National Park Service
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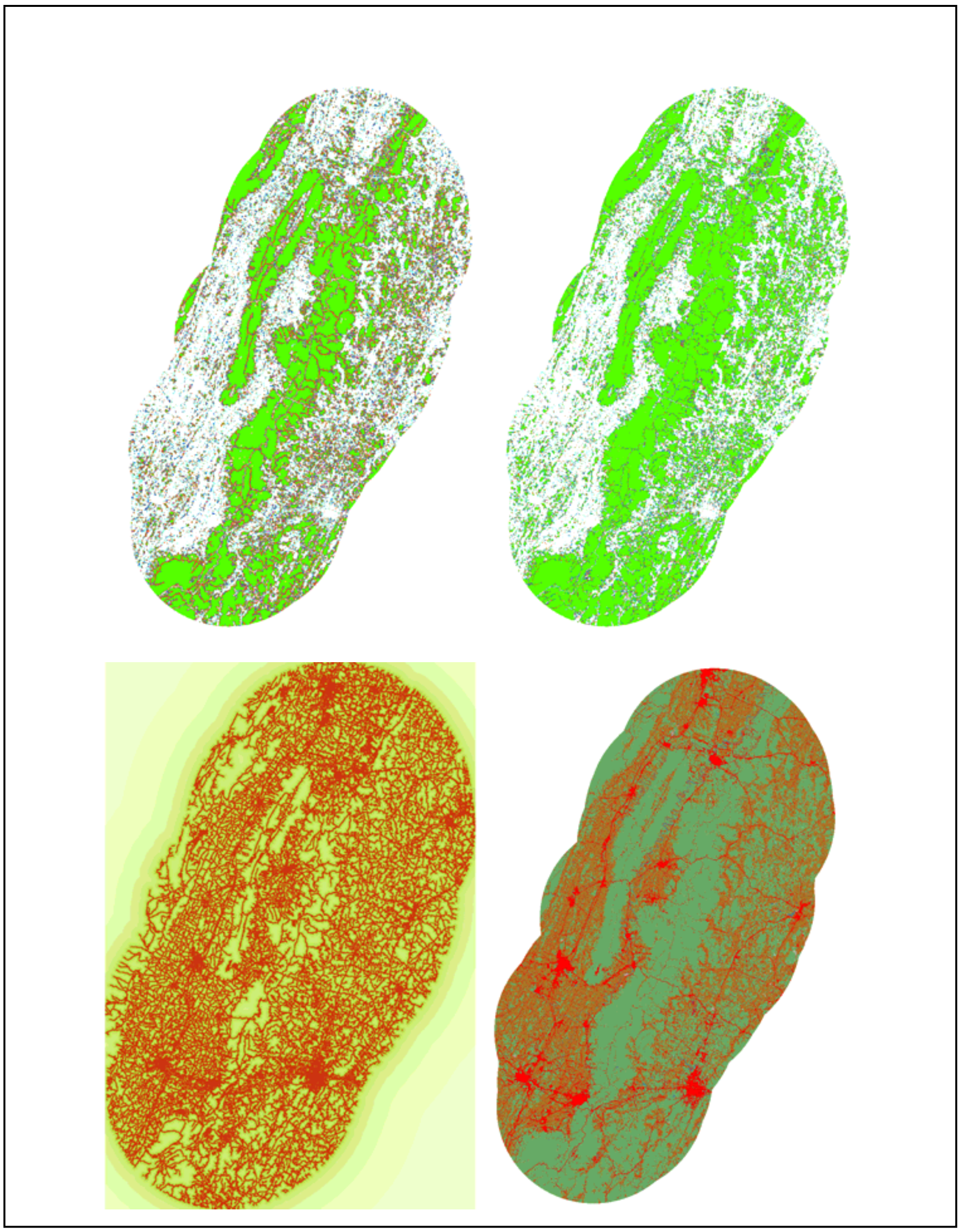
Natural Resource Program Center

A Guide to Interpreting NPScape Data and Analyses

Natural Resource Technical Report NPS/IMD/NRTR—2009/XXX



ON THE COVER

Four NPScape maps of Shenandoah National Park,Virginia, and the 30 km area around the park boundary. Top left: forest pattern with a 150 m edge width. Top right: landscape pattern with 30 m edge width. Bottom left: distance to a road. Bottom right: 2001 land cover.

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Abbreviations

Abbreviation	Full name
AOA	Area of analysis
BLM	Bureau of Land Management
CRI	Conservation Risk Index
DOI	Department of Interior
FWS	US Fish and Wildlife Service
GAP	Gap Analysis Program
HUC	Hydrologic unit code (USGS)
IBI	Index of Biological Integrity
IMD	Inventory and Monitoring Division
I&M	NPS Inventory and Monitoring Program
LCC	Land cover change
LULC	Land use/Land cover
MDS	Measurement Description Summary
MRLC	Multi-Resolution Land Characteristics Consortium
MSPA	Morphological spatial pattern analysis
NASA	National Aeronautics and Space Administration
NLCD	National Land Cover Dataset (see MRLC)
NPS	National Park Service
PAD	Protected Areas Database
USFS	United States Forest Service
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey

Acknowledgments

This report is just one product from the combined effort of the entire NPScape team. The authors are responsible for the text in this report, but the underlying data, analyses, and maps used as illustrations and examples are mostly the result of the very substantial efforts of other members of the NPScape team. Peter Budde, Lisa Nelson, Brent Frakes, Bill Hovanec, Ursula Glick, and Molly Thomas have worked extensively with most of the NPScape data sets. They were responsible for ingesting data, helping design analyses and procedures, processing enormous quantities of data, producing maps and other reports, and generally handling data management and data and product flows. Mike Story, Thom Curdts, and Shepard McAninch focused their considerable expertise and skills on land cover data and have led efforts to acquire, evaluate, and process land cover products. Allison Lundeby was a great help in the production of this report. The entire team participated in developing NPScape measures, the many other NPScape products that include data sets, Measure Description Summaries, Standard Operating Procedures, and various analyses and reports.

1 Introduction and Background

The goal of NPScape is to support NPS management of natural resources by providing relevant landscape-scale information to each of the NPS units with significant natural resources. NPScape products are designed to help resource managers, superintendents, and planners more effectively manage natural resources within parks.

The landscape context of a park can directly affect the condition of natural resources within that unit, and affect the importance or value of those resources to the public and for conservation. The area of suitable habitat near a park often affects wildlife dynamics within that park, and land use in upstream watersheds can affect aquatic resources within the park. The rarity of a species or habitat in the local and regional area can increase the value of that resource to the public. This interpretive guide focuses on the landscape context and effects of landscape-scale effects on the condition and potential future condition of natural resources. It provides little guidance on the consequences of changes to the value of rare or common resources at a specific location.

More than a decade ago, most park managers said resources in parks had already been damaged by activities outside park boundaries (GAO 1994). Projections of future population and land use trends indicate the impacts of anthropogenic activities to park resources will increase. These trends emphasize the need for the information provided by NPScape. To plan and implement actions that preserve park resources from broad-scale activities, we must identify important landscape-scale resources and places, assess levels of protection and risk, and figure out how best to act.

NPScape products were developed within a conceptual framework that links measurable attributes of landscapes to resources within parks. NPScape focuses on broad-scale factors and measures, where consistent data are available for at least the conterminous United States. Most NPScape data sources and products are best suited for analysis of areas that are hundreds to thousands of square kilometers. Most NPScape land cover data cannot be interpreted at the individual pixel level, and analyses are valid only when data are aggregated over larger areas. NPScape relies on US Census Bureau data, which is available at various scales. Much more information is available at the census block-group level (on average, an area with about 1200 people) than at the individual census block level. We expect these national-scale datasets to be consistently updated in the future, which permit us (or you) to estimate quantitative rates of change. Further, most projections of future landscape conditions are at these scales, or more commonly at the even broader county scale. Future projections may inform park staff of important changes long before they occur, facilitating mitigation, focused monitoring, and other management actions to be implemented in time to make a difference. Finally, centralized analyses of consistent data by NPScape reduced the duplication of effort that would have occurred if each I&M Network or larger park unit had independently acquired, processed, and analyzed those data for their own unit. Conversely, there is no logistical savings from centralized analysis of fine-scale measures where the availability, content, and format of data differ enough that each area requires a separate and often unique process for data acquisition, aggregation, and analysis.

The key products from NPScape are data, methods, analyses, and results. Technically adept users with a strong background in landscape ecology may be most interested in NPScape data, which we assembled for our analyses and provide in geographic subsets for each park, and methods, which we provide as documented scripts that can be easily modified for other uses.

This guide is for everyone else. That said, even the seasoned landscape ecologist is sure to find new and interesting information here.

This interpretive guide describes the scientific underpinning of NPScape measures, as well as literature summaries, citations, and examples that help readers put specific results in a broader context. We sought information that will help readers understand NPScape results, and how to use these results to evaluate condition and threats to resources and values where they live or work. In this guide, you will find our review and synthesis of scientific findings that help evaluate NPScape results, and citations that provide an entry into the vast scientific literature relevant to evaluating NPScape data. Our syntheses are not a comprehensive review; this task is well beyond our resources. We tried to write a thorough and balanced assessment of current knowledge. These accounts invariably have strengths and weaknesses, and we welcome comments, suggestions, pointers to especially good information, and other feedback that helps us improve this document.

1.1 Conceptual Foundation

In this section, we present a series of conceptual models and frameworks. These models illustrate connections among key environmental attributes and illustrate why and how NPScape measures are useful for evaluating the condition or context of parks. No single model or framework is sufficient to meet all needs; we present these selected models because they are easily understood, apply equally well to most systems, and they can easily be modified to better represent specific situations or needs. These models are presented in order from more general and broadest scale to the more specific and detailed.

At the most general level, Figure 1.1 (A) illustrates factors that are the primary determinants of ecosystem state and sustainability over broad scales of time and space. Five 'state factors' (upper case letters in Figure 1.1 A) impose fundamental constraints on ecosystem processes and these factors largely determine such things as dominant vegetation type (e.g., deciduous forest, tundra, grassland, etc.). These state factors are external to the ecosystem (as represented by the circles in Figure 1). The specific characteristics of a particular ecosystem are determined by the 'interactive controls', which regulate and respond to ecosystem processes. Implicit in this model is the assertion that a significant change to any interactive control will lead to a different ecosystem. Figure 1.1 (B) includes anthropogenic factors and activities that can significantly alter one or more of the interactive controls, and thus threaten the sustainability of the system. NPScape provides direct measures of some of these general stressors (e.g., land cover types), and strong indices to some others (e.g., road and population density are strongly related to vehicle emissions and to night lights).

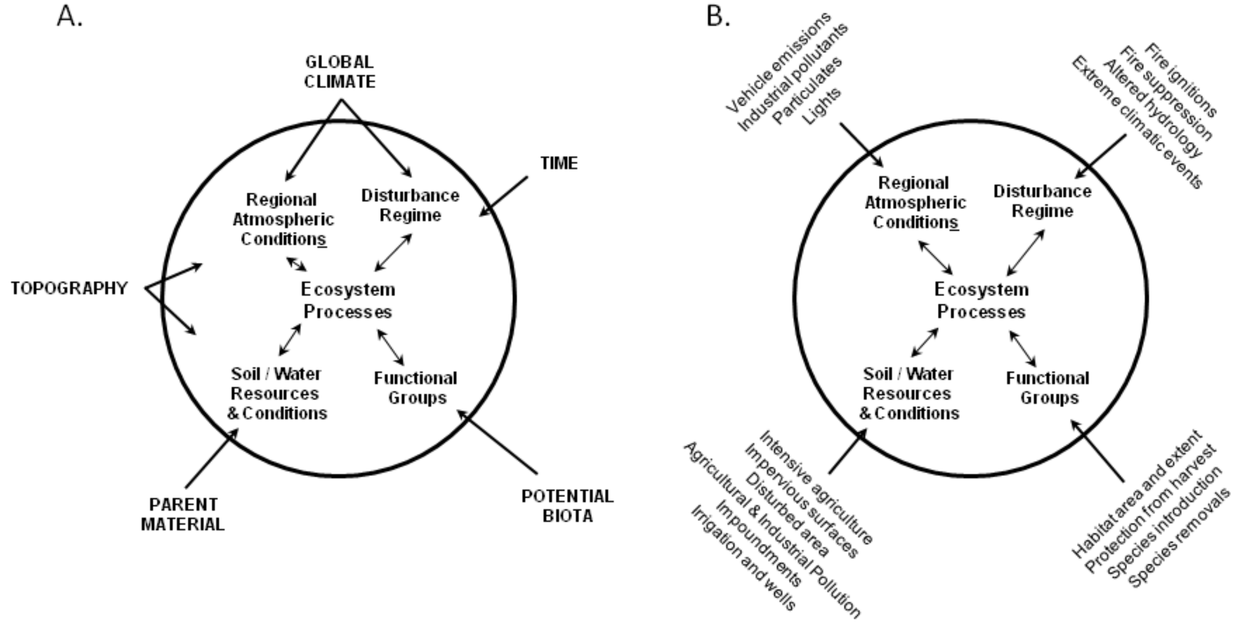


Figure 1.1. (A) General ecosystem model showing primary constraints (upper case) and ecosystem attributes or 'interactive controls'. (B) Factors that commonly alter ecosystems at local to landscape scale. Modified from Chapin et al. (1996) and Miller (2005).

At a somewhat finer level of resolution, the model in Figure 1.2 more directly focuses on the types of measures evaluated by NPScape and the interactions between them. Together, these measures describe landscape condition at a range of scales of space and time. Historical information provides a context for change - how, and how fast, did we get here? Historical rates and magnitudes of change help determine the urgency of decisions, and they provide a social context important for understanding and relating to people that have experienced and lived through these changes . Current information reflects status, and projections help identify trajectories in resource conditions or ecological drivers. Projections are a critical aid to planning for a future one desires, rather than a future that just happens.

Understanding context is important to identify and evaluate opportunities to prevent resource damage or loss, to assess actions for restoration, and to identify key area or threats to park resources. A clear understanding of the relationships between park natural resources, nearby protected areas, and the corridors linking them presents an opportunity to better preserve these resources. While many of the measures are, by themselves, insufficient to determine the status of a resource, the sum of this information can provide a rich picture of the history, status, risks, and opportunities for conservation decisions that can substantially alter the future condition of park resources.

While the conceptual models presented above provide a framework for considering landscape-scale factors that affect park resources, a more detailed picture is necessary to directly link landscape-scale measures to ecological processes or resources. This issue of linking pattern to process is one of the seminal challenges in landscape ecology (Turner 1989).

To explain how land use intensification can affect protected areas, Hansen and Defries (2007) developed the conceptual framework illustrated in Figure 1.3. This model links landscape changes to key ecological processes, effects on natural resources, and it identifies attributes

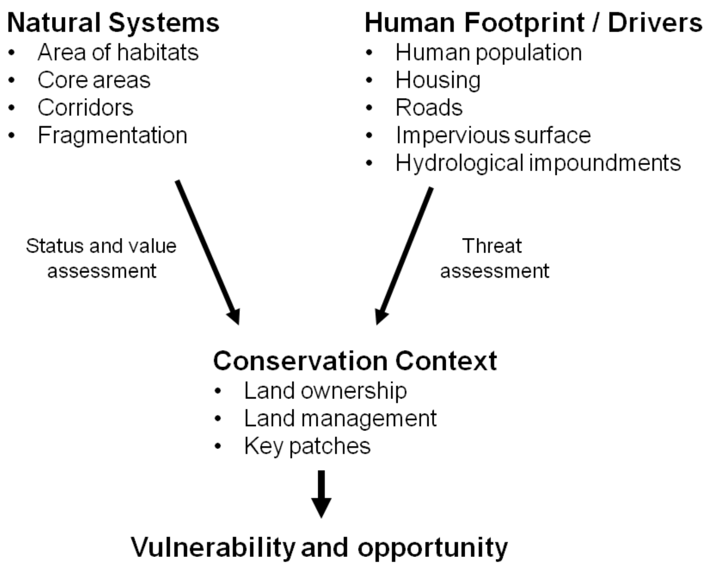


Figure 1.2. Broad categories of measures considered in this report, and how they contribute to understanding the landscape context of parks.

suitable for monitoring. Figure 1.3 identifies a number of variables that can be evaluated with NPScape data.

The ecological relationships and processes articulated in Figure 1.3 are common to many parks, and they provide a sound basis for evaluating the potential effects of landscape-scale measures on park resources. These include natural disturbances, (e.g., fire, floods), critical habitats, ecological flows (e.g., pollutants, nutrients), and both direct and indirect effects of humans (e.g., poaching, noise, behavioral disturbance).

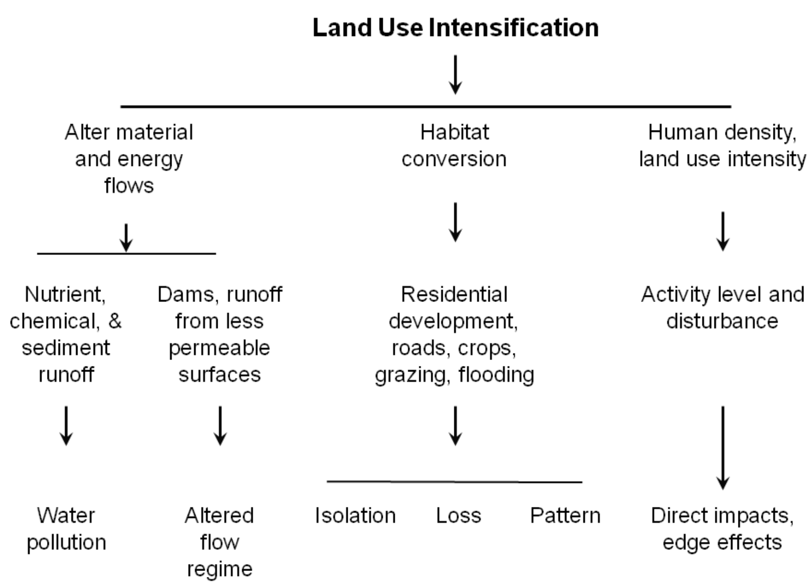


Figure 1.3. Mechanistic links between attributes of land use intensification and natural resources, depicting the relationships described by Hansen and DeFries (2007).

1.2 Spatial Scales for Analysis

Landscape attributes can affect park resources in fundamentally different ways at four broad spatial scales: the landscape within the park itself; the boundary layer adjacent to the park and perhaps 1-3 km wide; the local area, within approximately 15-40 km of the park; and the regional context, defined by ecoregion or watershed at some level. At the time of this report and data distribution (December 2009), NPScape analyses have only been completed at the scale of the DOI Geographical Framework regions and for area defined by a park and all areas within 30 km of the park boundary. During 2010, we anticipate an update to this report and completion of analyses of additional metrics and at other scales.

For many park decisions, the most relevant spatial scale of NPScape data will consist of the area within park boundaries. Wildlife may move among areas within the park; plants disperse and colonize, and development and visitor use have impacts that extend spatially from their locations with the park. This is also the scale where NPS management has the greatest control over landscape change. The size, spatial location, and geographical context of the park determine the relative importance of landscape factors within the park compared to those in the surrounding area. A few of the very largest NPS units may approximate self-contained or self-sufficient ecosystems. But the vast majority of park units are embedded in larger ecosystems, and relatively few wildlife populations or food webs are sustained solely by resources within the park boundary.

The landscape immediately adjacent to the park has a large effect on permeability or connections between the park and the surrounding landscape, and thus on the importance of landscape attributes in the broader local area. Roads and highly developed areas can act as barriers to movement of wildlife between the park and the surrounding landscape, isolating park resources from habitat outside parks (Figure 1.3). Land cover characteristics at this spatial scale can have a strong effect on the probability that wildfires ignited on the surrounding lands will burn into the park. Activities and processes occurring within the area immediately around a park are sufficiently close that even small effects can propagate into the park and affect park resources. Housing adjacent to the park boundary can increase pet – wildlife interactions, increase exotic plant introductions from yards and gardens, and increase the ignition frequency of small fires. Roads near park boundaries provide additional visitor access, and may increase the potential for poaching within the park.

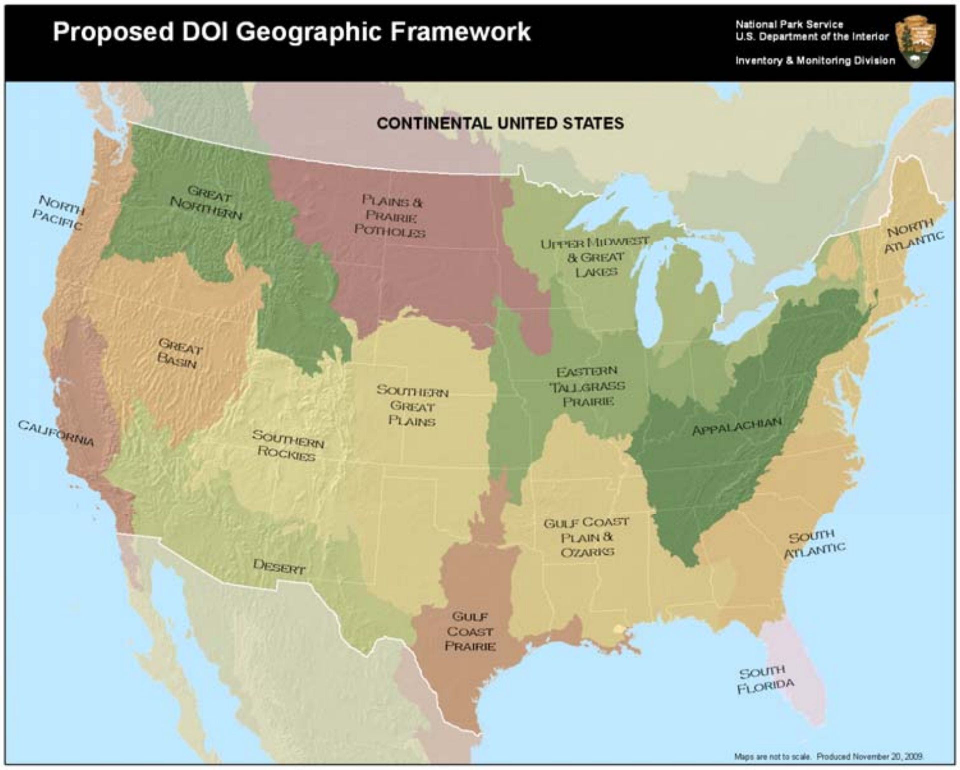
We acknowledge the importance of landscape attributes at these scales (within park and immediately adjacent), but many of the regional to national datasets available to NPScape are unsuited for analysis at such fine scales. Local aerial photography, county property records, and county or state road and traffic data are more suitable for analyses of landscapes within a few kilometers of a park. Because the format and content of these data vary widely across jurisdictions, too much time and effort was required for us to properly acquire or analyze data at this scale.

Many ecological processes operate at a local scale on the order of 15-40 km (10-25 mi). The propagation of wildland fires, and dispersal of local pollutants occur at this scale; land cover and fuel types in the local landscape affect the frequency of fires burning into the park and the influx of local pollution, or daily movements of vagile animals. This local landscape scale is the important spatial scale for many wildlife populations. It is the scale of daily to weekly movements of birds and larger mammals, and of yearly to multi-decadal dispersal of amphibians, reptiles, and smaller mammals. The generally larger population numbers of smaller animals roughly balances out their longer time scales for movement at this scale, so population dynamics within a park for a wide range of species can be affected by habitat availability and connectedness at this scale. While there is no magic number to define this local scale, we chose 30 km from the park boundary as a rough and hopefully robust distance for quantifying landscape effects of this type. If analyses are needed for particular species or areas of management concern, NPScape data and methodology can be used with species-specific definitions to determine a more appropriate “local” scale.

Finally, landscape attributes at a larger, regional scale affect natural resources within the park on a longer time scale of decades to centuries. A view at this broad scale is necessary to plan for or respond to issues such as the ability of altered landscapes to support regional biodiversity, and to evaluate impacts from broad-scale drivers like rapid climate change. Populations of species beyond the local area do not interact in terms of population dynamics, but do interact on the time scales of gene flow and long-term extinction and recolonization.

The concept of ecoregions is an attempt to define geographic boundaries of ecologically related systems at a range of ecological resolution (degree of ecological similarity or strength of ecological interaction)(Bailey 1995), and thus are hierarchical in nature. Most ecoregion definitions include both climatic factors and biotic factors such as vegetation or community composition, although edaphic (geologic and soil) factors are often included as well. The spatial scale or extent of these regional effects varies considerably depending on the context of parks, the size of a park, ecosystem properties, location of the park in a watershed, and myriad other factors. Therefore, there is no single ecoregion definition that will apply to all regional-scale effects for all parks.

We summarize landscape attributes and provide regional-scale data by USGS/FWS Geographical Framework units (Figure 1.4). These units are biologically based, incorporating the Bird Conservation Regions, Freshwater Ecoregions of the World, and Omernick Level II ecoregions (see http://www.fws.gov/science/SHC/lcc.html). For some management questions, finer-resolution ecoregion definitions, such as Omernick Level III, may be more appropriate. When parks are on or near the boundary of two or more Geographic Framework units, evaluation of the regional context will likely require consideration of all nearby Geographic Framework units.



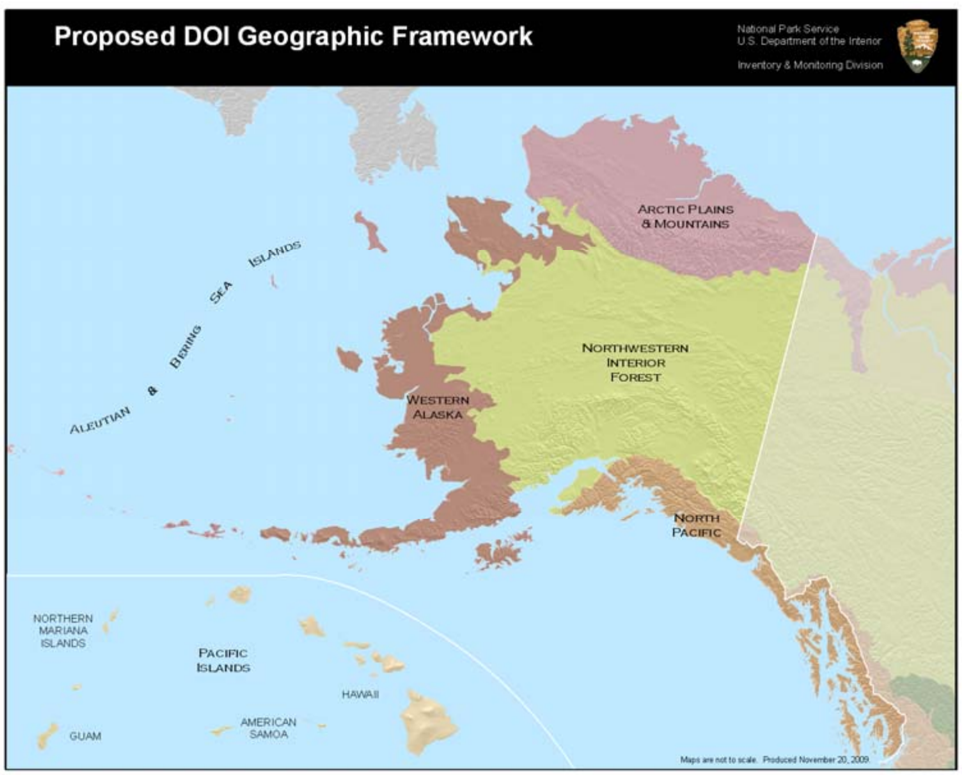


Figure 1.4. Proposed Department of Interior Geographical framework, labeled with names of Landscape Conservation Cooperatives (LCCs). See http://www.fws.gov/science/SHC/lcc.html for updates and additional information.

2 Issues Common to Many NPScape Analyses

Several issues and assumptions described below are inherent to the type of map-based analyses conducted by NPScape, and they need to be considered when using or interpreting NPScape data and results. Each of the measures (land cover/land use, landscape pattern, roads, population / housing, and conservation status), and many of the associated metrics (e.g., percent natural cover, road density, etc.), will be influenced these issues. We provide here a brief overview – additional details are in the Measure Description Summaries and, in some cases, in sections on particular measures.

The issues and assumptions can be broadly categorized as those related to the actual mapping (e.g., the land cover map) and those related to the statistical properties or calculation of the metric (e.g., proportions). Additional considerations for watershed-based evaluations are addressed later in the report.

2.1 Mapping

Issues and assumptions related to mapping are typically a result of 1) thematic scale, 2) spatial scale, 3) spectral scale, and 4) overall study objectives. The standards and methodologies employed in each mapping effort will differ, thus so will the overall applicability of those data to any given situation. In general, the following apply to all mapping products:

- Thematic classes, such as land cover, assume single homogeneous categories based on the dominant type or value, and do not necessarily differentiate finer characteristics such as stand age or understory composition. In many cases, the classes assume stability and do not isolate disturbances. For example, in a land cover map such as NLCD, an area with a great deal of forest clearing (e.g., logging) may be classified as forest, grass, or shrub. Similarly, private land is identified as a single homogeneous category and individual private land units, owners, or management status are not differentiated unless the information was provided voluntarily (typically to recognize a long-term commitment to biodiversity through binding conservation easements, covenants, or institutional dedication).
- Boundaries between thematic classes (e.g., land cover types) along real environmental gradients are seldom as sharp as implied by maps. Transition areas between classes represent gradients of condition that likely change with time. Similarly, aggregating thematic classes (e.g., 'natural' versus 'converted') implies simple relationships when realistically they are also gradients that vary in time and space.
- Products are not necessarily consistent across mapping extents. For example, state Gap Analysis Program (GAP) products may not be consistent across state lines and therefore, compiled datasets such as the PAD-US may also be inconsistent. Regional GAP products are developed across multiple states to minimize this issue, but there may still be inconsistencies between regions.
- Metrics will not account for land units smaller than the minimum mapping unit of the input datasets and are valid only at the point in time that the data were acquired. In addition, as with all spatial analyses, inaccuracies in the input data will multiply in the output.

In the case of land cover, mapping is generally done by adopting a land cover classification system, delineating areas of relative homogeneity, then labeling these areas using classes defined by the classification system. This mapping takes many forms – no single mapping strategy will work best in all locations. The usefulness, efficiency, and cost-effectiveness of any land cover map is dependent on the scope and scale of questions being addressed, the spatial and temporal scale of imagery, the comprehensiveness of field and classification methods employed, and the level of accuracy desired. Key to selecting an adequate scale for evaluating the status and trends of land cover composition, configuration, and connectivity is identifying what is being managed (e.g., what species or processes; Beatley et al. 2000) and the scales to which those species and/or processes respond. Appropriate application of remotely-sensed imagery (e.g., aerial photos, multispectral, hyperspectral, laser) and methods (e.g., photo interpretation, maximum likelihood classification, object oriented image analysis) at relevant spatial and temporal scales follows (Turner et al. 2003, Gross et al. 2006). All of which depend on the timeframe, cost allowance, and accuracy level desired. Story et al. (2009) evaluated the suitability of land cover data from National Land Cover Data (NLCD), LANDFIRE, GAP/ReGAP, and NatureServe landcover (from LandScope) for NPScape and similar analyses. Kennedy et al. (2009) provided detailed guidance and a framework for designing remote-sensing-based monitoring and determining the suitability of remotely sensed data sources for monitoring natural resources.

2.2 Calculations

Many of the metrics calculated in NPScape reflect a proportion of area (e.g., 10% of the 30 km area of analysis surrounding the park), and calculating the proportion of area in any manner is always a function of the sampling unit being measured (e.g., county, buffer, grid cell size) and must be interpreted in that context. This spatial measurement of the percent of land in a particular thematic class is dependent on the underlying map and, for land cover maps, Stranko et al. (2008) have shown that using various imagery methods (e.g., 2001 NLCD versus high-resolution aerial photography) interchangeably to assess potential thresholds can produce inconsistent results.

There are also assumptions regarding the distribution of the data. For example, the response of species to ecological thresholds is thought to be nonlinear and to occur at diverse spatial and temporal scales (Groffman et al. 2006, Utz et al. 2009), though some species exhibit linear responses at some scales (Morley and Karr 2002). Given this, it is difficult to generalize potential species or community responses (Karr and Chu 2000, Bledsoe and Watson 2001, Morley and Karr 2002).

Lastly, some calculations (such as the Conservation Risk Index; see Section 6), require some portion of the area of analysis to be under protection to avoid a divide by zero error when calculating the index. Two options are available for addressing this issue: 1) assign an extremely low value for the percent protected (e.g., 0.0001%) for those areas lacking protection or 2) calculate the inverse of the index. Doing the latter requires reversing the interpretation and requires that some portion of the land unit consist of a converted cover type.

2.3 Watershed-based Evaluations

Watershed boundaries often delineate a relevant and useful area for landscape analyses, largely because of the downhill and downstream transport of materials and energy. For example, landscape processes in the upstream watershed affect the hydrology, water chemistry, and aquatic biota; aboveground and belowground flows can rapidly transmit the effects of actions outside parks to aquatic resources within park boundaries. When there is intensive develop in the upstream contributing area to a park, the effects of upstream activities may constitute the most important stressors or drivers of aquatic conditions within the unit.

Understanding of the effects of landscape attributes and processes on aquatic resources is the goal of a rapidly growing science at the intersection of landscape ecology, aquatic ecology, hydrology, and remote sensing (Allan 2004, Goetz et al. 2009). In Section 3, we document the substantial influences that land use, populations, and roads can have on hydrology and water quality. However, other factors such as watershed size and steepness, base geology and soils, and precipitation regime not only affect hydrology and water chemistry, to a large extent they also modify the effects of the anthropogenic factors. This is illustrated by stark differences in hydrology and water chemistry of "pristine" southeastern blackwater streams, alpine headwater streams, and ephemeral streams in deserts. We think it's useful to place watershed attributes in a general conceptual framework, but all quantitative relationships must be evaluated on a region-specific basis, and even within regions quantitative relationship often vary with spatial and temporal scale (reviewed by Strayer et al. 2003, Allan 2004, Poff et al. 2006, see Section 3).

The importance of regional differences is illustrated by the relationship of hydrological flashiness to the percent of watershed in the least disturbed or natural condition. These attributes exhibit strong negative correlations in the southeastern and northwestern US, but a strong positive correlation in the central US (Poff et al. 2006). Similarly but at a different scale, Carlisle et al. (2009) investigated predictors of the biological conditions of streams east of the 100th meridian, and found that riparian land cover and road-stream intersections were the best predictors in higher elevations, but soil properties (clay and permeability), mean elevation, and riparian high density residential development were the best predictors in the lowlands. Analyses and models at the regional or basin scale control for, or at least minimize, the variation in precipitation regime, topography, geology and soils. Analyses at these scales are thus most likely to find strong and more consistent relationships between anthropogenic factors and the state of aquatic resources.

The non-random distribution of development with regard to watershed attributes further complicates watershed analyses (Allan 2004). Agricultural land use is more likely to occur in flatter areas with better soils; urban areas develop from transportation crossroads, or, historically, at fall lines or locations suitable for small-scale hydropower. Therefore, even within regions, anthropogenic and natural features covary, confounding estimates of the effects of anthropogenic factors. Worse, some of the covariation is not with current landscape attributes, but with past land use (Harding et al. 1998). For instance, the sediment "slug" responsible for natural levees that channel the Roanoke River within its floodplain is the result of colonial-era land clearing upstream in Virginia, much of which has since reverted to forest (Noe and Hupp 2009).

To some extent, much of the complexity of how natural and anthropogenic landscape attributes determine hydrology and water chemistry is irrelevant for management of NPS resources. The attributes of the upstream landscape, and the hydrology, water chemistry, and aquatic biotic resources within the park are what they are. What matters for management is predicting how changes in the upstream landscape are likely to affect park aquatic resources: which changes should be avoided, which can and should be mitigated within the park, and what vital signs and aquatic attributes might require enhanced monitoring to quantify the effect of those external forces on park resources. Therefore, it might be sufficient to understand how temporal changes in the landscape attributes of the upstream watershed are likely to change the status of the aquatic resources.

Unfortunately, nearly all of the region- or basin- specific quantitative relationships between landscape and aquatic attributes are based on cross-sectional covariation across sub-watersheds within the region. The long-term data on both landscape and aquatic attributes necessary to directly address the responses of aquatic resources to temporally changing landscape attributes are rarely available.

3 Land Cover and Land Use: Area and Pattern

Habitat loss and fragmentation are often cited as processes that have profoundly impacted biodiversity and other resources found in parks (Turner 1989, GAO 1994, Trzcinski et al. 1999, Fahrig 2003). Measures of habitat availability and pattern provide information on the suitability of landscapes for species, and the ability of an area to sustain a population. Many parks are too small, by themselves, to support long-term survival of all the species that once lived there (Newmark 1986, 1987). In these cases, species living in protected areas need adequate expanses of habitat outside parks and linkages to other protected areas to support and maintain biodiversity into the future (Hansen et al. 2001).

NPScape Area and Composition Metrics. NPScape metrics used to estimate the area and composition of habitat include percent and area of cover types, area and proportion of natural or converted (developed) land, and proportion of forest or grassland cover using a moving window analysis. For this analysis, a square neighborhood (window) is defined by the number of pixels on each side. The proportion of pixels in the window of a particular cover type (e.g., forest, grassland) is calculated and assigned to the pixel in the center of the window (Riitters et al. 2002, Wickham et al. 2007, 2008).

