P.virginiana*)	61.4	24.1	48.7	68.7	P	F

Table 6. Forest types where one or more ownerships are projected to have ≥ 50% of plots located in PBGs where housing density ≥ 6 units/km2 in 2030

[1] F = Federal, P = Private, S = State and local government.

Forest type	All owners	Federal	State & local	Private	Primary owner[1]	Secondary owner[1]
Port-Orford cedar *(C.lawsoniana)*	4.4	0.0	100.0	33.5	F	P
Misc. Western softwoods	9.9	5.1	100.0	12.8	F	P
Sand pine (*P.clausa*)	28.8	22.1	68.6	28.8	P	F
Table Mountain pine (*P.pungens)*	29.4	25.5	0.0	78.9	F	P
Atlantic white-cedar (*C.thyoides*)	30.7	0.0	39.8	54.4	S	F
Shortleaf pine (*P.echinata*)	30.9	14.0	56.2	44.1	P	F
Sitka spruce (*P.sitchensis)*	34.8	66.3	23.3	30.0	P	F
Sycamore/pecan/American elm (*P.occidentalis/ C.illinoensis/U.americana*)	34.9	13.4	54.8	34.8	P	S
Mangrove	39.0	0.0	41.5	50.0	S	P
Red pine (*P.resinosa*)	40.3	21.8	32.9	57.7	P	S
Other pine (*Pinus* spp.*)*/hardwood	42.0	24.4	33.3	54.2	P	F
Slash pine (*P.elliotii*)/hardwood	42.2	27.8	60.2	41.9	P	F
Bigleaf maple *(A.macrophyllum)*	42.8	0.0	44.8	56.4	P	F
Pitch pine (*P.rigida*)	44.5	21.5	37.4	63.0	P	S
White/red oak/hickory (*Q.alba/Q.rubra/Carya* spp.)	47.2	24.0	46.1	51.4	P	F
Eastern redcedar (*J.virginiana*)/hardwood	47.4	17.0	54.6	50.3	P	F
Northern red oak (*Q.rubra*)	48.2	26.4	42.9	56.0	P	S
Longleaf pine/oak (*P.palustris/Quercus* spp.)	48.7	15.8	47.4	58.5	P	F
Red maple (*A.rubrum*)/lowland	50.0	28.0	36.3	55.1	P	S
Sassafras/persimmon (*S.albidum/D.virginiana*)	50.2	35.4	57.8	50.7	P	F
Black locust (*R.pseudoacacia)*	50.2	5.4	97.0	51.4	P	F
Eastern redcedar (*J.virginiana*)	50.5	32.3	62.9	51.6	P	F
Black walnut (*J.nigra*)	50.7	57.7	56.7	49.8	P	S

Scarlet oak (*Q.coccinea*)	52.4	24.5	64.9	60.2	P	F
Sweetgum/yellow-poplar (*L.styraciflua/L.tulipifera*)	53.9	31.6	60.3	55.2	P	F
Chestnut/black/scarlet oak (*Q.prinus/Q.velutina/Q.coccinea*)	54.6	37.4	43.1	64.3	P	F
Bishop pine (*P. muricata)*	55.0	0.0	100.0	39.9	P	S
Eastern white pine (*P.strobus*)	55.6	26.4	41.9	65.9	P	S
Chestnut oak (*Q.prinus*)	55.7	37.0	46.9	75.5	P	F
Southern scrub oak (*Quercus* spp.)	57.4	7.1	45.5	69.0	P	S
Black cherry (*P.serotina*)	58.4	26.3	39.5	68.0	P	S
Eastern white pine/eastern hemlock (*P.strobus/T.canadensis*)	60.0	17.7	37.3	69.7	P	F
Red maple/oak (*A.rubrum/Quercus* spp.)	61.9	40.5	42.0	68.3	P	S
Yellow-poplar (*L.tulipifera*)	62.2	29.1	54.8	68.4	P	F
Yellow-poplar/white/North. red oak (*L.tulipifera/Q.alba/Q.rubra*)	64.0	35.2	68.4	71.7	P	F
East. white pine/North. red oak/white ash (*P.strobus/Q.rubra/F.americana*)	64.6	40.6	47.4	74.0	P	S
Cherry/ash/yellow-poplar (*Prunus* spp./*Fraxinus* spp./*L.tulipifera*)	67.0	23.3	47.9	71.7	P	S
Oregon ash (*F.latifolia*)	67.7	0.0	100.0	72.3	P	F
Virginia pine (*P.virginiana*)	72.1	41.2	62.0	78.1	P	F
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	72.5	49.7	55.6	79.4	P	F

Table 7. Forest types undergoing greatest increase in percentage of plots with housing densities $\geq$ 6.5 units/km^2: 2000-2030

[1] F = Federal, P = Private, S = State and local government.

Highlighted cells indicate those ownerships undergoing double-digit increases.

Forest type	**All owners**	**Federal**	**State & local**	**Private**	**Primary owner[1]**	**Secondary owner[1]**
Pacific madrone (*A. menziesii)*	3.9	6.6	16.9	0.0	P	F
Canyon live/interior live oak (*Quercus* spp.)	4.1	2.9	19.4	5.7	F	P
Cottonwood/willow (*Populus* spp./*Salix* spp.)	4.3	0.0	13.7	2.3	P	S
Misc. Western softwoods	5.3	5.1	14.6	0.0	F	P
Elm/ash/locust (*Ulmus* spp./*Fraxinus* spp./*R.pseudoacacia)*	6.0	13.0	4.7	5.8	P	S
Blue spruce *(P.pungens)*	6.1	0.0	0.0	13.8	F	P
Balsam poplar (*A.balsamea*)	6.8	13.5	2.4	7.6	P	S
Black locust (*R.pseudoacacia)*	6.9	0.0	29.7	5.9	P	F
Red maple/oak (*A.rubrum/Quercus* spp.)	7.1	12.9	6.8	6.6	P	S
Shortleaf pine (*P.echinata*)	7.6	5.5	21.8	8.5	P	F
Pitch pine (*P.rigida*)	7.7	9.5	0.0	13.8	P	S
Willow (*Salix* spp.)	7.9	12.3	0.7	8.9	P	S
East. white pine/North. red oak/white ash (*P.strobus/Q.rubra/F.americana*)	8.7	15.6	5.9	7.8	P	S
Intermountain maple woodland (*Acer* spp.)	9.0	5.9	0.0	14.8	F	P
Chestnut/black/scarlet oak (*Q.prinus/Q.velutina/Q.coccinea*)	9.3	13.9	5.8	8.6	P	F
Yellow-poplar/white/Northern red oak (*L.tulipifera/Q.alba/Q.rubra*)	10.0	10.6	14.0	9.5	P	F
Longleaf pine (*P.palustris*)	10.5	2.6	12.4	14.4	P	F
Virginia pine (*P.virginiana*)	10.7	17.1	13.3	9.5	P	F
Longleaf pine/oak (*P.palustris/Quercus* spp.)	11.1	0.0	13.1	13.9	P	F

Yellow-poplar (*L.tulipifera*)	11.6	0.0	9.1	13.7	P	F
Oregon ash (*F .latifolia)*	11.6	0.0	0.0	14.5	P	F
Gray pine (*P.sabiniana*)	12.0	4.0	0.0	19.8	P	F
Post oak/blackjack oak (*Q.stellata/Q.incana*)	12.0	9.3	6.1	13.0	P	F
Sweetbay/swamp tupelo/red maple (*M.virginiana/N.sylvatica/A.rubrum*)	12.2	3.9	12.5	13.2	P	F
Table Mountain pine (*P.pungens)*	13.2	14.5	0.0	0.0	F	P
Eastern redcedar (*J.virginiana*)/hardwood	13.2	5.1	0.0	14.4	P	F
Slash pine (*P.elliotti*)/hardwood	14.1	3.2	29.9	13.7	P	F
Eastern redcedar (*J.virginiana*)	14.3	12.9	12.2	14.6	P	F
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	15.1	25.7	13.8	13.2	P	F
Bishop pine (*P. muricata)*	25.3	0.0	100.0	0.0	P	S

Table 8. Summary of housing density changes by owner

(a) **Owner**	(b) **2000: FT ≥ 50% of plots**	(c) **2030: FT ≥ 50% of plots**	(d) **% FT increase > 5.9%**	(e) **% FT increase ≥10%**	(f) **% FT increase ≥ 25%**
All	7.7	16.9	51.5	17.7	0.7
Federal	1.5	1.5	23.1	13.1	0.7
S&L	10	14.6	33.1	16.2	2.3
Private	12.3	23.8	56.2	23.1	0

Table 9. Forest types with highest percentage of plots where housing density ≥ 6.5 hu/km2 in 2000 and 2030 -- all owners

Forest type	**%HD 00 ≥ 6.5**	**%HD 30 ≥ 6.5**
Yellow-poplar (*L.tulipifera*)	50.6	62.2
Black cherry (*P.serotina*)	51.0	58.4
Eastern white pine/eastern hemlock (*P.strobus/T.canadensis*)	52.2	60.0
Yellow-poplar/white/Northern red oak (*L.tulipifera/Q.alba/Q.rubra*)	54.0	64.0
Red maple/oak (*A.rubrum/Quercus* spp.)	54.8	61.9
Eastern white pine/Northern red oak/white ash (*P.strobus/Q.rubra/F.americana*)	55.9	64.6
Oregon ash (*F .latifolia)*	56.1	67.7
Cherry/ash/yellow-poplar (*Prunus* spp./*Fraxinus* spp./*L.tulipifera*)	57.1	67.0
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	57.4	72.5
Virginia pine (*P.virginiana*)	61.4	72.1

Table 10. Forest types projected to experience the greatest change in percentage of plots where housing density $\geq$ 6.5 hu/km2 between 2000 and 2030 -- all owners

Forest type	**Change 00-30 (%)**
Oregon ash (*F .latifolia)*	11.6
Gray pine (*P.sabiniana*)	12.0
Post oak/blackjack oak (*Q.stellata/Q.incana*)	12.0
Sweetbay/swamp tupelo/red maple (*M.virginiana/N.sylvatica/A.rubrum*)	12.2
Table Mountain pine (*P.pungens)*	13.2
Eastern redcedar (*J.virginiana*)/hardwood	13.2
Slash pine (*P.elliottii*)/hardwood	14.1
Eastern redcedar (*J.virginiana*)	14.3
Virginia pine/Southern red oak (*P.virginiana/Q.falcata*)	15.1
Bishop pine (*P. muricata)*	25.3

Figure 1. Map of plots used in this study.

