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Forest Fragmentation of the Conterminous United States: Assessing Forest Intactness through Road Density and Spatial Characteristics

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Over the past few centuries, widespread disturbance of native forests of the conterminous United States has dramatically altered the composition, structure, extent, and spatial pattern of forestlands (Curtis 1956, Whitney 1994). These forests have been either permanently replaced by other land uses or degraded to varying degrees by unsustainable forestry practices, forest fragmentation, exotic species introduction, or alteration of natural disturbance regimes.

Habitat fragmentation is generally defined as the process of subdividing a continuous habitat type into smaller patches, which results in the loss of original habitat, reduction in patch size, and increasing isolation of patches (Andrén 1994). Habitat fragmentation is considered to be one of the single most important factors leading to loss of native species (especially in forested landscapes) and one of the primary causes of the present extinction crisis (Wilcox and Murphy 1985). Although it is true that natural disturbances such as fire and disease fragment native forests, human activities are by far the most extensive agents of forest fragmentation (Burgess and Sharpe 1981). For example, during a 20-year period in the Klamath–Siskiyou ecoregion, fire was responsible for 6% of forest loss, while clear-cut logging was responsible for 94% (Staus et al. 2001). Depending on the severity of the fragmentation process and sensitivity of the ecosystems affected, native plants, animals, and many natural ecosystem processes (e.g., nutrient cycling, pollination, predator–prey interactions, and natural disturbance regimes) are compromised or fundamentally altered. For many species, migration between suitable habitat patches becomes more difficult, leading to smaller population sizes, decreased gene flow, and possible local extinctions (Wilcove 1987, Vermeulen 1993).

FOREST FRAGMENTATION CAN BE MEASURED AND MONITORED IN A POWERFUL NEW WAY BY COMBINING REMOTE SENSING, GEOGRAPHIC INFORMATION SYSTEMS, AND ANALYTICAL SOFTWARE

As native forests become increasingly fragmented, ecosystem dynamics switch from being predominantly internally driven to being predominantly externally driven (Saunders et al. 1991). Simultaneously, remnant patches become altered by changes within the patches themselves (Chen et al. 1995, Woodroffe and Ginsberg 1998) as the remnants become more and more isolated, thereby resulting in further ecological degradation across the landscape. Declines in forest species as a result of fragmentation have been documented for numerous taxa, including neotropical migrant songbirds

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(Whitcomb et al. 1981, Ambuel and Temple 1983), small mammals (Henderson et al. 1985, Verboom and Apeldoorn 1990), and invertebrates (Mader 1984). Forest fragmentation has also been associated with increased susceptibility to exotic invasion (Rejmánek 1989).

Concern over the widespread negative effects of fragmentation has led to calls for managing ecosystems at a regional scale (Franklin 1993), and it has led researchers to examine spatial patterns over large geographic extents (O'Neill et al. 1997, Jones et al. 1997, Riitters et al. 2000b). Quantitative methods have been developed to compare different landscapes, to identify landscape changes over time, and to correlate landscape pattern to ecological function (Turner 1989). Many indices can be calculated from the spatial patterning of land cover (Urban et al. 1987, Turner 1989, McGarigal and Marks 1995, Schumaker 1996), forming one of the major analytical pursuits of landscape ecology (Forman and Godron 1986).

Krummel and colleagues (1987) and O'Neill and colleagues (1988) examined landscape patterns based on high-altitude aerial photography and US Geological Survey (USGS) quadrangles (at 1:250,000 scale). The indices they chose to examine, which were found to be reasonably independent of one another, captured major features of landscape pattern. More recent assessments utilized a "sliding window" filter to reduce the complexity of the data and to draw out landscape patterns of interest (Jones et al. 1997, Riitters et al. 2000a, 2000b). Most landscape assessments have relied on land cover databases developed from coarse AVHRR (Advanced Very High Resolution Radiometer) satellite imagery (O'Neill et al. 1996, 1997, Loveland et al. 2000, Riitters et al. 2000a, 2000b). Although such assessments remain useful at continental scales, analysis of finer resolution imagery has been recommended when studying smaller geographic areas (O'Neill et al. 1997). Using classified Landsat Thematic Mapper (TM) imagery from National Land Cover Data (NLCD; Vogelmann et al. 1998), researchers have begun to examine spatial pattern at finer resolutions. Jones and colleagues (1997) examined numerous landscape indicators using the data set from NLCD for the mid-Atlantic states, with the primary research focus being water quality, and Riitters and colleagues (1997) employed multiple window sizes to examine landscape patterns of subwatersheds using the data set from NLCD for the Chesapeake Bay watershed.

Our objective was to build a forest fragmentation database for the conterminous United States by utilizing the high-resolution NLCD database, roads, and a series of fragmentation indices that quantify forest landscape patterns. Because of the numerous negative impacts that roads have on native forest ecosystems (Trombulak and Frissell 2000), roads data played a prominent role in the fragmentation assessment. We focused our analysis on forest ecoregions, as defined by the World Wildlife Fund (Ricketts et al. 1999), but we also summarized results at larger regional and national scales. Ecoregions can be defined as relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land use change (Olson et al. 2001). Because of the scope of the project and the lack of complete, uniform data sets, we conducted the analysis without consideration for ownership, forest type, stand age, forest health, or type of disturbance. In this article, after describing the assessment of forest fragmentation, we review the methodology that created this database and some of its potential uses for conservation scientists, restoration scientists, land managers, policymakers, and others. We then offer a review of the strengths and limitations of the database and make recommendations for future modification and research.