NPScape pattern metrics. NPScape pattern metrics derived from land cover type maps include the proportions of the Area of Analysis (AOA) that are edge and core habitat, the number and density of patches, the distribution of patch sizes, and the size of the largest patch(es). NPScape pattern metrics are evaluated at multiple scales; analyses were conducted for cover maps that consider edge widths of 30, 60, and 150 m. Pattern metrics are described in detail by Vogt et al. (2007a, 2007b, 2009), Riitters et al. (2007, 2009), and in the Measure Description Summary (Gross and Svancara 2009).

The analysis of roads data includes a distance-from-road metric that is useful for identifying core areas defined by a threshold distance from a road (e.g., patches greater than 1 km from a road).

3.1 Habitat Area and Biodiversity

Habitat loss is one of the most pervasive effects of land use intensification, and the loss and fragmentation of habitats has been extensively studied, debated, and reviewed (Andrén 1994, Harrison and Bruna 1999, Debinski and Holt 2000, Fahrig 2003, Ewers and Didham 2006). In summary, habitat loss is non-random in both location and in the cover types converted (Seabloom et al. 2002), and the effects of habitat loss and associated fragmentation vary with the habitat preferences and requirements of different species, effects on predators and competitors, parasites, scale, and the time and magnitude of habitat changes. The effects of habitat area or pattern on species can vary with spatial and temporal scale, and the effects may be strongly expressed as some scales, and absent at others (Wiens et al. 1987, Thompson et al. 2002). In the extreme, long-lived plant species or communities may show no response to landscape changes for decades or centuries (Kuussaari et al. 2009).

Habitat loss results in reduced natural area, creation of edges, and increased isolation of resulting remnants (Ewers and Didham 2006, Kupfer 2006). Each of these effects can influence species in a variety of ways depending on dispersal behavior, mode and scale of movement, habitat requirements, arrangement of the habitat, and type of landscape modification (O'Neill et al. 1988, Pearson et al. 1996, With and Crist 1995, McIntyre and Hobbs 1999, Kupfer 2006, Fischer and Lindenmayer 2007). The response of species to an overall reduction in habitat and/or loss of special habitats (winter or summer range, specific requirements) can include a change in distribution, abundance, behavior, physiological state or vital rates.

But how much habitat loss is too much? How does the conversion of land cover from natural to urban or agriculture impact species and communities? Along the gradient from completely natural to completely developed, we want to be especially sensitive to ecological discontinuities or thresholds, where a small change in habitat area or quality results in a large change in biodiversity or ecosystem function (Turner and Gardner 1991, Muradian 2001, Huggett et al. 2005, Groffman et al. 2006, Suding and Hobbs 2009). As summarized in Table 3.1, field-based studies have reported widely divergent results, but they generally suggest that species will almost certainly be affected when the landscape is composed of less than 60 or 70% natural habitat, although less sensitive species may exhibit no detectable response in landscapes with far less natural habitat. Depending on the context, species, and conservation objective, modeling studies have estimated a range of habitat area requirements covers almost the entire range from no habtat to 100% habitat (summarized in Appendix 1).

Thresholds in the extent of natural land cover have been used to define four broad types of landscapes, each associated with particular levels of habitat loss and connectivity (McIntyre and Hobbs 1999, 2000, Hobbs 2005). These landscapes cover a gradient from intact (more than 90% habitat remaining) to variegated (60-90% remaining), fragmented (10-60% remaining), and relictual (less than 10% remaining). The threshold of about 60% habitat is supported by percolation theory where, assuming a random distribution of habitat patches, landscapes rapidly switch from consisting largely of interconnected areas of habitat to consisting of a number of small, isolated patches (Stauffer 1985, Gardner et al. 1987, With and Crist 1995). With (2005) postulated that this loss of connectivity can initiate other ecological thresholds in a "threshold cascade". While there is strong theoretical support for a connectivity threshold at about 60% habitat, field studies have suggested that in real landscapes, the habitat area threshold is often much lower. At least in some habitats, 30% natural land cover may represent a more realistic threshold and below 30% natural land cover, loss of connectivity is particularly severe and there is a distinct loss of species dependent on natural land cover (Andrén 1994, With and Crist 1995, Fahrig 2003, Radford et al. 2005). The lower empirical threshold for 'natural' cover likely results, in part, from a methodological requirement to classify areas as 'habitat' or 'not habitat'. The limitations of a binary classification have always been appreciated by landscape ecologists, and now the importance of the background matrix is receiving more attention as recent analyses cast questions on the utility of conservation goals predicated on overly simplistic habitat classifications (Kupfer et al. 2006, Prugh et al. 2008, Franklin and Lindenmayer 2009).

While a single threshold value cannot adequately describe responses of all species to changes in landscape pattern or extent, certain levels of land cover conversion may act like "red-flags" for some species (Hansen and Urban 1992, Andrén 1994, With and Crist 1995, Bascompte and Solé 1996, Parker and Mac Nally 2002, Lindenmayer et al. 2005, Svancara et al. 2005). Among terrestrial species, Lande (1987) suggests that species with a large dispersal range, high fecundity, and high survivorship, may be able to persist when suitable habitat covers only 25-50% of the landscape, while species with low demographic potential may be lost when as much as 80% of the landscape remains suitable habitat.

Table 3.1. Percent of natural land cover or suitable habitat necessary for maintenance of various wildlife species in North America. Additional studies from Australia, Europe, and Africa are not included (modified from Svancara et al. 2005). See appendix 1 for area estimations from modeling studies.

Taxa	Area	% Habitat	Conclusions	Reference
Birds				
Kirtland's warbler	Michigan	30	Population of male Kirtland's Warblers increased fourfold when the proportion of suitable habitat on the landscape increased above 30% and patch size, age and distance to occupied patches were important variables.	Donner et al. 2009
Forest species	Panama	40	Bird species richness declined significantly when remaining forest cover dropped below 40% of the historical forest cover.	Rompre et al. 2009
Wetland species	South Dakota	50	Wetland species were more likely to occupy wetlands if < 50% of the upland matrix was tilled.	Naugle et al. 2001
Breeding birds	Seattle, Washington	48	Bird species richness was high and many native forest species were retained when urban land cover comprised < 52% of landscape	Donnelly and Marzluff 2006
Fish				
Trout	Maryland	96	Brook trout were almost never found in watersheds with > 4% impervious surface.	Stranko et al. 2008
Insects				
Beetles	Colorado	20	Tenebrionid beetles in experimental micro-landscapes exhibited a strong threshold in movement parameters when the proportion of grass was < 20%	Wiens et al. 1997
Butterfly	Ohio	40	Though results were species-specific, over half of the butterfly and skipper species surveyed were never observed in plots with < 60% suitable habitat remaining. Rare species were disproportionately more affected by habitat fragmentation.	Summerville and Crist 2001
Mammals				
Grizzly bear	US Northern Rocky Mtns	60	Predicted 60% of region in suitable habitat was necessary to maintain an effective population of 500 grizzly bears.	Metzgar and Bader 1992
American marten	Utah, Maine, Wyoming, Newfoundland	70-75	Compared results from different spatial scales and study sites and demonstrated that American marten populations are reduced to near zero density when only 25-30% of forest is lost.	Bissonette et al. 1997
Eastern chipmunk	SE Ontario	30	Predicted that 70% habitat loss was a critical threshold for population size and persistence of eastern chipmunks, though species-specific habitat dependencies produced different vulnerabilities to habitat loss.	Henein et al. 1998
Florida panther	SE US	60-70	Predicted 60-70% of historical range was necessary to maintain an effective population of 500 Florida panthers, actual population of 1000-2000.	Noss 1991
Reviews				
Multiple taxa	Literature Review	20-60	Recommended 20-60% suitable habitat necessary to sustain long-term populations of area-sensitive species and rare species.	Environmental Law Institute 2003
Birds, Mammals	Literature Review	10-30	10-30% suitable habitat in the landscape might be a critical threshold for birds and mammals.	Andrén 1994

3.2 Habitat Pattern

In the United States, patterns in land cover composition, configuration, and connectivity reflect the dynamics of natural ecological processes (Watt 1947), biophysical constraints (Whittaker 1967, Stephenson 1990), and extensive modification resulting from a long history of human occupation and habitat modification (Riitters et al. 2002, Heilman et al. 2002, Mann 2006, Schulte et al. 2007). In turn, these land cover patterns help shape overall biological diversity patterns including the complex array of species occurring in an area, movements of individual organisms, and energy and material flows (Levin 1981, Noss 1990, Dunning et al. 1992, Franklin 1993, Taylor et al. 1993, Turner 2005, Hansen and DeFries 2007).

All landscapes are more or less heterogeneous due to underlying variation in topography, geology, and soils, and natural disturbances like fire, windthrow, and floods, and anthropogenic disturbances like forest clearing, construction, or agriculture. The major difference between natural and anthropogenic disturbances is that habitats lost to natural disturbances usually 'recover', while anthropogenic disturbances usually result in permanent or semi-permanent habitat loss by converting habitat (e.g., forest) to a non-habitat type (e.g., road, agricultural field, parking lot, etc.). Changes in habitat availability and pattern are typically measured via independent methods, but they are usually correlated and this makes it difficult to separate effects of area and shape on species (Trzcinski et al. 1999). Landscape composition as described by proportion of cover types is a very important attribute of a system, and there are strong theoretical relationships between cover and attributes like landscape percolation (Gardner and O'Neill 1991).

Ecologically relevant landscape pattern metrics include land cover composition, patch area, patch shape, isolation or connectance of patches, habitat density, core habitat area, edge habitat, and the contrast between patches (edge contrast). Multiple metrics are required to adequately describe these features, and no single scales will suit all needs. A considerable degree of local knowledge may be necessary to evaluate how changes in landscape pattern will influence local biota and ecosystem processes.

3.2.1 Ecological Effects of Patchiness: Edges

An important consequence of increased habitat patchiness is the greater proportion of edges in the landscape. At a small scale, edges influence virtually every ecological property and there have thus been thousands of studies on the ecological roles of habitat edges. Forman (1995) and Reis et al. (2004) provide very good general syntheses and reviews. Chalfoun et al. (2002) focuses on edges and nesting success of birds. Ewers and Didham (2005) and Fletcher et al. (2007) investigate additional factors that influence species at the boundaries of ecological types and the interaction of edge and area effects.

Here, we follow the conceptual model developed by Reis and Sisk (2004). This model provides a general framework for evaluating the likely consequences of changes in patchiness and edge habitat at a specific site. Direct effects of edges are usually measured at small scales and across distinct boundaries (e.g., forest-grassland edges) because changes in the proportion of habitat edge accompanies changes in, for example, core habitat, proportional land cover, and habitat pattern (at a variety of scales).

Ries and Sisk (2004) identified four primary mechanisms to account for the vast majority of ecological effects of edges:

- ecological flows: changing the rate of transport of energy and materials across the boundary,
- access to spatially separated resources: a result of the juxtaposition of different cover types,
- resource mapping: where a species distribution is directly mapped to its resources ,
- species interactions: change in predator-prey, parasitism, etc. that results directly or indirectly from the contrast in adjacent cover types.

Ecological flows across edges

There are often sharp gradients in moisture, temperature, wind, and light across forest ecotones, and these can dramatically influence the compositing and structure of vegetation (Didham and Lawton 1999). Forest edges adjacent to open habitats are drier, lighter, and hotter than forest interior, and thus often suitable for a different suite of species (Chen et al. 1999). Open habitats near forests are likely to be cooler and more shaded than those farther from edges. Edges can facilitate or inhibit movement of species or their propagules (pollen, air-transported eggs or organisms).

Access to resources

Until the 1970s, habitat edges were widely considered to be beneficial because the higher density of animals and richness of species in these ecotones. Deer and other game species – the focus of many early studies – prefer mixed habitats, and thus management recommendations often included the creation or enhancement of edges. These species benefit from the close proximity of habitats that provide different resources – in this case cover and forage.

Resource mapping

Species distributions at edges are influenced by resource mapping via a multitude of pathways. Many plants and animals track abiotic gradients in moisture, light, or nutrients. Ries et al. (2004) noted that the most common studies of resource mapping focused on vegetation structure and habitat selection by birds. Another common form of resource mapping is the overlap in the distribution of animals. Spotted owls that prey primarily on wood rats are more abundant near forest edges, whereas spotted owls that feed mainly on flying squirrels exhibit no edge effect (Zabel et al. 1995, Ward et al. 1998).

Species interactions

Species interactions at edges include predation, parasitism, herbivory, and competition. Parasitism by birds is a well-know example. Hartley and Hunter (1998) found that rates of parasitism of nests in US forested habitats were higher near edges. Cowbirds are particularly well studied. The preferred foraging habitat for cowbirds is open pasture, but they exhibit a very strong preference for forested breeding sites within about 200 m of the forest edge (Howell et al. 2007). Nests of other bird species in edge habitats can experience very high rates of parasitism (Thompson et al. 2000, Howell et al. 2007).

Ries et al. (2004) discuss a variety of other situations where species interactions result in edge effects. These include bird predation on insects, mammals avoiding predation at near edges, higher predation of seeds and herbs (by mice) near edges. But the direction and magnitude of effects are inconsistent, and other studies revealed lower rates of seed predation, or herbivores that avoided edges due to higher predation rates.

A predictive model for edge effects

The mechanisms proposed by Ries et al. (2004, discussed above) are important to understand how organisms can respond to edges. Ries and Sisk (2004) incorporated this understanding into a predictive conceptual model of edge effects, based on resource distributions and the projected response of organisms to changes in resources across ecotones. In theory, the model can be applied to all species and all edges. Model predictions are founded on the relative distribution of resources in habitats on either side of the edge: when concentrated in one habitat there will be decreased abundance in the preferred habitat and an increase in the non-preferred habitat. When resources are divided between habitats, the model predicts an increase near both edges; when resources are spread equally among habitats, there will be a neutral edge response; and when resources are concentrated along the edge then an increase near the edge in both habitats. An initial test of the model using data for 52 bird species provided strong support (Ries and Sisk 2004). One advantage of the model is that it can account for apparently conflicting studies that have shown a positive, negative, or neutral response of a species to edges. The resource-based model readily accommodates differing responses to edges, depending on site-specific context.

Studies of edge effects emphasize the need for site- or study-specific objectives. For example, habitat managed to increase edge habitat to support high densities of deer may also unintentionally lead to excessively high rates of parasitism on other birds by cowbirds.

Table 3.2. Effects of landscape pattern on ecological functions and biodiversity.

Effect	Description	Reference
Movement between patches	Most strongly correlated with amount of habitat in a buffer around patch (simulation study)	Bender et al. 2003; Tischendorf et al. 2003
Percolation across a landscape	For random pattern, occurs when 60-70% of landscape is composed of habitat	Gardner et al. 1989
Number of forest interior bird species	Increases with patch size.	Forman et al. 1976; reviewed by Fahrig 2003
Sediment and nutrient absorption / buffering	Vegetation fragmentation and natural vegetation along waterways strongly affect stream biological condition.	Shandas and Alberti 2009
Forest cover in riparian buffers is associated with fish IBI	Examined effects at reach to watershed scale on fish IBI. Riparian forest and length slope (LS) were most important watershed-scale variables and mostly positively correlated with IBI scores in the eastern corn belt. Frimpong et al. (2005b) suggested a 30 m lateral buffer and 2000 m linear buffer were most strongly correlated with stream IBI, but noted 30 m scale matched input data.	Frimpong et al. 2005b

3.3 Land Cover effects on Water and Watershed Condition

The effects of landscape attributes in watersheds vary with both spatial and temporal scales. Some landscape factors are more important in small watersheds, others in large watersheds, but almost all factors and processes vary with respect to watershed size. Temporally, landscape factors can have transient effects on aquatic resources, such as sedimentation or temporarily altered flow during road or housing construction that will cease once roads or building are completed (Wheeler et al. 2005). Landscape attributes can also have persistent effects on aquatic resources: paved roads and developed areas will continue to affect hydrology and add nutrients and contaminants such as salts and metals to streams. Finally, land use can have historic impacts that last well beyond the duration of that use: mine runoff and sedimentation from colonial-era clearing can persist even though the land cover has long since reverted to forest (Harding et al. 1998, Noe and Hupp 2009).

Allan (2004) provides an excellent synthesis and review of the effects of land cover and land use changes on watersheds. He grouped the mechanisms of land use effects on stream ecosystems into six environmental factors. These factors are described in Table 3.3 and mapped to the most relevant NPSscape metrics in Figure 3.1.

Land cover in the watersheds upstream of parks can directly affect all six of Allan's (2004) environmental factors. Land cover types that provide more water storage capacity dissipate the pulse flow that can follow a storm, reducing the peak flow and extending the flow duration. Conversely, land covers such as impervious surfaces reduce the upland storage of precipitation and increase the rate of runoff into streams and rivers, thereby increasing the magnitude of peak flow and reducing the base flow between storms (Allan et al. 1997). Land covers such as forest have substantial capacity to hold nutrients as well as water. Other land covers may be susceptible to erosion and thus contribute to high sedimentation rates.

It can be difficult to evaluate independently the effect of an increase or decrease in the area of one specific land cover type on watershed conditions, because an increase in one type necessarily corresponds to decreases in other types of land cover. In some cases, it can be difficult to distinguish whether an observed response is due to the increase in one type or to the decrease in another type (King et al. 2005). For example, Poff et al. (2006) compared hydrological

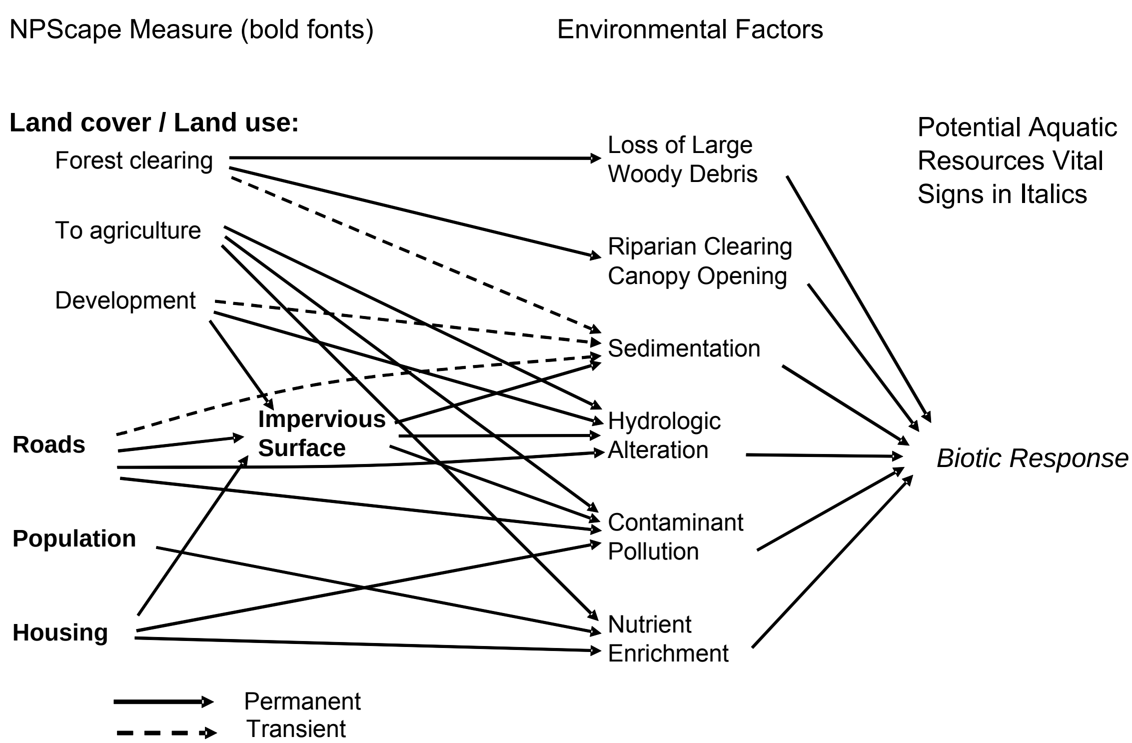


Figure 3.1. Mapping of NPScape landscape metrics affecting each of Allan's (2004) environmental factors, which affect aquatic ecosystems.

responses to land cover (urban, agriculture, or least disturbed) across catchments within different ecoregions, and found different responses in different ecoregions. They postulated the counterintuitive positive correlation of agricultural land cover with reduced flow flashiness in the Central US resulted from the negative relationship between agricultural and urban land covers in a region with little "least disturbed" area: watersheds with lower proportions of agricultural land had a greater proportional cover of urban land use

3.3.1 Impervious Surface

Impervious surface is a land cover classification that includes bare rock, paved roads, and most developed areas (note difference from NLCD/MRLC classification, which maps only _developed_ impervious cover types). Impervious surfaces prevent the infiltration of precipitation into thep ground. The consequences for hydrology are quicker runoff into streams, and thus more rapid rising and dropping of streamflow following storm events (flashiness of storm response), and reduced evapotranspiration and percolation to aquifers, and thus increased cumulative flow out of the catchment. The consequences for both nutrient enrichment and contaminants are increases, as chemicals picked up by the water are transported directly into the stream, without the opportunity for uptake or decomposition by soil organisms.

The effects of impervious surface on hydrology appear to be more important in smaller catchments. Where a storm can cover most or all of the catchment at the same time, and the time delay for stream flow from the top of the catchment is small relative to the duration of the storm event, increasing impervious surface can result in a large increase in peak flow. As watershed size and thus stream length increases, the effects of even a large storm will be spread out temporally by the difference in arrival times of flows from upstream versus downstream in the watershed. Increasing impervious surface still increases peak flow for storm events that last longer than the flow time in the watershed, but the flashiness is at least partially attenuated. In very large watersheds, larger than the largest storm system, impervious surface presumably should have even less effect, but the variation in percent impervious surface between such large watersheds is much smaller than the variation in precipitation patterns, so any such pattern cannont be quantified or confirmed.

The effects of impervious surface on nutrient enrichment and contaminants are a function of the availability of nutrients and contaminants. Therefore, the effects are greatest in smaller, highly-developed watersheds where impervious surfaces receive higher concentrations of both nutrients and contaminants. In smaller urban and suburban watersheds, storm water retention ponds and other engineered solutions are often required to mitigate both the hydrologic and nutrient / contaminant effects of impervious surface.

Multiple studies have quantified thresholds for the effects of the proportion of impervious surface. Paul and Meyer (2001) review studies of the effect of impervious surface from urbanization and report thresholds of 2-10% for effects on stream geomorphology, 10-15% for effects on fish diversity, and 1-33% for invertebrates. For example, when total impervious cover exceeds 10-15% stream biota are not maintained (Klein 1979, Schueler 1994, Wang et al. 2001). However, impacts to more sensitive species can occur at 3-5% impervious cover (Booth and Jackson 1997, Angemeier et al. 2004, Stranko et al. 2008), and thresholds vary geographically (Utz et al. 2009) and with a variety of physical and biotic factors (Allan 2004).

Geographical differences in biotic responses to habitat loss can also be important, and populations at the edge of a species' range may be more sensitive to disturbance (Stranko et al. 2008). For instance, aquatic invertebrates negatively affected by urbanization responded at lower threshold levels (10-45% converted) in the Piedmont compared to the Coastal Plain physiographic province (15-60% converted, Utz et al. 2009). Finally, Booth et al. (2002) question whether the varying "thresholds of effect" reflect differences in the systems studied or are functions of the imprecision of measurement, and argue that biological effects are more continuous rather than threshold effects, with small responses at lower levels of development than the inferred thresholds may suggest.

Table 3.3. Environmental factors altered by land use changes, and the mechanisms by which they alter stream ecosystems (from Allan 2004).

Factor	Effects	References
Sedimentation	Increases turbidity, scouring and abrasion; impairs substrate suitability for periphyton and biofilm production; decreases primary production and food quality causing bottom-up effects through food webs; in-filling of interstitial habitat harms crevice-occupying invertebrates and gravel-spawning fishes; coats gills and respiratory surfaces; reduces stream depth heterogeneity, leading to decrease in pool species	Burkhead & Jelks 2001, Hancock 2002, Henley et al. 2000, Quinn 2000, Sutherland et al. 2002, Walser & Bart 1999, Wood & Armitage 1997
Nutrient enrichment	Nutrient Increases autotrophic biomass and production, resulting in changes to assemblage composition, including proliferation of filamentous algae, particularly if light also increases; accelerates litter breakdown rates and may cause decrease in dissolved oxygen and shift from sensitive species to more tolerant, often non-native species	Carpenter et al. 1998, Delong & Brusven 1998, Lenat & Crawford 1994, Mainstone & Parr 2002, Niyogi et al. 2003
Contaminant pollution	Increases heavy metals, synthetics, and toxic organics in suspension associated with sediments and in tissues; increases deformities; increases mortality rates and impacts to abundance, drift, and emergence in invertebrates; depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance	Clements et al. 2000, Cooper 1993, Kolpin et al. 2002, Liess & Schulz 1999, Rolland 2000, Schulz & Liess 1999, Woodward et al. 1997
Hydrologic alteration	Alters runoff-evapotranspiration balance, causing increases in flood magnitude and frequency, and often lowers base flow; contributes to altered channel dynamics, including increased erosion from channel and surroundings and less-frequent overbank flooding; runoff more efficiently transports nutrients, sediments, and contaminants, thus further degrading in-stream habitat. Strong effects from impervious surfaces and stormwater conveyance in urban catchments and from drainage systems and soil compaction in agricultural catchments	Allan et al. 1997, Paul & Meyer 2001, Poff & Allan 1995, Walsh et al. 2001, Wang et al. 2001
Riparian clearing/canopy opening	Reduces shading, causing increases in stream temperatures, light penetration, and plant growth; decreases bank stability, inputs of litter and wood, and retention of nutrients and contaminants; reduces sediment trapping and increases bank and channel erosion; alters quantity and character of dissolved organic carbon reaching streams; lowers retention of benthic organic matter owing to loss of direct input and retention structures; alters trophic structure	Bourque & Pomeroy 2001, Findlay et al. 2001, Gregory et al. 1991, Gurnell et al. 1995, Lowrance et al. 1984, Martin et al. 1999, Osborne & Kovacic 1993, Stauffer et al. 2000
Loss of large woody debris	Reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability; influences invertebrate and fish diversity and community function	Ehrman & Lamberti 1992, Gurnell et al. 1995, Johnson et al. 2003, Maridet et al. 1995, Stauffer et al. 2000

3.4 Measuring and Monitoring Land Cover and Habitat Attributes

3.4.1 Habitat Loss

Critical habitat area thresholds, such as those supported by percolation theory, have been suggested as a landscape monitoring approach at the state level (O'Neill et al. 1997) and provide general guidance on broad management decisions related to total habitat area. For example, when the proportion of natural land cover is ≥60%, protective measures may be considered sufficient, while values between 40 and 60% might indicate a need for restoration (Wade et al. 2003). At the national level, such assessments may supply information useful from both a management and policy perspective (Kupfer 2006). However, regional and national assessments can provide context for more local evaluations, but they cannot replace local level monitoring (Svancara et al. 2009a).

Proportional change in natural land cover is possibly the simplest indication of biotic condition (O'Neill et al. 1997). Calculating the proportion of natural land cover remaining in an area provides a general indication of overall landscape condition surrounding protected areas and offers insight into potential threats (i.e., how much land has been converted and how much natural habitat remains?) it may offer some insight to opportunities for conservation. Calculating the proportion of converted (agriculture and urban) land, also known as the U-index (O'Neill et al. 1988), can be used to measure general land use pressure by humans. The definition of "natural" will vary depending on the scope of question asked and typically requires aggregating the original land cover/land use data to broad 'converted' versus 'natural' categories. Table 3.4 is the aggregation used by NPScape for NLCD land cover categories.

Table 3.4. Example aggregation of 2001 NLCD for calculating percent of natural and converted land cover.

General Category	NLCD Classes (class number)
Converted	Low intensity developed (22), Medium intensity developed (23), High intensity developed (24), Open space developed (21), Pasture/hay (81), Cultivated crops (82)
Natural	Grassland/herbaceous (71), Shrub/scrub (52), Mixed forest (43), Evergreen forest (42), Deciduous forest (41), Barren land (31), Perennial ice/snow (12), Woody wetlands (90), Emergent herbaceous wetlands (95), Open water (11)

The local (30 km) area of analysis for most parks still includes considerable natural habitat (Figure 3.2), although there is a large variation between both parks and regions. Regions with parks that generally are surrounded by a lower proportion of natural cover either have high densities of people (e.g. North Atlantic), or large proportions of agricultural development (e.g. grassland regions).

3.4.2 Habitat Pattern

Pattern metrics can describe individual patches (size, shape), properties of a cover classes (connectivity, isolation), or landscape-level metrics that are functions of the composition and arrangement of patches throughout the entire area of analysis (richness, evenness, diversity). While metrics of landscape composition (abundance, variety of patches) are relatively straight-forward (see section 3.1.3), they do not explicitly consider the location, size, shape, or spatial configuration of patches. Conversely, spatial configuration is difficult to quantify in simple metrics, and most metrics of spatial configuration are of little value by themselves. Some metrics, like patch size distribution,

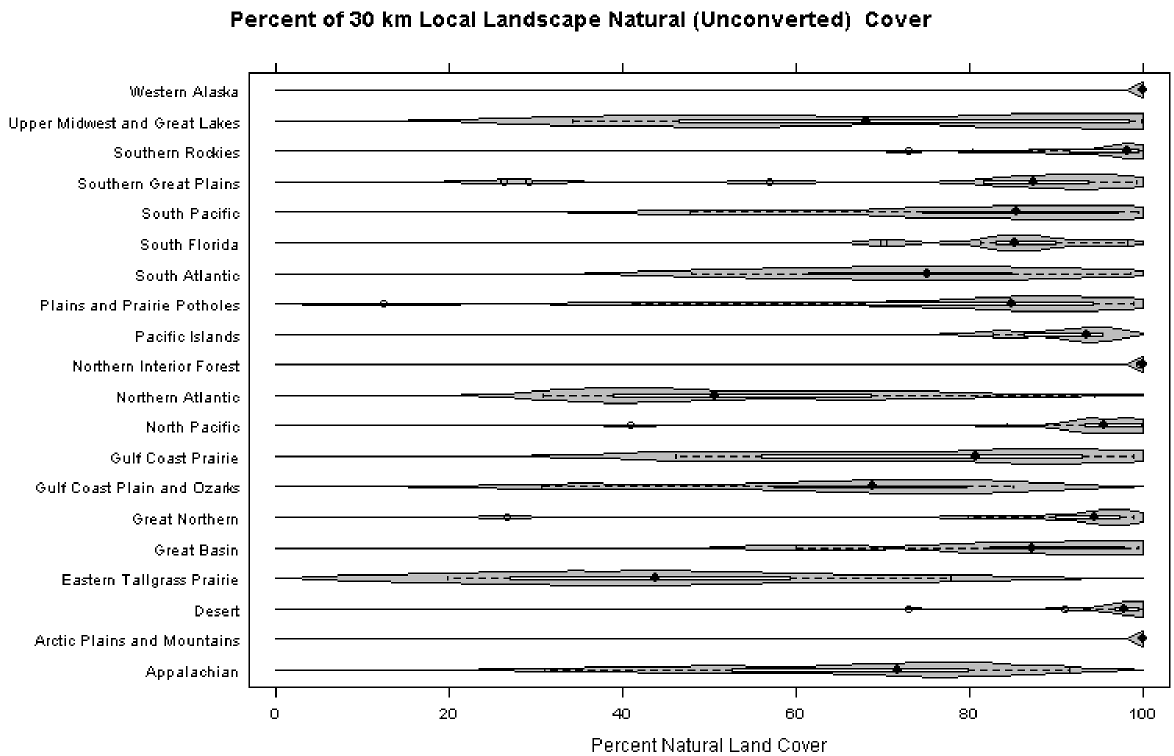


Figure 3.2. Percent of natural land (see above) in an NPS unit and 30 km area of analysis surrounding the park, grouped by DOI Geographical Framework areas (see figure 1.4). The thickness of the 'violin' is proportional to the frequency of observations. Dots indicate the median and outliers. See appendix 2 for additional results.

reflect the aggregate of individual patch characteristics across an area of analysis, while others metrics, like isolation or connectivity, describe the specific spatial relationship between patches.

Description of pattern metrics can further be divided between those that describe structural versus functional characteristics of the landscape. Structural metrics, such as patch size distribution and nearest-neighbor distances between patches, measure physical attributes without regard to any specific species or ecological function. The determination of structural pattern metrics is (in theory) invariant across applications. By contrast, functional pattern metrics measure spatial landscape attributes that are relevant to properties of specific species or processes. Functional metrics thus require parameters that apply to focal species or processes, and different applications may yield different results. Connectivity metrics are typically functional, and they include parameters that account for dispersal abilities of a species or group of species. For example, graph analyses (Townsend et al. 2009) can evaluate landscape connectivity at a scale relevant at the relatively short movements characteristic of amphibians (Lookingbill et al. 2008), or at broader scales that reflect the movement abilities of species such as spotted owls (Keitt et al. 1997). A key factor in all these analyses is the selection of what constitutes the 'patch' and what constitutes the 'matrix'.

Many landscape pattern metrics are easily calculated using widely available software such as r.le (Baker and Cai 1992), FRAGSTATS (McGarigal and Marks 1995), and Patch Analyst (Rempel 2009) with either vector or raster-based land cover maps. The basic attributes of most of these pattern metrics have been extensively studied and reviewed, and it is apparent that many metrics are highly correlated (Cain et al. 1997, Riitters et al. 1995, Calabrese and Fagan, Cushman et al.

2008). The FRAGSTATS web site provides a particularly lucid overview of the type of pattern metrics. Most of the information on pattern can be conveyed by a small number of metrics, and the key challenge is to figure out which metrics best apply to the problem.

Important considerations when evaluating pattern are selection of metrics, the methods used to create the land cover map, and the techniques for distinguishing a patch from the background matrix. In particular, we want to identify opportunities or threats where a small difference in management (or management of a small area) may have a large impact on park resources. An example would be the management or preservation of a small corridor that connects much larger patches of habitat (Beier 1993, Hilty et al. 2006).

NPScape provides land cover pattern metrics based on Morphological Spatial Pattern Analysis (MSPA), a pixel-level analysis of cover maps using image segmentation to classify individual pixels into a set of pattern types, including core, islet, perforation, edge, loop, bridge (=corridor), branch, and background (Vogt et al. 2007a, 2007b, 2009). The behavior of these metrics across scales and in landscapes with various statistical properties has been examined (Riitters et al. 2007, 2009, Ostapowicz et al. 2008), as has their ability to characterize landscape connectivity and identify corridors (Vogt et al. 2007b, 2009). Vogt et al. (2007a) describe the process of MSPA, provide an example application, and highlight the advantages of this method over approaches that require identification and delineation of specific patches. Software used to calculate the landscape metrics is part of the GUIDOS package, available from the European Commission Joint Research Centre (http://forest.jrc.ec.europa.eu/download/software/guidos/).

The pattern metrics provided in NPScape (Table 3.5) are can be objectively repeated over time, and they are sensitive to the more common types of landscape change. For NPScape, two key advantages of MSPA are (1) an ability to apply the algorithms over very large spatial extents, and (2) the greater sensitivity of pixel-based maps and analyses to detect changes in patterns over time. The number of pixels in each morphological pattern class are easily summed and the composition of an area of interest described in terms of the proportion or area of core and edge habitats (areas of other morphological types can also calculated, but interpretation of the significance or ecological function of some types is difficult). NPScape used date from K. Ritters (personal communication) and the December 2009 NPScape data distribution includes spatial data with edges defined as the area within 30 and 150 m of a core area of forest or grassland (i.e., there are four data layers: forest with 30 m edge, forest with 150 m edge, and the same edge definitions for grassland).

Table 3.5. Landscape pattern metrics and their importance. These metrics can be sensitive to both areas of analysis, and to the scale at which the metric is defined.

Metric	Description	References
Composition	Proportion of area. Usually an aggregate class (forest/non-forest, natural/converted). Can be elegantly expressed at multiple scales.	Heinz Center 2008a (pattern report); Riitters et al. 2009
Patch size distribution	Simple and understandable metric. Does not describe spatial relationships between patches.	Turner 1989; Gardner et al. 2008 (and many others)
Landscape morphology	A pixel-based, structural approach to identify specific types of landscape elements. The key types are described below. Analyses used by NPScape are based on maps that aggregate classes in to forest/non-forest, or natural grassland/non-natuarl-grassland. See text for a more detailed and complete description.	Vogt et al. 2007a, 2007b, 2009; Riitters et al. 2007, 2009.
- Core	Pixels that are within a patch, and more than a defined 'edge' distance from a non-patch type.	
- Edge	Pixels within a defined distance from the boundary of a patch (e.g., forest) and non-patch (e.g, non-forest). We report analyses with edges of 30 m (1 pixel) and 120 m (4 pixels).	
- Connector	A cluster of non-core pixels connected at two or more locations to edge.	

4 Roads: Habitat alteration and other effects

Roads provide remarkable access to lands within the continental United States, and the system of roads in the US is the envy of nations all over the world. On the flip side, roads and associated activities can profoundly effect a broad range of physical, ecological, and social attributes important to parks. Comprehensive reviews have documented extensive direct and indirect impacts of roads on both terrestrial and aquatic environments (Spellerberg 1998, Ercelawan 1999, Trombulak and Frissell 2000, Forman et al. 2002), the effects of which may be undetectable in some taxa for decades (Findlay and Bourdages 2000). Even in areas where human population densities are relatively low and landscapes are perceived as natural, the impacts of roads are pervasive (Saunders et al. 2002) and may extend hundreds to thousands of meters from the roadside (Forman 2000, Forman and Deblinger 2000, Forman et al. 2002).