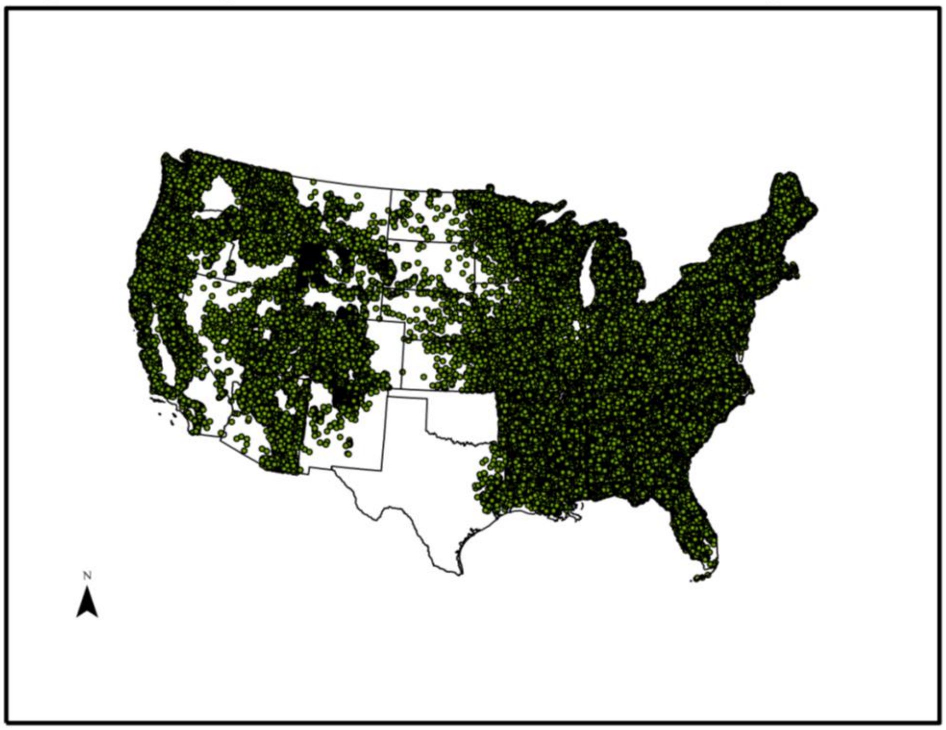


Figure 2. Housing density distribution of FIA plots by ownership category (housing units per km^2)

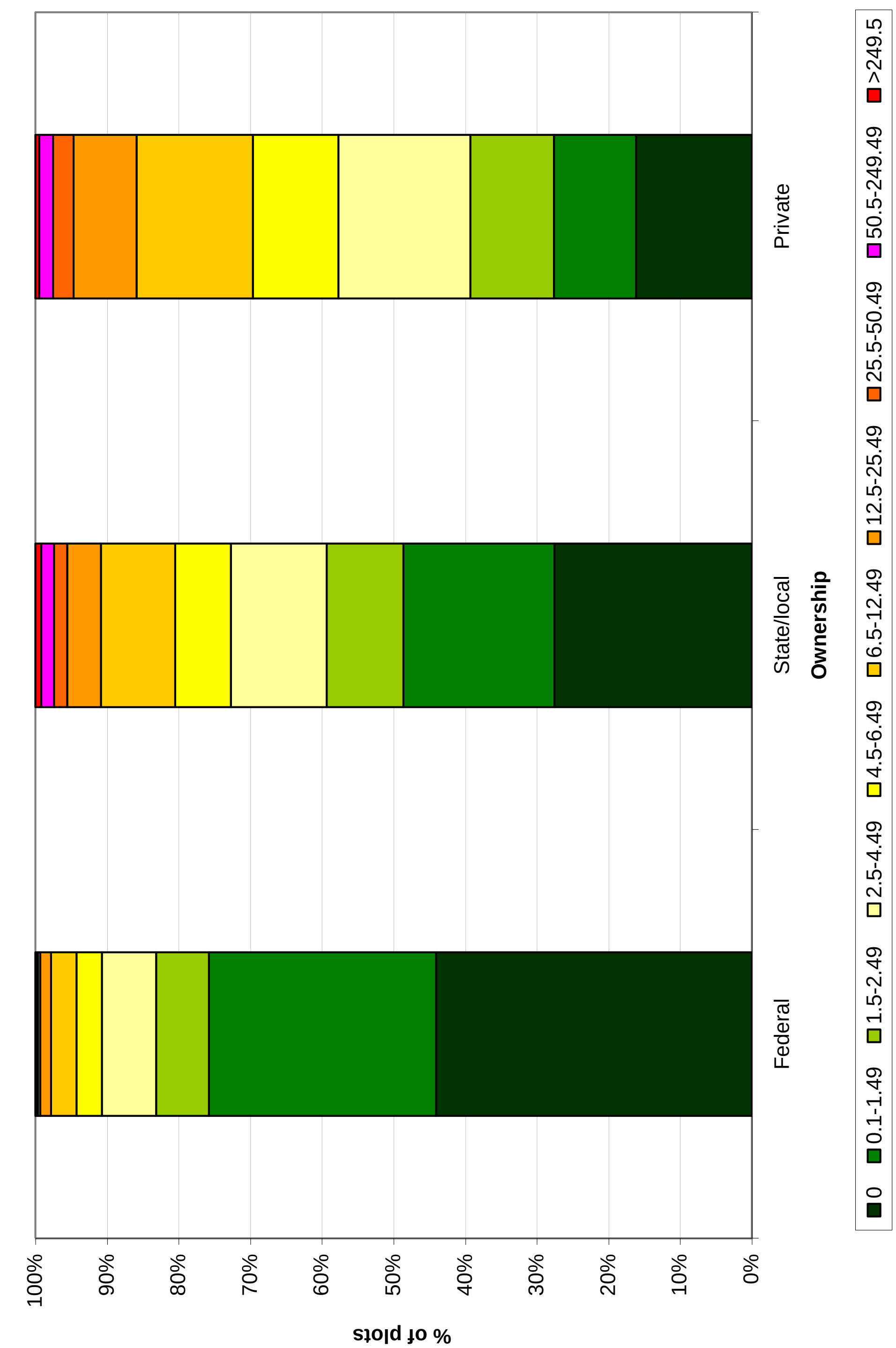
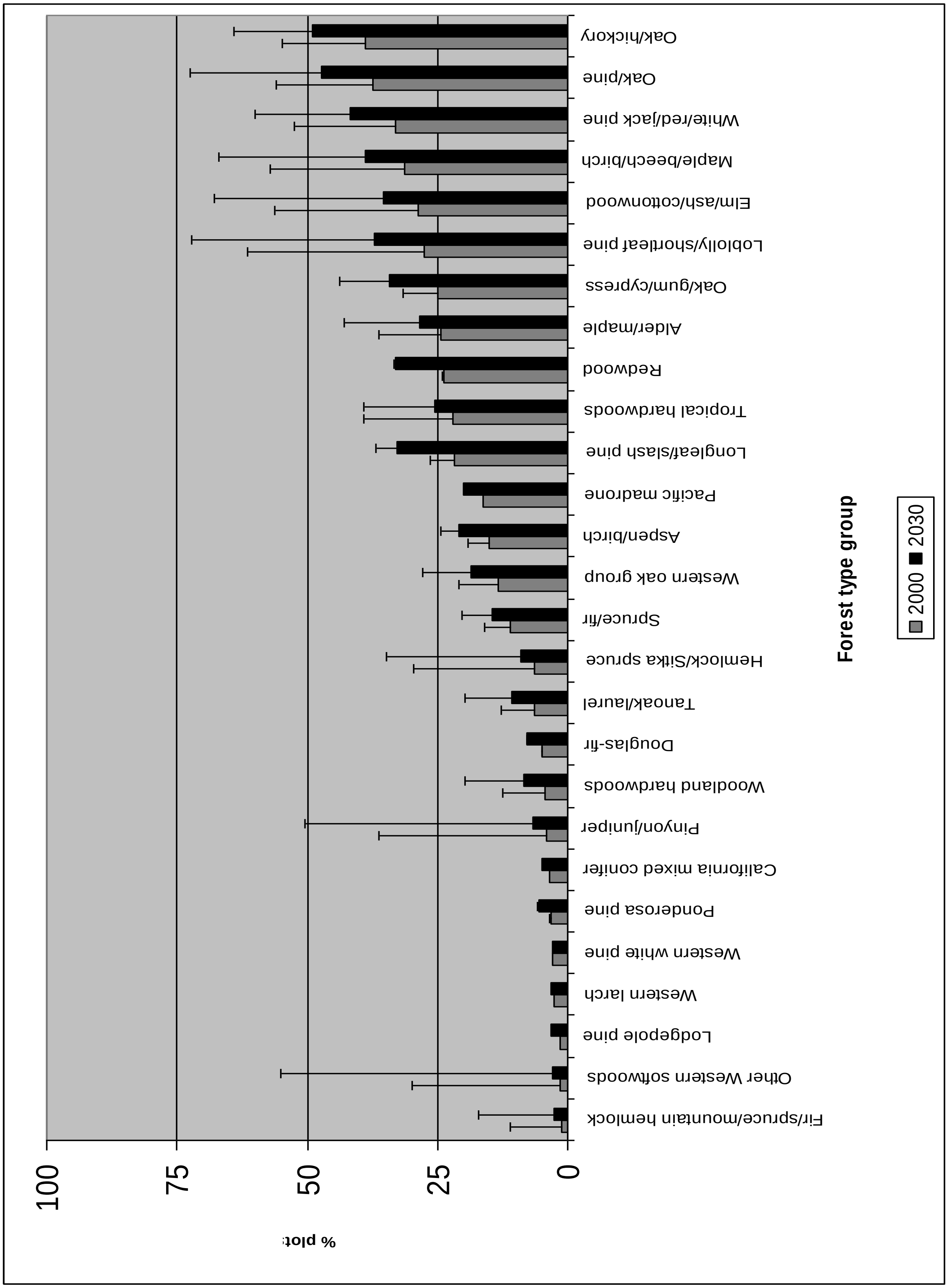


Fig 3. Percent of plots in each forest type group with housing density $\geq$ 6.5 units/km^2 in 2000, with projections for 2030.