Analyzing and mapping forest fragmentation

We used six basic geographic information systems (GIS) data layers from five separate sources: (1) national land cover data based on 30 meter (m) resolution Landsat 5 TM satellite imagery (Vogelmann et al. 1998), (2) USGS 1:100,000 scale roads, (3) US Census Bureau Topologically Integrated Geographic Encoding and Referencing (TIGER) 1:100,000 scale highways and US boundaries, (4) Bureau of Transportation Statistics (BTS) 1:100,000 scale boundaries for urbanized areas with a population of greater than 50,000, and (5) World Wildlife Fund ecoregions (Ricketts et al. 1999). We used the TIGER roads and BTS urban boundaries to define our units of analysis and the data set from NLCD and USGS roads data for the fragmentation analysis.

Choosing the unit of analysis. In general, the better the ecological subdivision of a region, the more sensitive and interpretable any landscape pattern index will be (O'Neill et al. 1996). Of the few ecological assessments that have analyzed large regions, most employed the watershed as the basic unit of study (Jones et al. 1997), which may be a reasonable subdivision for some ecological research questions, particularly regarding effects of land use on aquatic ecosystems. For regional assessments of forest spatial pattern and fragmentation, however, dissecting the landscape by watersheds can be considered to artificially sever intact forest patches and alter analytical results. For example, many forest organisms have no difficulty moving from one watershed to another within the same forest patch, in effect treating watershed boundaries as highly permeable. Roads, however, have been shown to be a significant barrier to movement for many forest organisms. Units of study should be defined according to a significant source of forest fragmentation, such as major roads and highways (Trombulak and Frissell 2000). For example, Anderson and colleagues (1999) used an analytical unit they termed an "ecoblock," which was defined by paved and unpaved roads, railroads, power lines, and bodies of water.

We defined our units of analysis, termed *land units,* using the TIGER highway data (US interstates, US routes, and state and county highways) and the borders of the conterminous United States. We used TIGER highway data instead of USGS highway data to delineate land units, because TIGER data on highways were more complete and up-to-date. Only those areas that were at least 2000 hectares (ha) were included as land units. We decided on 2000 ha after exploring a number of size limits, because this size reduced the amount of land units to a manageable number, yet was sufficiently small in comparison with the average land unit size. We used BTS data to identify and remove urban areas from the analysis, assuming that the amount of intact forest would be minimal in those areas. A final land units GIS data layer was created to which fragmentation analysis results could be linked.

Assessing fragmentation. For the purposes of calculating fragmentation statistics, we combined the 21 potential NLCD classes into two classes: forest (including woody wetlands) and nonforest (including water). Only portions of the largest interstates were delineated in the NLCD data set. Thus, to account for the fragmenting effect of roads, we superimposed a 30 m resolution raster version of the USGS roads data set onto the NLCD forest–nonforest data set. We used the USGS roads data, because this data set presented smaller roads in more detail than did the TIGER roads data set. All forest and nonforest patches smaller than 1 ha were reclassified to match the surrounding land cover type to decrease the number of very small patches and thus the time required for processing data. The resulting land units were at least 2000 ha, did not include urban areas, and contained both forest and nonforest patches that were at least 1 ha in size.

Because highways defined the land units, land unit boundaries did not match up directly with the ecoregion boundaries. In every case, the outermost land unit boundaries extended outside the ecoregion. For most ecoregions, the land unit area was a fairly close approximation of the ecoregion area (see figure 4b). For five ecoregions made up of smaller forest ecoregions surrounded by large nonforest ecoregions, we matched the land unit boundaries to the ecoregion boundaries to avoid skewing the fragmentation results by including large areas of nonforest habitat. These "island" ecoregions (figure 1) were the Great Basin montane forests, Wasatch and Uinta montane forests, Colorado Rockies forests, Arizona Mountains forests, and Madrean Sky Islands montane forests.

We conducted spatial analyses for the conterminous US portion of 39 forest ecoregions, as defined by the World Wildlife Fund (figure 1; Ricketts et al. 1999), 21 in the East and 18 in the West. To quantify landscape patterns, we calculated 33 class-level and 39 landscape-level metrics (or indices) using FRAGSTATS, a software program for analyzing spatial patterns (McGarigal and Marks 1995). Additionally, we calculated road density directly from the 1:100,000 scale USGS roads data set, which included all size classes of roads except for four-wheel drive roads. Results for the 72 indices were then spatially linked back to the land units GIS database. (See box 1 for a list of the attributes associated with each land unit.) Because of the lack of compatible, nationwide data sets for natural fragmentation, such as fire, windthrow, or flooding, we did not attempt to distinguish natural and anthropogenic fragmentation within the land units.