NPScape road metrics. NPScape road metrics are derived from readily available spatial road maps and include road density, distance to roads, and effective mesh size (described below). Road density (km / km^2) and distance-to-a-road are perhaps the most common and convenient overall measures of the amount of road in an area and how far it is to the nearest road of various types (Forman et al. 2003). The effective mesh size is the area or diameter of patches enclosed within a road network and provides a measure of the spatial arrangement of roads. This metric may be a better measure of potential impacts of roads by determining the relative size, shape, and arrangement of enclosed patches (Forman et al. 2003) and has been shown to be a useful measure for monitoring landscape fragmentation (Jaeger 2000, Jaeger et al. 2008). Together, these metrics provided by NPScape can be used to answer useful question such as, "How many acres of accessible forested area within 1000 m of a pond could be accessed by amphibians without crossing a major road? " (see Eigenbrod et al. 2008).

By physically altering the landscape, roads result in the direct loss of habitat, fragmentation of the remaining habitat (Carr et al. 2002), altered landscape structure (Saunders et al. 2002), increased influence of edge effects (Carr et al. 2002), and disruption of hydrological processes (Jones et al. 2000, Trombulak and Frissell 2000). In fact, Noss (1993) asserted that roads may be the single most destructive element in the process of habitat fragmentation. Table 4.1 describes some of the more pervasive effects of roads. There is a huge literature on effects of roads, and in Table 4.2 we list references we found to be particularly useful.

Table 4.1. Pervasive effects of roads relevant to natural resources, park visitors, and park operations.

Physical Effects	Biological Effects
Alter temperature, humidity, and other climate attributes	Collisions between animals and cars
Increase rate and amount of water runoff	Physical barrier to movement
Alter surface and ground water flows	Habitat loss
Alter rates of sediment and nutrient dispersal	Habitat fragmentation
Runoff of chemicals applied to road surface	Behavioral avoidance of disturbances
Alter geological and soil substrates	Corridor for invasive species
Increase production and propagation of noise	Indirect effects like poaching, fire ignition, trash
Alter light	Noise interference with species communication
Physical barrier to many species	Habitat alteration

Table 4.2. Especially useful sources of information on roads and their effects on resources important to parks.

Topic	Description	Citation
Road Ecology	This book is the 'go-to' source and is a comprehensive guide to ecological and other aspects of roads. More than 1000 citations. A bit dated.	Forman et al. 2003
Anurans	Web site with extensive list of references on effects of roads on reptiles and amphibians. See: http://community.middlebury.edu/~herpatlas/roads_biblio.php	Carter and Andrews 2007
Many species and communities	Broad review of ecological effects of roads on terrestrial and aquatic communities	Trombulak and Frissell 2000
Wildlife	Short, concise review of multiple effects of roads on wildlife	Jackson 2000
Wildlife (books)	Detailed treatment and review of effects on wildlife	Sherwood et al. 2002; Spellerberg 2002
Ecological effects	Very broad scope of topics with examples (a few to many citations) of studies for about 25 types of effects	Spellerberg 1998
Ecological effects	A review of the impacts of road, powerline, gasline and canal fragmentation on tropical forest ecosystems. Also provides a solid review of the road ecology literature in all ecosystems.	Laurance et al. 2009
Small mammals	Generally found no effect on abundance, density, or diversity in desert community near I-25. Good concise literature review for small mammals.	Bissonette and Rosa 2009
Breeding birds	A comprehensive review of noise impacts to bird behavior and populations in relation to overall traffic load and thus noise intensity.	Reijnen and Foppen 2006
Traffic noise and birds	Noted response of birds with higher frequency song; Very recent literature review of traffic noise effects.	Parris and Schneider 2009
Noise and terrestrial organisms	Comprehensive review of effects of chronic noise exposure on a variety of terrestrial species.	Barber et al. *in press*
Research agenda	Rauischholzhausen agenda for road ecology. Outstanding consideration of key research questions related to 'road ecology', and evaluation of research designs.	Roedenbeck et al. 2007

4.1 How is terrestrial biodiversity affected by roads?

Numerous studies have documented effects of roads on a variety of vertebrate taxa (Table 4.3). Wildlife populations are impacted directly by increasing mortality of organisms from vehicles (e.g., Fahrig et al. 1995), modifying behavioral patterns such as home ranges and migrations (e.g., Reijnen et al. 1996, Forman et al. 2002), interfering with species communications (Barber et al. in press), and imposing physical barriers which limit or prevent access to resources (Trombulak and Frissell 2000, Forman et al. 2003, Jaeger et al. 2005), limit gene flow and subdivide and isolate populations (e.g., Gerlach and Musolf 2000, Keller and Largiader 2003). Loss of connection between complementary habitats (e.g., breeding and feeding) can be particularly detrimental (e.g., Pope et al. 2000).

Table 4.3. A summary of results from studies of road-related mortality.

Type of effect	Description
Direct mortality	Reduced abundance of anurans as a function of distance to road varied by species; thresholds from 250 to > 1000 m from road. Heavy truck traffic (Eigenbrod et al. 2009).
Noise	Interferes with auditory communication and ability of orient to calls among birds (Reijnen et al. 1996, Rheindt 2003) and amphibians (Sun and Naris 2005, Bee and Swanson 2007)
Release from predators	Moose cows with calves preferentially forage near roads, apparently to avoid wolves and bears (Berger 2006). Some mammalian predators – foxes, wolves, and bears – avoid roads (see below).
Avoidance	Bears in NC (Brody and Pelton 1989) and grizzly bears elsewhere shift home ranges away from roads (McLellan and Shackleton 1987). Elk prefer feeding away from roads (Grover and Thompson 1986). Elk and mule deer in Colorado prefer winter feeding area more than 200 m from roads (Grover and Thompson 1986, Rost and Bailey 1979)
Prefer roads	Caribou in AK use roads during migration (Banfield 1974); Caribou are reputed to rest on roads in summer due to lower insect harassment. Turkey and black vultures establish home ranges in areas with higher road density (Coleman and Fraser 1989).

Fahrig and Rytwinski (2009) conducted a comprehensive review of literature on the effects of roads on animal abundance and evaluated the direction of effects on approximately 150 species or guilds of invertebrates, anurans, birds, and mammals (see Table 1 in Fahrig and Rytwinski 2009). From this synthesis, Fahrig and Rytwinski developed the conceptual model in Figure 4.1.

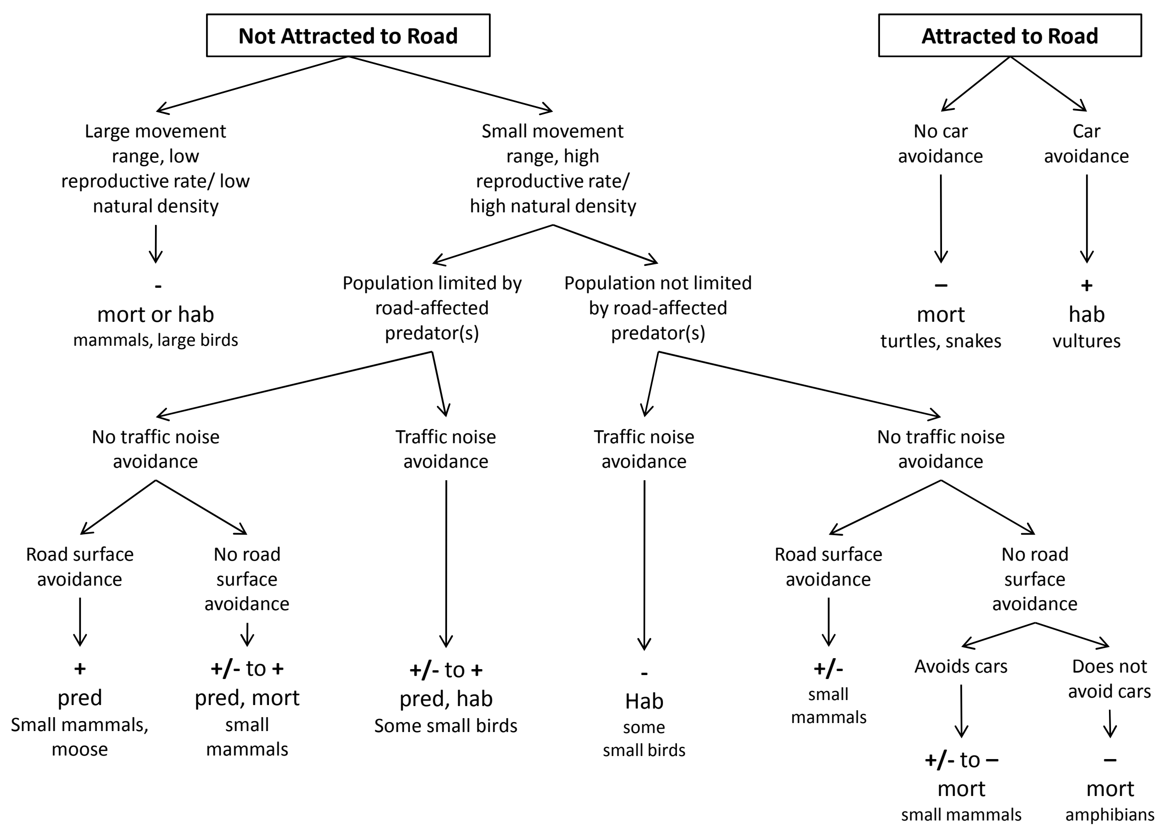


Figure 4.1. Conceptual model of species characteristics, attributes of roads and traffic, and the consequences on mortality and abundance. (-), (+), and (+/-) are negative, positive, and neutral, respectively. Effects are mortality (mort), habitat loss/increase (hab), and release from predation (pred). Figure and modified caption from Fahrig and Rytwinski (2009).

One of the most important direct effects is increased mortality from vehicle collisions. Road-related deaths can have significant effects on wildlife populations, particularly for smaller organisms. For example, in studying four amphibian species, Hels and Buchwald (2001) found that the probability of getting killed ranged from 0.34 to 0.61 when crossing a low to moderately traveled road (3200 vehicles / day) and from 0.89 to 0.98 when crossing a busy road (>15,000 vehicles / day). In their study area, about 10% of the adult populations of these species were killed annually by traffic.

Road noise can also have important effects on wildlife, generally affecting physiological or behavioral responses. Even on sparsely traveled roads, the noise from a transiting vehicle is a potential disturbance. For example, a radio telemetry study of elk in relation to all-terrain vehicles documented responses at distances of in excess of 1 km (Preisler et al. 2006); responses were more likely when the animal was closer to a forest road. On roads with heavy traffic, chronic noise exposure can degrade auditory awareness by masking sounds that would otherwise be heard. Masking can inhibit perception and recognition of intentional and adventitious sounds, and can affect a variety of social and ecological processes (Barber et al. *in press*). If noise is a primary factor, then increased traffic will be problematic, even if road networks do not proliferate.

Many times, even in 'pristine' protected areas, wildlife populations are affected by several, if not all, of these impacts resulting in the highest risk of mortality in areas of high road density (e.g., grizzly bears - Boyce et al. 2001, Johnson et al. 2005). For example, research on grizzly bears in Yellowstone National Park indicates that the effects of roads and associated development are disproportionately high on adult females and subadults resulting in higher mortality and lower productivity (Mattson et al. 1987). Similarly, in Banff and Yoho National Parks (Alberta, Canada), where grizzly bears are protected from hunting, as much as 100% of known adult grizzly bear mortalities occurred within 500 m of roads or 200 m of high use trails (Benn and Herrero 2002).

Indirect effects of roads can be as important, or more so, than direct effects. Roads have been associated with an increased incidence of fire ignitions (Cardille et al. 2001, DellaSala and Frost 2001, Brown et al. 2004) and higher probability of large fires in the upper Midwest (Cardille et al. 2001, but see Dickson et al. 2006). Roads are also associated with an increased occurrence of invasive species (Tromulak and Frissell 2000) with improved roads having a greater proportion of exotic species occurrence and cover than unimproved (4WD) roads (Gelbard and Belnap 2003). Wildlife may experience increased legal hunting pressure and poaching (Trombulak and Frissell 2000) and chronic disturbance from human activity such as recreation (Thurber et al. 1994, Reijnen et al. 1996). More recent studies have identified other indirect species-specific effects such as impacts of road salt runoff on amphibians (Karraker et al. 2008).

Like the other measures described in this report, the effects of roads are frequently confounded with other attributes, such as the loss or fragmentation of habitats. Eigenbrod et al. (2008) proposed the use of 'accessible habitat' as a means to uncouple the effects of total habitat area, road density, and the area of habitat most available to organisms. By explicitly accounting for the location of roads with respect to habitat areas around ponds, Eigenbrod et al. (2008) found 'accessible habitat' better predicted anuran species richness than total habitat area or distance to a road. For their study, Eigenbrod et al. (2008) defined accessible habitat as the forested area within 1000 m of a pond that could be accessed without crossing a major road. The NPScape road mesh variable can be used to estimate this sort of metric.

4.2 How far do the effects of roads penetrate?

Recent reviews suggest that most road impacts occur within 1 km (0.6 mi) of roads (Forman et al. 2003), but effects on species behaviors can extend well beyond this distance. The size of a 'road zone' will depend on the species of interest, ecosystem characteristics, season, time of day, road width, road surface, proximity to water, and traffic density. For example, in Canada national parks where grizzly bears are protected from hunting, as much as 100% of known adult grizzly bear mortalities occurred within 500 m of roads or 200 m of high use trails (Benn and Herrero 2002). In Massachusetts, presence and breeding of grassland birds was decreased within 1200 m of heavy traffic areas (multilane highway, >30,000 vehicles / day) and within 700 m of moderately heavy traffic areas (two-lane highway, 15,000 - 30,000 vehicles / day). Moderate traffic (8000-15,000 vehicles / day) had no effect on bird presence but reduced regular breeding for 400 m from a road (Forman et al. 2002). For macroinvertebrate soil fauna, however, even relatively narrow, lightly travel roads through continuous forest have significant impacts up to 100 m away (Haskell 2000).

A sufficient body of knowledge has accumulated to allow one to estimate broad bounds on many specific effects of roads, although these obviously differ depending on the specific circumstances (Figure 4.2).

4.3 Road Metrics Related to Park Resource Condition

4.3.1 Road Density

Road density, measured as the average total road length (km) per unit of area of landscape (km^2), is perhaps the most common and convenient overall measure of the amount of road in an area (Forman et al. 2003). Road density must be estimated for a given area and therefore, is sensitive to the area over which it is calculated. Refined values based on road type (e.g., primary and secondary roads versus rural, 4WD roads) may provide additional insights.

Various thresholds of road densities have been reported as important for wildlife. Gibbs and Shriver (2002) suggest that areas with road densities >1 km / km^2 and with traffic volumes of >100 vehicles/lane/day, typical of the eastern and central US, may jeopardize population persistence of some land turtles. Similar high road density (>1.5 km / km^2) resulted in male-biased sex ratios in painted and snapping turtles, potentially indicating incipient changes in turtle populations (Steen and Gibbs 2004). For wolves in the Great Lakes region (Mladenoff et al. 1999) and mountain lions in Utah (Forman and Alexander 1998), populations appear to thrive only where road density is less than 0.6 km / km^2. Wolf packs were most likely to occur in areas with < 0.23 km / km^2, nearly all wolves occurred in areas with < 0.45 km / km^2, and no wolf pack territory was bisected by a major highway (Mladenoff et al. 1999).

4.3.2 Effective Mesh Size

Network form, or the spatial arrangement of roads, may be a better measure of potential impacts of roads by determining the relative size, shape, and arrangement of enclosed patches (Forman et al. 2003). Mesh size, the area or diameter of patches enclosed within a road network, is inversely related to road density and focuses on the enclosed habitat fragments.

Effective mesh size has been shown to be a useful measure for monitoring landscape fragmentation (Jaeger 2000, Jaeger et al. 2008) and has been applied in a variety of landscapes (Padoa-Schioppa et al. 2006, Moser et al. 2007). The effective mesh size is related to the

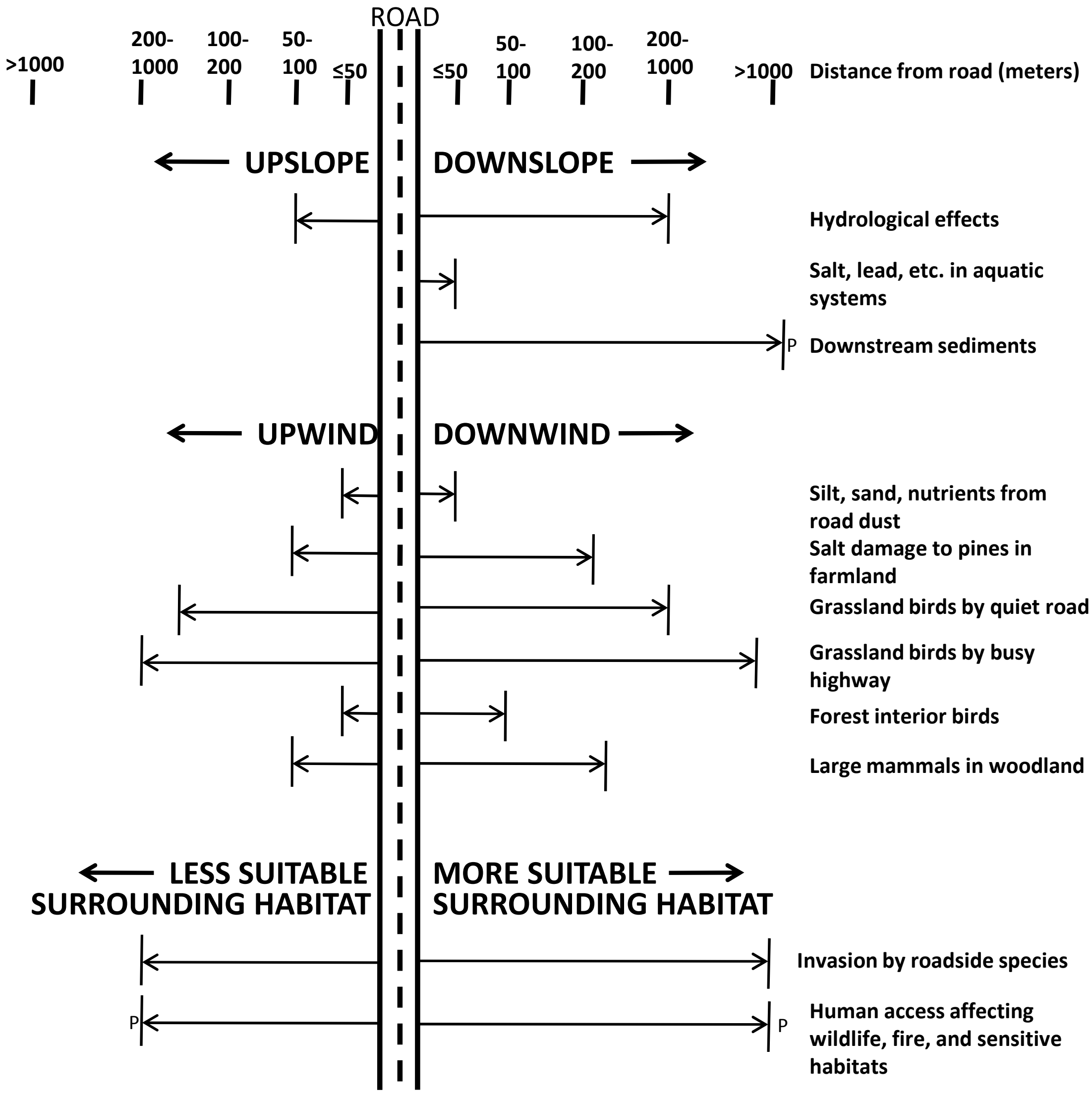


Figure 4.2. Some reasonable estimates of the size of a 'roadzone.' From Forman and Alexander 1998.

probability of two random points in a region being connected, i.e., not separated by barriers such as roads, railroads, or other features depending on the criteria selected (Jaegar 2000, Jaegar et al. 2008). This metric explicitly addresses the movement of individuals between habitat patches in the landscape and can be interpreted as the average size of the area that an animal placed randomly in the landscape would be able to access without crossing barriers (Girvetz et al. 2007).

The calculation of effective mesh size requires specifying the landscape elements that cause fragmentation (e.g., roads, railroads) and definition of the scale over which fragmentation will be determined (e.g., counties, ecoregions). These parameters will apply to a particular species, or to a group of species that have similar capabilities for movement.

5 Population and Housing

Impacts on resources may originate directly from the behaviors of humans (e.g., poaching, noise), or indirectly from the roads, houses, landscaping, and other infrastructure used to support humans. Although the extent of impact is often difficult to measure directly, data on human population characteristics (e.g., population change, economic status, housing) usually provide information that is a direct indication of the magnitude of anthropogenic impacts to lands adjacent to parks. Cultural, political, and socioeconomic factors all contribute to land use decisions (Naveh 1995, Nassauer 2005, With 2005) and are widely used indicators of landscape quality or threats to biodiversity (Nassauer 2005, Cincotta et al. 2000). Because human land uses tend to expand over time (Wade et al. 2003), and we can make projections into the future, these data provide a window into potential threats to park resources.

NPScape population and housing metrics. NPScape analyses are based on historical US Census Bureau data and projections from state offices. Metrics include total population and population density from 1790 and projected to at least 2030. Spatial resolution differs, but is no broader than county level. US Census Bureau data is used to estimate historical housing density, and SERGoM (v3; Theobald 2005) is used to estimate future trends in housing. Housing density usually is strongly correlated with other factors, including population density, road density, and developed impervious surface (e.g., Theobald et al. 2009). Nonetheless, housing density may be a better indicator of environmental impacts than population density alone because it accounts for declining household size and second-home ownership (Liu et al. 2003, Radeloff et al. 2001, 2005). Svancara et al. (2009b) describe these NPScape metrics in detail.

5.1 Population and Settlement Effects on Terrestrial Biodiversity

High human population density has been shown to adversely affect the persistence of habitats and species (Kerr and Currie 1995, Woodroffe 2000, Parks and Harcourt 2002, Luck 2007). In an assessment of the coterminous U.S., Svancara et al. (2009a) found that counties near parks had higher population densities and experienced a greater change in population between 1990 and 2000 than distant counties. They suggested that, even with more intact landscapes surrounding parks, species in these counties may be at greater risk. Counties near parks also had significantly higher per capita income than distant counties (Svancara et al. 2009a). While economic activity has been shown to impact biodiversity (Naidoo and Adamowicz 2001, McKinney 2002), there is debate whether increasing per capita income fuels land conversion (Sisk et al. 1994) or is necessary to solve environmental problems (Beckerman 1992).

Increasing human population results in development, and that development often leads to more development, with more land being converted to expand transportation networks, build schools, and accommodate businesses, which in turn may create a demand for still more housing (Heinz 2008). Conversion of natural landscapes to agriculture, suburban, and urban landscapes is generally permanent (Heinz 2008) and this loss of habitat is the primary cause of biodiversity declines (Wilcove 1998). Thus, there is great concern about the rate of development of rural landscapes around parks (Hansen et al. 2005). For example, growth in low-density, exurban areas (1 home / 0.4-16.2 ha (0.99-40 ac), Brown et al. 2005) has been shown to have numerous biological impacts (Hansen et al. 2002, Hansen et al. 2005) and is increasingly recognized as a primary driver of ecological processes and threat to biodiversity (McKinney 2002, Miller and Hobbs 2002). In the Greater Yellowstone Ecosystem, exurban development has occurred disproportionately in low elevation, riparian areas and is predicted to result in up to 40% habitat conversion by 2020 (Gude et al. 2007).

Table 5.1. Selected studies of the effects of human settlements on plants and vertebrates.

Taxa	Geographical	Result or conclusion	Source
Birds	California foothills	Compared suburban, exurban (4-16 ha (9.9-39.5 ac)/house), and natural areas. Results varied between guilds of ground/shrub nesters, temperate migrants, and species found only in large natural patches. There were clear effects at all housing densities.	Merenlender et al. 2009
Birds	Chesapeake Bay	Examined 28 watersheds. A single-variable model with % developed land was ≥ 13 more likely than more complex models to fit an index of waterbird community integrity.	DeLuca et al. 2008
Birds	Colorado front range	Compared ranches, dispersed housing (avg 16 ha/house), and nature preserves. Human-commensal species (e.g., ravens, blackbirds, starling) had highest densities near houses; ground and shrub-nesting species most abundant on ranches or reserves or both.	Maestas et al. 2003
Birds	Colorado oak-shrubland	At rural houses, compared densities at 30, 180, & 330 m from houses and undisturbed area, and at two densities of houses. Effects differed between human-sensitive and tolerant guilds, and clearly present to more than 180 m, but most densities did not differ between low and high density housing.	Odell and Knight 2001
Birds	Colorado, riparian	Settlement intensity best explained variance in community composition, especially building density within 1.5 km. Found that (especially) ground-feeding species were intolerant of areas with high-use trails.	Miller et al. 2003
Birds	New York: Hudson Valley	Sampled 72 sites along gradients of development, fragment size, and perimeter/area ratio. All species declined at the percent developed within 150 m increased.	De Wan et al. 2009
Birds	Rhode Island	Disturbance intolerant species predominated below 12% residential development and 3% impervious surface, whereas tolerant species predominated above these levels.	Lussier et al. 2006
Birds	Arizona - SE	Overall, low density housing generally increased the number and diversity of birds, but most effect was at very low housing density. Some species negatively affected; grazing effect minor compared to housing.	Bock et al. 2008
Birds	Arizona - Tucson	Housing density had strongest effect on native bird abundance. Study sites were in urban and suburban areas.	Germaine et al. 1998
Mammals: medium-sized	Colorado oak-shrubland	At rural houses, compared densities at 30, 180, & 330 m from houses and undisturbed area, and at two densities of houses. Cats and dogs observed closer to houses; foxes and coyotes farther away and in undisturbed areas.	Odell and Knight 2001
Mammals: predators	Colorado front range	Compared ranches, dispersed housing (avg 1 house/16 ha), and nature preserves. Domestic dogs and cats almost exclusively near houses. Coyotes most commonly seen on ranches.	Maestas et al. 2003
Mammals: rodents	Arizona - SE	Exurban development had no or virtually no effect on rodent communities in grassland, mesquite, and savanna habitats. Grazing did affect rodents.	Bock et al. 2006
Plants	Colorado front range	Compared ranches, dispersed housing (avg 1 house/16 ha), and nature preserves. Ranches had highest diversity of native plants and lowest cover of invasive species	Maestas et al. 2003
Plants: vascular, and vertebrates	Worldwide	At large scales, a positive correlation between population density and species richness; at small scales, a negative correlation. Break in scales at a ~ 1 km study grain size.	Pautasso 2007
Macro-invertebrates, benthic	Chesapeake Bay	98% probability of change-point (impediment) at 20% development in 14-digit HUC; at 2% development, 60% probability of change-point, and 77% probability at 10% developed. Extensive study examined forested and grassland watersheds; biotic indices, etc. Considerable variation in responses across sites.	Bilkovic et al. 2006

5.2 Effects of Housing on Terrestrial Biodiversity

Human settlements can alter ecosystems and affect biodiversity by:

- Loss of habitat to structures and non-habitat cover types
- Fragmenting habitat
- Provisioning of food and water
- Increasing disturbance by people and their animals (dogs, cats, horses, etc.)
- Altering vegetation types
- Increasing light and noise

Settlements, as measured by housing density, generally result in simultaneous changes that all affect native biodiversity, and most studies are thus unable to isolate and identity the unique effect of a single factor. While this confounding of effects is of academic interest, the important questions for resource managers are more likely to focus on the cumulative effects of developments in or near protected areas, and how these alter species, ecosystem functions, or other important values (e.g., viewshed, dark sky, natural sounds, water quality, etc.).

Many studies have examined changes in species abundance or community composition along gradients from natural to rural to urban densities of human settlement; McDonnell and Hahs (2008) provide a recent synthesis of more than 200 studies and Table 5.1 summarizes effects of settlements on of birds, mammals, and plants. Despite a very large number of studies, there are surprisingly few that provide clear quantitative data on housing density or the percentage of land converted for human habitation. Nonetheless, the following (sometimes obvious) conclusions can be drawn from this literature.

Some native species are intolerant of settlements.

Many studies show that some species are highly sensitive to settlements, and the abundance of these species decline. Similarly, commensal species like crows and magpies are generally more abundant in settlements.

Effects of settlements on species can extend a long distance.

The extent of effects varies with species (Table 5.1) and with the magnitude and kinds of disturbance associated with housing. In some areas, the impacts of domestic predators – dogs and cats – are sufficiently strong to drive species in nearby habitat to extinction. Crooks and Soule (1999) documented the reduced density or extirpation of a larger predator (coyote) with development, and the subsequent increase in abundance of mesopredators (e.g., domestic cats, foxes, etc). The high abundance of small predators led to the extinction of scrub-nesting birds. In this case, the impacts of even dispersed housing can extend a long distance.

Settlements as an 'oasis.'

Especially in arid areas, settlements may provide enhanced sources of water, food (via bird feeders), and habitat for nests or breeding. Bock et al (2008) observed an 'oasis' effect of settlements compared to other areas, but noted the effect disappeared within settled areas – i.e., within settled areas, bird species richness was negatively correlated with housing density. These effects are not limited to deserts (Maestas et al. 2003), and they appear to be scale-dependent (Pautasso 2007).

5.3 Human Population Density and Watershed Condition

Population *per se* within a watershed has a limited direct effect on downstream aquatic resources. Human waste can increase nutrients in streams, with the nutrient flux proportional to the population size. It is not clear whether this effect is greater in very small watersheds, where population density can reach high levels, or in large watersheds, where small nutrient inputs can be aggregated over a large area. In small watersheds in rural areas, nutrient inputs from human waste can be substantial due to poorly sited septic systems, as the effect decreases with distance from the stream and increases with belowground connection to the hydrolic flow. However, most higher population densities have some form of municipal waste treatment, which removes the majority of nutrients before dumping effluent into watercourses. Given current water quality regulations, population growth in upstream watersheds should have minimal effect on aquatic nutrient levels.

Population has indirect effects via housing density and road traffic, both of which should be highly correlated with local population size. Because of the paucity of comprehensive road traffic data, temporal change in population size may be the best available surrogate for change in traffic volume.

Housing density within a watershed can influence aquatic resources while houses and infrastructure are constructed, and chronically thereafter via changes in infiltration rates, land cover conversion, and transport of contaminants. Housing construction disturbs the soil surface and may increase sedimentation in streams. The degree to which sediment is actually added to waterways is a function of distance from the construction to the nearest stream, the erodibility of the soil, and the use of management practices such as sediment fences, and precipitation or other sources of water that result in runoff. Landscaping around dwellings stabilizes the soil typically and makes sediment runoff a transient effect.

Maintenance of lawns and other landscaping can contribute to direct effects of housing on nutrient enrichment and contaminant pollution of streams. Lot sizes and prevalent types of landscaping affect the magnitude of this effect: typical suburban lawns receive large inputs of fertilizers, pesticides, and herbicides. Again, regional variation is great enough to preclude generalizations. The only empirical studies are for small urban watersheds and in the context of local-scale water quality mitigation. No quantitative data exist for the effects of landscaping in exurban housing.

5.3.1 Developments and hydrology

While the general relationship between development and increased flashiness of runoff is robust, few data are available to understand changes in this relationship at a specific site. In one of the few longitudinal studies of landscape effects on hydrology, Olivera and DeFee (2007) analyzed changes in hydrologic flow parameters, parcel-level development status, and impervious surface estimated from parcel size and development type from 1949 - 2000 in a small (223 km^2) urbanizing watershed in southeastern Texas. Developed area increased from 10% in 1950 to 30% in 1970 and 75% in 2000; estimated impervious surface increased from approximately 4% in 1950 to 10% in 1970 and 31% in 2000.

The increases prior to the early 1970s were primarily via development of new patches in the watershed; subsequent development was primarily infill, expanding and connecting earlier patches. Cumulative annual runoff and annual peak flow increased over both periods, but more rapidly after breakpoints around 1972 for cumulative and 1968 for peak flow. Both the annual precipitation and the percent developed area were significant predictors of cumulative runoff and peak flow. However, while the coefficients for predicting peak flow did not change across the 52 years, the coefficient for the effect of percent developed area predicting cumulative annual runoff was best fit as no significant relationship prior to a breakpoint in 1972, and a significant positive effect after 1972. Their interpretation was that cumulative runoff in the urbanizing watershed was unaffected by development until a threshold was reached, then increased significantly with further development.

5.4 Housing and Population Measures

Housing density in parks and the surrounding landscapes varies from very low to extremely high densities, and our understanding of the quantitative effects of a particular density of houses is limited. Furthermore, the actual effect of one or more dwellings varies depending on the intensity of landscaping, geographical position and location, whether the landowner has pets, etc. While NLCD and other data used by NPScape identify urban centers and other built-up areas, delineation of ex-urban development – the low-density housing so prevalent near many parks – is much more difficult to identify and describe. It is this low-density development that has exploded in recent decades (Brown et al. 2005, Theobald 2005, Wittemeyer et al. 2008) and that has a disproportionate effect on fragmentation of natural areas (Irwin and Bockstael 2007). Theobald (2004) reviewed approaches to describing population and housing density for ecological purposes, but did not provide specific recommendations for categorizing density measures.

NPScape products provide housing density data categorized into 11non-uniform housing density categories (Svancara et al. 2009b). These 11 categories follow Theobald (2005) and the non-uniform ranges permit a much finer delineation of areas of low-density housing than is common for non-ecological studies. Theobald (2005) defined development as rural (0-0.0618 units/ha), exurban (0.0618-1.47 units/ha), suburban (1.47-10.0 unit/ha), and urban (> 10.0 units/ha).

Housing density can be a major factor in more complex indices of the intensity and impact of land use (see Section 7 of this report), but multivariate or ‘transformation’ methods are still rapidly changing, and they are still not well suited for monitoring. In these indices, it will generally be more useful to estimate the amount or proportion of a landscape that has been physically or functionally modified. Ideally, ecologists will have access to region or land-use specific functions to convert readily available data on housing density to these more ecologically-relevant metrics. Leinwand et al. (in review) have attempted to do this, using high-resolution data for Colorado.

6 Evaluating Resource Protection and Risk

The traditional response to habitat loss and potential impacts on biodiversity has been the creation or expansion of protected areas. Yet the conservation status or stewardship of land surrounding these protected areas often dictates and directs potential changes in land use and can have profound impacts on park resources (GAO 1994, Hansen et al. 2005). Impacts – positive or negative – can be categorized based on changes in the effective size of reserves, changes in ecological flows, loss of critical habitat, and increased exposure to humans (Hansen and DeFries 2007). Within each category, Hansen and Defries (2007) distinguished the types of mechanisms that drive changes, such as species area effects, trophic structure, migration habitats outside parks, and hunting and poaching. A common feature of these effects is that they are known, or strongly postulated, to be directly related to land use intensification.

Knowing changes in land stewardship and resulting land use near and adjacent to parks is important for assessing current threats and impacts, and for evaluating how the situation around parks might change in the future. For example, broad scale patterns of habitat conversion and protection (stewardship) have been used to estimate conservation risk and help identify areas that were at greatest risk both globally (Hoekstra et al. 2005) and nationally (Svancara et al. 2009a). Combined with patterns of potential threats (e.g., roads, developments), assessments of the level of resource protection have also helped identify areas at risk and refine conservation strategies on a statewide basis (Theobald 2003).

NPScape conservation status metrics. NPScape conservation status metrics are derived from currently available land ownership and management maps and include the percent of land area protected and percent of land in broad ownership categories (e.g., federal, state, tribal, etc). Protected lands are those considered to have permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a primarily natural state (i.e., GAP Status 1 and 2). NPScape analyses of these proportions are conducted at 30 km buffer and regional extents.

6.1 Defining What Is Protected

In the United States, the U.S. Geological Survey's Gap Analysis Program (GAP) is responsible for assessing the conservation status of biodiversity with the laudable goal of "keeping common species common." As such, GAP has been the primary developer of land stewardship and protected areas information and is now an integral part of the PAD-US Partnership along with Bureau of Land Management, U.S. Forest Service, Conservation Biology Institute, The Nature Conservancy and the Land Trust Alliance. In GAP, land stewardship combines attributes of ownership, management, and a measure of intent to maintain biodiversity. The term "stewardship" is used in place of "ownership" in recognition that legal ownership of a land area does not necessarily equate to the entity charged with managing the resource, and that the mix of ownership and managing entities is complex and can change rapidly. At the same time, it is necessary to distinguish between stewardship and management status in that a single land steward, such as a national forest, may contain several degrees of management for biodiversity (Crist et al. 2007).

GAP currently uses a scale of 1 to 4 to denote the relative degree of management committed to maintaining biodiversity in each land unit. A status of "1" denotes the highest, most permanent level of protection, and "4" represents the lowest level or unknown status. In assigning conservation status, the gap analysis process follows two principles: first, prescribed management, not land ownership, is the primary determinant in assigning status and second, while data are imperfect, and all land is subject to changes in both ownership and management, the intent of a land steward as evidenced by legal and institutional factors can be used to assign status. The criteria used in assigning a status rank include:

- Permanence of protection from conversion of natural land cover to unnatural (human-induced barren, arrested succession, cultivated exotic-dominated).
- Relative amount of the land unit managed for natural cover, with 5% allowance for intensive human use.
- Inclusiveness of the management, i.e., single feature or species versus all biota
- Type of management (e.g., suppresses or allows natural disturbance) and degree that it is mandated through legal and institutional arrangements.