Bars represent mean for the forest type group. Barred lines represent the maximum percent of plots for any forest type in that type group. (Next page).


% plots
100
75
50
25
0
Fir/spruce/mountain hemlock
Other Western softwoods
Lodgepole pine
Western larch
Western white pine
Ponderosa pine
California mixed conifer
Pinyon/juniper
Woodland hardwoods
Douglas-fir
Tanoak/laurel
Hemlock/Sitka spruce
Spruce/fir
Western oak group
Aspen/birch
Pacific madrone
Longleaf/slash pine
Tropical hardwoods
Redwood
Alder/maple
Oak/gum/cypress
Loblolly/shortleaf pine
Elm/ash/cottonwood
Maple/beech/birch
White/red/jack pine
Oak/pine
Oak/hickory
Forest type group
2000
2030

Chapter 3: Effects of housing density on standing and down dead wood in the upper Midwest

Abstract

Standing and downed dead wood are vital components of forest ecosystems, providing habitat and foraging substrate as well as playing important roles in carbon sequestration, fuel loading, and soil geomorphology. Residential development is one broad-scale factor that may affect the amount and characteristics of dead wood, but these impacts have not been well explored for terrestrial environments. Here, we examined the relationships between standing and down dead wood and residential development in the upper Midwest (Michigan, Minnesota, and Wisconsin), which has experienced considerable housing growth over the past three decades, particularly in and around forested areas with abundant recreational and aesthetic amenities. Logistic regression and general linear models indicated that housing density, along with ecological section, ownership, stand age, live basal area and forest type were all important in predicting the abundance of snags, maximum DBH of snags, fine woody debris, and both the length and the diameter of coarse woody debris. However, these relationships are complex and patterns were not consistent between states, highlighting the need to exercise caution when generalizing results to the regional scale.

Introduction

Standing and down dead wood is an important structural component of forest ecosystems. Dead wood provides habitat for a variety of organisms, including arthropods (Ulyshen et al. 2004), birds (Swallow et al. 1988), mammals (Maser et al. 1979, Bull et al. 1997), and fungi (Rubino and McCarthy 2003). Dead wood also plays an important role in carbon cycling (Kueppers et al. 2004, Woodall et al. 2008) and fuel loads in fire-prone areas (Burgan 1988, Rollins et al. 2004). The amount, structure, and dynamics of dead wood in forested ecosystems can influence ecological processes, species composition, nutrient cycling, productivity, and soil geomorphology for hundreds of years (Harmon et al. 1986, Spies and Cline 1988). Given the vital role dead wood plays in the structure and function of healthy forest ecosystems, it is important to understand the factors that affect the quantity and quality of standing and down dead wood in order to enact effective conservation and management practices.

Residential development is one broad-scale factor that may exert influence over the quantity and characteristics of terrestrial dead wood. Even low-density residential development can have pronounced impacts on a variety of ecological processes, including the amount of forest interior habitat (Theobald et al. 1997), habitat connectivity (Gonzalez-Abraham et al. 2006), biodiversity (Pidgeon et al. 2007), the number and distribution of exotic species (Gavier-Pizzaro et al. 2010), the structure of native vegetation (Riemann and Tillman 1999), and the range of forest management practices that are socially acceptable (Barlow et al. 1998, Sabor et al. 2003, Ward et al. 2005). Residential development has increased considerably over the last 35 years, increasing at a rate faster than that of population growth (Hammer et al. 2009), particularly in rural areas with abundant forest and waterbodies, and this trend is expected to continue over the next 20 years (Gustafson et al. 2005, Hammer et al. 2009).

The effects of residential development on terrestrial dead wood have not been well-studied, but research on aquatic coarse woody debris and historic anthropogenic disturbance suggest that these impacts may be significant and long-lasting. Residential development has strong negative effects on the abundance of down dead wood (Jennings et al. 2003, Francis and Schindler 2006, Marburg et al. 2006) and standing dead trees (Christensen et al. 1996) in lakes and on lakeshores with higher levels of housing density, and in lakes with abundant boat docks (Kratz et al. 2002). Protected areas with a land use legacy of prior concentrated or diffuse human settlement contain significantly less down dead wood and smaller-sized standing dead wood than primary forests without a history of residential development even decades after such settlement had ended (Webster and Jenkins 2005).

The three states of the upper Midwest – Michigan, Minnesota, and Wisconsin – have undergone significant landscape change due in large part to rural housing growth since the 1970s (Gobster et al. 2000). Although the tri-state area contains approximately 52 million acres of forestland (Shifley and Sullivan 2002), most forests in the upper Midwest are located less than 25 km from human settlements and contain at least some houses (Figure 1; Radeloff et al. 2005), making this an excellent area to study the impacts of housing density on standing and down dead wood. Our first goal in this research was to examine the abundance of standing dead trees (snags) and down dead wood in relationship to housing density and other factors known to affect the quantity of dead wood in forest ecosystems. In particular, we hypothesized that dead wood would have a negative relationship with housing density. In addition, because larger diameter (i.e., >15 cm) snags are more ecologically important (Bunnell et al. 2002, Fan et al. 2003, DeLong et al. 2004) and size diversity in down dead wood is important to forest structure and function (Heilmann-Clausen and Christensen 2004), our second goal was to examine the size of dead wood in this region, particularly as related to housing density.

Methods

Study area

The study region included Michigan, Minnesota, and Wisconsin, an area covering 494,014 km^2. Within the study area, approximately 210,000 km^2 are designated as forestland, defined by the USDA Forest Service as land at least 10% stocked by forest trees of any size, including land that formerly had such tree cover, will be naturally or artificially regenerated, and is not currently developed for a nonforest use (Smith et al. 2007). Predominant forest types in these three states include maple/beech/birch, aspen/birch, spruce/fir, and oak/hickory (Shifley and Sullivan 2002). This region exhibits a great variety of forest types, stand ages, and housing densities, making it particularly suitable for a study of this type.

Data sources

Forest Attributes

Analyses were conducted using Forest Inventory and Analysis (FIA) data collected in Michigan, Minnesota, and Wisconsin. FIA Phase 2 (P2) plot data were used for analyses of snags and Phase 3 (P3) plots for analyses of down coarse and fine woody materials. P2 plot inventories include assessments of all standing live and dead tree boles > 2.5 cm in diameter at breast height (dbh)(Bechtold and Patterson 2005). P2 plots are 0.41 ha in size and occur at an intensity of one plot per 1,200 ha in Minnesota and Wisconsin and one plot per 800 ha in Michigan (R. McRoberts, pers. comm). Plots may include one or more conditions, as defined by changes in type of ownership, forest type, stand age, reserve status, and other factors, and each condition on the plot is assigned a condition proportion (USDA 2009).

In Phase 3 (P3), a 1/16 subset of P2 plots are measured for a variety of forest health indicators, including down woody materials (Woodall and Nagel 2006). Data on down woody materials are collected only on P3 plots. Coarse woody debris (CWD) is defined as dead tree and shrub boles, large limbs, and other woody materials > 7.62 cm in diameter that are severed from their original source of growth and on the ground (USDA Forest Service 2005). CWD are sampled using twelve 7.32 m transects per plot (Woodall and Williams 2005). All coarse woody materials are measured for bole diameter at transect, diameter at large end, diameter at small end, total length, species, and decay class (Stolte et al. 2002). Fine woody debris (FWD) are divided into small (< 0.6 cm), medium (0.6 – 2.5 cm), and large (2.51 – 7.6 cm) fractions. For small and medium FWD, field crews tally the number of pieces for each size class encountered along four 1.8 m transects per plot, while for large FWD the count is tallied on four 3.0 m transects per plot (Woodall and Williams 2005).

All P2 data were downloaded from the FIA website (*http://199.128.173.17/fiadb4-downloads/datamart.html*). Although the web-available FIADB includes geographic coordinates for every plot location in the database, these are not the precise location of the plot centers. FIA is legally required to perturb the plot coordinates in the web-available data to ensure that individual landowner and other proprietary information cannot be determined with certainty by those outside the program (USDA 2009). However, prior research indicates that housing densities assigned to FIA plots using either true or altered plot locations are rarely significantly different (Sabor et al. 2007).

Web-available P3 data do not include geographic coordinates and use a plot numbering convention that prevents them from being linked to the P2 plot with which they are associated. As a result, all analyses on CWD and FWD were conducted at the FIA Spatial Data Services Center in St. Paul, MN using actual plot coordinates. Only data from 2001-2002 were used in these analyses since data from subsequent years were not yet available at the time this study was conducted.