Interpretation of fragmentation results. This GIS data set was designed to help address a wide range of ecological inquiries pertaining to forest fragmentation. As an example, we provide one possible interpretation of the results by combining 5 of the 72 indices using an unweighted additive scoring method. The indices used included road density (kilometers per kilometers squared [km/km^2]); total core area index (percentage of all forest area within a land unit that is considered core area, based on a 90 m edge buffer distance); mean nearest neighbor (the average distance in meters from one forest patch to the nearest forest patch); class area (total amount of forest in hectares within each land unit); and percentage of landscape (percentage of a land unit that is composed of forest). We calculated these five indices for each land unit and aggregated the results by ecoregion using natural breaks. This method, natural breaks, uses the Jenks's optimization method, which identifies breakpoints that minimize the sum of variance within each class and maximize the variance between classes (Jenks and Caspall 1971). In this case, each land unit received a score for each of the five indices, ranging from 1 (highest fragmentation outcome) to 5 (lowest fragmentation outcome). The individual scores were then combined into one composite score for each land unit, ranging from 5 (highest possible level of fragmentation) to 25 (lowest possible level of fragmentation).

Forest fragmentation of the conterminous United States

A total of 19,953 land units (18,659 in the East and 1294 in the West) were delineated, which covered approximately 3.6 million km^2 (2.5 million km^2 in the East and 1.1 million km^2 in the West). The mean area of land units was 13,297 ha for eastern forest ecoregions and 86,851 ha for western forest ecoregions. The number of land units ranged from 9 in the North Cascade Forest (ecoregion 23) to 2777 in the Southern Great Lakes Forest (table 1; ecoregion 36). Slightly over 50% of the forest ecoregions were actually covered by forest, and approximately 33% of the ecoregions were covered by core (or interior) forest, with a 90 m edge buffer distance. The percentage of core area values ranged from 9.8 in the Southern Great Lakes Forest (ecoregion 36) to 68.1 in the Eastern Forest–Boreal Transition (ecoregion 14). The number of forest patches differed considerably between East and West, with nearly four times as many patches in the East as in the West. The mean forest patch size ranged from 21 ha in the Southern Great Lakes Forest (ecoregion 36) to 268 ha in the Central Pacific Coastal Forest (ecoregion 9). The mean forest patch size was approximately 92 ha in the West and 67 ha in the East.

The land unit database was constructed to give users a variety of quantified forest fragmentation results. Summaries could be made over a number of geographic extents, including country, region, biome, state, or ecoregion. For this study, we compiled results at the country (conterminous United States), region (East versus West), and ecoregion levels and included them as separate files in the database. Fragmentation