Using the above criteria, the four biodiversity management status ranks can generally be defined as follows (after Scott et al. 1993, Edwards et al. 1994, Crist et al. 1996):

Status 1: An area having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a natural state within which disturbance events (of natural type, frequency, intensity, and legacy) are allowed to proceed without interference or are mimicked through management.

Status 2: An area having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a primarily natural state, but which may receive uses or management practices that degrade the quality of existing natural communities, including suppression of natural disturbance.

Status 3: An area having permanent protection from conversion of natural land cover for the majority of the area, but subject to extractive uses of either a broad, low-intensity type (e.g., logging) or localized intense type (e.g., mining). It also confers protection to federally listed endangered and threatened species throughout the area.

Status 4: There are no known public or private institutional mandates or legally recognized easements or deed restrictions held by the managing entity to prevent conversion of natural habitat types to anthropogenic habitat types. The area generally allows conversion to unnatural land cover throughout.

Typically, status 1 and 2 lands are considered “protected” while “multiple-use” lands, such as most areas managed by BLM or USFS, are considered status 3. NPScape metrics of conservation status followed this approach and calculated percent protected as the area of land protected (status 1 and 2) divided by the total area of land. Open water may or may not be included in the total, depending on the management status and jurisdiction of the water which is often not actively owned or managed by any one entity. Various combinations of the measurement can be used to gain insights into the protection status of different species and habitat types as is done with the gap analysis process (see Scott et al. 1993).

6.2 How Much Habitat is Enough?

How much of a particular area or habitat needs to be conserved to ensure the long-term protection of biodiversity? This question has spawned endless debate in the literature as many have sought to establish quantitative targets or goals for conservation. Generic, *a priori* targets of 10 and 12 percent protected land are often seen as conservation goals for political entities, such as the World Commission on Environment and Development (WCED) and the International Union for Conservation of Nature (IUCN), while values based on scientific evidence are nearly three times as high (Svancara et al. 2005). Although the arbitrary goals of 10 and 12% were considered bold when first proposed (Soulé and Sanjayan 1998), they are now often deemed inadequate (Rodrigues and Gaston 2001, Wright et al. 2001, Solomon et al. 2003).

The relationship between habitat loss and species loss (i.e., species-area curve) is well established (Williams 1943) and suggests that at the 10% target level of habitat protection, 50% of species could be lost. However, species vary widely in their space requirements based on factors such as threat, natural or induced rarity, and genetic heterogeneity. Even for a single species, the area required to maintain minimally viable populations may differ greatly from those required for ecologically or evolutionarily viable populations (Peery et al. 2003, Soulé et al. 2005). After extensive review of the literature, Groves (2003) suggests that protection of 30-40% of any given community or ecosystem type is likely to conserve an average of 80-90% of species. Similar literature reviews suggest that protection of 10-60% of suitable habitat is necessary to sustain long-term populations of area-sensitive and rare species (Andrén 1994, Environmental Law Institute 2003).

While no single percentage of protected area can be used to ensure protection or maintenance of biodiversity (Lindenmayer and Franklin 2002, Groves 2003, Svancara et al. 2005), setting quantitative conservation targets such as 10%, 30%, or 50% can have utility. Conservation targets provide a means for guiding and evaluating conservation plans, measuring success, and bringing together partnerships (Groves 2003, Pressey et al. 2003). Such targets, however, need to be informed by conservation planning processes based on the biological needs of species, communities, and ecosystems, as well as social and economic considerations, not simply arbitrary and capricious values selected *a priori* (Svancara et al. 2005).

6.3 Measuring and Monitoring Conservation Status

The percent of land area protected provides an indication of conservation status and offers insight into potential threats (how much land is available for conversion and where is located in relation to the park boundary) as well as opportunities (connectivity and networking of protected areas). For the majority of parks, <20% of the surrounding local (30 km) landscape is protected (Figure 6.1, Appendix 2). This suggests that many species and communities may be at risk given expected rates of habitat loss and associated species loss. However, many of these parks still have significant amounts (≥60%) of natural land cover in surrounding landscapes (Figure 3.2), indicating potential opportunities for collaborative conservation partnerships.

Using the Department of Interior Geographical Framework units (Figure 1.4), all regions have at least one park with ≥60% natural land cover in the surrounding (30 km) landscape yet only 8 of the regions (primarily in the western US, southern Florida, and Great Lakes) have surrounding landscapes with ≥20% protected. Of those, the Desert, South Pacific, Great Northern, and Great Basin regions have the highest percent of protected area.

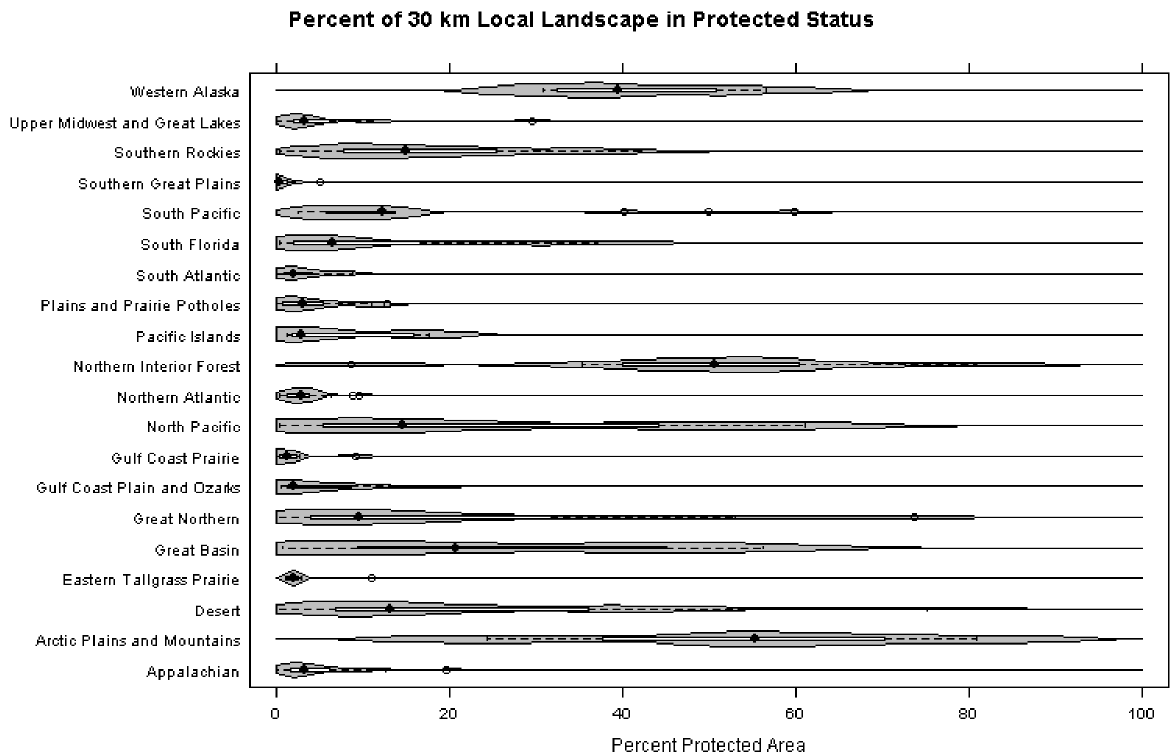


Figure 6.1. Violin plot of percent protected natural land cover in landscapes within 30 km of park boundaries, grouped by DOI Geographical Framework areas (see figure 1.4). The thickness of of the 'violin' is proportional to the frequency of observations. Dots indicate the median and outliers. See appendix 2 for additional results.

Another way to measure whether particular land units (e.g., states, counties) protect their natural environments on the same scale as those it converts is with the Conservation Risk Index (CRI) put forth by Hoekstra et al. (2005). Measured as the ratio of percent area converted to percent area protected, the CRI is most easily interpreted as for every acre (hectare) converted, "x" acres are protected. For example, in Edmonson County, KY the area outside of Mammoth Cave NP is 35% converted with only 0.8% protected (based on the 2001 NLCD and 2006 PAD data) resulting in a CRI of 44 (see Svancara et al. 2009a). So, for every 35 acres that have been converted outside the park, only 0.8 have been protected. In contrast, lands outside of Craters of the Moon National Monument and Preserve in Blaine County, ID are 5% converted and 21% protected resulting in a CRI of 0.2. In this county, for every 5 acres converted, 21 have been protected.

Hoekstra et al. (2005) further classifies areas with >20% conversion and CRI >2 as 'vulnerable,' those with >40% conversion and CRI >10 as 'endangered' and those with >50% conversion and CRI > 24 as 'critically endangered.' Combining this information with the percent urban area and human population change can help identify potential conservation and educational opportunities (see Svancara et al. 2009a). Various modifications can also be calculated to assess the potential risks from agriculture (percent agriculture to percent protected) and urban (percent urban to percent protected) separately. Similarly, multiple-use lands (GAP status 3) can be assessed separately from those traditionally considered protected (GAP status 1, 2) depending on the question being addressed.

7 Multivariate Indices and Approaches

A consistent theme in this report is that individual landscape metrics tell only one part of a potentially complex story. Common reasons to develop and use mathematical indices are to:

- Identify high-quality conservation areas
- Communicate cumulative impacts of a variety of different kinds of stresses
- Reduce complexity by statistically aggregating variables
- Forecast effects of specific land use or cover changes on biota

In this section, we introduce the approaches most commonly used to combine multiple landscape-scale variables. The relevant scale of multivariate indices can vary from global to local, but we think NPS staff will be most interested in indices that are relevant at local to regional scale. At these finer scales, local expertise and data are very important, and our focus is thus on providing an introduction. Some parks and/or I&M Networks may decide that development of locally-relevant multivariate indices is a desirable use and embellishment of NPScape data. We do not think multivariate indices are appropriate for use, by themselves, as a Vital Sign. However, the indices may effectively use Vital Signs as inputs and provide an effective means for communicating landscape condition. Many of these indices are at an early stage of development, and frequent changes are likely.

The general approaches to developing multivariate indices can be characterized as 'top down' or 'bottom up'. The 'top down' approach relies on ecological theory or a variety of empirical relationships to derive an integrative index. These indices are usually calculated in an equation that includes such things as distance to a road, housing and/or population density, land conversions, and impervious surfaces to (e.g., Sanderson et al. 2002, Leu et al. 2008, Svancara et al. 2009a, Riitters et al. 2009). The 'bottom-up' approach typically relies on an extensive base of field observation, and it uses statistical techniques to define the relationship of environmental variables to the observations (e.g., Danz et al. 2007, Carlisle et al. 2008). The 'observations' are often themselves indices, and perhaps the most common application of this approach is to use a stream or fish Index of Biological Integrity (IBI) as the response variable.

7.1 Top-Down Approaches

These indices share a basic approach of using known or strongly postulated relationships between metrics that are generally available and resources of interest. All of these indices we located require land cover maps that distinguish natural from converted or developed land, and most include a measure of access (roads, railroads), and one or more variables that relates to human population density and/or housing structures. Riitters et al. (2009) take a somewhat different approach, relying entirely on land cover and land use data aligned along axes of percentage natural, agricultural, and developed land covers.

We located surprisingly few publications that developed and documented top-down approaches (Table 7.1). The most consistent category of top-down indices are a group based on the concept of the 'human footprint'. Other top-down indices address a variety of different needs. We're confident that many, many more top-down indices exist, often as a result of site or issue specific needs. But these have apparently not been published, or they address very specific needs (e.g., spotted owl core habitat), and they are not included in this review. Similarly, there are many applications relying on the concept of the 'ecological footprint', which accounts for the amount of biologically productive land and water area required to support an activity or population.

Klitzes and Wackernagel (2009) provide an excellent and concise introduction to this related concept and some of the applications that use it.

A common trait of top-down approaches is the use of weighting or conversion factors that are largely based on heterogeneous studies and expert opinion. Despite the plethora of studies on the effects of e.g., roads and houses on biodiversity, we still find it remarkably difficult to identify specific impacts of these factors on resources, and even more difficult to establish general and defensible quantitative thresholds. Effects of most or all landscape-scale variables are dependent on the species of most interest, and there are usually interactions between variables. Top-down indices can have a profound effect just by illustrating the ubiquity of human impacts, but there are substantial limits to the situations where it is appropriate to base decisions on these indices.

7.1.1 The Human Footprint

Humans dominate virtually all of the world's ecosystems, and we appropriate more than 40% of the world's net primary production (Vitousek et al. 1986). The magnitude of these effects is unimaginable to most people, and environmental scientists have struggled to communicate the cumulative effects of humans on ecosystems. The concept of the 'human footprint' have proven to be an effective means of communication, and recent advancements are focused on increasing the applicability of the concept to address park-scale management concerns.

Sanderson et al. (2002) provided the first broad-scale, quantitative evaluation of the human footprint. They published a global analysis and map that illustrated the distribution and intensity of human impacts on terrestrial, based on population density, land transformations (land cover, built-up centers, settlements, roads and railways), access, and electrical infrastructure. While the analyses are clearly very crude approximations of cumulative human impacts, this work provided an important conceptual basis for more recent and much more detailed studies (Table 7.1).

NPScape data can be used to calculate a coarse approximation of cumulative human impacts, such as that provided by Sanderson et al. (2002). Where the primary impacts to biodiversity and other resources are the result of land conversion, roads or houses, NPScape data will likely reflect the magnitude of the human footprint at broader scales. We feel that NPScape data does not yet provide the breadth or resolution of information needed to generate quantitative and credible estimates of the cumulative impacts of all human activities at local scales. NPScape measures and underlying data can and usually should be used in aggregate to evaluate land condition. More recent and detailed estimations of the human footprint have relied on many more measurements than currently provided by NPScape (e.g., Leu et al. 2008). Having said this, it is worth noting that Wolmer et al.'s (2008) detailed evaluation of the human footprint found that human settlements, roads, and land use/land cover accounted for most of the variation in the magnitude of the human footprint. Other attributes, such as mines, utility lines, and railroad lines has a notable effect in some rural areas, but in general these things were usually associated with roads or other land uses. Wolmer et al. (2008) concluded that useful ecoregional human footprint analyses could thus be obtained using NPScape or NPScape-like data and very inexpensive tools.

7.1.2 Other top-down indices

Table 7.1 includes several examples of top-down indices that address more specific issues, rather than cumulative human impacts. Theobald (in preparation) is developing an additional index that addresses the multi-scale nature of impacts, and that relies on variables provided by

NPScape. We expect to provide a detailed description of this index and its estimation once development stabilizes.

7.2 Bottom-up Approach

Multivariate indices in the bottom-up category rely on statistical relationships, and development of these indices thus generally requires an extensive data base of observations. When the number of variables is not excessive, the statistical approach may be relatively simple and just require some form of regression such as linear regression (Jones et al. 2001), logistic regression (Hale et al. 2004), or classification and regression trees (CART; Carlisle et al. 2008). When a large number of predictor variables is used, then principal component analysis (PCA) or a similar technique is first used to reduce the parameter space and help accommodate variables that may be strongly correlated.

Many bottom-up indices have been developed, at scales that can be characterized as reach (a section of a stream or river), riparian (a bit larger – the stream and near-bank terrestrial area), or at the still broader scales of landscape or watershed. Because the parameters in a purely statistical model are strongly determined by site-specific characteristics, these models generally are not to be used outside the area in which they were developed.

Bottom-up models and indices have provided many valuable insights to the often complex relationships between landscape characteristics and health or condition of aquatic resources. They have and will be important to resource managers and support a variety of site-specific decisions on land management. However, except for rare exceptions, these indices do now have a mechanistic basis, and users are often unaware of the limitations of their use.

Allan (2004) includes a particularly accessible and eloquent summary of issues that complicate our understanding and use of statistical relationships between landscape characteristics and stream integrity. Allan's "four challenges" address complexities introduced by covariation of anthropogenic and natural landscape features, issues of scale, effects of legacies, and thresholds. Anyone that anticipates using a bottom-up index to evaluate streams or terrestrial habitat condition is strongly encouraged to consult Allan (2004).

Table 7.1. Studies that proposed landscape indices or multivariate relationships between landscape attributes and condition.

Indicates	Predictors	Description	Citation
Top-Down Approaches			
Human footprint – Global analysis of terrestrial ecosystems	Population density, built-up areas, roads, railways, rivers, power infrastructure	Converted inputs to common units; extrapolated area affected (e.g., up to 15 km from roads), and summed scores on global scale.	Sanderson et al. 2002
Human footprint – western US	Land use and cover, infrastructure, models of species effects, landscape pattern, etc.	Detailed and relatively sophisticated approach to estimating the 'human footprint' across the western US. YELL, DEVA, ROMO, and MORA used as reference areas. Perhaps most comprehensive study of this sort currently available.	Leu et al. 2008
Human footprint – New England, US	Human settlements, access, land use change, electric power infrastructure	Region-specific application to New England at 90 m resolution. Compared results to Sanderson et al. (2002).	Woolmer et al. 2008
Human footprint - Global analysis of marine systems	Quantified threat based on 17 anthropogenic drivers.	Produced quantitative evaluation of a single index, based on expert judgement. Identified large, little-impacted areas (mostly near poles)	Halpern et al. 2008
Human footprint - California current marine systems	Quantified threat based on 25 classes of anthropogenic drivers, including 10 land-based criteria, for 19 marine ecosystems.	Most land-based data at 1 km scale; fishing and other marine data at 1 km to 1 degree scale. Concluded that coastal ecosystems near high population centers off Oregon/Washington coast are most heavily impacted. Climate change is greatest threat, but most area impacted by multiple threats. Results highly correlated with global results from Halpern et al. (2008)	Halpern et al. 2009a, b
Core habitat – eastern US	National-level datasets: roads, developed areas, tree cover, stewardship category, large water features	Estimated extent and location of core natural areas in eastern US states from North Carolina through main. Evaluated effects of unimproved roads, buffer size, and a connectivity scenario. Goetz et al. (2009) embellished and extended analysis and used graph theory to identify relative importance of core areas to connectivity.	Jantz and Goetz 2008; Goetz et al. 2009
Conversion risk and context – lower 48 states	Land cover, land ownership, population density and change, housing density, income	For natural land covers, estimated patch sizes, edge, and isolation to describe context. Conversion risk a function of proportion protected in surrounding areas and population changes. Estimated for entire contiguous US.	Svancara et al. 2009a
Landscape composition relevant to biodiversity and integrity – 48 states	Proportion of natural, developed, and agricultural land cover-types in neighborhood	Classified lands at 30-m scale into one of 19 landscape mosaic classes, based on proportional composition of 4.41 ha (10.9 ac) neighborhood. Can be visualized as position with triangular relationship of cover types. Evaluated 'risk' based on changes over 9 year period in southern US.	Riitters et al. 2009
Spatial cohesion for multiple species	Habitat quality, habitat area, spatial distribution, and matrix permeability	Developed a rather complicated index to address fragmentation due to urbanization, and provide a sample application to a moderate-sized landscape in the Netherlands.	Opdam et al. 2003
Bottom-up Approaches			
Anthropogenic stress in coastal Great Lakes watersheds	86 variables in 5 stress classes, including land cover, human population (incl. housing), and agriculture	Targeted 762 Great Lakes coastal watersheds. Final index used 5 PCA axes. Agriculture, human population density and development, and cover types (forest Found strong correlation between stress index and bird & fish communities, and water chemistry. Included forest and residential vs commercial development.	Danz et al. 2007
Potential human disturbance to stream, river or	Land use and land cover	Described as scalable to river, stream, or lake watershed. Based on Odum's emergy concept, with many examples from Florida. Identifies sufficient 'area	Brown and Vivas 2005

Indicates	Predictors	Description	Citation
lake watershed – Florida		of influence' as about 100 m for Florida's relatively flat terrain around isolated forest patches or wetlands.	
Benthic macroinvertebrate community condition and biodiversity – mid-Atlantic	% riparian urban, wetlands, agriculture on steep slopes, riparian land use and cover.	Examined 58 pairs of mid-Atlantic coastal plains watersheds with small estuaries. Land use and land cover variables alone were able to identify degraded bottom communities. Using logistic regression, the mean riparian urban cover for low condition category was 14%, or 7% agriculture on steep slopes. Logistic regression odds ratio showed that a 1% increase in the urbanization index increased the odds of a degraded benthic index by 10%.	Hale et al. 2004
Impacts of land cover and use to Chesapeake Bay	Land cover, land use (especially agriculture), impervious surface, watershed hydrological model	Evaluated spatial distribution of impacts across the entire Chesapeake Bay watershed, and examined relationship to ground-based measurements	Goetz et al. 2004
Stream biological condition - Wisconsin		Factors related to condition in 920 streams in Wisconsin area.	Carlisle et al. 2008

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9 Appendix 1. Modeling Studies to Estimate Suitable Habitat Requirements for Wildlife

Percent of natural land cover or suitable habitat necessary for maintenance of various wildlife species in North America, estimated from modeling studies (modified from Svancara et al. 2005). Studies from Australia, Europe, and Africa not included.

Taxa	Area	% Habitat	Conclusions	Reference
Models				
Multiple	Model	75-98	For species with poor dispersal abilities and low reproductive potential, thresholds varied from 75% to 98% suitable habitat as fragmentation increased.	With and King 1999*
Birds	Model	60-100	Model results indicate the numbers of individuals a location can support depends not only on the amount of habitat loss but, whether the loss is restricted to the best or worst patches. If best patches were removed first, decline in number of individuals began immediately. Birds could withstand 40% loss of worst patches.	Sutherland and Anderson 1993
Multiple	Model	30-40	Landscape structure and dispersal behavior affected dispersal success in landscapes with <30-40% suitable habitat; spatial pattern was generally not a factor in dispersal success when habitat > 40%.	King and With 2002
Multiple	Model	20-30	Occupancy probability of single hypothetical species decreases with the percentage of habitat loss due to biological parameters.	Bascompte and Sole 1996
Northern spotted owls	Model	19-23	Predicted that extinction of Northern spotted owls would result if suitable habitat (old growth forest) is reduced to less than 19-23% of the total area in a large region.	Lande 1988
Multiple	Model	10-30	Model results suggest that real landscapes may have a lower threshold than the theoretical value of 60%, population performance likely declines past threshold of 70-90% habitat loss.	Gardner et al. 1987
Multiple	Model	1-99	Determined minimum amount of habitat needed for persistence varies among regions with species reproductive potential and dispersal strategy, quality of the matrix also has strong influence. These extinction thresholds should not be confused with the 20% fragmentation threshold (see Fahrig 1997, 1998; Andrén 1994)	Fahrig 2001
Multiple	Model	10-80	Results indicated that thresholds depend on demographics of species of interest with 80% suitable habitat required for species with low demographic potential.	Lande 1987
Multiple	Model	59.28	In an infinite, random landscape, percolation theory predicts an organism can move freely if its critical resource or habitat occupies 59.28% of the landscape.	Stauffer 1985, Orbach 1986
Birds, Ants	Model	55	Predicted that 45% habitat loss led to extinction of army ants, consequently bird species (n=50) dependent on the ants also showed threshold responses.	Boswell et al. 1998
Multiple	Model	50	Expected occupancy dropped below 0.6 with all simulations when the proportion of suitable habitat was < 50%.	Keymer et al. 2000

Taxa	Area	% Habitat	Conclusions	Reference
Grass-hoppers	Model	40	Models of 2 grasshopper species indicated that >40% of suitable habitat was needed for habitat specialists to maintain dispersed populations and >35% for habitat generalists. Thresholds are "not just a property of landscapes, but one that emerges from species' interactions with landscape structure".	With and Crist 1995*
Trees	Model	25	Simulated migration rates for the tree species (*Tilia cordata)* slowed markedly when habitat availability fell below 25%, though patch size and connectivity were also important.	Collingham and Huntley 2000
Northern spotted owls	Model	25	Model predicted that >25% of suitable habitat was necessary for an 80% probability of survival of Northern spotted owl for 250 years with environmental variance.	Lamberson et al. 1992
Multiple	Model	20	Model results suggest that when breeding habitat covers more than 20% of landscape, species survival is virtually ensured no matter how fragmented and the effects of habitat loss far outweigh effects of fragmentation.	Fahrig 1997, 1998
Beetles	Model	20	Predicted that the ability of ladybird beetles to track prey populations was affected when suitable habitat dropped below 20%.	With et al. 2002

* References are included in the literature cited section of this report.

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS XXXXXX, December 2009

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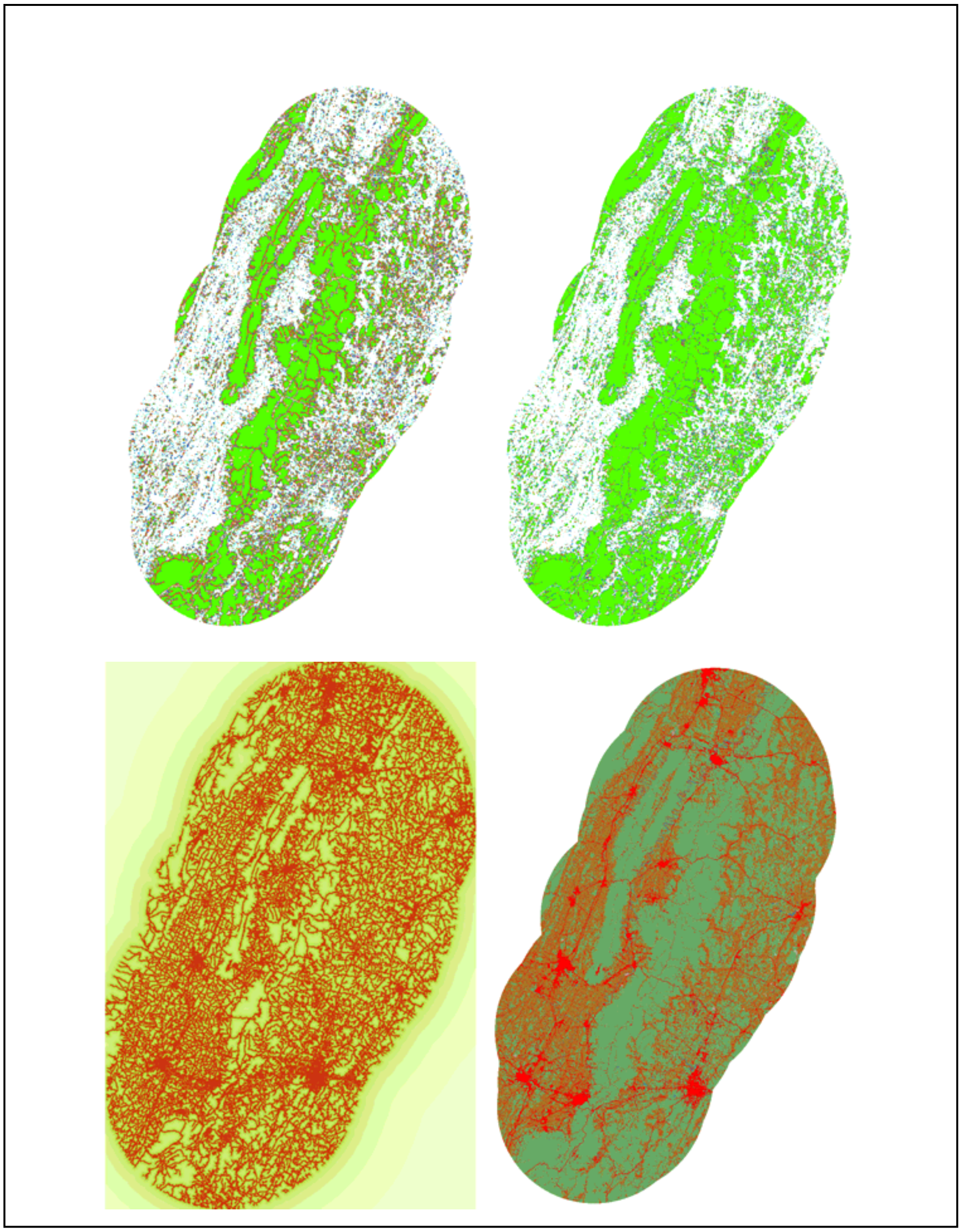
A Guide to Interpreting NPScape Data and Analyses

Appendix 2: Distributions of Quantitative NPScape Metrics Among NPS Units

Natural Resource Technical Report NPS/IMD/NRTR—2009/XXX



ON THE COVER
Four NPScape maps of Shenandoah National Park,Virginia, and the 30 km area around the park boundary. Top left: forest pattern with a 150 m edge width. Top right: landscape pattern with 30 m edge width. Bottom left: distance to a road. Bottom right: 2001 land cover.

A Guide to Interpreting NPScape Data and Analyses

Natural Resource Technical Report NPS/IMD/NRTR—2009/XXX

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All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

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Appendix 1. Distributions of Quantitative NPScape Metrics Among NPS Units

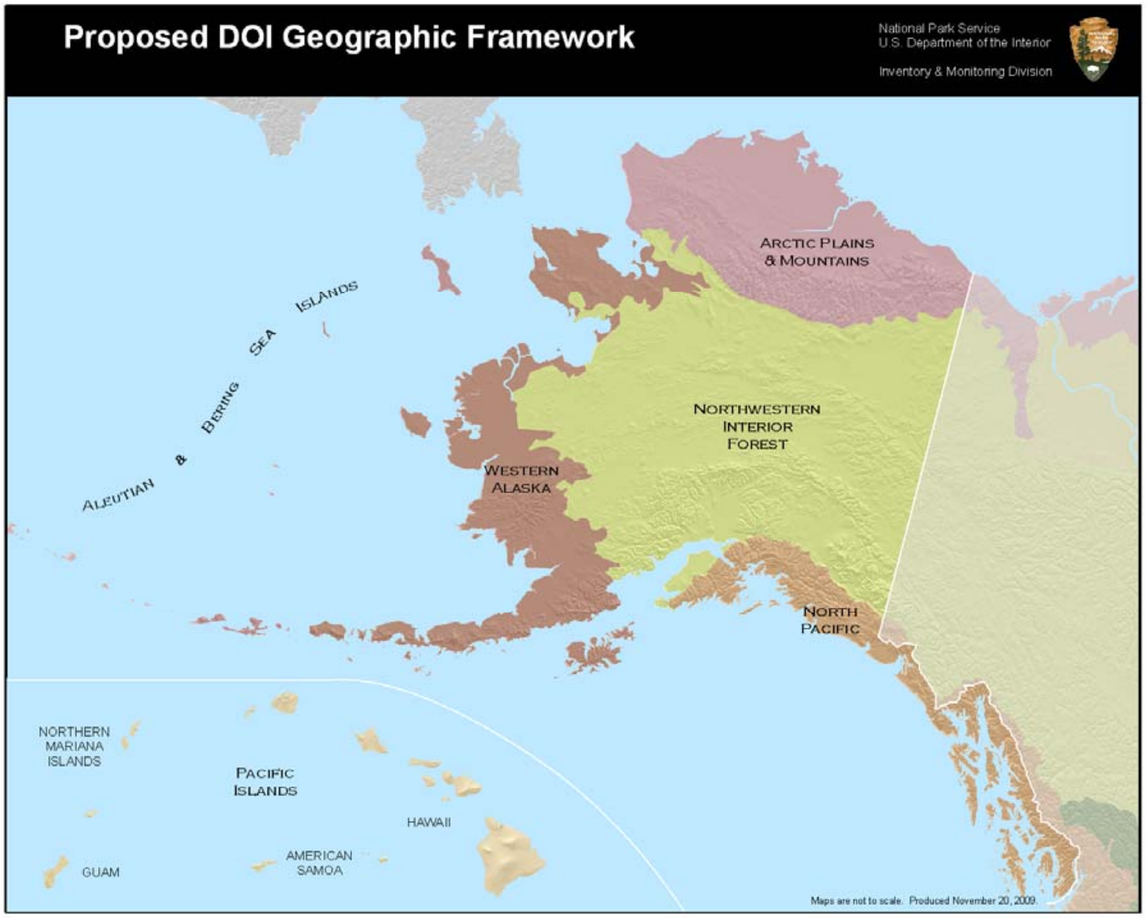
Introduction

These figures summarize the variation among NPS units in the mean values of NPScape metrics within their 30 km local landscapes, including the unit itself. The Land Cover metrics for both 1990 and 2000 are percentages of the local landscape in each of several cover classes. In order to present the most informative summaries relevant to habitat availability, we only present figures for the percent natural cover and for percent forest cover.

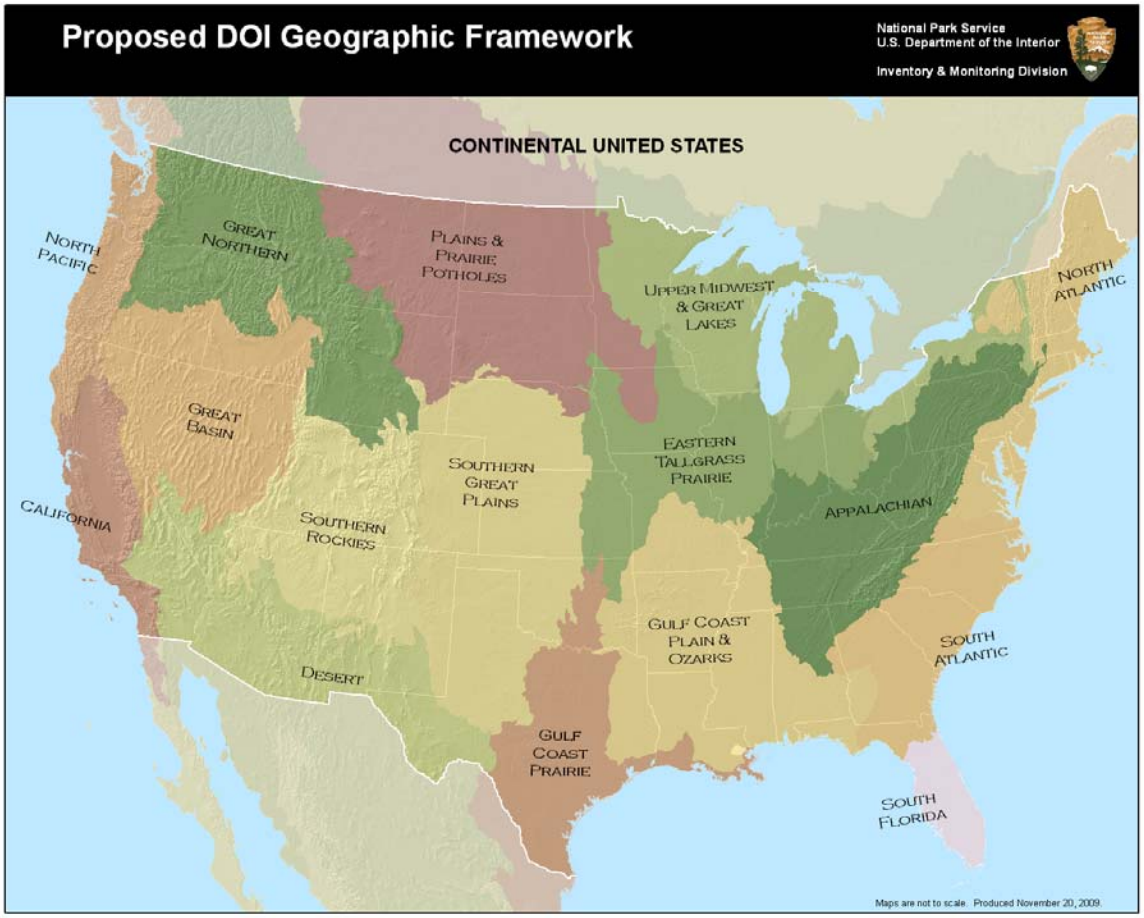
Historic and projected housing density are available for each decade from 1950 through 2100; however they are only available as percentages of the landscape in 10 density categories. We only provide figures for 1950, the earliest date, 2000, the data of the most current observational data, and the projected 2030 densities. Projections beyond 2030 are too uncertain to inform most decision making. Instead of producing a large series of figures for percent of the local area below each of the 9 housing density cutpoints, we provide figures for 2 density groupings: 0 – 1.5 houses per km^2, which is the sum of the areas in the lowest two density categories, and greater than 495 houses per km^2, which is the sum of the top two density categories.