Housing density

Housing density for the year 2000 was estimated using U.S. Decennial Census data at the partial block group (PBG) level via methods developed by Hammer et al. (2004). Due to concerns about privacy and sampling error, certain Census data are released only for aggregations of census blocks (block groups). However, block groups are transected by a variety of political boundaries, such as Congressional districts and minor civil divisions, which permit division into multiple partial block groups. PBGs have a mean size one-tenth that of block groups and they therefore provide a much better spatial resolution while including the complete array of Census population and residential attribute information available at the block group level. Sizes of PBGs in the study region range from <0.01 to 1,640 km^2 (mean = 2.5 km^2), while residential densities range from 0.0-16,945 units/ km^2 (mean = 71.45 units/km^2).

Data analysis

Snags

Housing density and FIA data were compiled using a geographic information system (GIS), and each FIA plot was assigned the housing density of the PBG in which it was located. In order to preclude issues of spatial autocorrelation within plots and allow us to treat each plot as one sample unit, statistical analyses included only those P2 plots that were at least 75% in a single condition. We also checked the model residuals for spatial autocorrelation among plots using variograms.

The response variables we modeled were the number (DEADCOUNT) and maximum dbh (MAXDBH) of snags on FIA plots. A common feature of data collected on the abundance of organisms is that it often contains large numbers of zeroes (McCullagh and Nelder 1989). Since these data do not conform to the standard error models used with general linear models, the data may best be modeled in two steps. The presence-absence component of the data is modeled using a logistic regression, followed by modeling the observed abundance in situations where the observed abundance is greater than zero (Welsh et al. 1996, Barry and Welsh 2002). Since initial summary of the dataset indicated that DEADCOUNT included many zeroes, we used the two-step approach to modeling this response. Because MAXDBH was only measurable on plots that had at least one dead tree, we modeled this response using only general linear models (Table 1).

We used six explanatory variables to model DEADCOUNT and MAXDBH: housing density, land ownership, ecological section, stand age, forest type group, and live basal area. Standing and down dead wood are known to generally increase with both stand age (Goodburn and Lorimer 1998, Hale et al. 1999, Spetich et al. 1999) and live basal area (Ferguson and Archibald 2002, Woodall and Westfall 2009) and to vary among forest types (Harmon 1993, Pedlar et al. 2002). Since dead wood attributes are influenced by forest management practices (McGee et al. 1999, Kenefic and Nyland 2007, Gronewold et al. 2009), we included ownership as a proxy for different management objectives. We also included ecological section to account for broad geographic homogeneity in environmental and biotic characteristics (Figure 2; McNab et al. 2005).

Data for all explanatory variables except housing density were taken directly or derived from the FIA database. Live basal area was calculated by summing the basal area of all live trees on the plot. Since logistic regression requires a minimum number of presence and absence observations for each level of a categorical variable (Hosmer and Lemeshow 2000), ecological subsection and forest type data from the FIA were aggregated to the level of ecological section and forest type group (Table 2). Prior to regression analysis, we log transformed the response and explanatory variables as necessary in order to ensure compliance with model assumptions. We also calculated a Pearson's correlation coefficient matrix for all continuous explanatory variables to measure collinearity and found that no variables were correlated above 0.51.

We performed both logistic and general linear regression models for all possible models and used Akaike's Information Criterion (AIC) to choose the best subset of 10 models for each dependent variable (Burnham and Anderson 2002). This approach highlights variables that appear repeatedly in the best models and indicates whether they have a consistently positive or negative relationship to the response variable. Since we did not know a priori if model results would be consistent among states, we ran regressions for each response variable separately for each state in the study area. We then counted the number of times each variable was included in the best subset for each state and the direction of the effect for each continuous variable. We ranked the estimate for each level of the categorical variables for each model and compared rankings for all models in the best subset to determine if the rankings were consistent across models.

Down woody materials

To ensure adequate estimation of means, we used only those P3 plots with ≥ 5 observations of CWD/plot in the analyses of down woody materials, resulting in a final sample of 129 plots including approximately 2,100 observations of CWD/FWD. For CWD, we fitted general linear models using total length and transect diameter as our response variables. For FWD, we fitted general linear models using counts of small diameter, medium diameter, and large diameter FWD as the response variables. For both CWD and FWD regression analyses, we used forest type, ecological subsection, stand age, ownership and housing density as explanatory variables and included interaction terms between main effects in our models. Prior to regression analysis, we log transformed the response and explanatory variables as necessary in order to ensure compliance with model assumptions. Backwards elimination was used to select the best model for each response variable. We also calculated means for all response variables by housing density and ownership.

Results

Snag abundance and maximum DBH

Snags accounted for an average of 8.2% of all standing trees across the study region. The ratio of snags to live trees was greater in Minnesota (10.2%) than in Michigan or Wisconsin (7.3-7.7%). For plots that had snags, the average DEADCOUNT was 4 (range 1-123).

In the logistic regressions for DEADCOUNT, forest type group and live basal area were the most strongly related variables and included in 100% of the best models. Stand age and ecological subsection were each included in approximately two-thirds of the best models, while housing density and ownership were present in half the models. The effects of stand age and live basal area were always positively and significantly related to DEADCOUNT. Housing density was not consistent in either the direction of its effect or statistical significance (Table 3). There was no consistent rank order in the estimates for different ownerships and ranks were not entirely consistent between models for ecological section.

Sections that showed up frequently at the high end of the ranking were 212J and 212L, and those at the low end included 212H, 212Y, and 222K. Forest types groups that were consistently ranked highly were Elm/Ash/Cottonwood and Aspen/Birch, while Red/White/Jack Pine was the lowest ranked forest type group in all models.

In general linear models where DEADCOUNT $\geq$ 1, ecological section, forest type group, and live basal area were included in >90% of the best models. Stand age, housing density, and ownership were each included in approximately 50% of the best models. Overall, these models explained 5-9% of the variance in DEADCOUNT (Table 4). Live basal area was always positively and significantly related to DEADCOUNT, while stand age was always significant but the relationship was positive for Michigan and Minnesota and negative for Wisconsin. Housing density was never statistically significant, and the direction of the effect varied by state. The effects of ownership were never statistically significant but the rank order of the effects of ownership highly consistent, with Federal ownership ranking highest and private ownership lowest. Both forest type group and ecological section were always significant when they entered the model. Jack/Red/White Pine, Spruce/Fir, and Aspen/Birch were frequently highly ranked among forest type groups, but there was no consistent pattern at the high end of the rank order. However, Oak/Hickory and Maple/Beech/Birch were consistently at the bottom of the rank order. There was again no clear pattern in the rank order for ecological section, although sections 212R and 212L were often highly ranked while sections 212Y and 222L tended to rank low.

The average MAXDBH of snags in the study sample was 59.6 cm (range 12.7-122.7). In general linear models for MAXDBH, ecological section, forest type group, and live basal area were included in >85% of the best models. Stand age and ownership were each included in approximately 2/3 and housing density in approximately half of the best models. These models explained 8-14% of the variance in MAXDBH (Table 5). Stand age and live basal area were always positively and significantly related to MAXDBH. Although the effects and significance of housing density was consistent within each state, they each varied among states. The effects of ownership varied in significance within and among states and the rank order of the effects was not consistent between models. Both forest type group and ecological section were always significant when they entered the model. There was no regularity among high-ranking forest type groups, but Jack/Red/White Pine and Spruce/Fir were consistently the two lowest ranked groups. Similarly, there was no clear pattern in the rank order for ecological section, although sections 212H, 212T, and 212Y were generally ranked low.

Down dead wood

The average total length of CWD was considerably longer on plots under Federal ownership than on those that were privately owned. Partial block groups with zero housing units had the longest average total length for CWD, while the lowest average length was found in the second-highest housing density category. Mean transect diameter was more than twice as large for plots in Federal ownership or with no houses as it was for those that were privately owned or in the highest housing density category (Table 6). Conversely, small diameter FWD was more than twice as abundant on private and high housing density plots as on those where there were no houses or that were in Federal ownership. There was no consistent pattern between housing density or ownership and the abundance of either medium or large diameter FWD (Table 7).

Forest type was the only explanatory variable included in the best model for total length of CWD, explaining 22% of the variation. For transect diameter, housing density was not significant by itself but was significant in interaction terms with ecological subsection and forest type. Stand age, ownership, and ecological subsection were all significant predictor variables, as was the interaction term between stand age and ownership. This model explained 86% of the variation in CWD transect diameter (Table 8).

Similarly, forest type was the only variable significantly related to the count of small diameter FWD, and this model explained 31% of the variation. None of the explanatory variables were significant in explaining the count of medium diameter FWD. Housing density and stand age were significant alone and in an interaction term in the model for large diameter FWD, and this model explained 6% of the variation (Table 9).

Discussion

Our results paint a complex picture of the relationship of standing and down dead wood to housing density in the upper Midwest. Although housing density appeared in approximately half of the best models for snag abundance and maximum DBH, its effect was generally not significant and the direction of its effect varied among states. While housing density appeared to have positive and sometimes significant influence on whether snags are present on a plot, the relationship to the number of snags present was not as strong or consistent. Housing density was also a significant term in models predicting the diameter of

CWD and the quantity of large diameter FWD. While housing density affects the amount of dead wood at the site level (e.g., Webster and Jenkins 2005, Marburg et al. 2006), it is likely that at the landscape scale these patterns begin to average out and therefore become less detectable. However, as areas experiencing dispersed, low- and mid-level housing densities continue to increase in the upper Midwest in the coming decades, this effect may become more noticeable.

The ratio of snags to standing live trees in this region was consistent with those from studies conducted in other parts of the Midwest (Spetich 1999, Shifley et al. 2003). In addition, the average maximum DBH of snags in the study area was remarkably consistent across states, housing densities, and ownerships. Although there was a decrease in the size of the largest snags as housing density increased (data not shown), the maximum DBH of snags in all housing densities fell within or exceeded the threshold size for cavity formation, which only occurs at larger sizes and renders the snags suitable habitat for a broad variety of wildlife (Shifley et al. 2003).