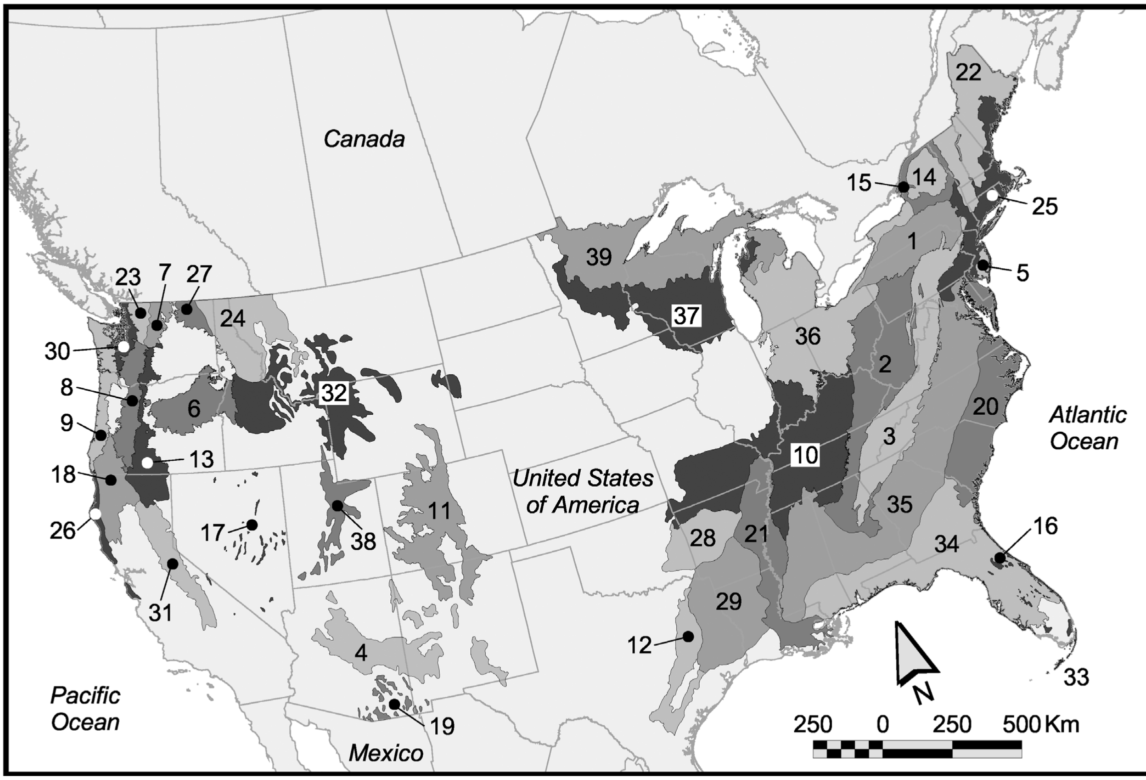

Figure 1. Forest fragmentation was analyzed for 39 forested ecoregions: (1) Allegheny Highland Forest, (2) Appalachian Mixed Mesophytic Forest, (3) Appalachian/Blue Ridge Forest, (4) Arizona Mountain Forest, (5) Atlantic Coastal Pine Barren, (6) Blue Mountain Forest, (7) Cascade Mountain Leeward Forest, (8) Central and Southern Cascade Forest, (9) Central Pacific Coastal Forest, (10) Central US Hardwood Forest, (11) Colorado Rockies Forest, (12) East Central Texas Forest, (13) Eastern Cascade Forest, (14) Eastern Forest/Boreal Transition, (15) Eastern Great Lakes Lowland Forest, (16) Florida Sand Pine Scrub, (17) Great Basin Montane Forest, (18) Klamath–Siskiyou Forest, (19) Madrean Sky Island Montane Forest, (20) Middle Atlantic Coastal Forest, (21) Mississippi Lowland Forest, (22) New England/Acadian Mixed Forest, (23) North Cascade Forest, (24) North Central Rockies Forest, (25) Northeastern Coastal Forest, (26) Northern California Coastal Forest, (27) Okanogan Forest, (28) Ozark Mountain Forest, (29) Piney Wood Forest, (30) Puget Lowland Forest, (31) Sierra Nevada Forest, (32) South Central Rockies Forest, (33) South Florida Rockland, (34) Southeastern Conifer Forest, (35) Southeastern Mixed Forest, (36) Southern Great Lakes Forest, (37) Upper Midwest Forest/Savanna Transition, (38) Wasatch and Uinta Montane Forest, and (39) Western Great Lakes Forest. (See Ricketts et al. 1999 for a discussion of ecoregion.)

metrics summarized for the country using ordinal scores for our five example indices show the national pattern of forest fragmentation (figure 2). In figure 2, it is easy to see the differences in land unit size between East and West, as well as regions in the country where forests appear more intact. Moving east to west, some of the larger, more intact areas include the Northwoods of Maine, Adirondack Park in New York, the Boundary Waters area of northern Minnesota, Glacier National Park and the Bob Marshall Wilderness area of Montana, the Selway–Bitterroot region of Idaho, the North Cascades and Olympic Mountains of Washington, and the Klamath–Siskiyou region of southwest Oregon and northwest California. Higher levels of forest fragmentation can be seen in southern New England; portions of the mid-Atlantic states; the Piedmont of the Southeast; and large sections of Ohio, Indiana, Michigan, Wisconsin, southern Florida, and the Mississippi Valley. All of these examples are located in the eastern United States, where the size of the land units is much smaller than in the West. Some land units in western Wyoming also received low scores, mostly in regions where naturally occurring nonforested lands intermix with forested areas.

Forest fragmentation at the ecoregion level

Although it is useful to consider forest fragmentation at the national level, the strength of the land unit database is realized best when focusing on smaller geographic extents. Examining forest fragmentation at the ecoregion level is particularly important for several reasons. General forest type, ecology, and disturbance histories are far more similar for land units within ecoregions than they are between them. This similarity helps considerably when trying to choose appropriate

Table 1. Summary of results for road density and selected fragmentation metrics for each ecoregion, the western and eastern portion of the study area, and the entire conterminous United States.