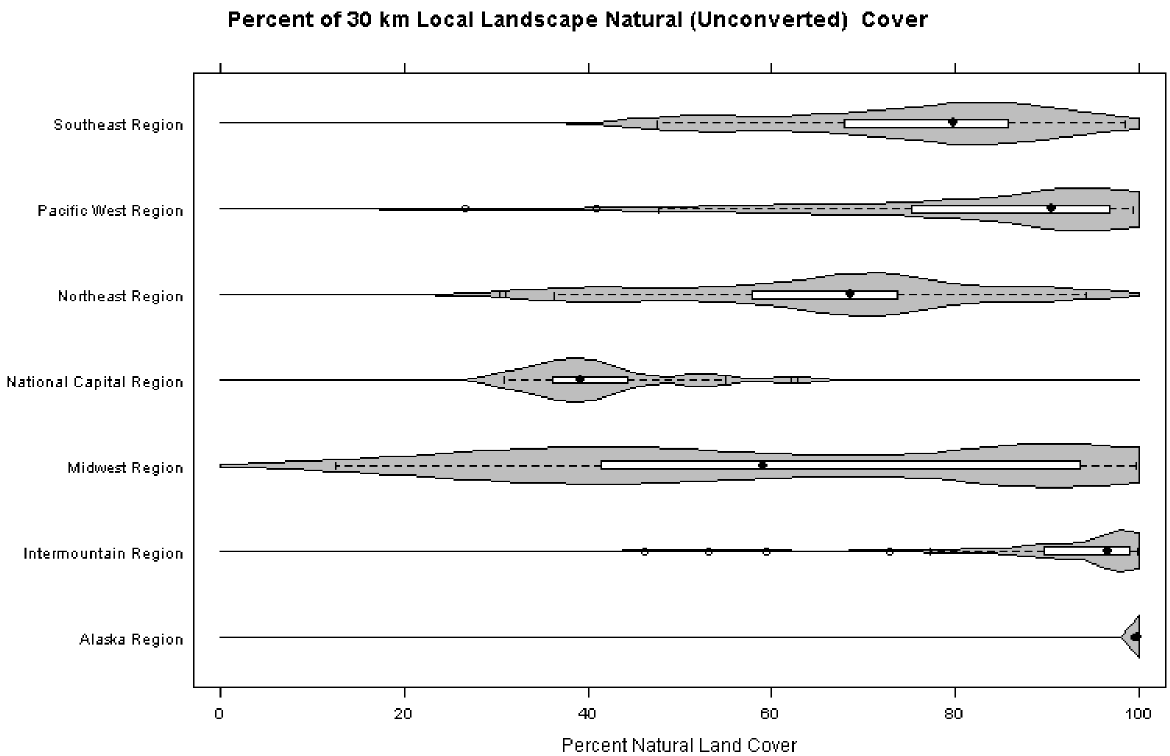
The first two figures for each metric illustrate the distribution of that metric among units grouped by NPS Region or FWS Geographic Area. Violin plots are enhanced boxplots, with the thickness of the violin proportional to the frequency of units with similar values. The major advantage of violin plots is that they can illustrate bimodality in the distribution and indicate modal values, while boxplots reflect central tendencies even of bimodal data. Because the violin part is based on kernel density estimation, the violin bands extend slightly beyond the range of the observed values. Therefore, for metrics in units of percent of the local landscape area, several values at or near 100% produce density estimates that extend beyond 100%. The boxplot components indicate the median (dot), interquartile range (box), range of values not considered outliers (whiskers), and outliers (open circles).

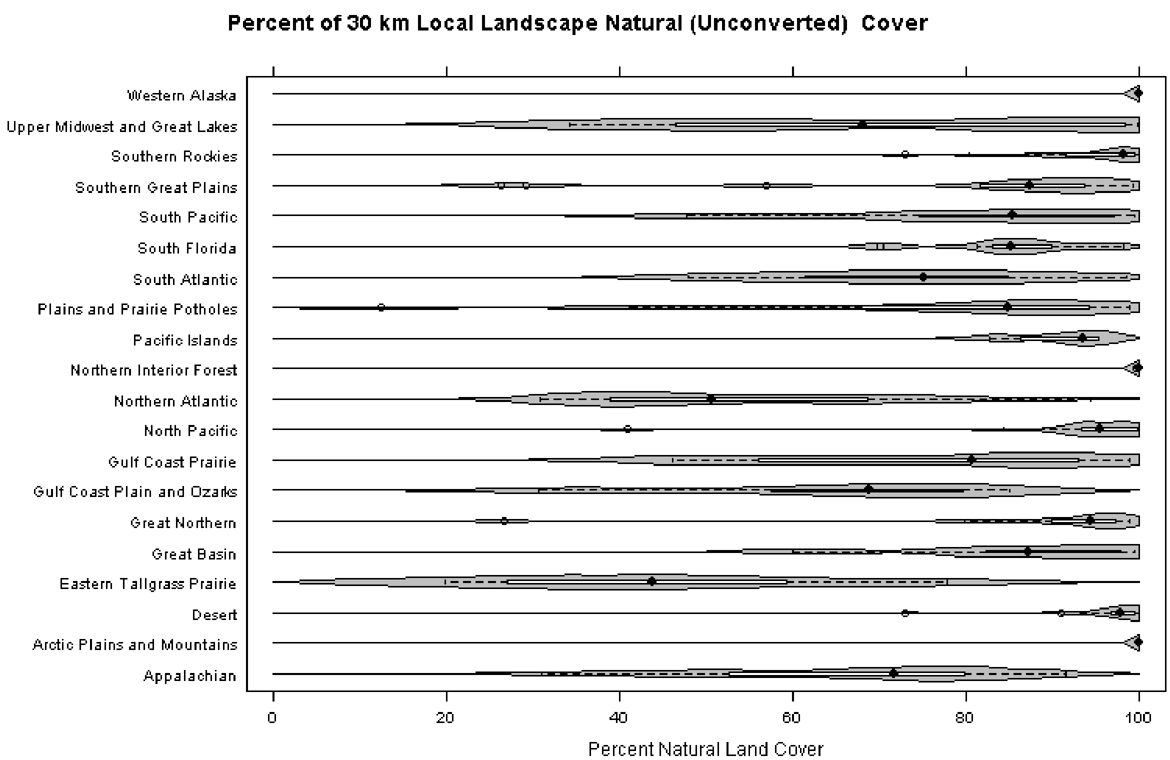
The third figure presents variation in the metric as a function of unit size. Rather than providing boxplots or violin plots for arbitrary categories “small”, “medium”, “large”, and “very large”, we use scatterplots with the base 10 logarithm of the unit area in square kilometers on the X axis. Symbols and figures differentiate among NPS Regions. These figures, plus boxplots and other formats, are available as .png files from the NPScape website, or by contacting tom_philippi@nps.gov.

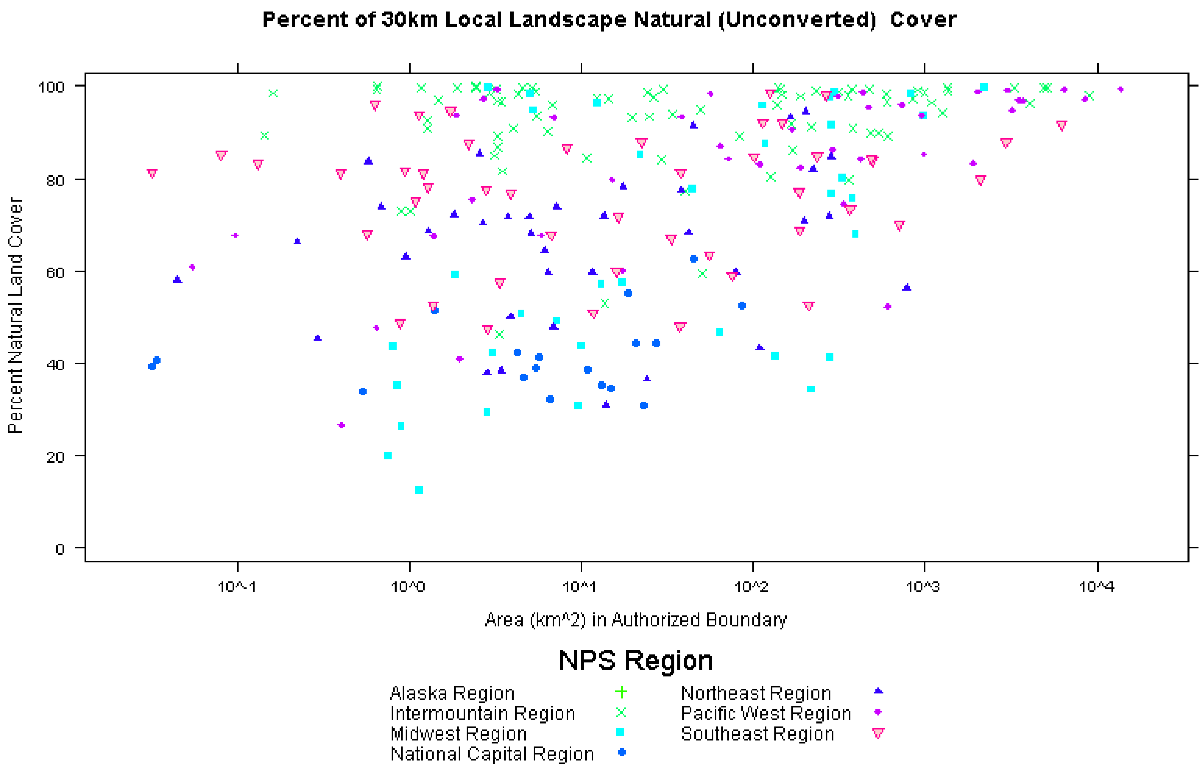


Maps of the proposed DOI Geographic Framework. This appendix presents NPScape metrics summarized by these ecosystem-based areas as well as by NPS regions.

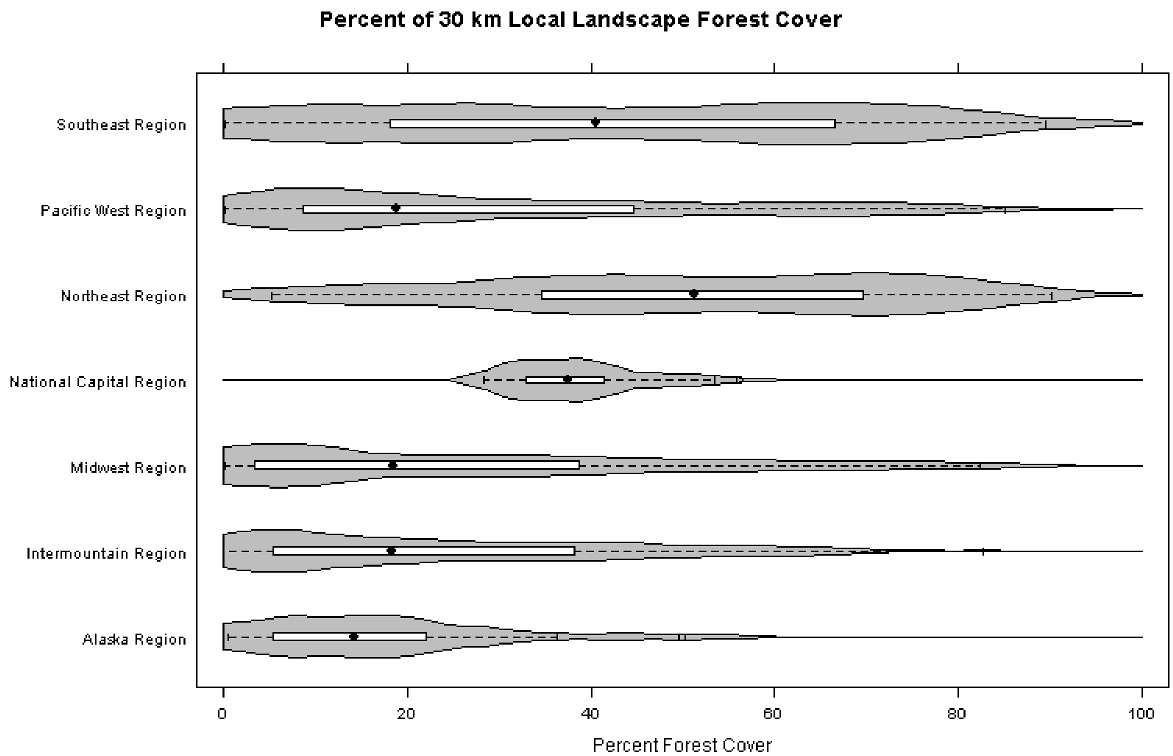

Proposed DOI Geographic Framework
National Park Service
U.S. Department of the Interior
Inventory & Monitoring Division
CONTINENTAL UNITED STATES
North Pacific
Great Northern
Plains & Prairie Potholes
Upper Midwest & Great Lakes
North Atlantic
Great Basin
California
Southern Rockies
Southern Great Plains
Eastern Tallgrass Prairie
Appalachian
Desert
Gulf Coast Plain & Ozarks
South Atlantic
Gulf Coast Prairie
South Florida
Maps are not to scale. Produced November 20, 2009.

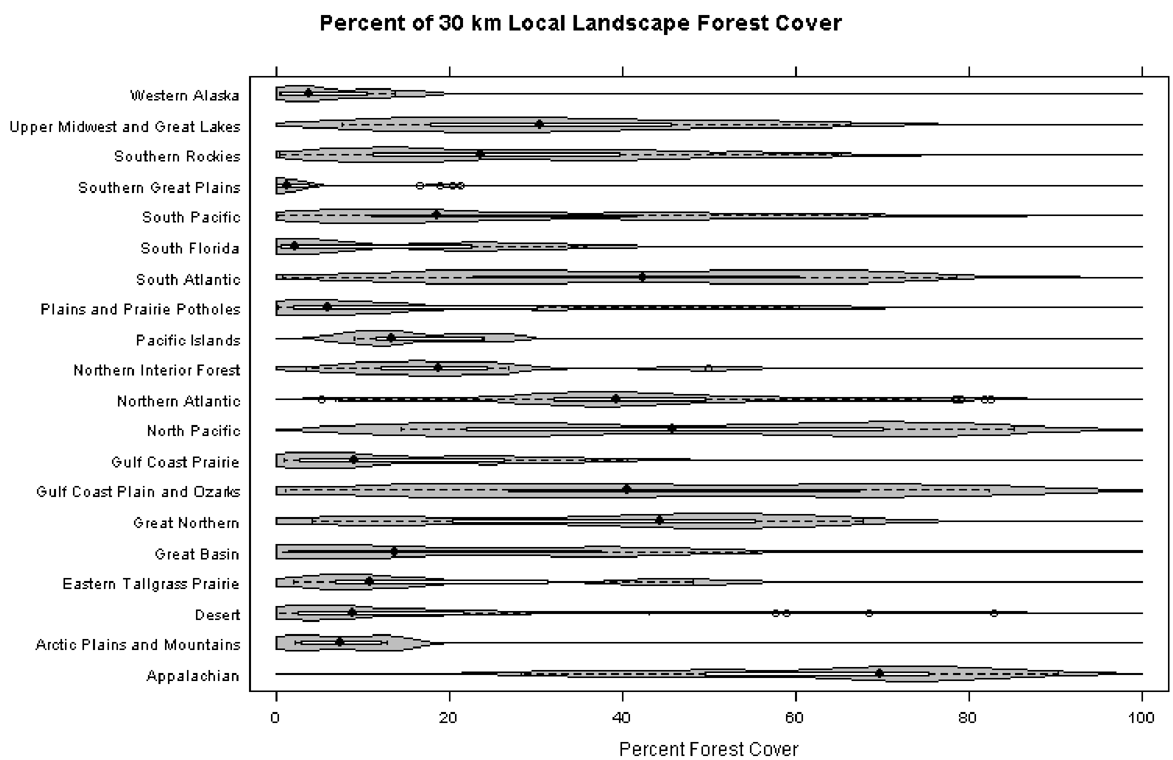

Percent of 30 km Local Landscape Natural (Unconverted) Cover
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0
20
40
60
80
100
Percent Natural Land Cover

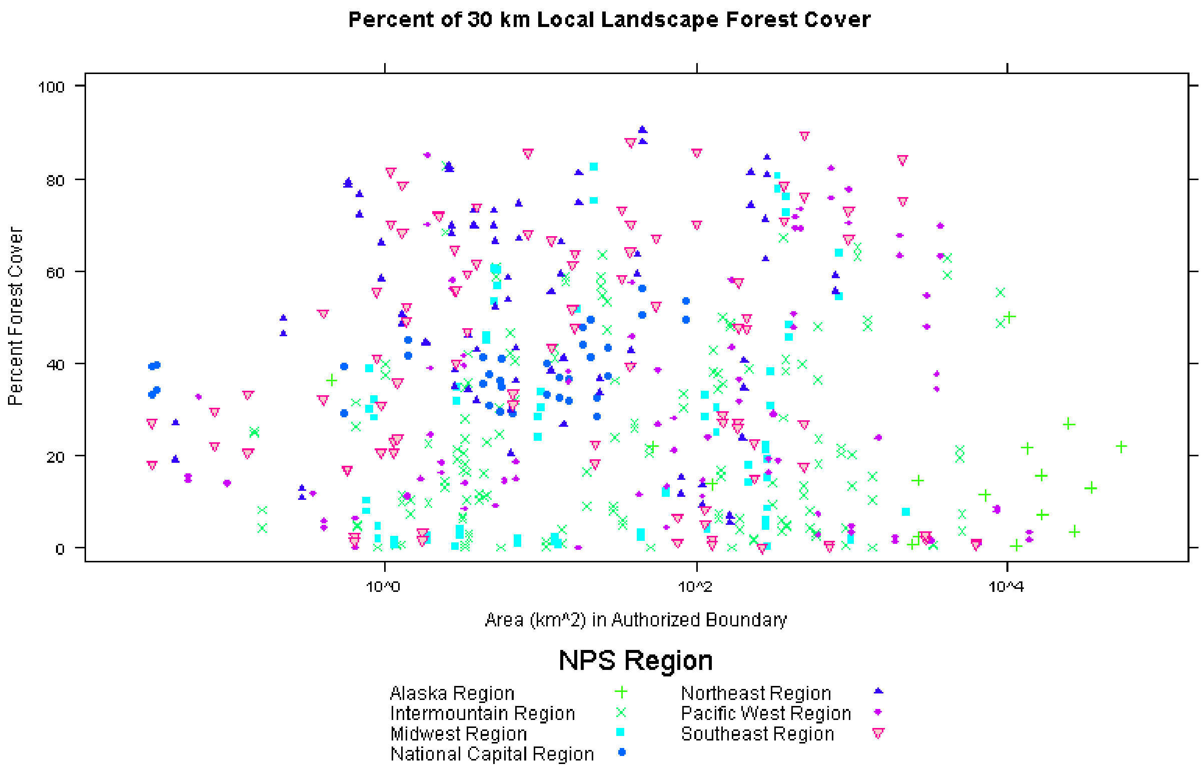

Percent of 30 km Local Landscape Natural (Unconverted) Cover
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0
20
40
60
80
100
Percent Natural Land Cover



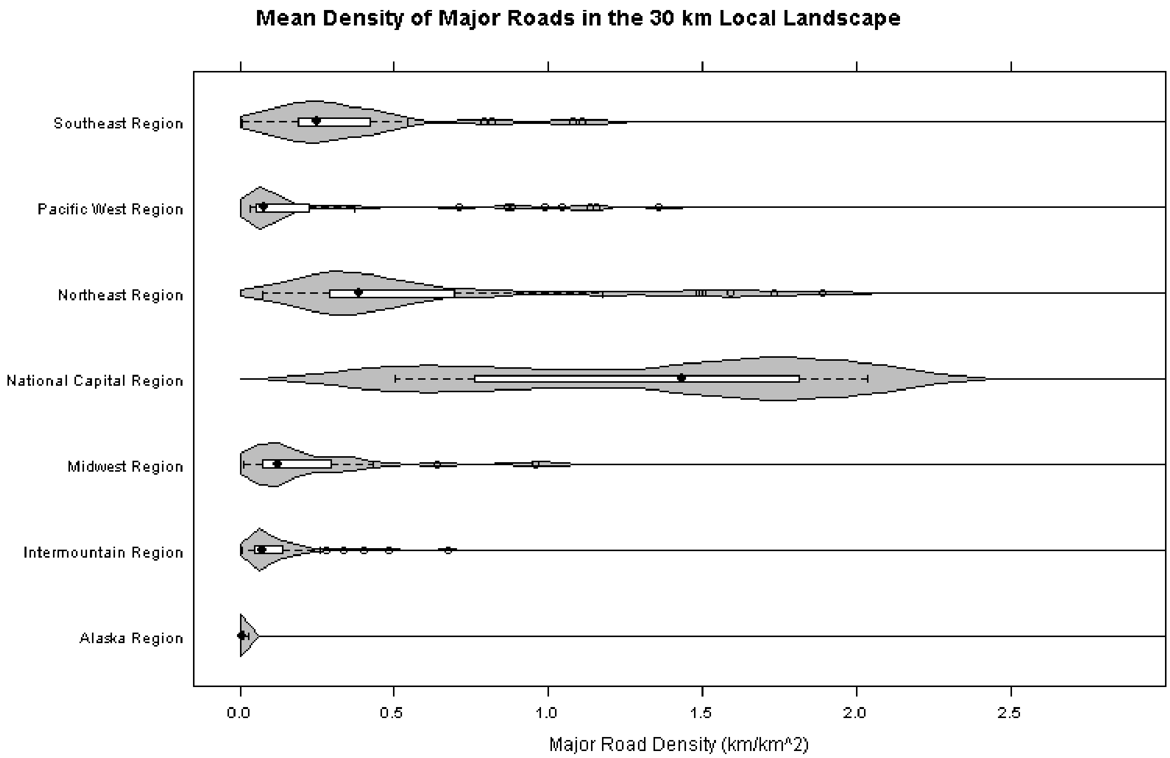
Percent of the 30 km local region land cover classified as natural cover from 2001 NLCD. Metric LNC. While the National Capital Region has no units with greater than 65% natural cover in their local area, it has no units with less than 30% natural cover, either. The Midwest Region has units with the lowest percent natural cover. The one low value for the Pacific West Region is Whitman Mission NM, surrounded by agricultural fields near Walla Walla, WA.

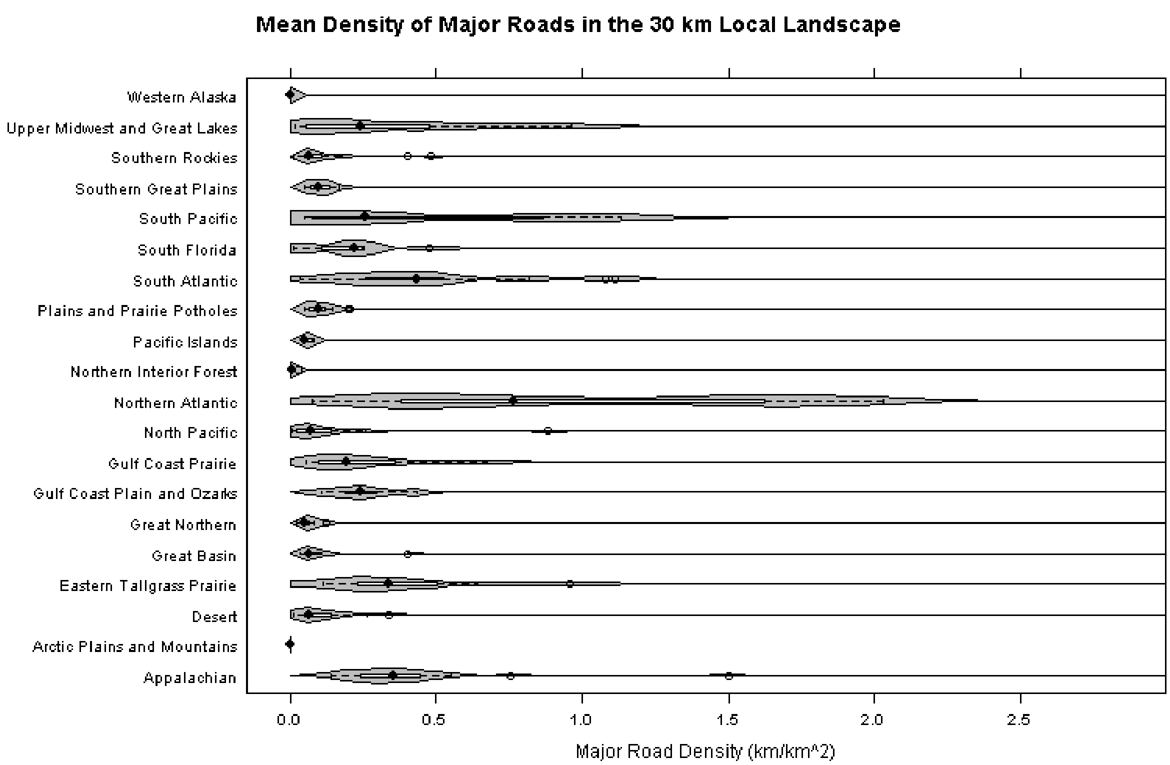

Percent of 30 km Local Landscape Forest Cover
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0
20
40
60
80
100
Percent Forest Cover

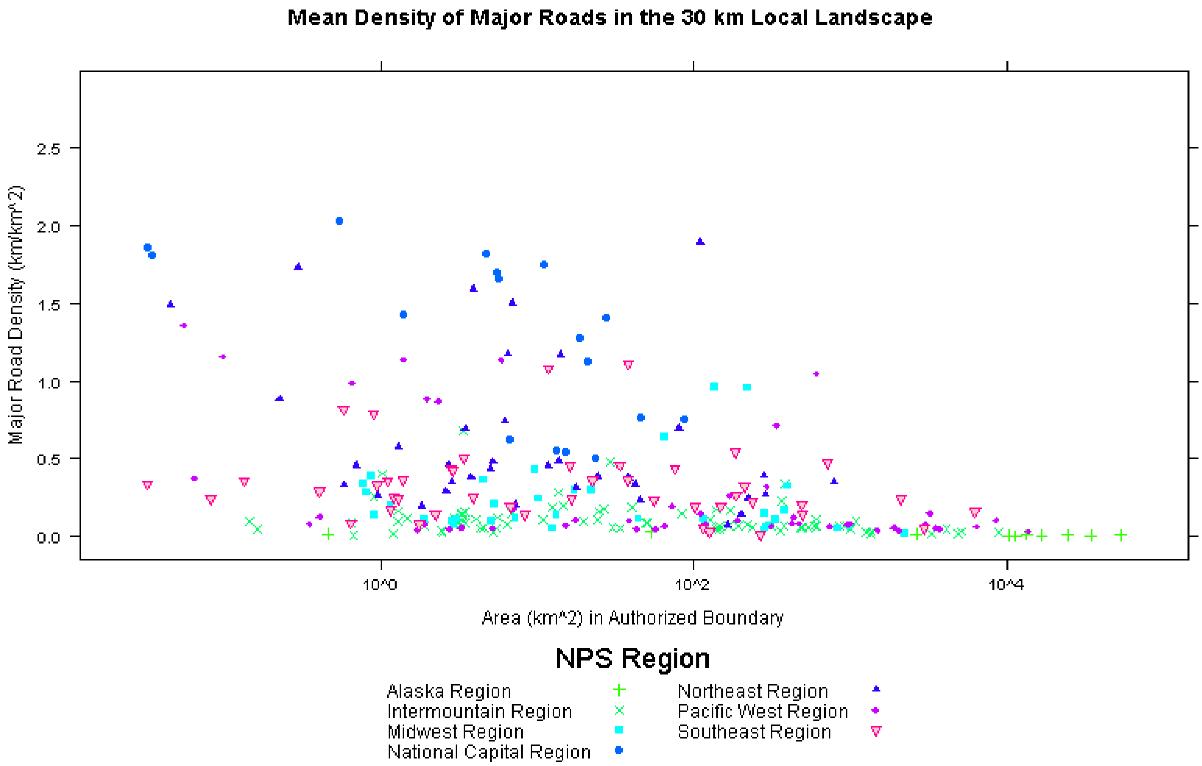

Percent of 30 km Local Landscape Forest Cover
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0
20
40
60
80
100
Percent Forest Cover



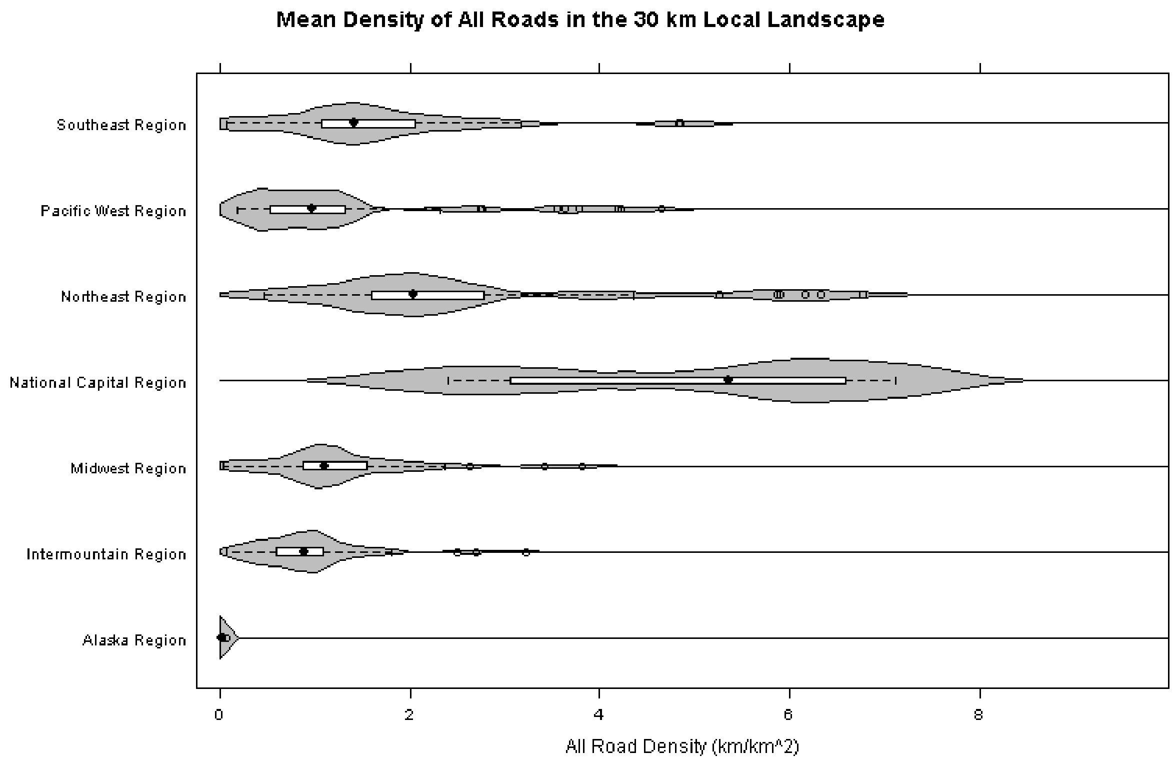
Percent of the 30 km local region land cover classified as forest cover from 2001 NLCD. Metric LAC1, ClassName="Forest".

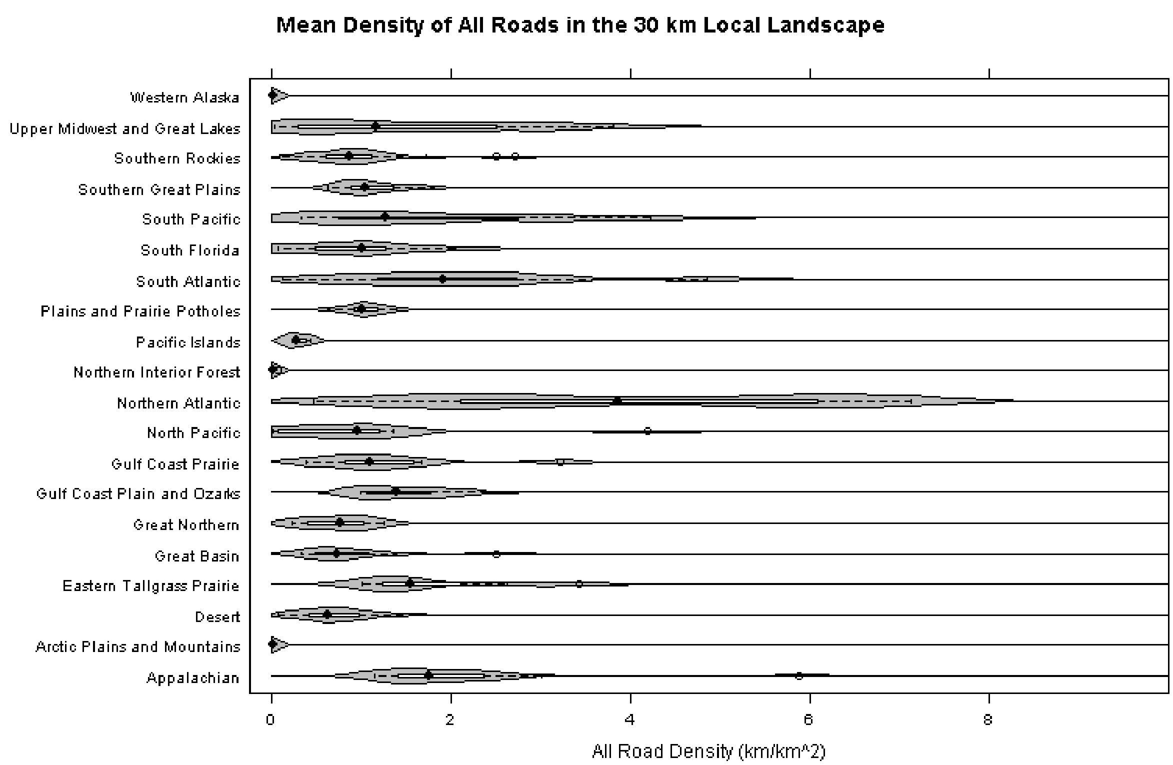

Mean Density of Major Roads in the 30 km Local Landscape
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0.0
0.5
1.0
1.5
2.0
2.5
Major Road Density (km/km^2)

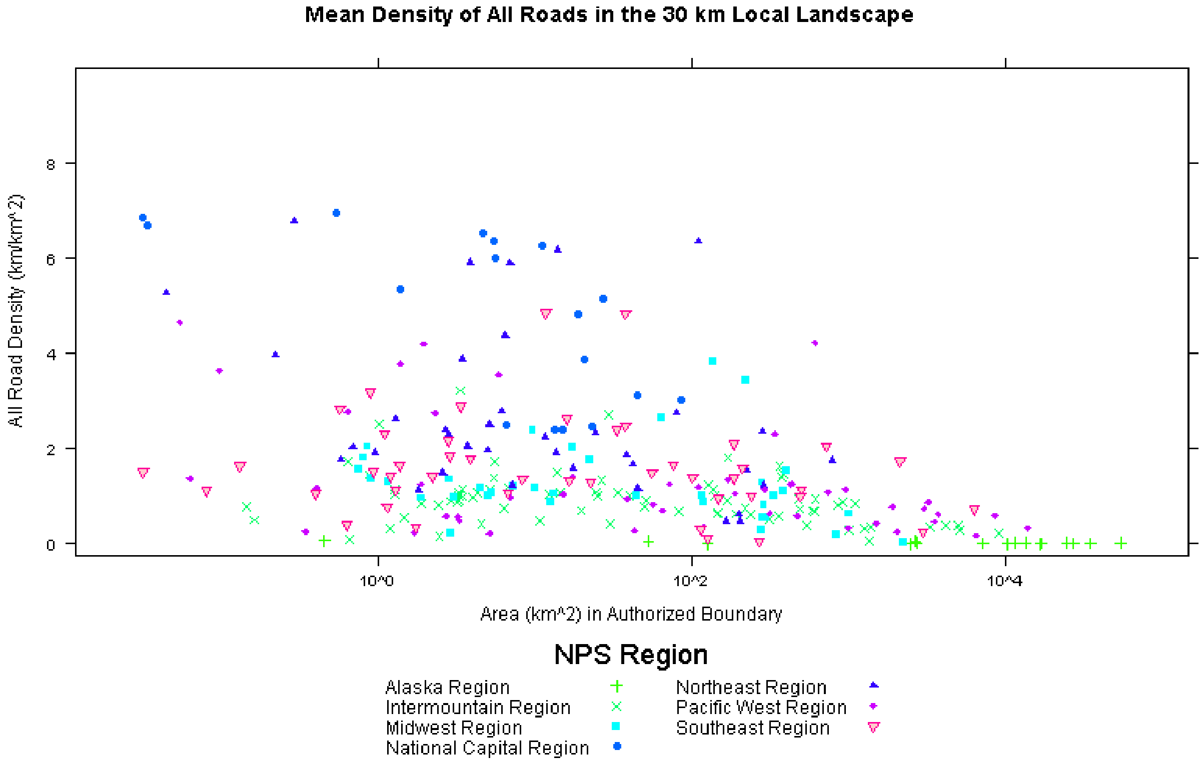

Mean Density of Major Roads in the 30 km Local Landscape
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0.0
0.5
1.0
1.5
2.0
2.5
Major Road Density (km/km^2)



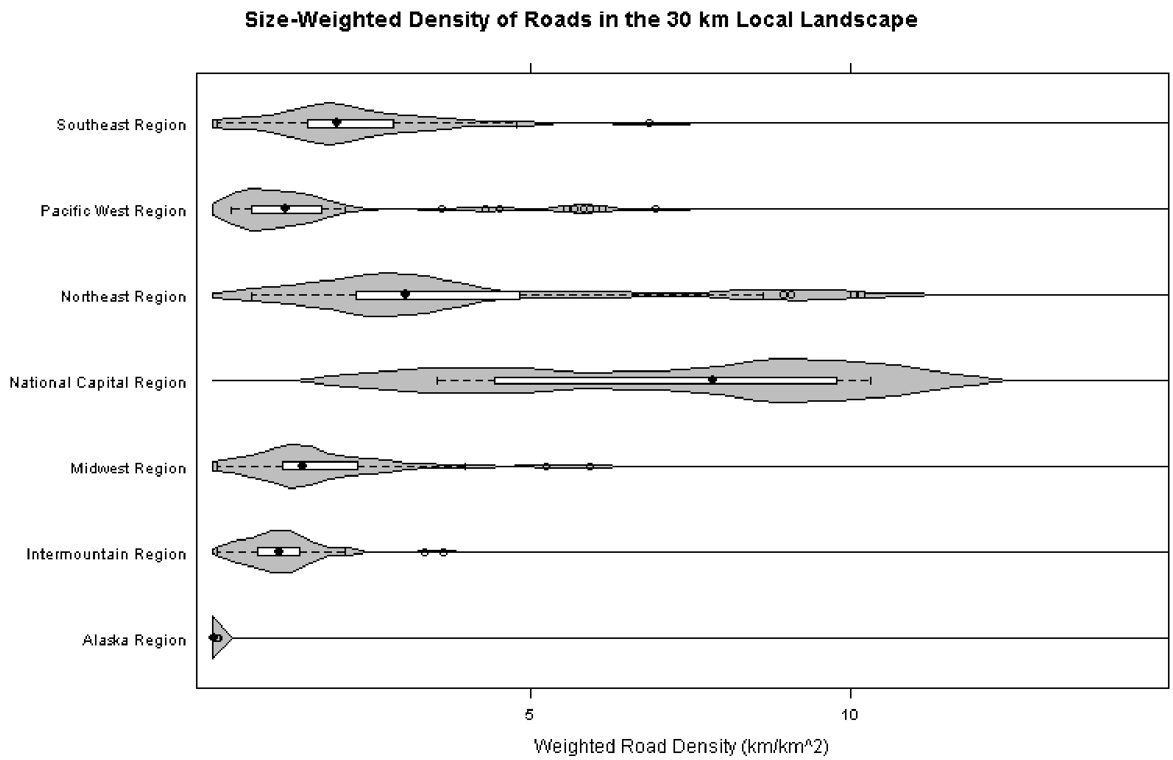
Density of major roads in km / km^2 in the 30 km local region. Metric RDDmr.