Ecological section, another variable that was significant in many of the best models, may also reflect the effects of housing density. Although the section rankings did not always follow this pattern, a number of the sections that ranked highest in estimates of snag abundance were areas where housing density was quite low. For example, much of the area of section 212L is taken up by the Boundary Waters Wilderness Area in Minnesota and section 212J is located in the western Upper Peninsula of Michigan, where there are few areas of higher housing density. Conversely, numerous sections with relatively high housing density ranked low in snag abundance, such as 212H (the northern part of the Lower

Peninsula of Michigan) and 222K (the corridor between Milwaukee and Madison in Wisconsin). Sections that spanned more than one state did not necessarily rank the same in all models, and often these appear to be related to differences in housing density. For example, section 212Y ranked at the bottom of snag abundance for Minnesota, where it is in the vicinity of Duluth, but did not in the other two states, where it is located in portions of those states that have low housing density.

Live basal area was almost always present and had a highly significant, positive effect in the best models for snag abundance and maximum DBH. This result concurs with those of previous studies that found a positive relationship between live basal area and various measures of snag abundance and volume (e.g., Ferguson and Archibald 2002, Woodall and Westfall 2009). Forest type group was another variable that was extremely significant in models of snag abundance and size, CWD, and FWD. When snags are present, our results closely approximate those of Harmon (1993) in finding that softwood forest type groups have larger amounts of dead wood than hardwood forest type groups such as Maple/Beech/Birch.

Ownership was present in about half of the best models but was not always statistically significant. While there was no consistent pattern in ownership in predicting whether snags were present or their maximum size, Federal ownership was consistently ranked highest and private ownership lowest when we modeled only plots where snags were present. This finding is consistent with Federal mandates to manage for multiple uses, including wildlife habitat, by which private owners are not constrained.

The explanatory power of our models was not as high as that of other studies of dead wood, which have often found much greater explanatory power based on live basal area alone (e.g., Ferguson and Archibald 2002). One possible explanation for this is that the amount of dead wood in forests is known to vary widely in space and time (Harmon 1986), and most studies of dead wood have been conducted on small numbers of plots that are relatively homogeneous in forest type, stand age, and/or management practices (e.g., Goodburn and Lorimer 1998, Sweeney et al. 2010). Thus, the results of our models and the general trends in our summary data may more fully capture the natural variability of dead wood than has previous research. A second reason may be that the upper Midwest is currently a landscape in transition. Only since the 1950s have early successional forest types begun to transition to later successional types (Schulte et al. 2003), and major housing density changes began twenty years after the forest type transition began. The majority of forests included in this study were quite young, with nearly 20% under 40 years old and 56% less than 65 years old at the time they were sampled. More work is needed to tease apart relationships between down dead wood and the other factors we examined, and it may be possible to do by focusing more closely on areas where forests types and housing growth have come closer to an equilibrium state.

Our results also point to the value in exercising caution when extending the results from a limited number of sites or even a whole state to the regional scale. While many management issues need to take place at a regional or national scale (e.g., Gavier-Pizzaro et al. 2010), finer scale factors may change the relationships at the local level. Our research did not find the consistent, strong relationship between dead wood levels and housing density that have been found on individual lakefronts (e.g., Schindler 2006, Marburg et al. 2006), and while our results were generally consistent within individual states they sometimes varied considerably in both significance and direction of the effect among states. Had we examined the region as a whole, without consideration of individual states and ecological sections, the picture painted by our results would have been incorrect for any one state and for each of our analyses. As data on dead wood volume and type are important to forest managers seeking to maximize biodiversity, it is important to consider local ecological and biotic differences when extrapolating from other research.

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Table 1. Number of plots included in analyses of snags trees.

State	# of plots used in logistic models of DEADCOUNT	# of plots used in general linear models of DEADCOUNT & MAXDBH
Michigan	7940	6017
Minnesota	3737	2838
Wisconsin	4682	3439

Table 2. Levels of categorical explanatory variables included in the analysis.

Ecological Section	Forest Type Group	Ownership
212H – Northern Lower Peninsula	Aspen/Birch	Federal
212J – Southern Superior Uplands	Elm/Ash/Cottonwood	State & Local Government
212K – Western Superior Uplands	Oak/Hickory	Private
212L – Northern Superior Uplands	Oak/Pine	
212M – Northern MN & Ontario	Maple/Beech/Birch	
212N – Northern MN Drift & Lake Plains	Spruce/Fir	
212Q – North Central WI Uplands	White/Red/Jack Pine	
212R – Eastern Upper Peninsula		
212S – Northern Upper Peninsula		
212T – Northern Green Bay Lobe		
212X – Northern Highlands		
212Y - Southwest Lake Superior Clay Plain		
212Z – Green Bay-Manitowac Upland		
222K - Southwestern Great Lakes Morainal		
222L – North Central U.S. Driftless & Escarpment		
222M – MN & Northeast IA Morainal		
222N – Lake Agassiz-Aspen Parklands		
222R – WI Central Sands		

Table 3. Summary of logistic regression for snag abundance (DEADCOUNT).

Sign of effect: '+' = positive, '-' = negative, 'Mixed' = effects were sometimes positive and sometimes negative.

Where p-values differed between models they are given as a range. (Next page).

		Ownership	Ecological section	Forest type group	Housing density	Stand age	Live basal area
Michigan	No. of models	4	6	10	5	8	10
	Sign of effect				Mixed	+	+
	P-values				0.45-0.95	<0.0001	<0.0001
Minnesota	No. of models	7	6	10	4	6	10
	Sign of effect				+	+	+
	P-values				0.52-0.82	<0.0001	<0.0001
Wisconsin	No. of models	5	6	10	6	8	10
	Sign of effect				+	+	+
	P-values				0.002-0.14	<0.0001	<0.0001
	Total no. of models	16	19	30	15	22	30
	Percent of total models	53.3	63.3	100.0	50.0	73.3	100.0

Table 4. Summary of regression analysis for snags where DEADCOUNT > 1.

Sign of effect: '+' = positive, '-' = negative, 'Mixed' = effects were sometimes positive and sometimes negative.

Where p-values differed between models they are given as a range. (Next page).

		Ownership	Ecological section	Forest type group	Housing density	Stand age	Live basal area	Model p-values	Adjusted R^2
Michigan	No. of models	5	8	10	4	6	10	<0.0001	0.09
	Sign of effect				+	+	+		
	P-values	0.21-0.39	<0.0001	<0.0001	0.35-0.56	<0.0001	<0.0001		
Minnesota	No. of models	4	10	10	5	6	8	<0.0001	0.06-0.08
	Sign of effect				-	+	+		
	P-values	0.15-0.38	<0.0001	<0.0001	0.22-0.48	<0.0001	<0.0001		
Wisconsin	No. of models	6	10	8	4	5	10	<0.0001	0.05-0.06
	Sign of effect				-	-	+		
	P-values	0.06-0.20	<0.0001	<0.0001	0.34-0.59	0.22-0.96	<0.0001		
	Total no. of models	15	28	28	13	17	28		
	Percent total models	50.0	93.3	93.3	43.3	56.7	93.3		

Table 5. Summary of regression analysis for maximum snag DBH (MAXDBH).

Sign of effect: '+' = positive, '-' = negative, 'Mixed' = effects were sometimes positive and sometimes negative.

Where p-values differed between models they are given as a range. (Next page).

		Ownership	Ecological section	Forest type group	Housing density	Stand age	Live basal area	Model p-values	Adjusted R^2
Michigan	# of models	4	10	6	5	10	8	<0.0001	0.07-0.08
	Sign of effect				+	+	+		
	P-values	0.69-0.99	<0.0001	<0.0001	0.20-0.32	<0.0001	<0.0001		
Minnesota	# of models	6	8	10	5	6	10	<0.0001	0.13-0.14
	Sign of effect				-	+	+		
	P-values	0.001-0.15	<0.0001	<0.0001	0.02-0.12	0.008-0.02	<0.0001		
Wisconsin	# of models	8	8	10	5	8	8	<0.0001	0.08-0.09
	Sign of effect				+	+	+		
	P-values	<0.0001	<0.0001	<0.0001	0.31-0.70	<0.0001	<0.0001		
	Total number of models	18	26	26	15	24	26		
	Percent of total models	60.0	86.7	86.7	50.0	80.0	86.7		

Table 6. Summary statistics for CWD total length and transect diameter.

Housing density *(units/km^2)*	# of plots	$\bar{x}$ total length (m)	Stdev (m)	$\bar{x}$ transect diameter (cm)	Stdev (cm)
0	41	7.7	4.4	16.0	5.6
0.1-2.0	20	6.6	2.8	5.9	4.1
2.1-4.0	22	5.8	1.4	7.1	3.3
4.1-8.0	22	6.2	3.1	5.8	9.7
8.1-16.0	9	5.0	0.9	6.8	2.0
16.1-32.0	12	5.3	2.3	6.5	1.7
Ownership					
Federal	58	8.5	5.2	15.8	10.2
State & Local Government	22	5.3	5.1	6.5	11.4
Private	44	5.6	4.7	6.6	9.7

Table 7. Summary statistics for FWD small, medium, and large diameter counts

	# of plots	$\bar{x}$ small diameter count	Stdev	$\bar{x}$ medium diameter count	Stdev	$\bar{x}$ large diameter count	Stdev
Housing density							
0	41	10.7	10.1	5.7	5.5	3.5	3.1
0.1-2.0	20	16.8	21.0	3.6	3.4	2.3	2.3
2.1-4.0	22	13.4	10.5	4.6	3.2	2.1	1.3
4.1-8.0	22	14.8	12.5	4.6	4.1	3.0	2.3
8.1-16.0	9	23.8	44.9	6.4	9.9	3.8	3.7
16.1-32.0	12	20.5	25.8	3.8	2.1	1.8	1.3
Ownership							
Federal	58	9.1	8.6	4.9	4.7	3.2	2.9
State & Local Government	22	12.2	12.5	4.2	4.4	2.4	1.8
Private	44	23.2	40.8	5.1	5.4	2.7	3.4

Table 8. Summary of results for linear models for CWD.