Region	Ecoregion area (ha)	Number of land units	Percent forest[a]	Percent core area[b]	Number of forest patches	Mean forest patch size (ha)
Allegheny Highland Forest	7,675,748	602	69.7	46.5	66,514	90
Appalachian Mixed Mesophytic Forest	17,854,294	1,602	76.7	52.8	125,894	123
Appalachian/Blue Ridge Forest	14,827,932	1,301	72.5	50.8	142,238	97
Arizona Mountain Forest	10,330,107	101	48.5	29.9	76,303	65
Atlantic Coastal Pine Barren	825,117	113	54.6	27.0	19,918	28
Blue Mountain Forest	5,898,031	47	48.3	28.6	71,800	83
Cascade Mountain Leeward Forest	1,456,954	17	62.3	39.9	14,914	142
Central and Southern Cascade Forest	4,090,056	65	68.0	46.5	31,894	163
Central Pacific Coastal Forest	3,745,165	88	84.0	62.4	15,401	268
Central US Hardwood Forest	27,580,236	1,886	50.3	28.3	327,957	49
Colorado Rockies Forest	12,283,430	134	58.9	38.5	69,452	105
East Central Texas Forest	5,119,185	445	29.5	10.9	98,364	22
Eastern Cascade Forest	5,045,576	103	37.8	22.9	76,509	75
Eastern Forest/Boreal Transition	2,659,400	78	82.3	68.1	11,236	242
Eastern Great Lakes Lowland Forest	2,374,371	224	66.5	47.4	32,640	93
Florida Sand Pine Scrub	386,176	90	34.0	16.5	21,452	33
Great Basin Montane Forest	534,324	27	48.3	24.5	3,942	63
Klamath–Siskiyou Forest	4,610,238	110	77.1	52.1	32,762	182
Madrean Sky Island Montane Forest	1,097,147	31	21.3	10.1	7,756	28
Middle Atlantic Coastal Forest	12,624,046	1,055	58.2	33.4	126,351	62
Mississippi Lowland Forest	10,690,623	675	25.4	14.5	86,938	42
New England/Acadian Mixed Forest	10,741,731	611	83.2	64.2	41,023	231
North Cascade Forest	1,304,363	9	72.1	49.8	14,775	188
North Central Rockies Forest	9,313,772	71	66.1	46.3	52,617	180
Northeastern Coastal Forest	8,217,277	1,185	69.4	46.5	101,060	77
Northern California Coastal Forest	1,223,314	85	75.5	50.5	13,690	160
Okanogan Forest	1,303,530	38	55.5	32.7	13,359	93
Ozark Mountain Forest	5,836,909	253	67.2	46.9	44,875	105
Piney Wood Forest	13,523,604	957	69.0	46.3	101,365	102
Puget Lowland Forest	1,496,320	163	71.7	48.2	31,865	136
Sierra Nevada Forest	4,889,313	166	46.4	25.4	55,452	71
South Central Rockies Forest	14,530,308	107	37.7	23.4	121,234	77
South Florida Rockland	219,994	17	34.1	12.3	12,573	29
Southeastern Conifer Forest	23,103,750	1,695	53.5	31.0	229,194	59
Southeastern Mixed Forest	32,933,256	2,606	68.4	42.3	268,860	92
Southern Great Lakes Forest	20,178,698	2,777	25.4	9.8	255,440	21
Upper Midwest Forest/Savanna Transition	15,150,620	1,602	31.5	14.4	204,508	28
Wasatch and Uinta Montane Forest	3,817,489	79	54.5	27.8	33,129	62
Western Great Lakes Forest	18,232,102	861	72.0	49.6	106,790	135
Western Conterminous United States	86,969,605	1,294	50.9	31.9	595,252	92
Eastern Conterminous United States	250,755,098	18,659	56.5	35.3	2,100,742	67
Entire Conterminous United States	337,724,703	19,953	54.8	34.3	2,695,994	72

Note: Number of land units for the western, eastern, and entire conterminous United States is less than the sum of land units for each ecoregion because some land units are shared by two or more ecoregions.

a. *Percent forest* is the amount of the entire land unit area that is composed of forest.

b. *Percent core area* is the amount of forest cover composed of core forest area using a 90 m edge effects distance.

fragmentation indices and interpret them in an ecologically meaningful fashion. For example, comparing a deciduous forest type in the eastern United States, which is more likely to be naturally contiguous but heavily disturbed by humans, with a dry conifer forest type in the western United States, which may be naturally patchy and minimally disturbed by humans, can cause serious problems in the interpretation of the calculated results.

Forest fragmentation profiles can be created and compared for each ecoregion. For example, using ordinal scores for our five indices, we generated individual ecoregion fragmentation profiles (figure 3). These histograms were calculated by using the amount of land represented in each cumulative ordinal score class as a percentage of the total land unit area for each ecoregion. Starting from the eastern seaboard (ecoregion 20) and heading west to the final forest ecoregion before the Plains states (ecoregion 10), forest fragmentation profiles show different conditions. Among these five ecoregions, forest fragmentation is high along the coast (ecoregion 20) and in the Piedmont region (ecoregion 35),