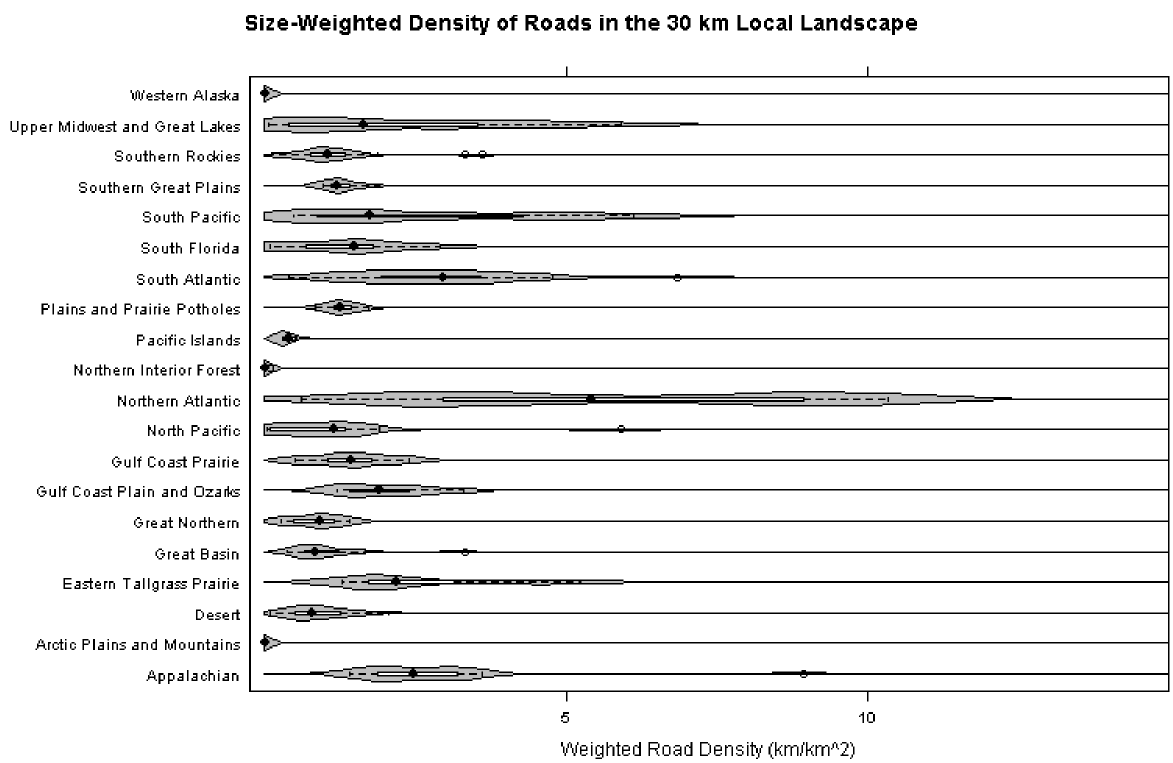

Mean Density of All Roads in the 30 km Local Landscape
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0
2
4
6
8
All Road Density (km/km^2)

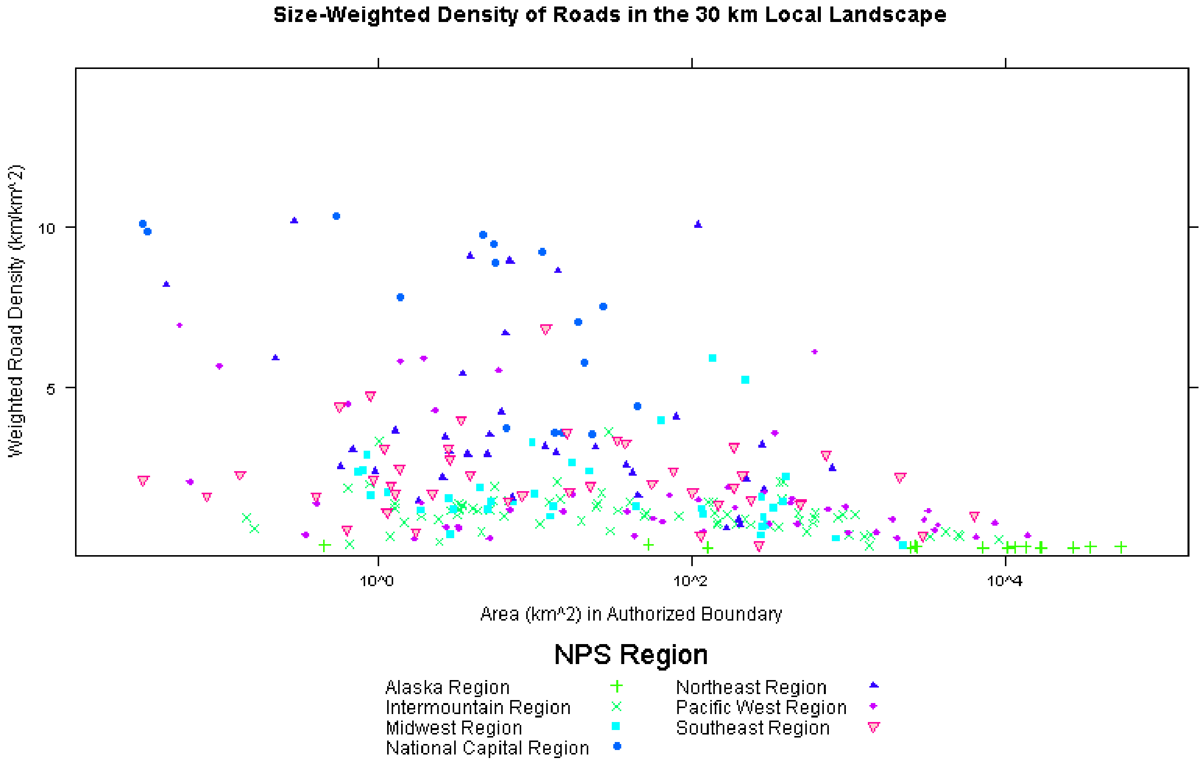

Mean Density of All Roads in the 30 km Local Landscape
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0
2
4
6
8
All Road Density (km/km^2)



Density of all roads in km / km^2 in the 30 km local region. Metric RDDall.

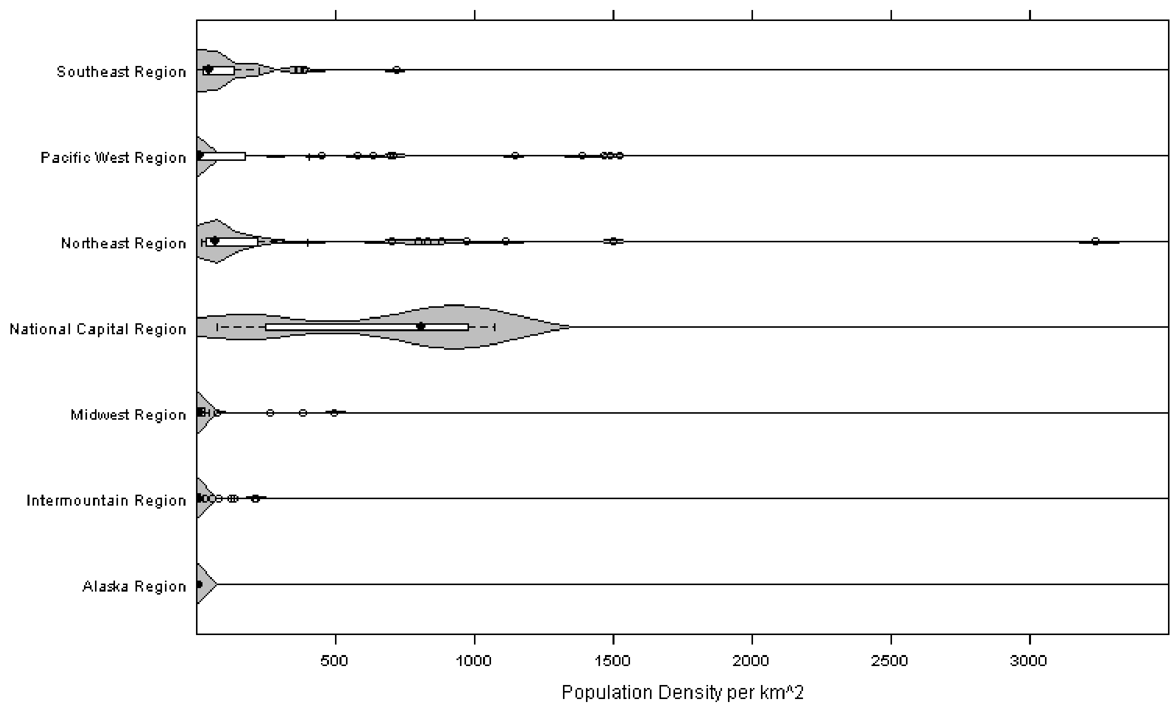

Size-Weighted Density of Roads in the 30 km Local Landscape
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
5
10
Weighted Road Density (km/km^2)


Size-Weighted Density of Roads in the 30 km Local Landscape
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
5
10
Weighted Road Density (km/km^2)

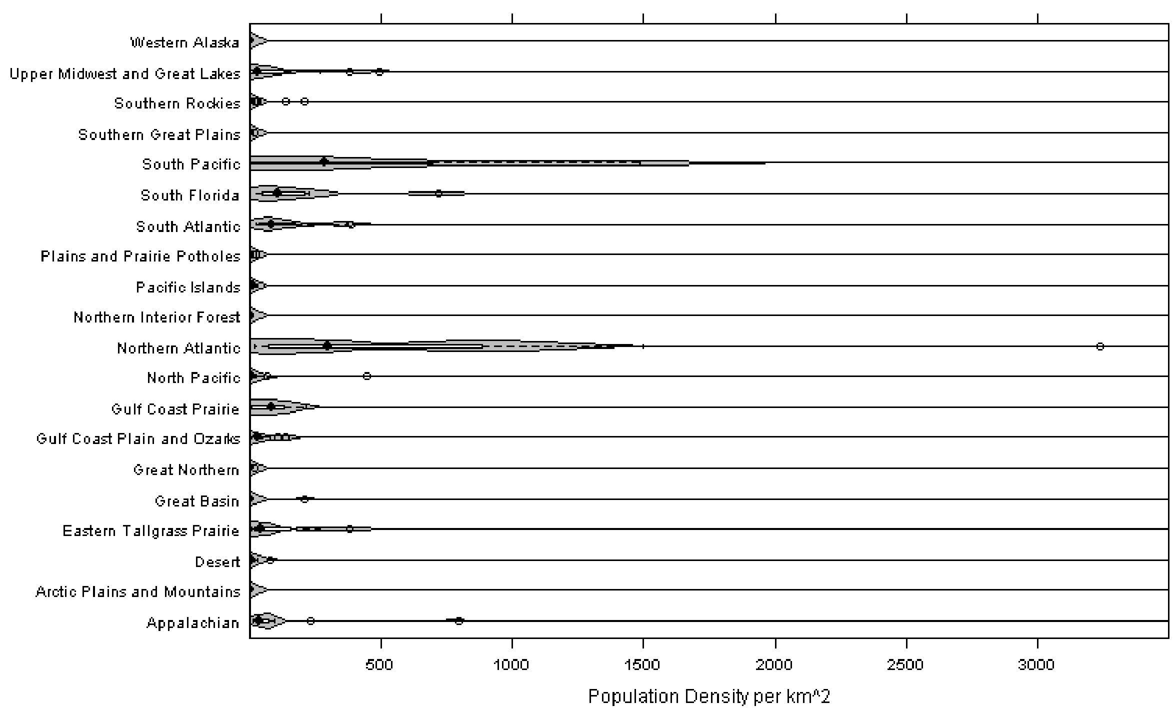


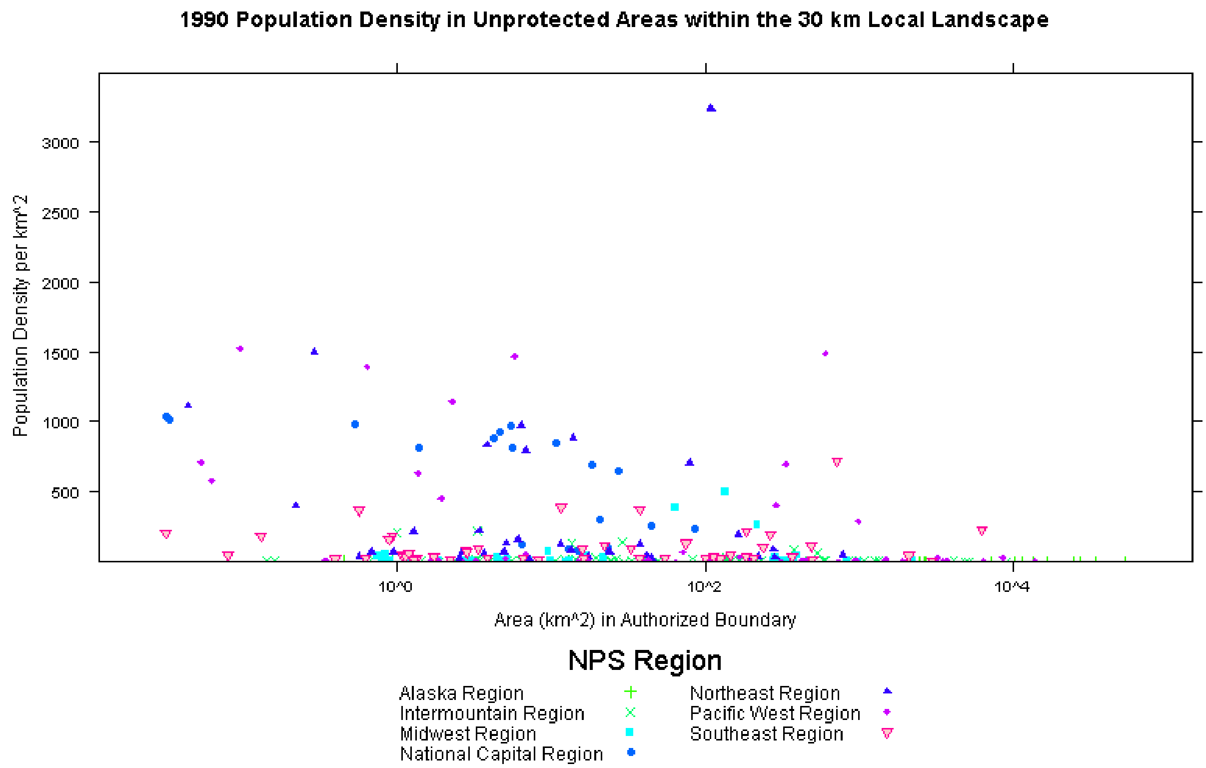
Density of roads weighted by type in km / km^2 in the 30 km local region. Metric RDDwt.

1990 Population Density in Unprotected Areas within the 30 km Local Landscape



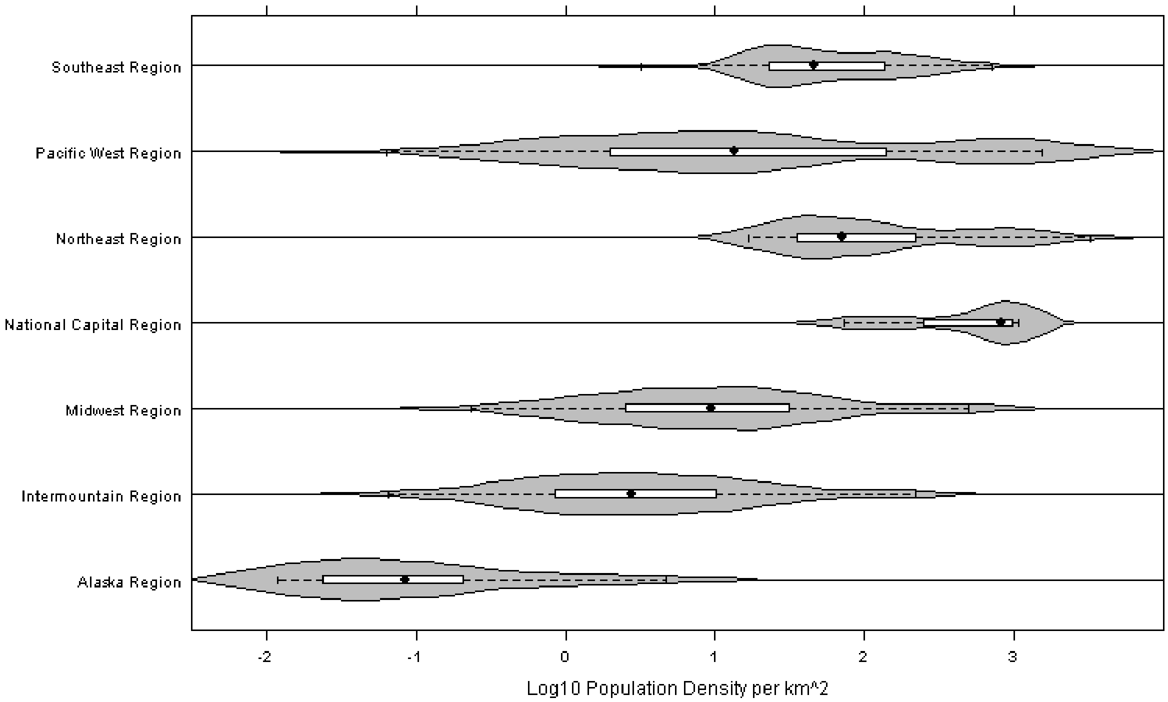
1990 Population Density in Unprotected Areas within the 30 km Local Landscape



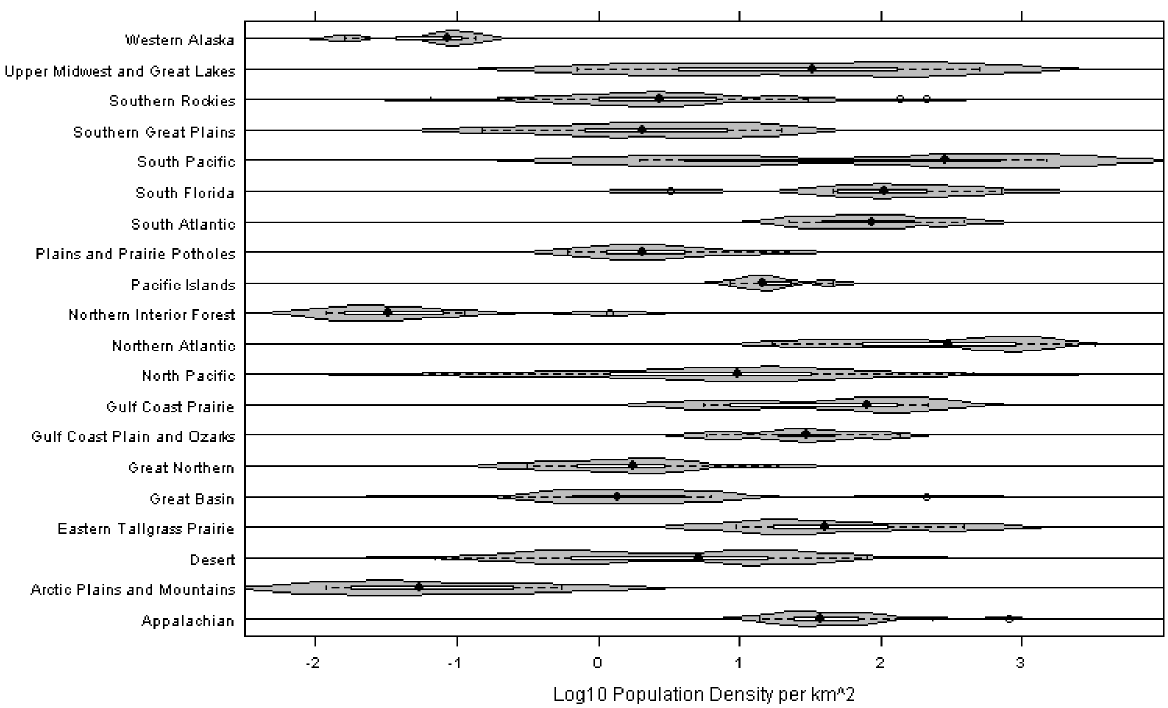


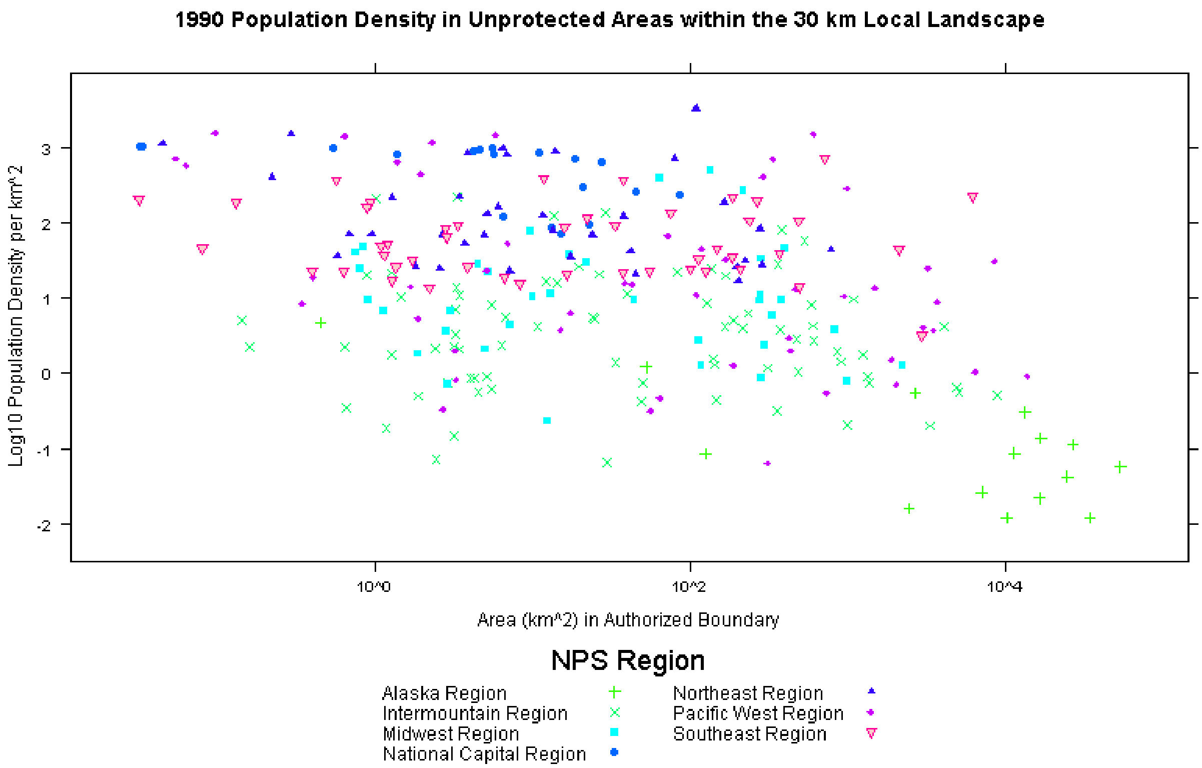
1990 Population density per km^2 in the 30 km local area. Metric PDD; year=1990.

1990 Population Density in Unprotected Areas within the 30 km Local Landscape

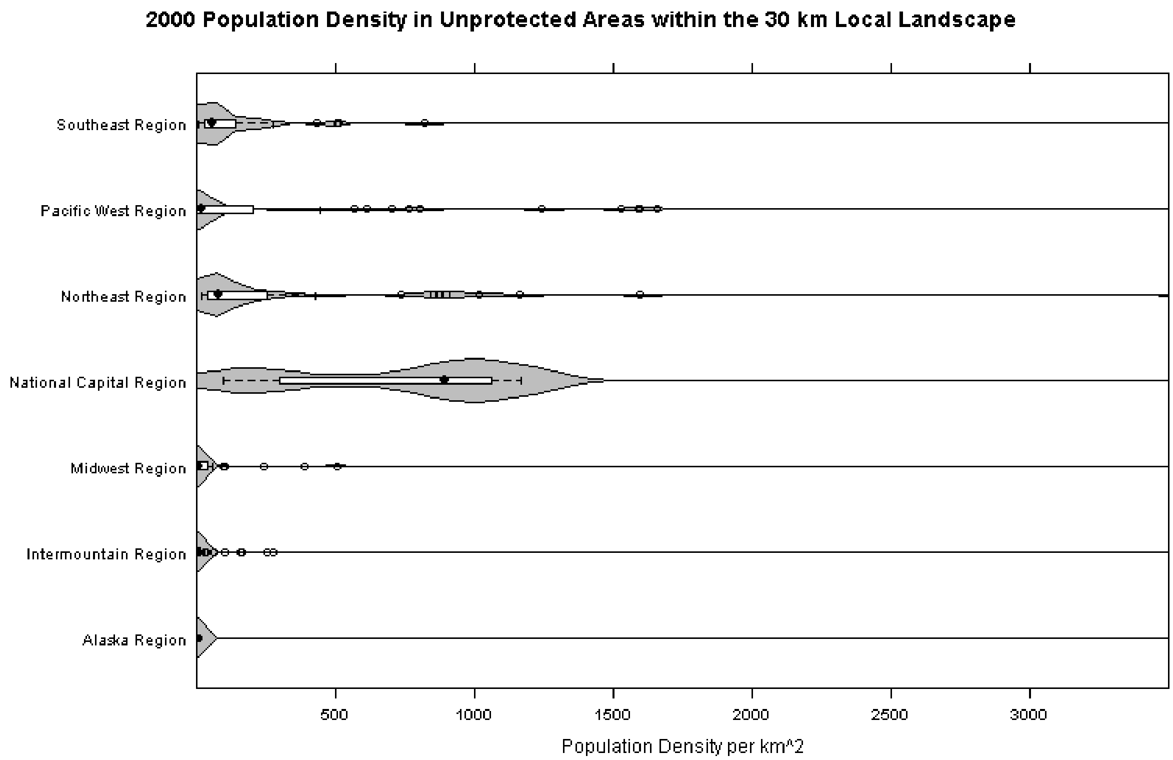


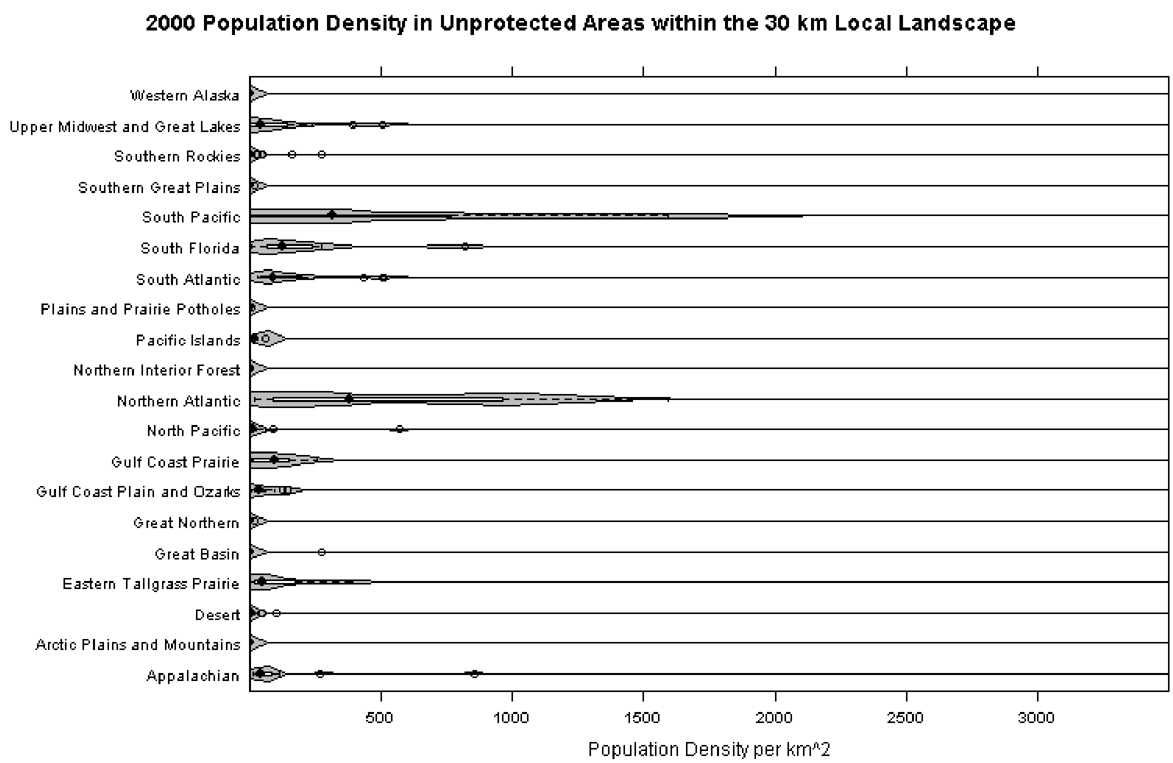
1990 Population Density in Unprotected Areas within the 30 km Local Landscape

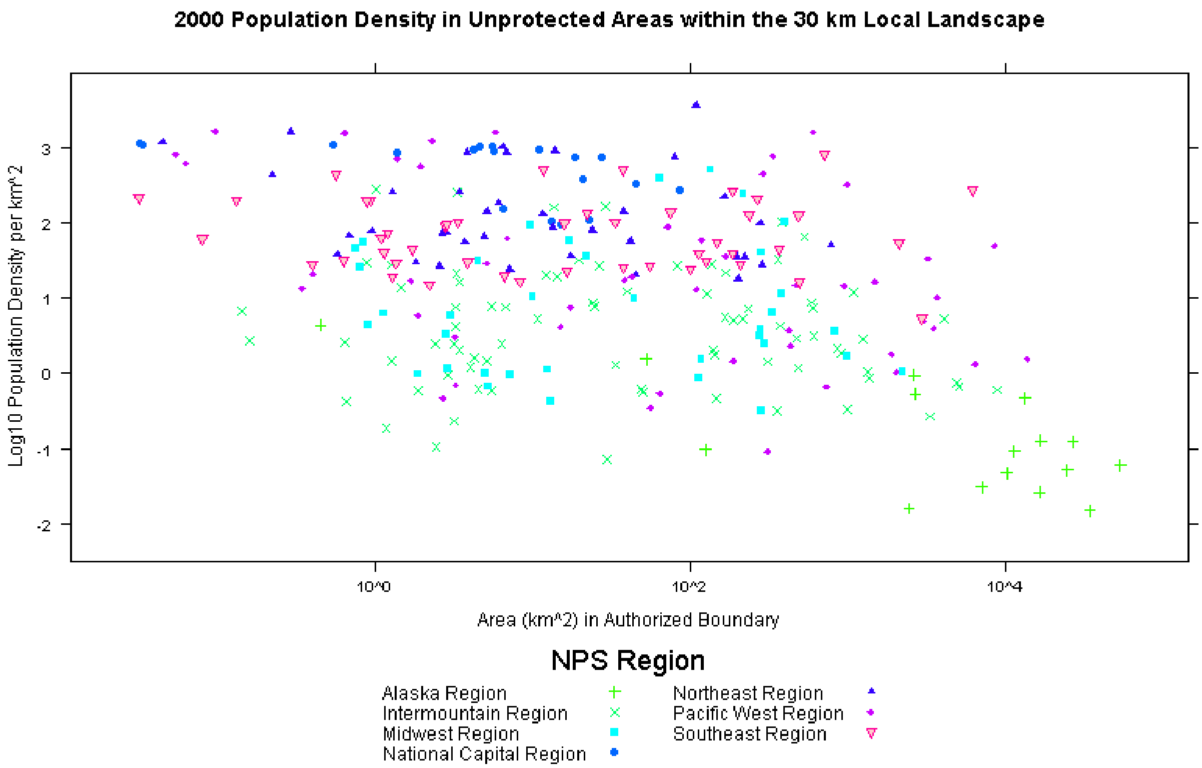




1990 Population density per km^2 in the 30 km local area, on log base 10 scale.. Metric PDD; year=1990.

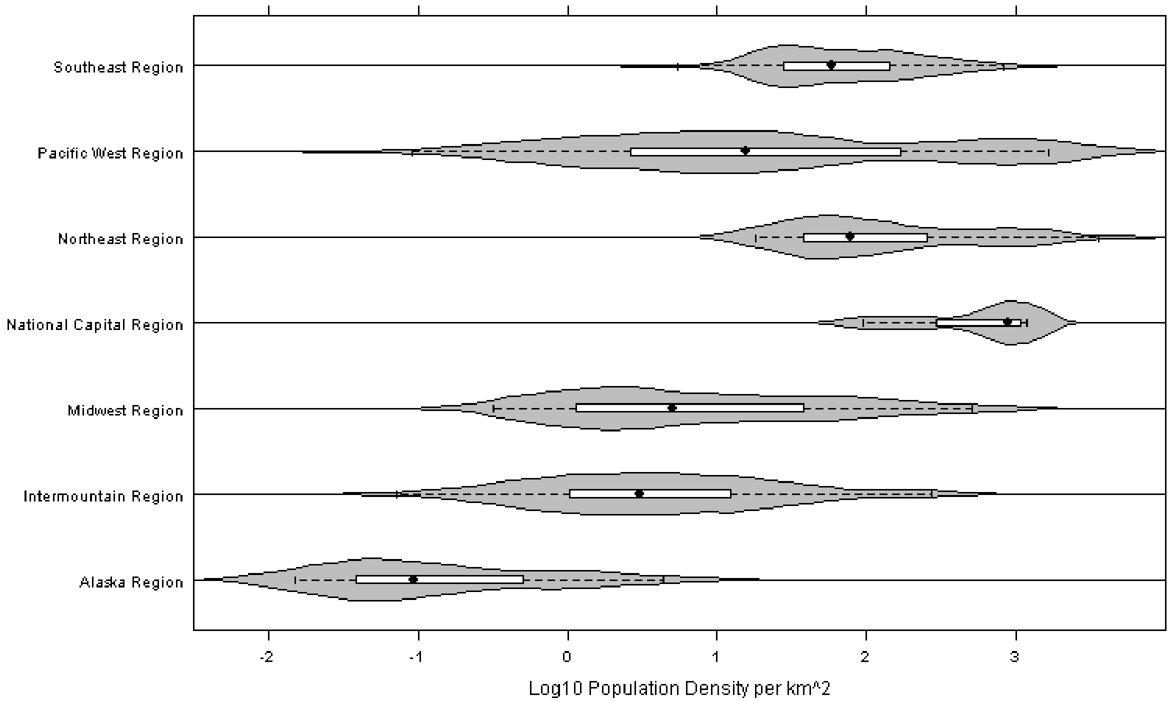

2000 Population Density in Unprotected Areas within the 30 km Local Landscape
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
500
1000
1500
2000
2500
3000
Population Density per km^2


2000 Population Density in Unprotected Areas within the 30 km Local Landscape
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
500
1000
1500
2000
2500
3000
Population Density per km^2

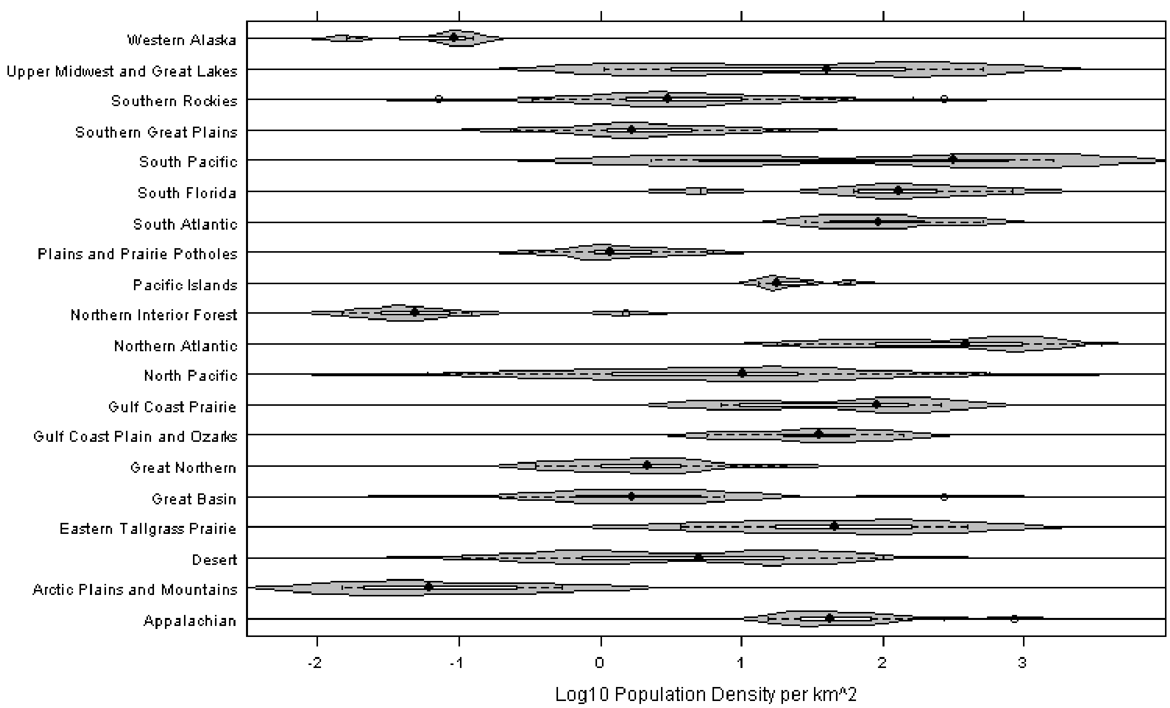


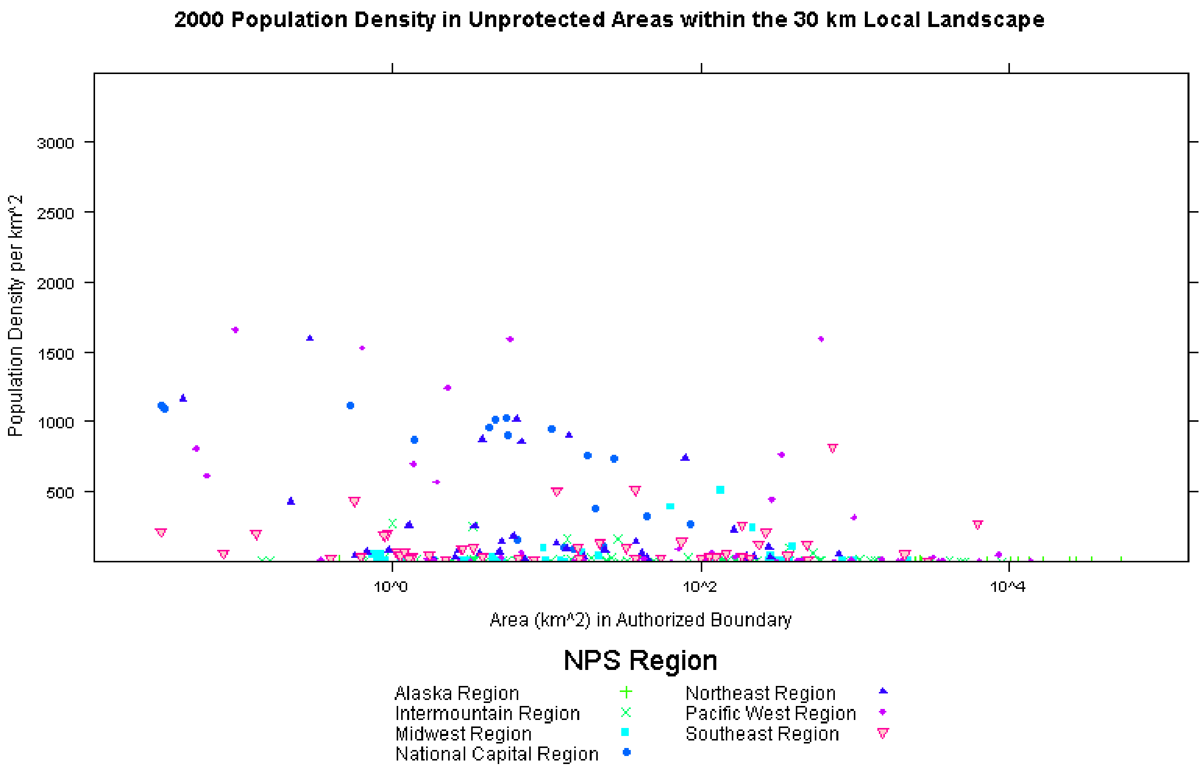
000 Population density per km^2 in the 30 km local area. Metric PDD; year=2000.