P-values for significant response variables

ns = not significant as a main effect but significant in an interaction term

	Mean Length (ln)	Mean Transect Diameter (ln)
Ownership		0.02
Ecological subsection (ES)		0.002
Forest type (FT)	0.04	0.001
Housing Density (HD)		ns
Stand age (SA)		<0.0001
FT*SA		0.004
HD*ES		0.0002
HD*SA		0.02
Model p-value	0.04	<0.0001
Adjusted R^2	0.22	0.86

Table 9. Summary of results for linear models for FWD

ns = not significant

	Small diameter count (ln)	Medium diameter count (ln)	Large diameter count (ln)
Ownership		ns	
Ecological subsection (ES)		ns	
Forest type (FT)	0.0004	ns	
Housing Density (HD)		ns	0.003
Stand age (SA)		ns	0.006
HD*SA		ns	0.004
Model p-value	0.0004	ns	0.02
Adjusted R^2	0.31		0.06

Figure 1. Housing density in the study area, 2000.

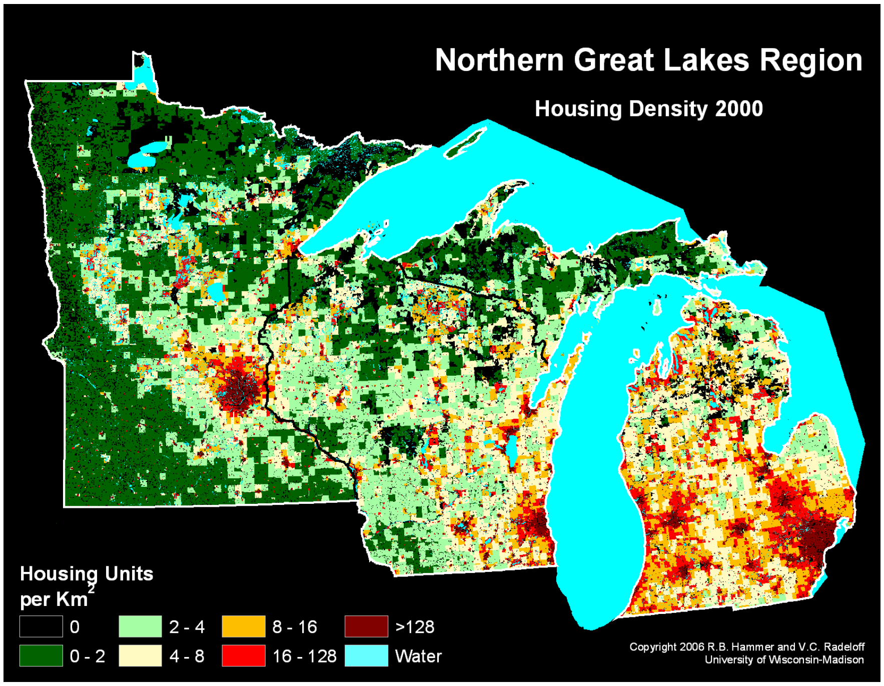
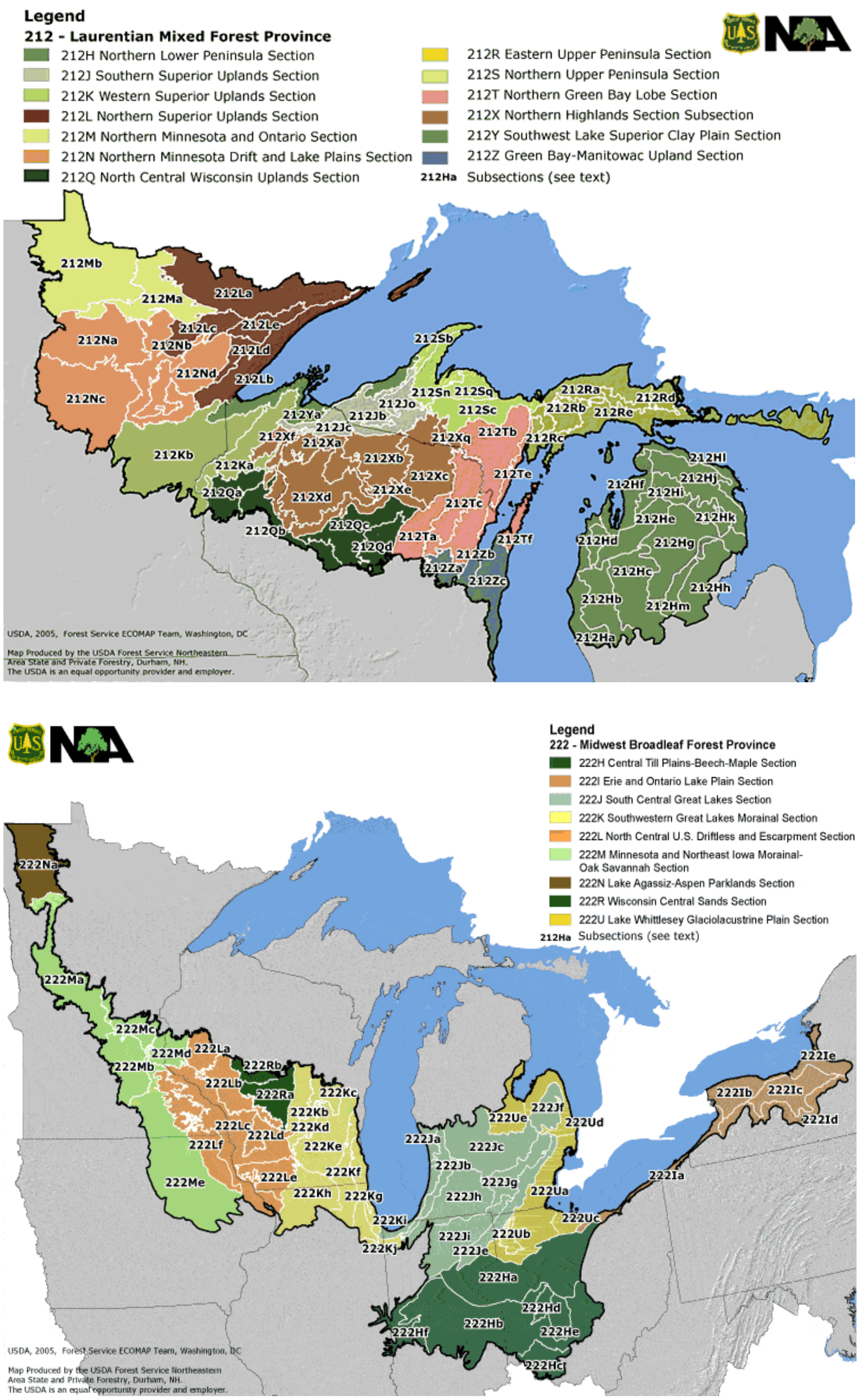


Figure 2. Ecological sections of Michigan, Minnesota, and Wisconsin.



Source: USDA Forest Service ECOMAP
http://www.na.fs.fed.us/sustainability/ecomap/provinces/sec_212/s212.shtm
http://www.na.fs.fed.us/sustainability/ecomap/provinces/sec_222/s222.shtm

Chapter 4: On the Cutting Edge? Evaluating the effects of housing density and other factors on timber harvest in the upper Midwest

Abstract

Housing density has been increasing in numerous regions throughout the United States, particularly in areas rich in environmental amenities. Demand for second and vacation homes is particularly high in and around forested areas, but the effects of housing growth on timber harvest are not well understood. We examined the relationships timber harvest and housing density in the upper Midwest (Michigan, Minnesota, and Wisconsin), an area that has experienced considerable housing growth over the past three decades. Our analyses indicated that ownership is the most important variable in predicting whether timber harvest occurs, with private landowners most likely to harvest. Housing density and stand age were the variables most strongly related to removal volume, but they were not consistent in either their statistical significance or the direction of their effects. Our results provide empirical evidence that increases in housing density may affect timber harvesting, but this activity is strongly influenced by other factors as well. Since housing densities in this region are expected to continue increasing in the coming years, forest managers and policy makers should continue to monitor the potential economic and ecological impacts of housing growth on forest resources.

Introduction

Due to declining average household size and other factors, over the past several decades numerous regions of the U.S. have been undergoing considerable increases in housing density, often at the urban fringe and in rural areas with high amenity values (Beale and Johnson 1998, Vesterby 2002, Nowak et al. 2005). Natural resource managers throughout the United States frequently cite the increasing proximity of human dwellings and other anthropogenic disturbances to forestland as a growing concern (Wear et. al. 1996, Riemann and Tillman 1999, Egan et al 2007). Negative ecological effects associated with increases in housing density include changes in the distribution or behavior of wildlife (Theobald et al. 1997, Cincotta et al. 2000), increased invasion of exotic species (Gavier-Pizzaro et al. 2010), decreased forest interior habitat (Hawbaker et al. 2006, Gonzalez-Abraham et al. 2007), and changes in forest ecosystem structure and functioning (Matlack 1997, Cincotta et al. 2000). However, the influence of housing growth on forest management in the United States, while thought to be significant, is not well understood.

In particular, the potential effects of housing development on timber harvest are a significant concern for forest managers and the public alike. Changing conditions on and around U.S. forests have raised concerns about their ability to meet the goals of sustainable, multiple-use management (Haynes et. al 2007). Timber has always been one of the primary goods provided by forests in the U.S., and many local economies have traditionally relied on timber harvesting and processing forest products (Haynes 2003). Reductions in timber harvest may also have significant environmental consequences, including effects on carbon sequestration and global climate change (McWilliams et al. 2009). Thus, an understanding of the effects of residential development on timber harvest is crucial to achieving the multiple goals associated with sustainable forest management (Stein et al. 2007).