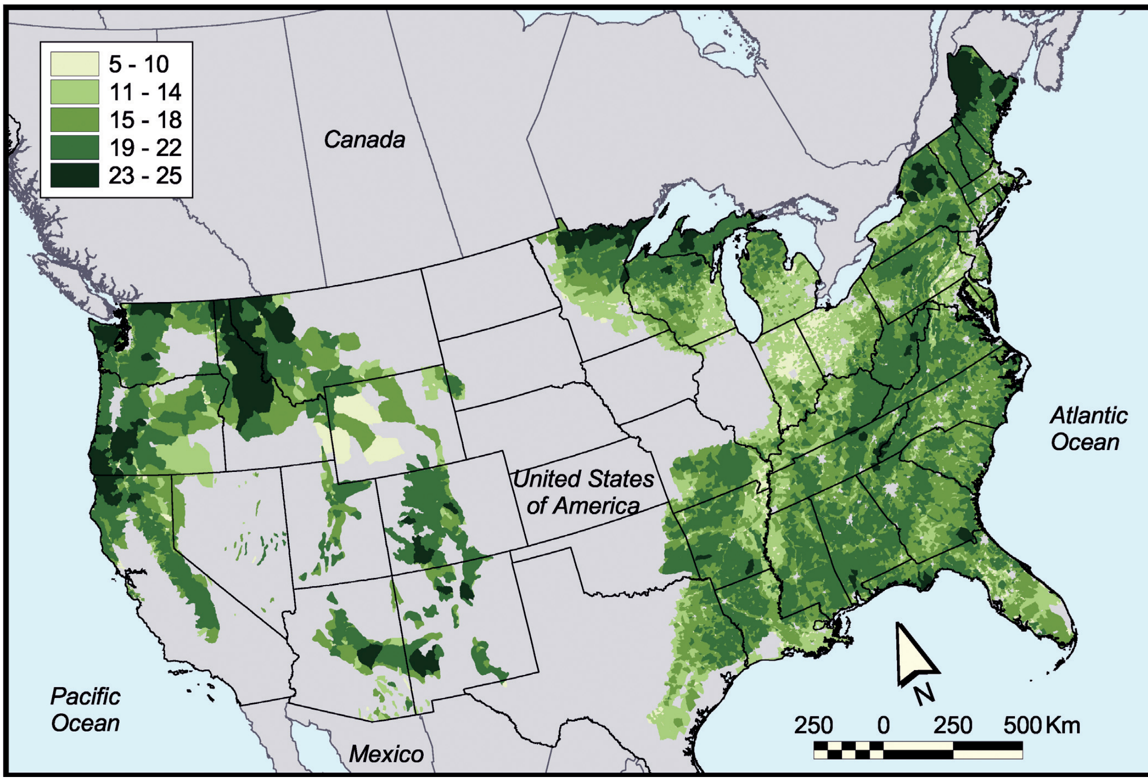

Figure 2. Map of cumulative ordinal scores results for all land units in the conterminous United States. Note that ordinal score ranges were determined using fragmentation results for all land units. Higher scores (darker areas) denote less fragmented forest landunits.

decreases in the Appalachian region (ecoregions 2 and 3), and increases again in ecoregion 10. These are obviously general results; more specific attributes could be tracked within ecoregions over time. One of the strengths of this database and methodology, however, is that it can be replicated cost-effectively as a tool for monitoring forest fragmentation. For example, forest fragmentation is one of nine indicators included in the conservation of biological diversity criteria for the Montréal Process (Montréal Process 1996). The Montréal Process was convened to develop and implement internationally agreed criteria and indicators for the conservation and sustainable management of temperate and boreal forests. Numerous technical challenges regarding the assessment, reporting, and monitoring of identified criteria and indicators still exist. For example, land cover and road data sets are often unavailable, lack appropriate detail, or are outdated. For forest fragmentation, the methodology outlined in this article, or a modified version of it, might serve as a foundation for ongoing monitoring for member nations, particularly where roads are numerous across the landscape. As a parallel process, periodic updates of the underlying data sets would be required to produce a more accurate assessment.

Looking more closely at just one ecoregion (figure 4), the Middle Atlantic Coastal Forest, further observations can be made and the potential utility of the land unit database explored. Figure 4a shows the forest–nonforest land cover upon

Table 2. Data ranges used to determine ordinal ranking for each selected fragmentation metric for ecoregion 20 (Middle Atlantic Coastal Forest).

Fragmentation metric	Ordinal score data range 1	2	3	4	5
Road density (km/km^2)	3.583 – 6.418	2.318 – 3.582	1.740 – 2.317	1.301 – 1.739	0.208 – 1.300
Class area (ha)	153 – 5099	5099 – 11855	11855 – 22977	22977 – 42416	42416 – 77981
Percentage of landscape	7.37 – 31.66	31.67 – 46.41	46.42 – 58.78	58.79 – 71.51	71.52 – 92.78
Total core area index (%)	7.13 – 31.13	31.14 – 43.93	43.94 – 54.08	54.09 – 64.28	64.29 – 86.66
Mean nearest neighbor (m)	145.57 – 285.55	89.46 – 145.56	63.00 – 89.45	45.76 – 62.99	30.00 – 45.75

Note: Ranges were determined using natural breaks classification, which is based on Jenks's optimization method (Jenks and Caspall 1971).