2000 Population Density in Unprotected Areas within the 30 km Local Landscape



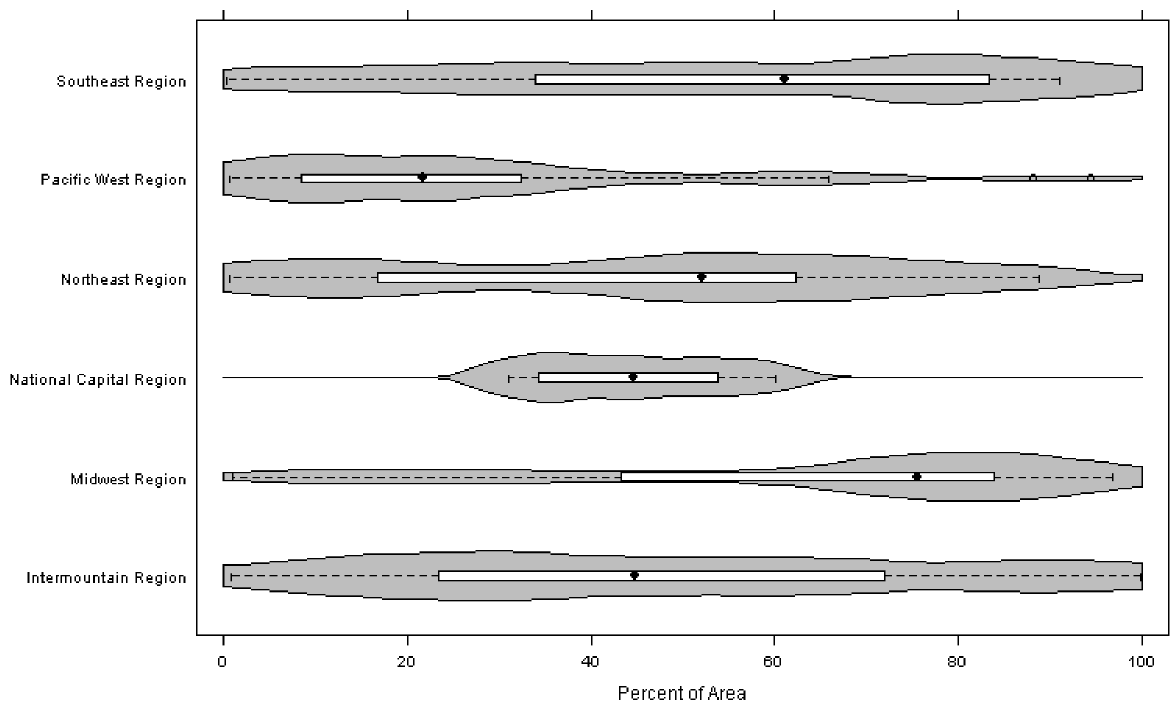
2000 Population Density in Unprotected Areas within the 30 km Local Landscape



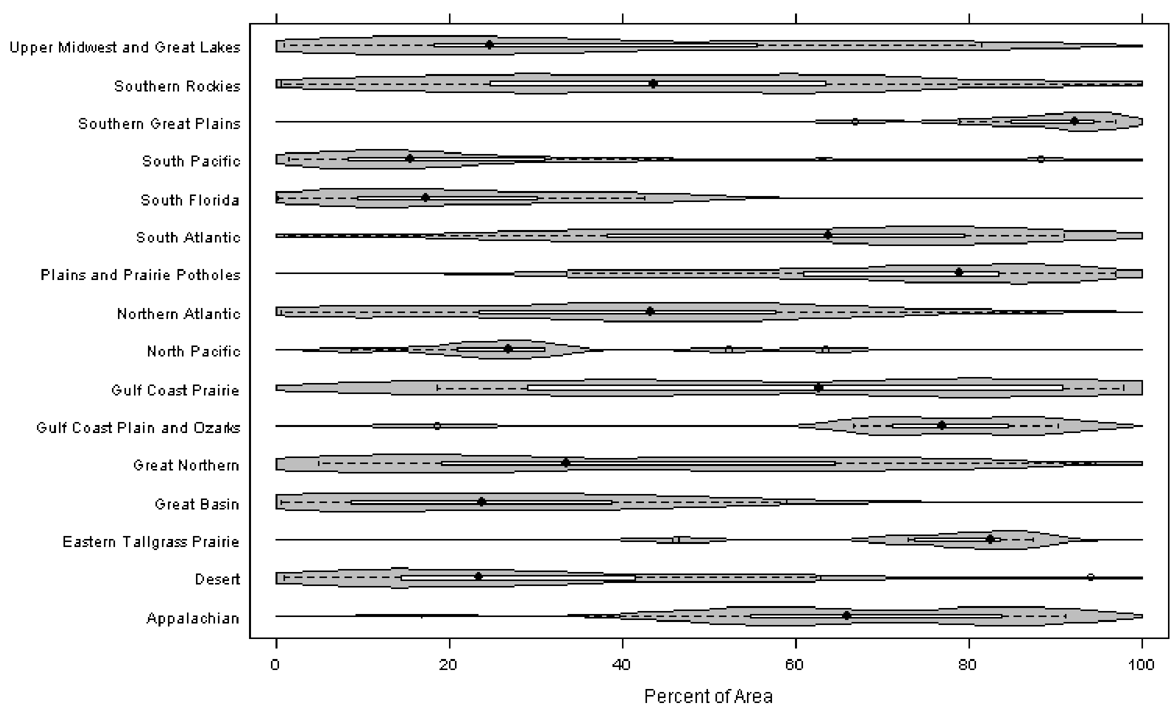


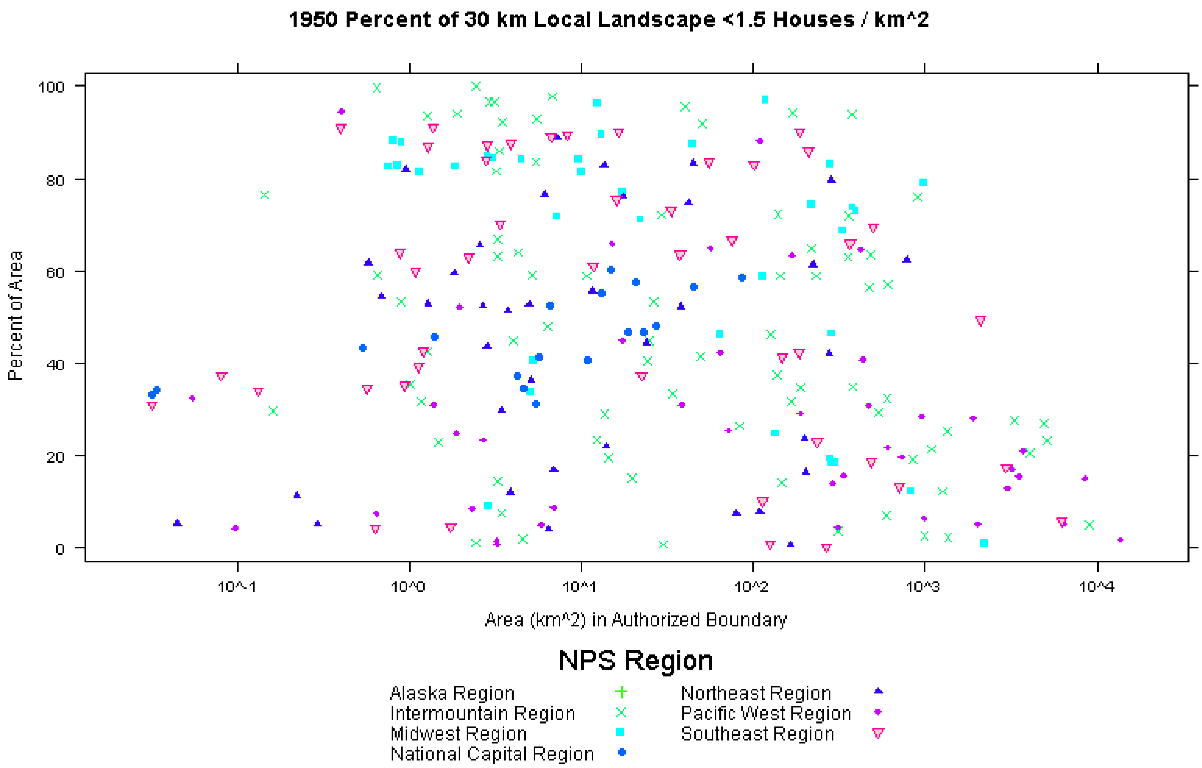
2000 Population density per km^2 in the 30 km local area, on log base 10 scale. Metric PDD; year=2000.

1950 Percent of 30 km Local Landscape <1.5 Houses / km^2



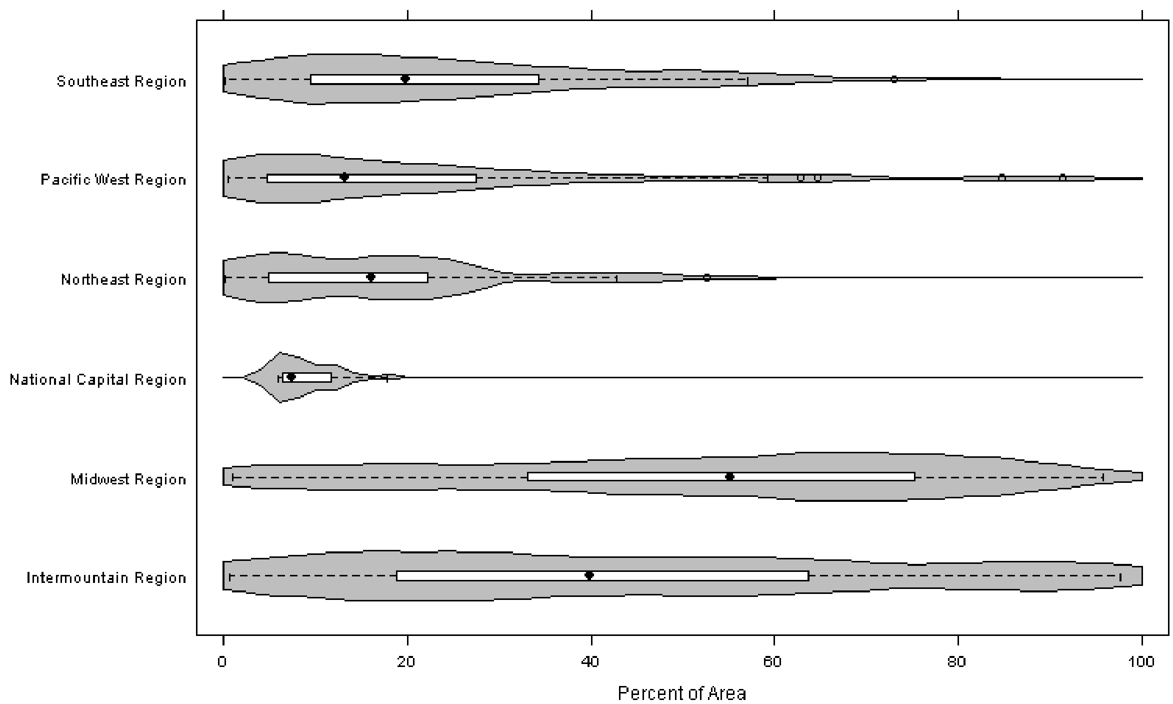
1950 Percent of 30 km Local Landscape <1.5 Houses / km^2



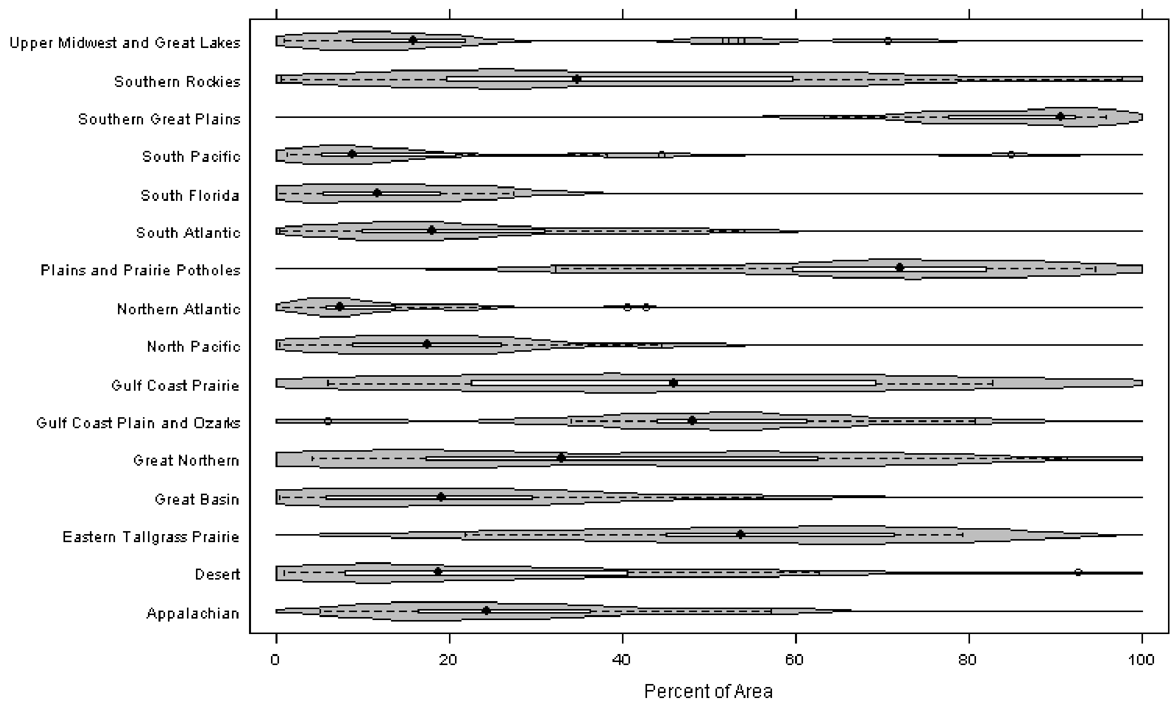


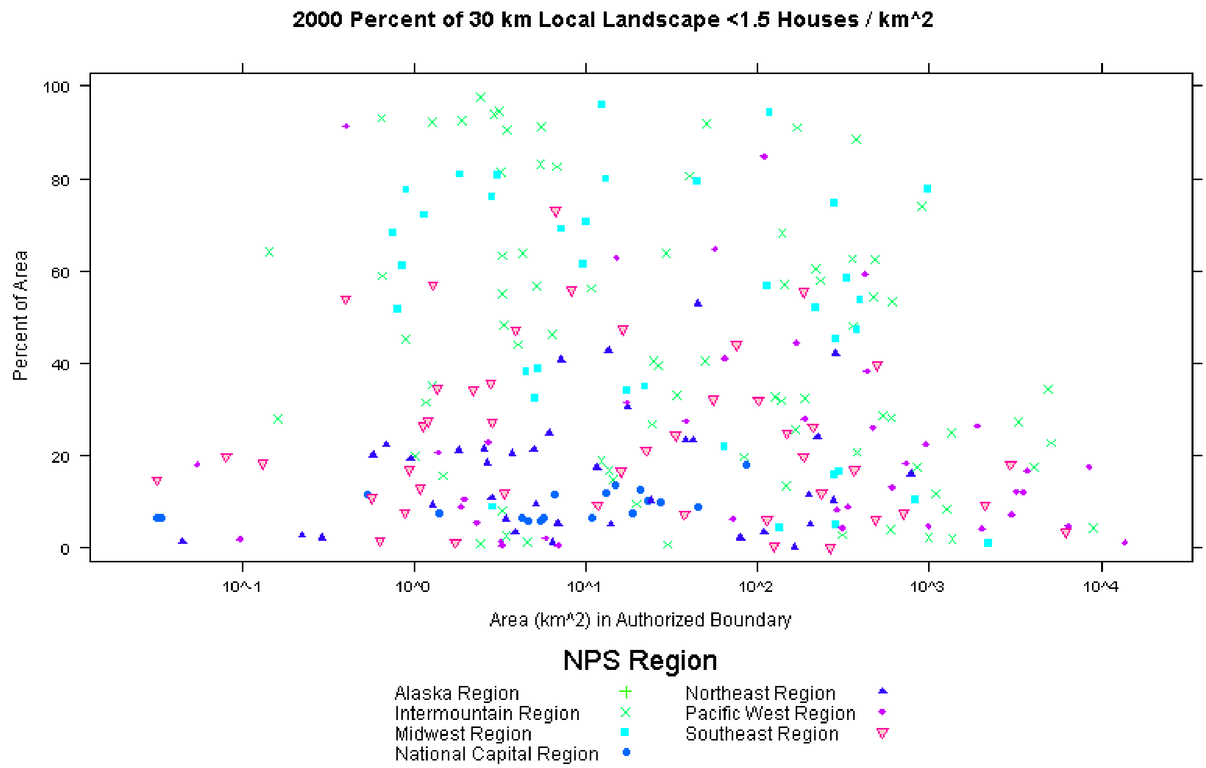
1950 Percent of the developable local 30 km area with housing density less than 1.5 units per km^2 . Metric HDD; year=1950, value = 1 or 2.

2000 Percent of 30 km Local Landscape <1.5 Houses / km^2

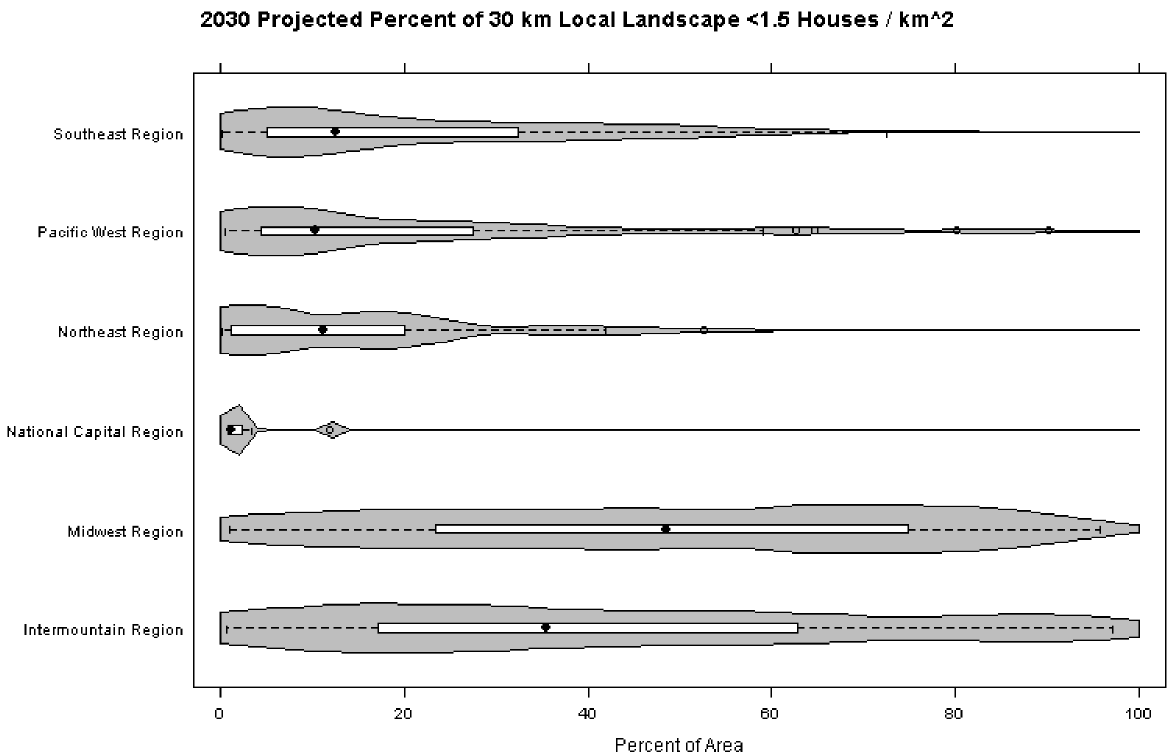


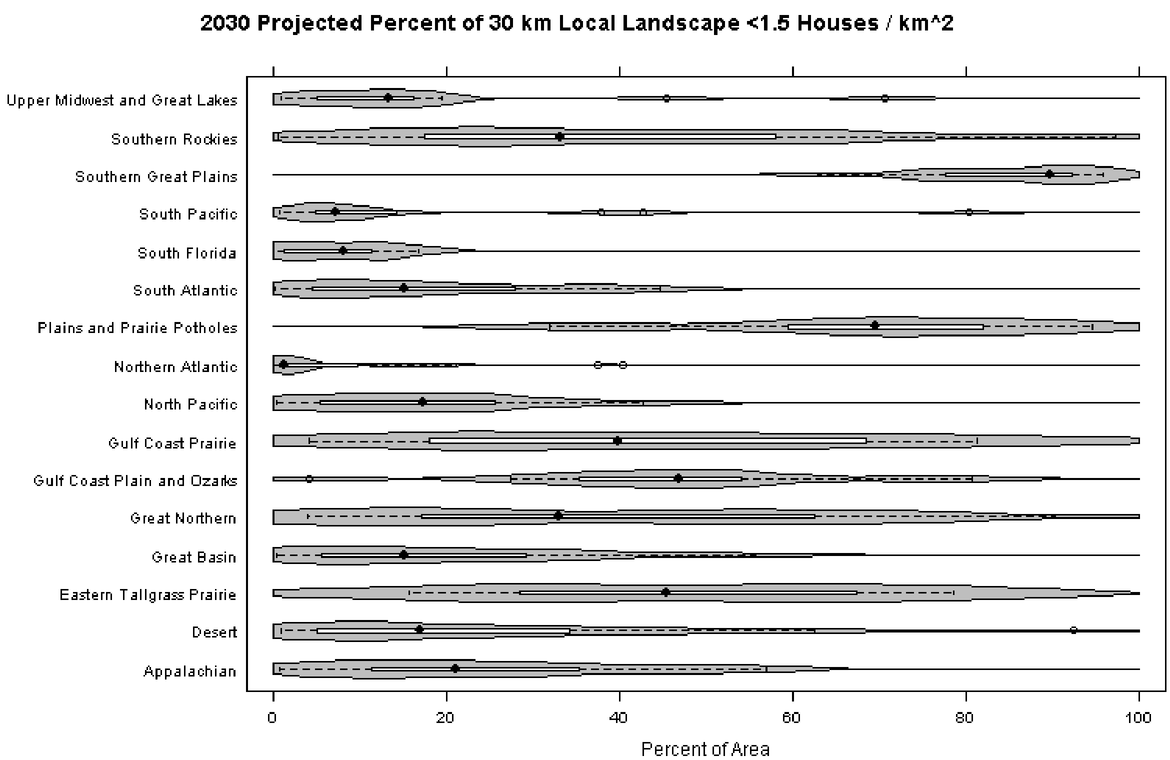
2000 Percent of 30 km Local Landscape <1.5 Houses / km^2

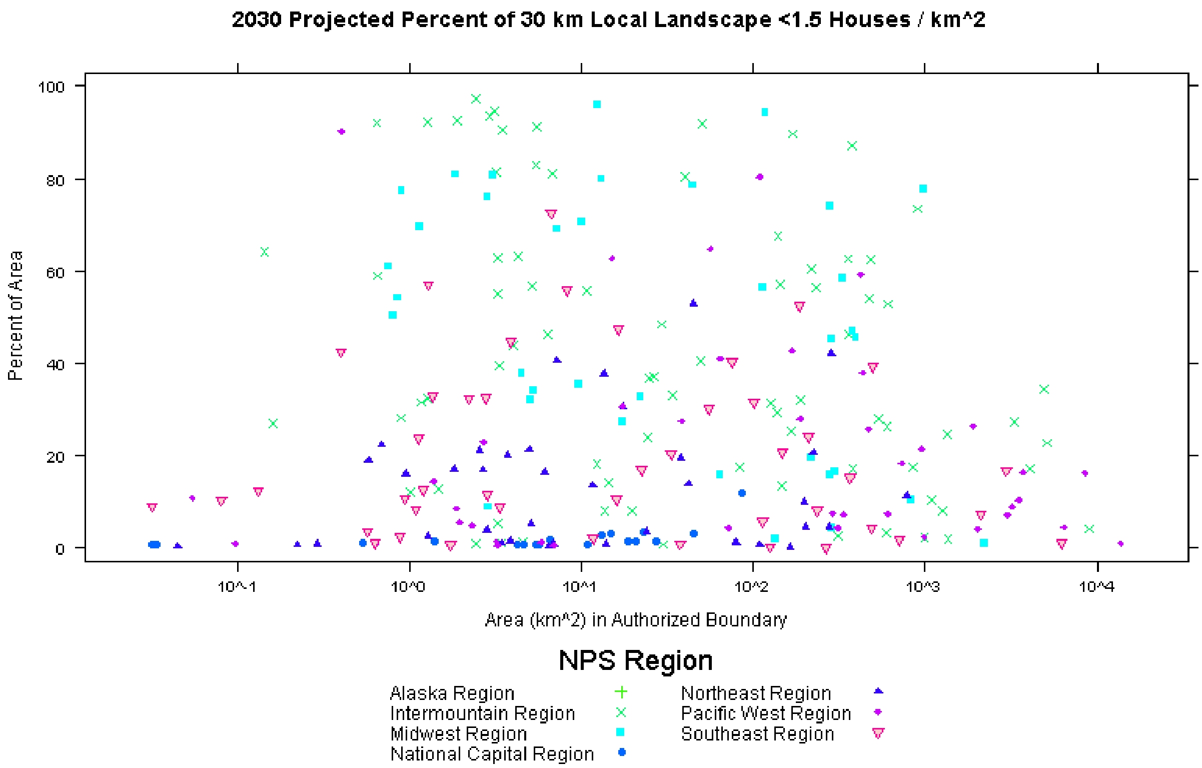




2000 Percent of the developable local 30 km area with housing density less than 1.5 units per km^2 . Metric HDD; year=2000, value = 1 or 2.

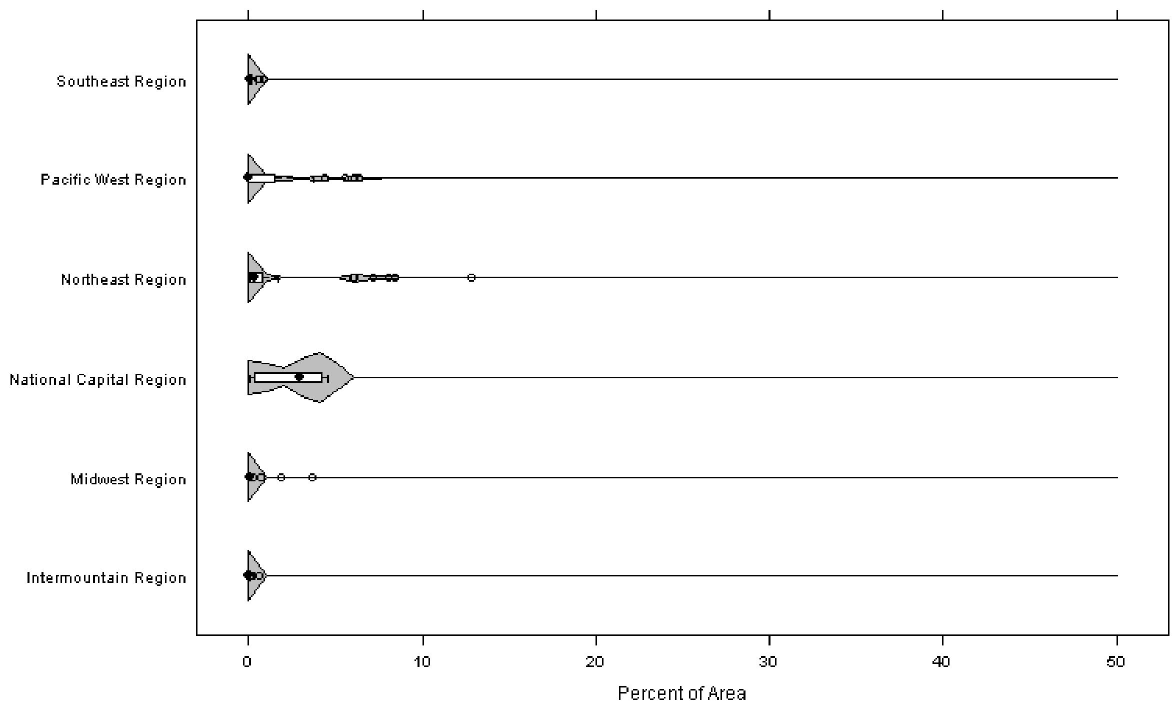

2030 Projected Percent of 30 km Local Landscape <1.5 Houses / km^2
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
0
20
40
60
80
100
Percent of Area


2030 Projected Percent of 30 km Local Landscape <1.5 Houses / km^2
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Appalachian
0
20
40
60
80
100
Percent of Area

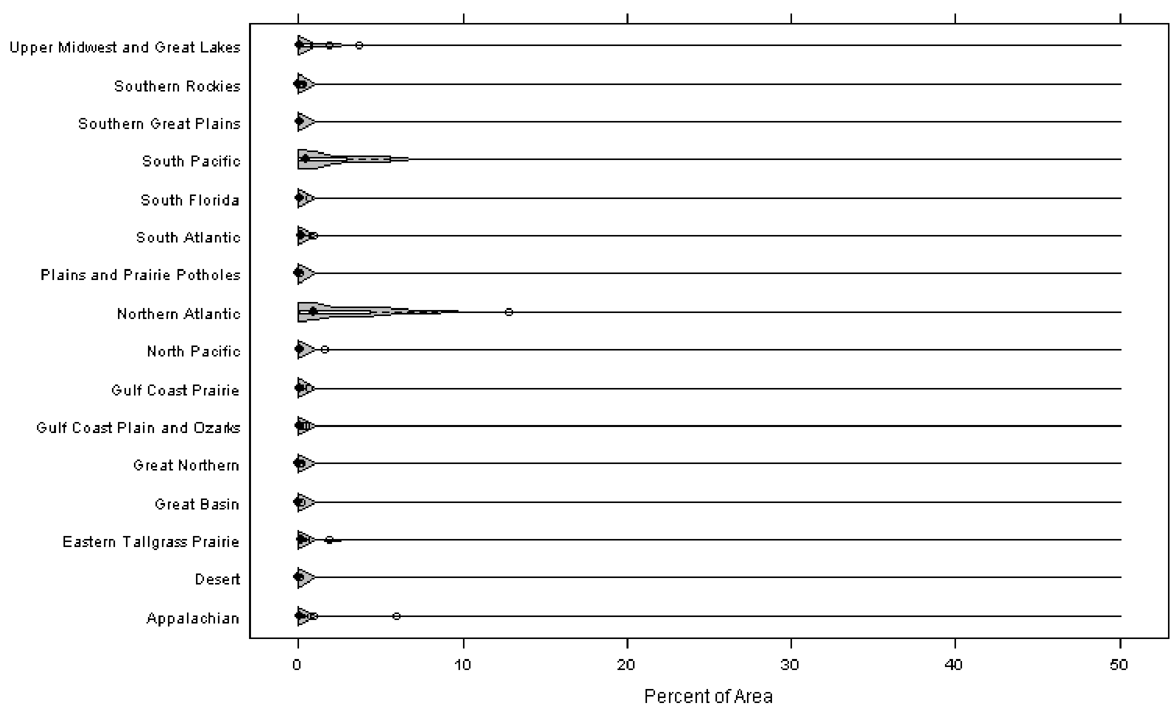


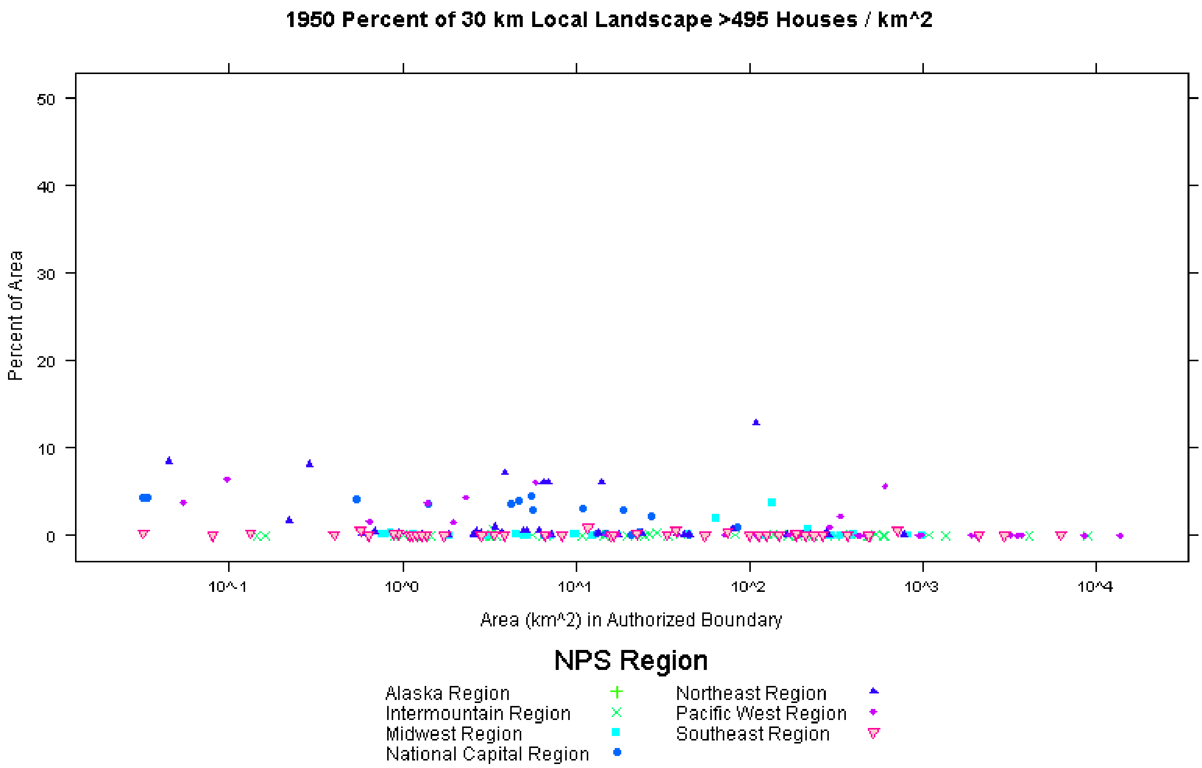
2030 Percent of the developable local 30 km area projected to have housing density less than 1.5 units per km^2 . Metric HDD; year=2030, value = 1 or 2.