The states of the upper Midwest – Michigan, Minnesota, and Wisconsin – have experienced significant housing growth over the past several decades (Gobster et al. 2000). Housing densities have increased at a rate exceeding that of population growth since the 1970s (Gustafson et al. 2003), due in part to the growing prevalence of seasonal and retirement homes, particularly in areas with attractive recreational and aesthetic amenities (Fig.1; Green et al. 1996, Radeloff et al. 2000, 2001, Hammer et al. 2004). As a result, the vast majority of forests in this region are affected by housing development, exhibiting either at least low-level housing density or located less than 25 km from human settlements (Radeloff et al. 2005). While each single new house causes negligible impact, the cumulative effects of these individual changes over time and within a landscape or region may constitute a major impact (Theobald et al. 1997).

As housing densities increase, timber harvest and other forest management practices may be altered in response to a changing social context (Hull and Stewart 2002). On private land, forest owners may decide against silvicultural treatments such as thinning if they expect to develop their properties within the foreseeable future (Wear et al. 1999), and once new property owners have constructed a home they often want to protect the scenic beauty of their surroundings, potentially precluding timber harvest (Martus et. al. 1995, Green et al. 1996). Housing densities are often relatively high in the vicinity of protected areas (Radeloff et al. 2010, Wade and Theobald 2010), and thus demands for recreation and open space, negative public attitudes toward forest harvesting, and shifts in forest management objectives may result in decreased timber harvest on public land even though it is not available for development (Martus et al. 1995, Barlow et al. 1998). Increases in housing density may also cause practical barriers to timber harvest as subdivision results in parcel sizes that are not economically efficient to harvest (Barlow et. al. 1998, Gobster and Rickenbach 2004). While such parcelization may occur slowly over decades (Hawbaker et al. 2006), it can also occur rapidly when forest industry lands are divested and purchased by private individuals and nonforest industry corporations, as has happened in the upper Midwest and other parts of the U.S. in recent years (Woodard 2006).

Prior research has indicated that proximity to development and higher population densities result in lower rates of timber harvest even when forest land area remains stable (Barlow et al. 1998, Munn et al. 2002), with commercial timber harvest ceasing when population densities exceed 58 people/km^2 (Wear et al. 1999). Such results highlight the potential effects of housing change on timber harvest, but because they use population density as the predictive variable these studies may miss the effects of housing density increases that occur in nonmetropolitan counties with high recreational and environmental amenity values (Radeloff et al. 2000). In such regions, an increase in housing density may occur without corollary increases in population density. Probability of timber harvest has been found to be negatively correlated with road density (McDonald et al. 2006), a landscape attribute that is highly correlated with housing density (Hawbaker et al. 2006). Building density has also been negatively correlated with management activities such as precommercial thinning and post-harvest planting (Kline et al. 2004, Kline and Azuma 2007). Although these studies found no statistically significant relationship between harvest activities and building density, they were conducted in areas that are among the least densely populated in the continental U.S. and where overall building densities are very low. Thus, variability in housing density may not have been sufficient to produce an observable effect on timber harvest.

The three states of the upper Midwest contain approximately 52 million acres of forestland (Shifley and Sullivan 2002), have been a center of timber industry activity since the 1800s (Stearns 1997, Mendel 2006), and exhibit considerable variation in housing densities (Fig. 2), providing an excellent opportunity to examine the impacts of housing growth on timber harvest. Our goal in this research was to examine the relationship of housing density to the likelihood and volume of timber harvest in the upper Midwest, hypothesizing that both of these responses would be negatively correlated with increases in housing density.

Data sources

Forest Attributes

We conducted all analyses using USDA Forest Service Forest Inventory and Analysis (FIA) Phase 2 (P2) data collected in Michigan, Minnesota, and Wisconsin. P2 plot inventories include assessments of all standing live, dead, and harvested trees > 2.5 cm in diameter at breast height (dbh)(Bechtold and Patterson 2005). P2 plots are 0.41 ha in size and occur at an intensity of one plot per 1,200 ha in Minnesota and Wisconsin and one plot per 800 ha in Michigan (R. McRoberts, pers. comm). Each plot may include one or more conditions, which are defined by changes in ownership, forest type, stand age, reserve status, and other factors, and each condition on the plot is assigned a condition proportion (USDA 2009).

All P2 data were downloaded from the FIA website (*http://199.128.173.17/fiadb4-downloads/datamart.html*). Although the FIADB available for download include geographic coordinates for every plot location in the database, these are not the precise location of the plot centers. In order to protect individual landowner and other proprietary information, the Forest Service is legally required to alter the plot coordinates for the web-available FIA data (USDA 2009). Our own prior research, however, indicates that housing densities assigned to FIA plots using altered plot coordinates are rarely significantly different from those assigned using true plot locations (Sabor et al. 2007).

Housing density

We used housing density data for the year 2000 derived from U.S. Decennial Census data at the partial block group (PBG) level (Hammer et al. 2004). Due to privacy concerns, certain Census data are only released at the block group level. These block groups are transected by political and geographic boundaries that permit their division into multiple partial block groups, which have a mean size one-tenth that of block groups. PBGs therefore provide much better spatial resolution while including the complete array of Census population and residential attribute information available at the block group level. Sizes of PBGs in the study region range in size from <0.01 to 1,640 km^2 (mean = 2.5 km^2), and include residential densities ranging from 0.0 to 16,945 units/ km^2 (mean = 71.45 units/km^2).

A more complete description of PBG delineation and housing projection methods is available in Hammer et al. 2004 and Radeloff et al. 2010.

Data analysis

Housing density and FIA data were compiled using a geographic information system (GIS), and each FIA plot was assigned the housing density of the PBG in which it was located. In order to preclude issues of spatial autocorrelation within plots and allow us to treat each plot as one sample unit, statistical analyses included only those P2 plots that were at least 75% in a single condition. We also checked for spatial autocorrelation among plots in the model residuals using variograms.

Beginning in 1999, FIA began phasing in a nationally standardized plot design and annualized inventory system (USDA 2009). As a result of this sampling redesign, many previously inventoried plots were replaced. Since harvest events are recorded and removal volumes estimated only for remeasured plots, analysis was necessarily limited to these plots (Table 1). We also excluded from the analysis any plots that were not classified as timberland, which is defined as forestland that is producing or is capable of producing crops of industrial wood and that is not withdrawn from timber utilization by reserve status (Smith 2009).

The response variables we modeled were harvest occurrence and removal volume on FIA plots. Since the data on harvest occurrence contained large numbers of zeroes and therefore did not conform to the standard error models used with general linear models, we modeled these data using logistic regression (Barry and Welsh 2002). To determine if the probability of harvest was the same across all housing densities, we summarized the percentage of plots that were harvested above and below three housing density categories. The first category, 6.17 housing units/km^2, equates to approximately 1 house/40 acres and is considered by some to be a threshold beyond which land is no longer rural (Stein et al. 2005, Theobald 2005). However, because there is no universally agreed-upon level of housing density at which the landscape transitions to nonrural, we also doubled and quadrupled that figure to derive the other two categories, 12 housing units/km^2 and 25 housing units/km^2. We used general linear models to examine harvest volume removal using only plots that had at least one tree that had been recorded as harvested (Table 1). Harvest removal volume for each plot was calculated by summing the removal volumes of individual trees that had been harvested.

We used five explanatory variables describing stand and site characteristics to model harvest occurrence and removal volume: housing density, land ownership, stand age, site class code, and timber type (Table 2). Timber type was assigned a value of '0' if the forest type on the condition was either softwood or aspen/birch and a value of '1' for any hardwood forest type other than aspen/birch. Data for all explanatory variables except housing density were taken directly or derived from the FIA database. Prior to regression analysis, we log transformed the response and explanatory variables as necessary in order to satisfy model assumptions.

An initial regression analysis indicated that responses were not uniform among states, so we ran regressions for each response variable separately for each state. We ran all possible valid models for both the logistic and linear regression analyses and subsequently used Akaike's Information Criterion (AIC) to choose the best subset of 10 models for each dependent variable (Burnham and Anderson 2002). Rather than resulting in one single "best" model, this approach highlights variables that appear repeatedly in the best models and indicates whether they have a consistently positive or negative relationship to the response variable. We then counted the number of times each variable was included in the best subset for each state and the direction of the effect for each continuous variable. We ranked the estimate for each level of the categorical variables for each model and compared rankings for all models in the best subset to determine if the rankings were consistent across models.

Results

The percentage of plots where harvest occurred was generally similar between plots that fell above and below selected housing density thresholds (Table 3). In logistic regressions for harvest occurrence, ownership was the most strongly related variable, occurring in 93.3% of the best models. Stand age was included in 70% of the best models, while housing density, site class code, and softwood were present in approximately half the models. The effects of ownership were always significant and the rank order of ownerships was consistent across all models, with privately owned lands ranking highest and Federal lands ranking lowest. Stand age was always negatively related to harvest incidence but the effects were rarely significant. Housing density was not statistically significant or consistent in the direction of its effect. Although there was some inconsistency in the rank order of site class codes, the most productive sites (class code 1 or 2) in each state were always negatively related to timber harvest occurrence. The rank order of timber type was also consistent, with hardwoods ranking higher than softwoods and aspen/birch (Table 4).

Housing density, stand age, and timber type were the variables most strongly related to removal volume, appearing in more than half of the best models. Both housing density and stand age varied in the statistical significance and direction of their effects. The rank order for timber type was not consistent and the effect of this variable was not significant. Ownership and site class index appeared in more than one-quarter of the best models but these effects were not significant or consistent in rank order. Explanatory power for these models was low, ranging from 1-5% (Table 5).