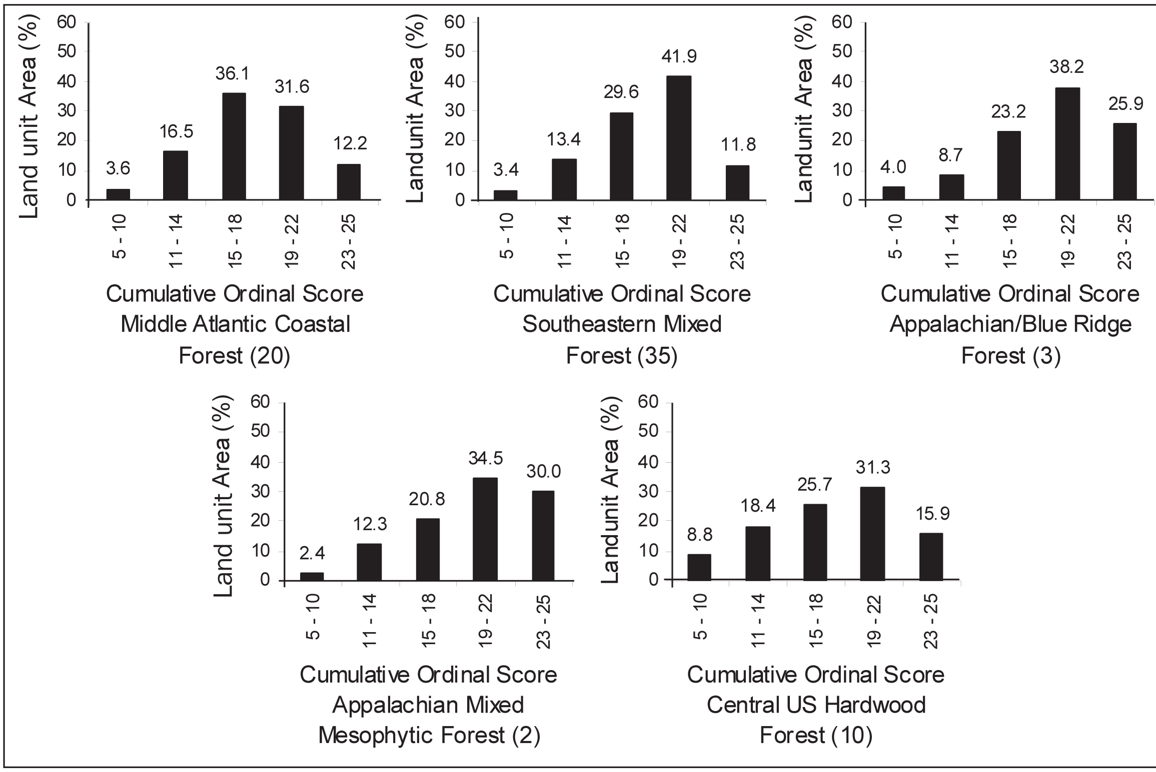

Figure 3. Amount of land represented in each cumulative ordinal score class as a percentage of the total land unit area for each of five eastern US ecoregions. (Please refer to figure 1 for ecoregion locations.)

which fragmentation indices, except road density, were calculated. Figure 4b shows the cumulative ordinal score results for this ecoregion using our five indices. Data ranges for each of the five indices used to determine ordinal ranking are presented in table 2. Note that the range in ordinal scores in figure 2 and figure 4b is identical, but the mapped results of each figure appear very different. This difference is due to differences in scoring within each figure: Figure 2 scores are based on all 19,953 land units in the conterminous United States; the results in figure 4B were generated by scoring only the 1055 land units that made up that particular ecoregion.

Other important features in figure 4b differ from those of figure 2. First, the irregular size and shape of land units is evident. Second, the spatial distribution of the cumulative results provides important information. Most of the higher scoring land units are located along the coast, while lower scoring land units reside in the western half and northernmost portions of the ecoregion. Connected land units of similar score are evident as are isolated, high-scoring land units surrounded by lower scoring land units. It is important to remember that this initial analysis does not distinguish among various forest quality attributes such as native versus plantation or late seral versus early seral forests.

Figure 4c demonstrates an extended utility of the database. This figure shows the cumulative ordinal score results along with existing protected areas taken from a protected areas database (DellaSala et al. 2001). GAP status codes pertain to the USGS GAP Analysis Program, in which “GAP” refers to a geographic approach to planning for diversity (Scott et al. 1994). GAP status 1 and status 2 lands (in blue) are essentially protected from conversion to nonnatural land cover, with GAP 1 lands emphasizing more management to promote native biodiversity and GAP 2 lands emphasizing less. GAP 3 lands (in orange) are also protected from conversion to nonnatural land cover, but they are subject to various extractive uses.

Many of the GAP 1, 2, and 3 protected areas correspond to some of the highest-scoring land units in this ecoregion; however, other high-scoring land units remain outside these existing protected areas. With this information, conservation planners can focus on areas that have more intact forests from which they can design and prioritize conservation activity. For example, the area with high forest intactness between Hofmann State Forest, Bladen Lakes State Forest, and Green Swamp could receive a higher priority for protection as a link between existing protected areas. Planners can gain a perspective on regional forest loss and fragmentation, and possibly forecast future problem areas, once a time-series analysis is completed. By repeating the assessment periodically, changes in forest condition at the regional scale could be tracked with empirical data routinely reported and ongoing management actions updated to reflect current information.

Ecological thresholds

In developing the forest fragmentation data presented in this article, we made no attempt to include known ecological thresholds in the scores. Thus, all scoring was intentionally un-