1950 Percent of 30 km Local Landscape >495 Houses / km^2

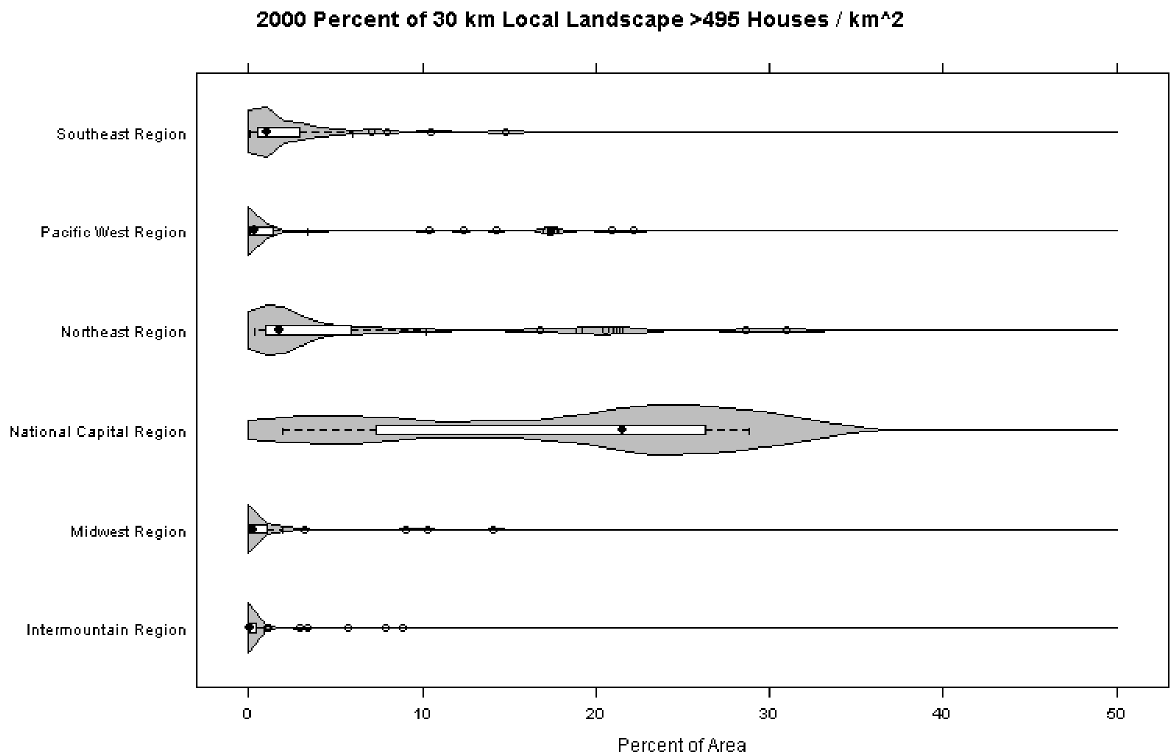


1950 Percent of 30 km Local Landscape >495 Houses / km^2

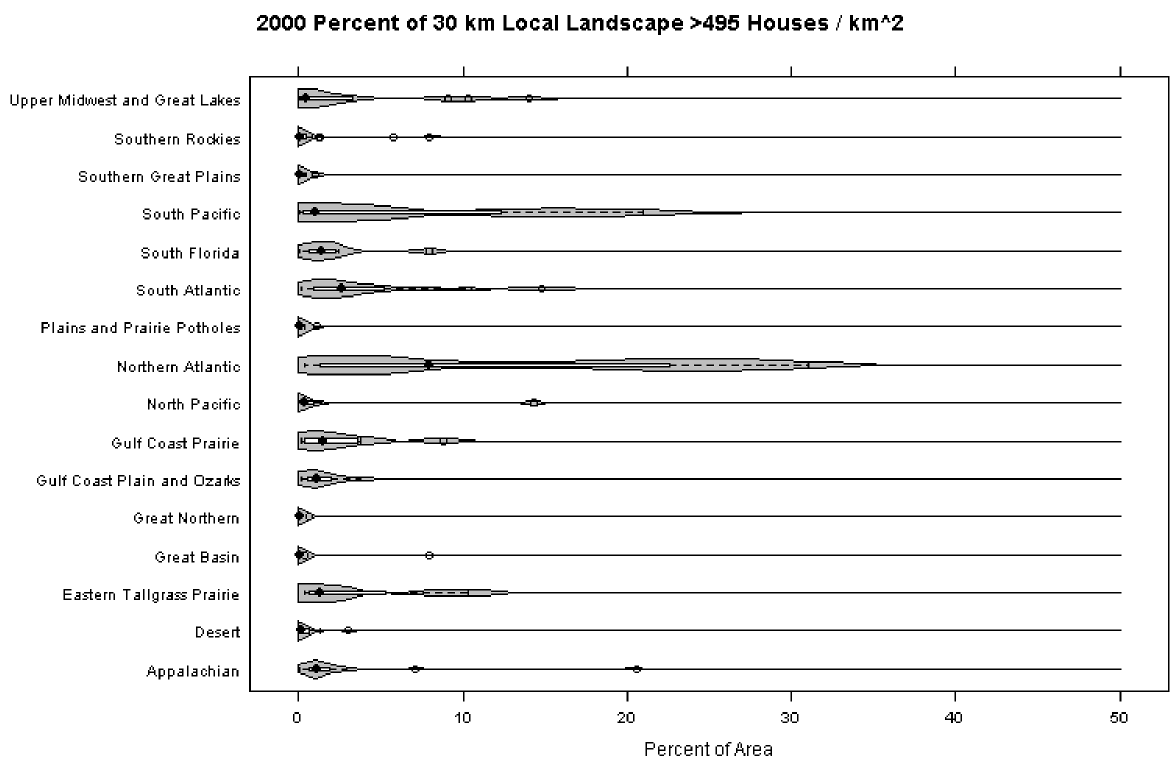




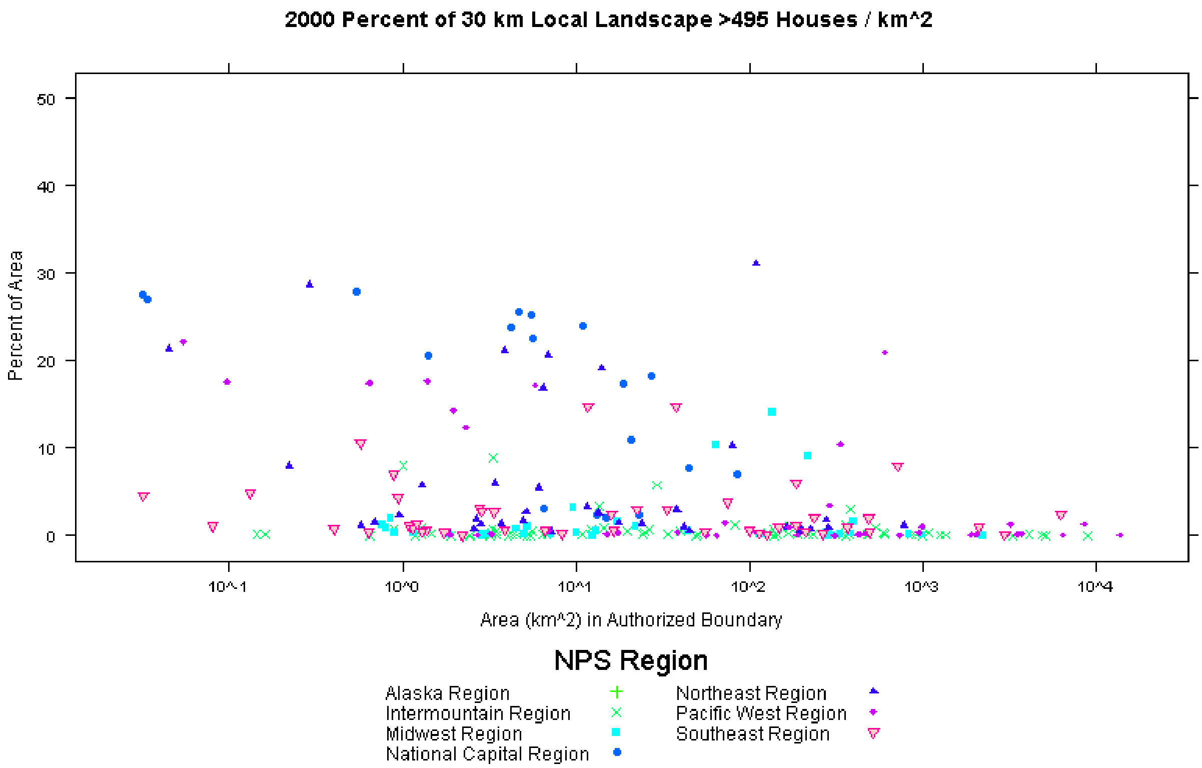
1950 Percent of the developable local 30 km area with housing density greater than or equal to 495 units per km^2. Metric HDD; year=1950, value = 9 or 10.



2000 Percent of 30 km Local Landscape >495 Houses / km^2
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
0
10
20
30
40
50
Percent of Area

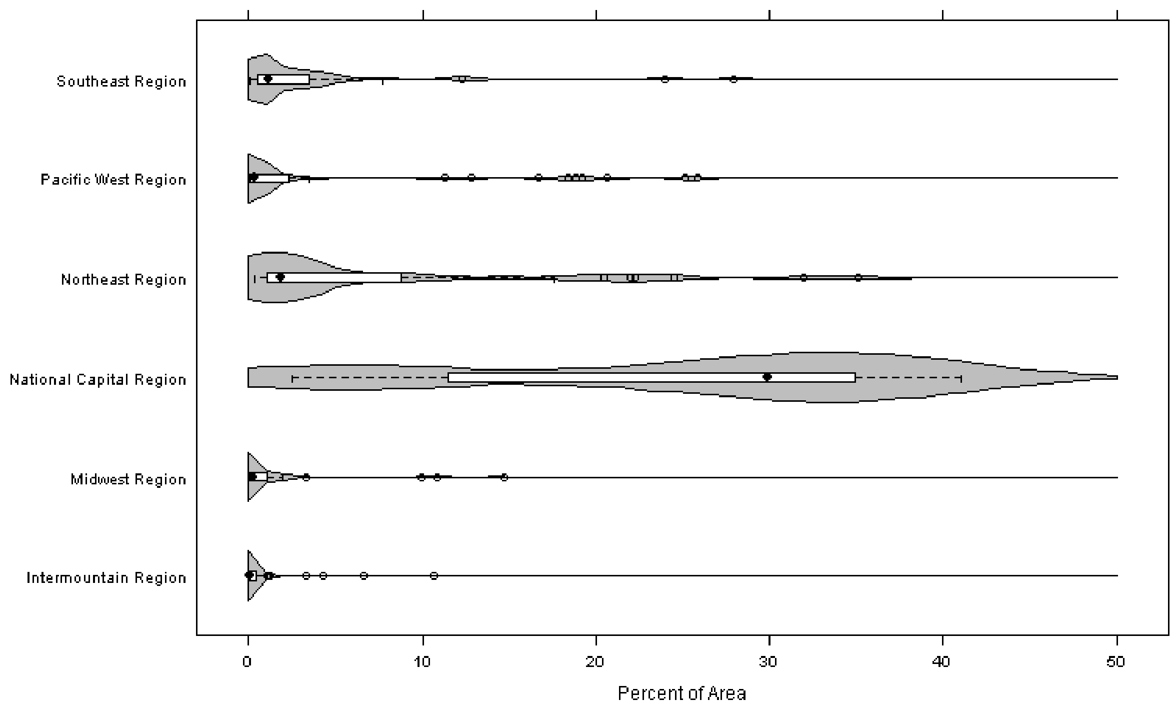


2000 Percent of 30 km Local Landscape >495 Houses / km^2
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Appalachian
0
10
20
30
40
50
Percent of Area

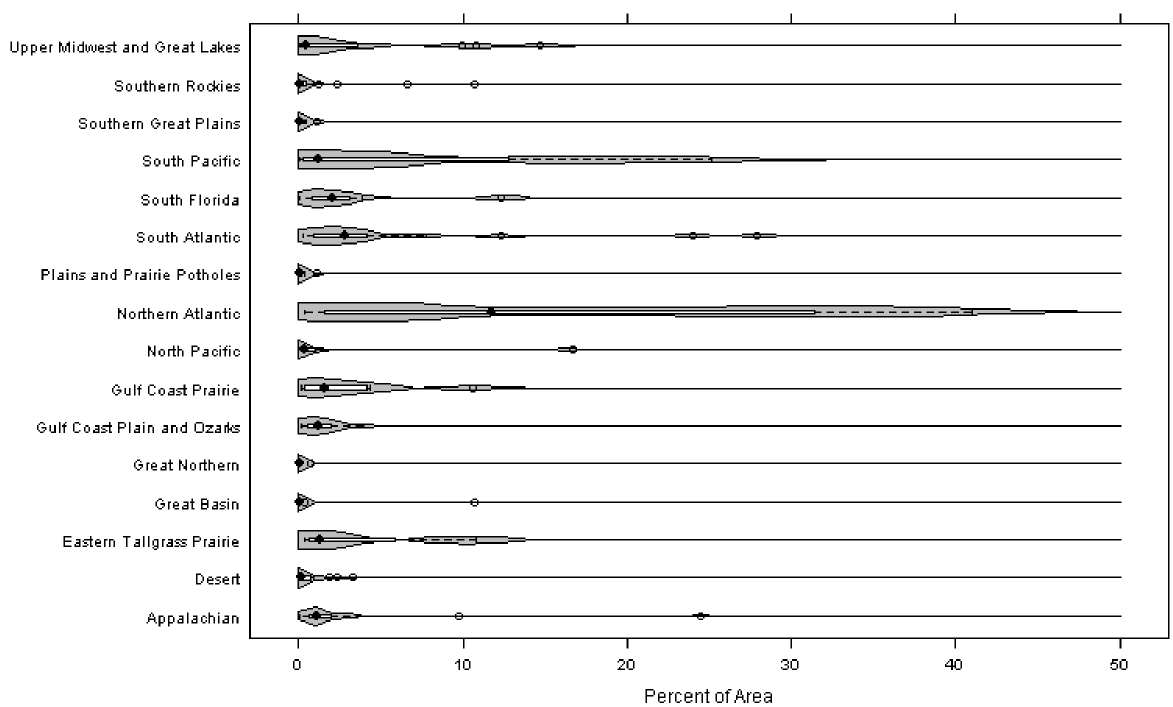


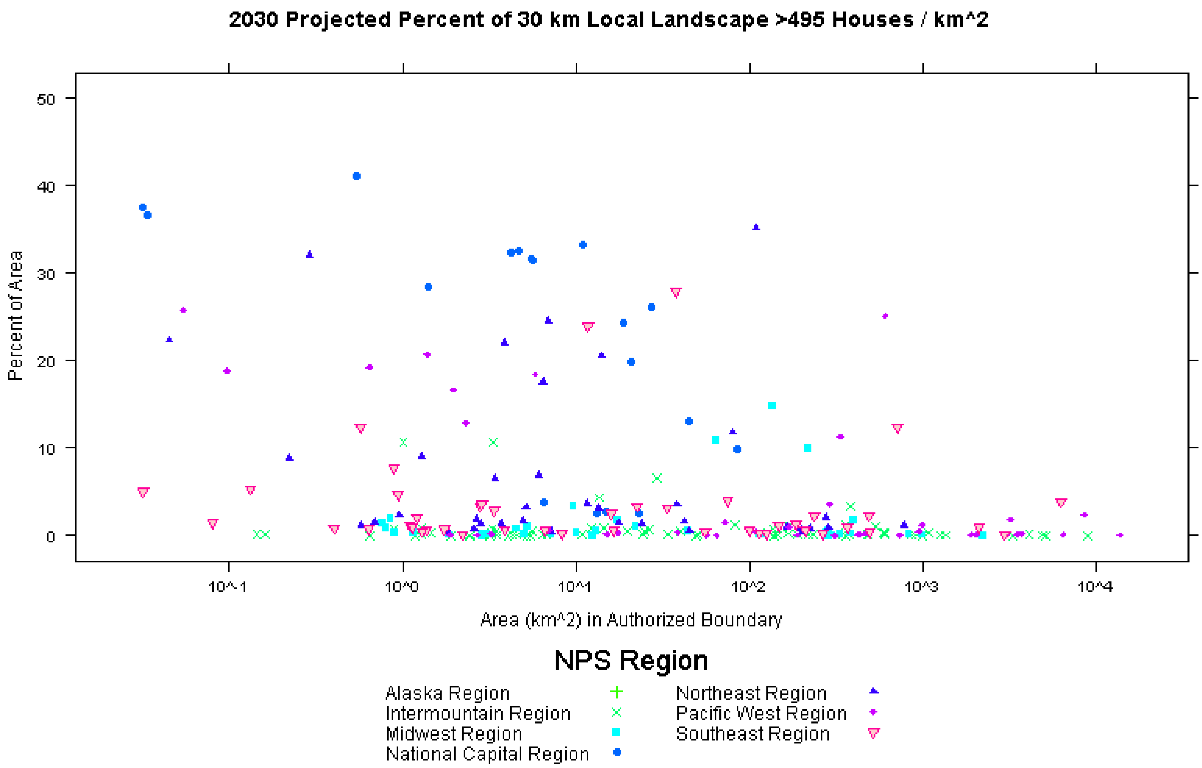
2000 Percent of the developable local 30 km area with housing density greater than or equal to 495 units per km^2. Metric HDD; year=2000, value = 9 or 10.

2030 Projected Percent of 30 km Local Landscape >495 Houses / km^2

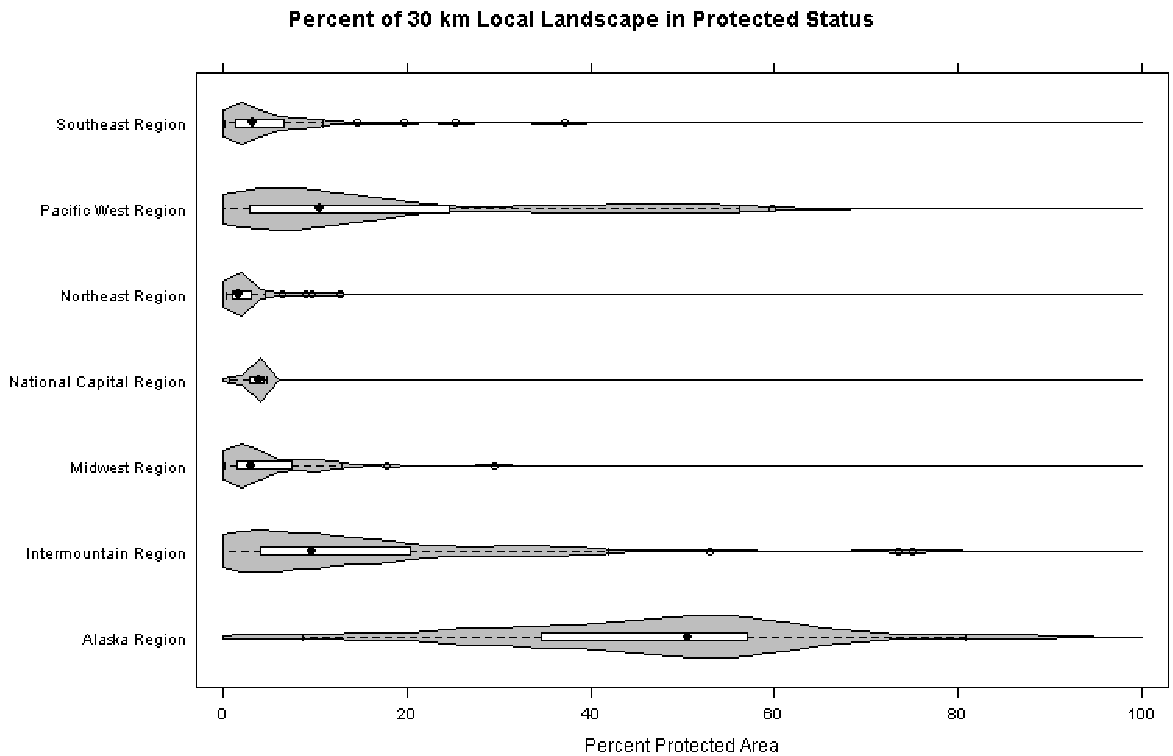


2030 Projected Percent of 30 km Local Landscape >495 Houses / km^2

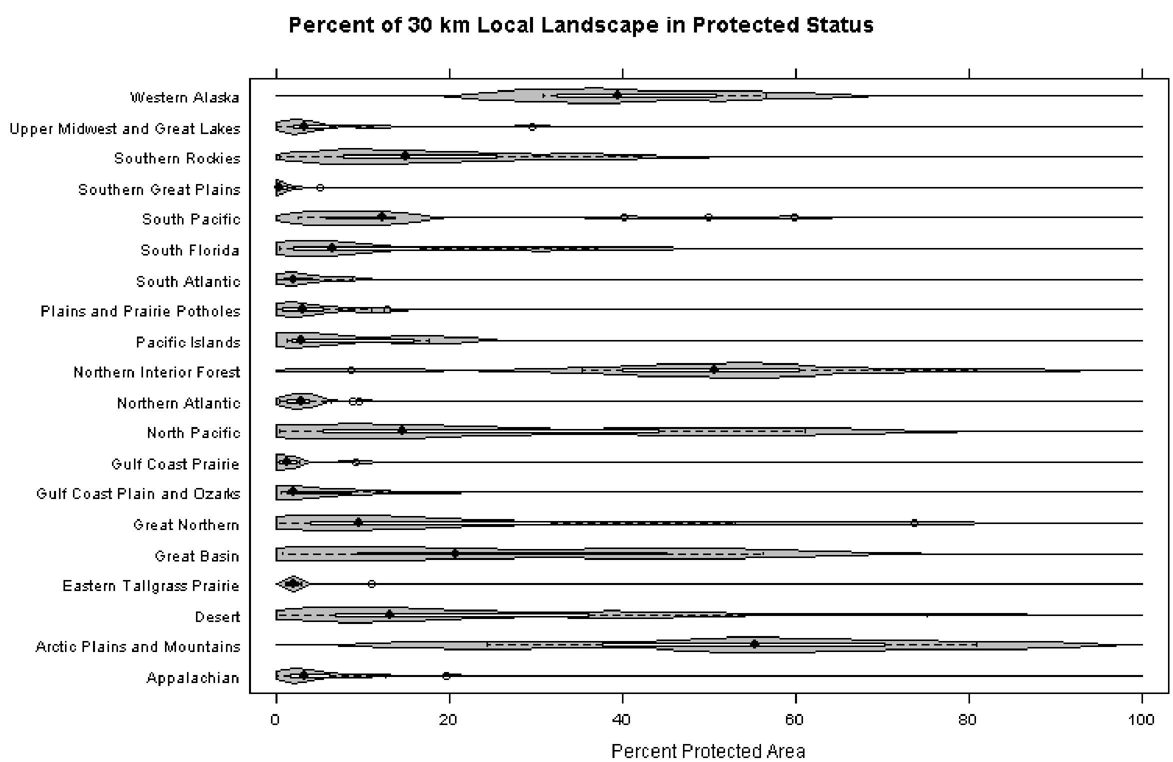




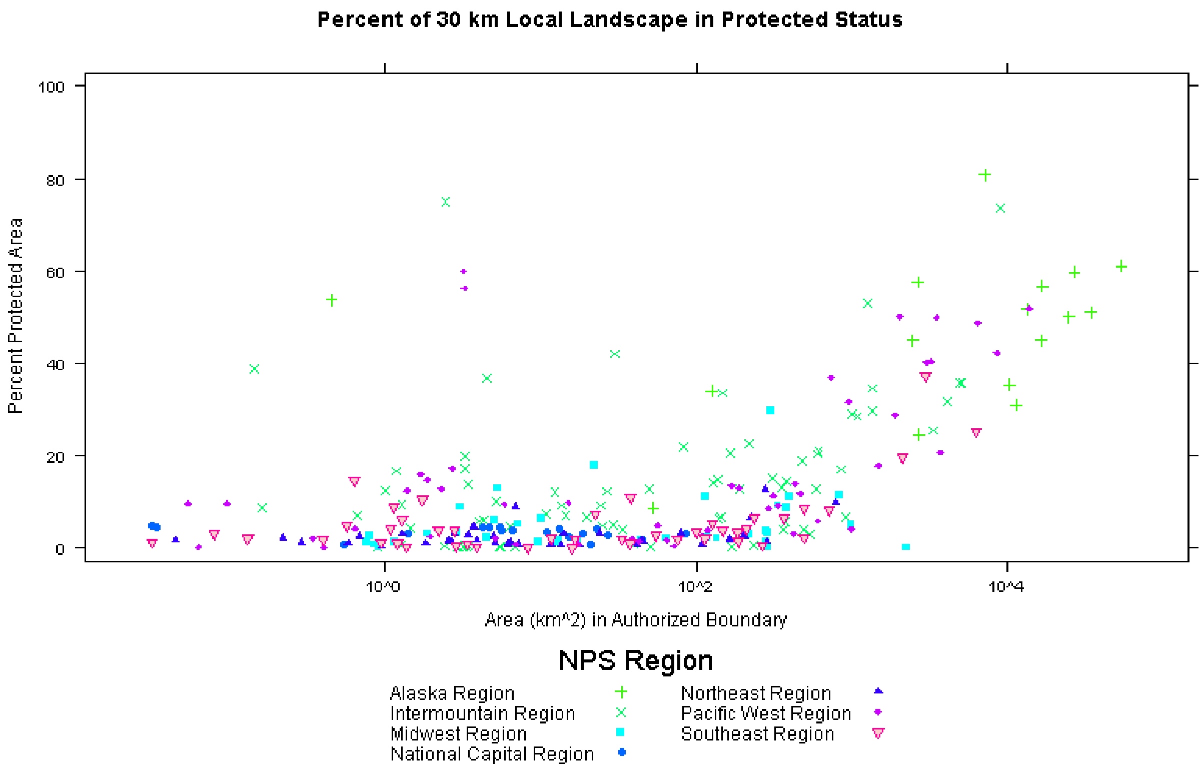
2030 Percent of the developable local 30 km area projected to have housing density greater than or equal to 495 units per km^2. Metric HDD; year=2030, value = 9 or 10.



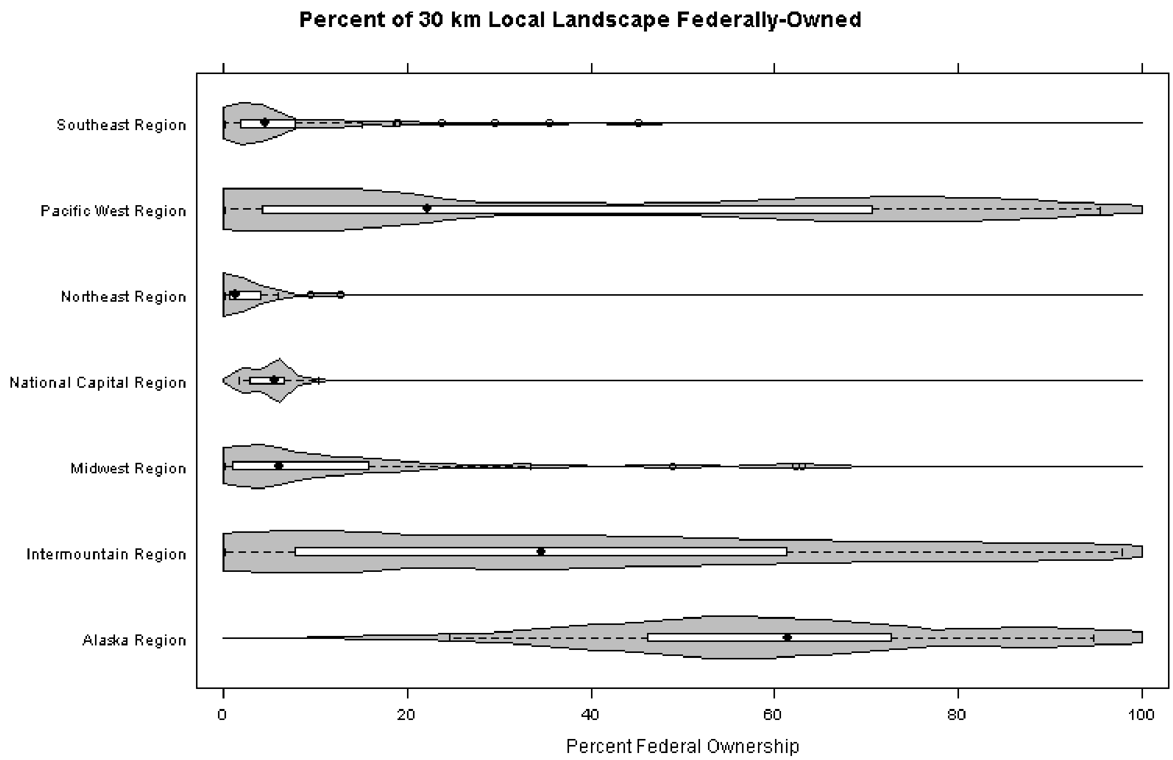
Percent of 30 km Local Landscape in Protected Status
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0
20
40
60
80
100
Percent Protected Area

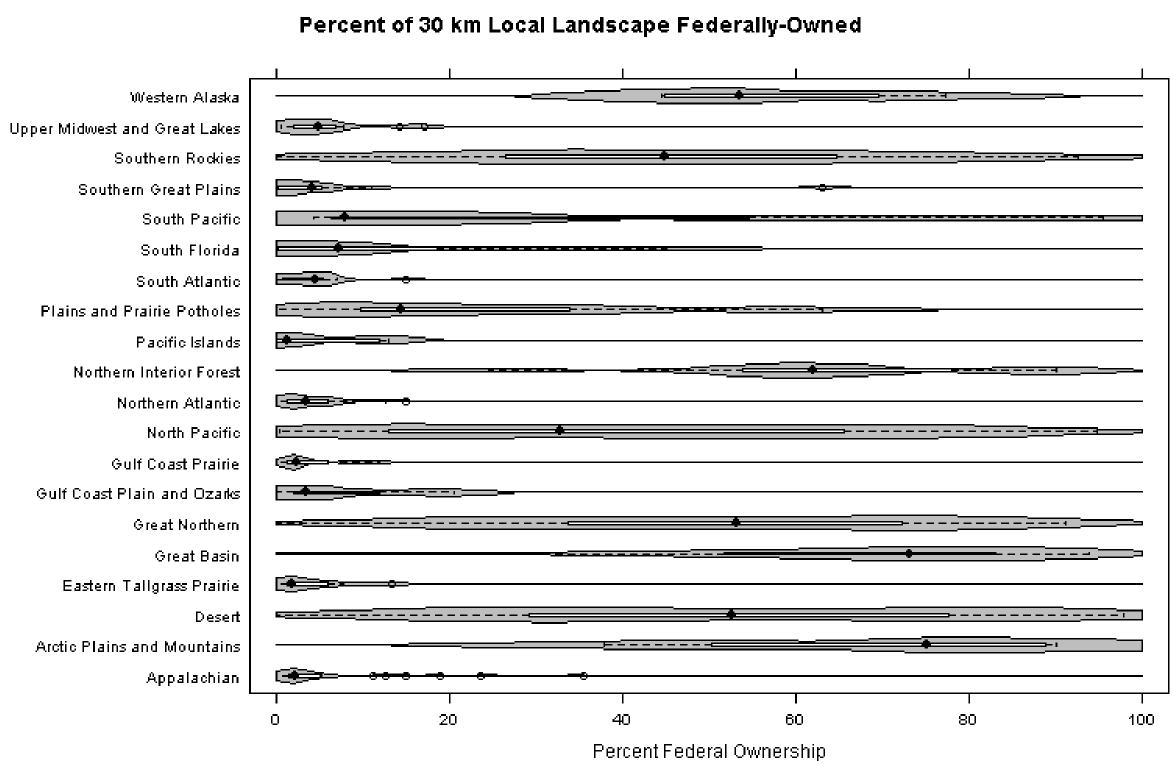


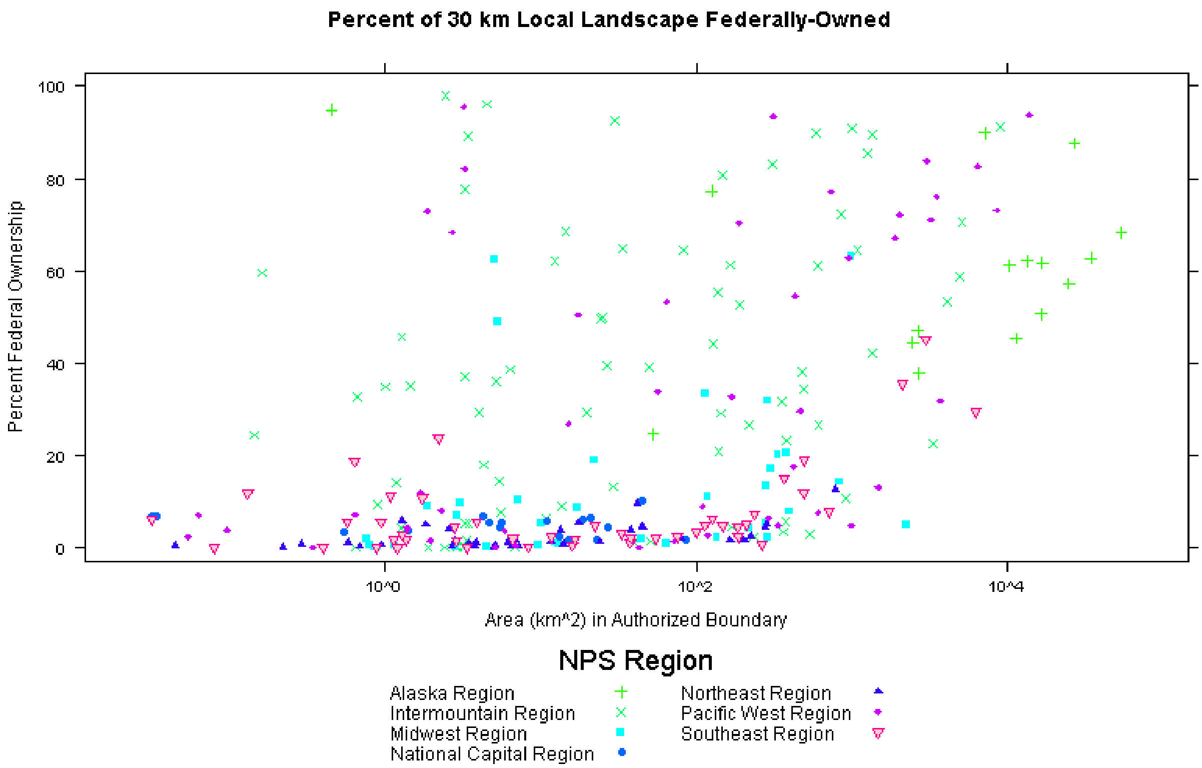
Percent of 30 km Local Landscape in Protected Status
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0
20
40
60
80
100
Percent Protected Area



Percent of the 30 km local area in protected status. Metric CAP.


Percent of 30 km Local Landscape Federally-Owned
Southeast Region
Pacific West Region
Northeast Region
National Capital Region
Midwest Region
Intermountain Region
Alaska Region
0
20
40
60
80
100
Percent Federal Ownership


Percent of 30 km Local Landscape Federally-Owned
Western Alaska
Upper Midwest and Great Lakes
Southern Rockies
Southern Great Plains
South Pacific
South Florida
South Atlantic
Plains and Prairie Potholes
Pacific Islands
Northern Interior Forest
Northern Atlantic
North Pacific
Gulf Coast Prairie
Gulf Coast Plain and Ozarks
Great Northern
Great Basin
Eastern Tallgrass Prairie
Desert
Arctic Plains and Mountains
Appalachian
0
20
40
60
80
100
Percent Federal Ownership



Percent of the 30 km local area owned by the US Federal government. Metric CAC; Owner_Type=Federal.

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS XXXXXX, December 2009

National Park Service
U.S. Department of the Interior

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