Discussion

Our results provide empirical evidence that increases in housing density may affect timber harvesting, but they strongly suggest that many other factors influences this activity as well. In particular, our analyses indicated that the factors most likely to exert a strong influence on the decision to initiate timber harvest differ from those that affect the decision of how much to harvest once the choice to engage in silvicultural management has been made.

Ownership was the most significant variable explaining whether harvest occurred or not, with private lands always most likely to undergo harvest. This result makes sense given that changing management goals resulted in decreased timber harvest from federal lands in the 1990s and 2000s (Haynes et al. 2007). Private lands did not operate under such constraints and would be expected to experience higher initial levels of timber harvest as housing density increased. However, once a harvest decision had been made, ownership played neither a consistent nor a strong role in determining how much timber was removed.

Surprisingly, stand age had a uniformly negative impact on the decision to harvest timber. Since stand age is correlated with live basal area, we expected the opposite relationship. However, this relationship may reflect the overall difference in time to maturity for most hardwoods as compared with softwoods and relatively fast-growing hardwoods such as aspen/birch. The landscape transition to later, hardwood successional types such as maple/beech/birch is relatively recent in this region (Schulte et al. 2003), and the majority of forests in the study area were less than 65 years old at the time they were sampled.

Housing density appeared in approximately half the models in the best subsets for both harvest occurrence and timber volume removal, indicating that it likely does influence forest management in some way. However, the significance and direction of this effect was not consistent among either response variables or states. Housing density was more likely to be significant in determining how much timber was removed rather than whether a harvest decision was made, but the influence was positive for Michigan and Minnesota and negative for Wisconsin. One possible explanation for the differences in the effects of housing density is that our results reflect which stages of the building cycle were more locally active at the time these data were sampled, since timber harvest might be expected to go up as land is initially cleared for housing and then level off once houses are built.

Our models had low explanatory power even though they included multiple variables thought to influence timber harvest activities. Due to the privacy restrictions placed on publicly-available FIA data, we were not able to evaluate certain complex interactions between the explanatory and predictor variables that may have influenced our results. For example, we were not able to differentiate among different types of private ownership, but levels of housing density and the intensity of timber harvest may vary between and among corporate and nonindustrial private landowners. Timber Investment Management Organizations (TIMOs) and Real Estate Investment Trusts (REITs) have both become increasingly common classes of corporate forest ownership in the Midwest over the past two decades (Mendell 2006), but the consequences of each of these types of ownership on housing and timber harvest may differ markedly. TIMOs are more likely to engage in forest management that resemble traditional forest products industries while REIT-owned land is typically divided into smaller parcels and sold for housing development (Fernholz et al. 2007). At the other end of the spectrum, nongovernmental organizations are private owners that may own large tracts of land where neither timber harvest nor housing growth occurs.

There may also be factors influencing timber harvest that are not included in FIA data and thus were not measurable in this study. Possible explanatory factors not accounted for in these analyses include distance to the nearest mill, parcelization, stumpage prices, and changing landowner attitudes. Global market conditions may also affect timber harvest. Throughout the 1990s and early 2000s, expanding global production of timber and timber products stabilized prices (Haynes et al. 2007) and a strong dollar made U.S. products less competitive in the market (Haynes 2003), which may have resulted in decreased domestic timber harvest.

Domestic timber production has been declining over the past 20 years, while consumption has remained relatively stable (Morgan et al. 2009). While the results of this study did not find that housing growth is currently a major contributor to declines in domestic timber supply in the upper Midwest, they do suggest that housing plays a role. As low- and mid-density housing growth continues to rise in this region over the next several decades, it is likely that the influence of housing density on timber harvest will continue to grow. Forest managers and policy makers should continue to monitor the potential economic and ecological impacts of housing growth on the region's forests.

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Table 1. Number of plots included in analyses of harvest occurrence and removal volume .

	Michigan	**Minnesota**	**Wisconsin**
Total plots	1482	2001	1497
Plots with harvest	269	313	274
Percent plots harvested	18.2	15.6	18.3

Table 2. Levels of categorical explanatory variables included in the analysis.

Site class code	Ownership	Timber type
1 – 15.8+ m^3/ha/year	Federal	**0** – Softwood forest and Aspen/Birch types
2 – 11.5-15.7+ m^3/ha/year	State and Local Government	**1** – Hardwood forest types
3 – 8.4-11.4 m^3/ha/year	Private	
4 – 5.9-8.3 m^3/ha/year		
5 – 3.5-5.8 m^3/ha/year		
6 – 1.4-3.4 m^3/ha/year		

Table 3. Summary of timber harvest occurrence by housing density.

(Next page).

	Plots w/harvest	HD ≤ 6.17	HD > 6.17	HD ≤ 12	HD > 12	HD ≤ 25	HD > 25	Max harvest housing density
Michigan	18.1%	18.8%	16.8%	18.8%	14.1%	18.0%	21.2%	272.1
Minnesota	15.6%	15.2%	20.3%	15.6%	18.8%	15.7%	13.0%	33.3
Wisconsin	18.3%	18.2%	19.0%	18.1%	22.5%	18.2%	25.0%	334.5

Table 4. Summary of logistic regression for timber harvest occurrence.

Sign of effect: ‘+’ = positive, ‘-‘ = negative.

Where p-values differed between models they are given as a range. (Next page).

		Ownership	Timber type	Site Index class	Housing density	Stand age
Michigan	# of models	10	4	2	6	5
	Sign of effect				-	-
	P-values				0.11-0.16	0.07-0.17
Minnesota	# of models	8	5	6	6	10
	Sign of effect				-	-
	P-values				0.09-0.93	0.07-0.17
Wisconsin	# of models	10	5	8	4	6
	Sign of effect				+	-
	P-values				0.92-0.99	0.04-0.19
	Total # of models	28	14	16	16	21
	Percent of total models	93.3	46.7	53.3	53.3	70.0

Table 5. Summary of general linear models for timber harvest removal volume.

Sign of effect: '+' = positive, '-' = negative, 'Mixed' = effects were sometimes positive and sometimes negative.

Where p-values differed between models they are given as a range. (Next page).

		Ownership	Timber type	Site Index class	Housing density	Stand age	Model p-values	Adjusted R^2
Michigan	# of models	0	4	6	3	3	0.11-0.49	0.01
	Sign of effect				+	+		
	P-values		0.30-0.49	0.08-0.13	0.38-0.63	0.33-0.52		
Minnesota	# of models	6	6	2	7	9	<0.005	0.04-0.06
	Sign of effect				+	-		
	P-values	0.07-0.21	0.004-0.20	0.26	0.05-0.11	0.001-0.07		
Wisconsin	# of models	2	5	1	7	7	0.03-0.15	0.01-0.03
	Sign of effect				-	Mixed		
	P-values	0.85-0.87	0.09-0.75	0.63	0.04-0.06	0.01-0.11		
	Total number of models	8	15	9	17	19		
	Percent of total models	26.7	50.0	30.0	56.7	63.3		

Figure 1. Seasonal homes as a percent of total housing units in the upper Midwest.

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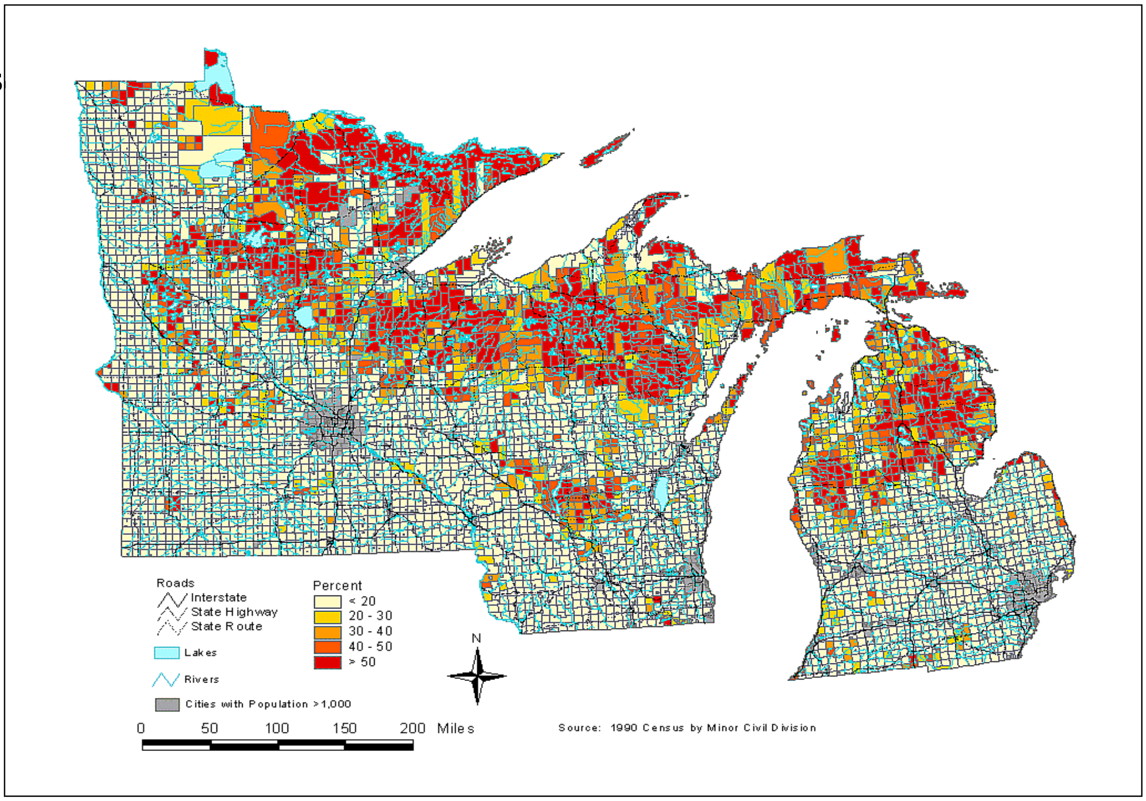

Figure 2. Housing density in the study area, 2000.