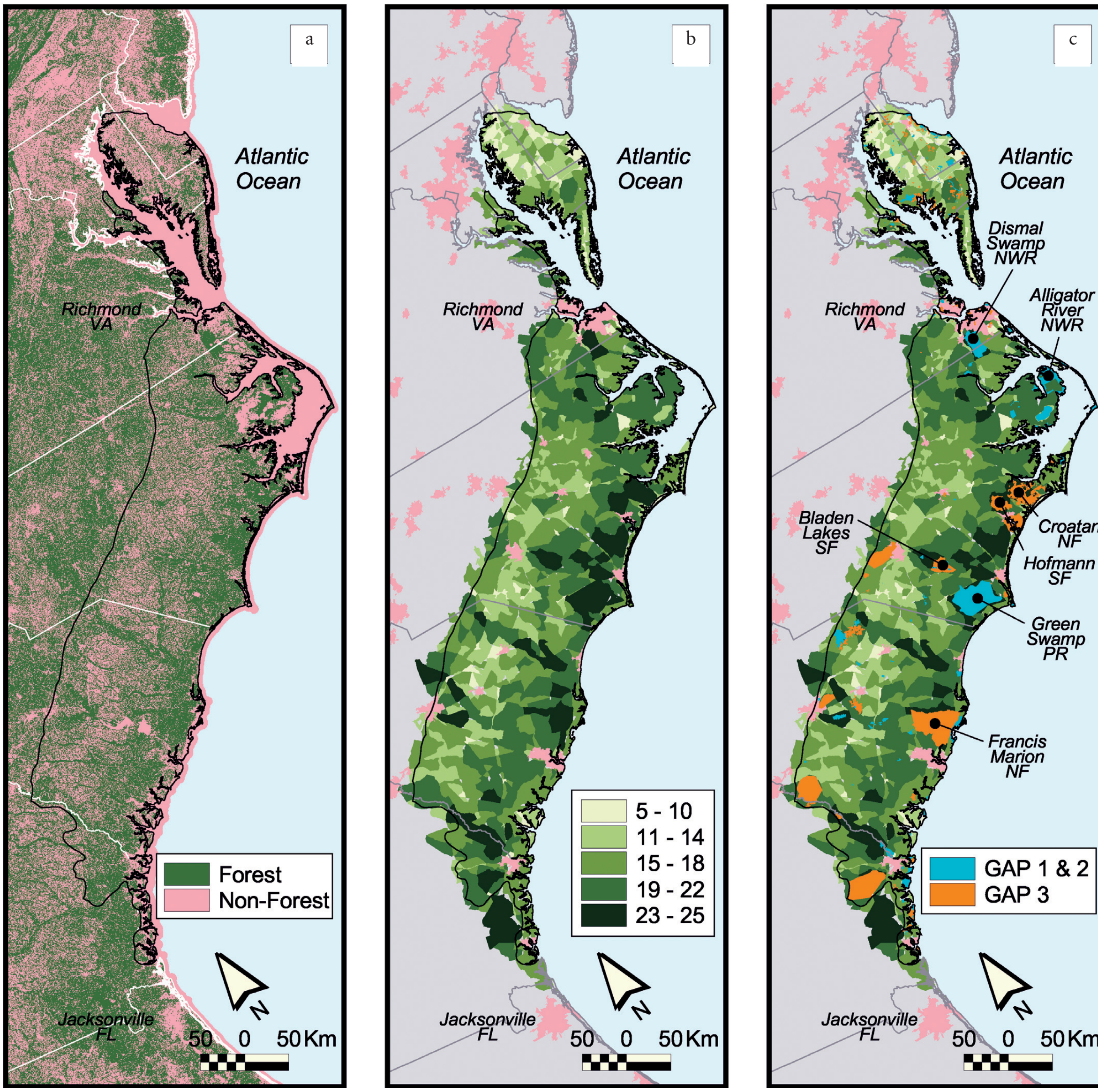

Figure 4. (a) National Land Cover Data reclassified as either forest or nonforest for ecoregion 20 (Middle Atlantic Coastal Forest), with a 30 meter resolution raster version of USGS 1:100,000 scale roads added as nonforest. Forest included coniferous forest, deciduous forest, mixed forest, and forested wetland classes. (b) Cumulative ordinal score results for all land units in ecoregion 20. Please refer to table 2 for the data ranges used to determine ordinal ranks for selected fragmentation metrics. (c) Protected areas for ecoregion 20 overlaying cumulative ordinal score results. GAP status 1 and 2 are lands protected from conversion to nonnatural land cover with greater emphasis on conserving native biodiversity for GAP 1. GAP 3 lands are also protected from conversion to nonnatural land cover, but subject to various extractive uses. For (b) and (c), pale red areas denote cities with a population of at least 50,000 people.

weighted and relative. We did not try to include ecological thresholds because of the general lack of reliable threshold data. However, that does not preclude use of the land units database to address specific conservation issues where ecological thresholds are better understood. For the Middle Atlantic Coastal Forest ecoregion, for example, we offer two different representations of the data (figures 5a, 5b). Figure 5a shows road density scores for each land unit using three natural breaks in the data. The best range for road density was 0.0–1.8 km/km^2 and included the majority of the ecoregion. In comparison, conservation planners in charge of the eastern red wolf *(Canis rufus)* recovery effort, which is centered in and around the Alligator River National Wildlife Refuge (figure 4b), could be concerned about the impact of roads on recovery efforts. Although there has been some variability based on species and geographic location, the scientific literature reports an approximate road density threshold of 0.5 km/km^2 for long-term persistence of wolves (Thiel 1985, Mladenoff et al. 1995). Reviewing road density results for the Middle Atlantic Coastal Forest ecoregion with this ecological threshold tells a very different story than that presented by natural breaks. There are very few places where road density in this area is below the threshold that is required for successful long-term existence for large carnivore populations in the Middle Atlantic Coastal Forest ecoregion, although those areas that do exist are near the wolf recovery area (figure 5b).

Critical assessment and research recommendations

Roads. The emphasis on roads in the establishment of an analytical unit and as an index for fragmentation is unusual for a forest fragmentation analysis of this scope. Roads have been included in other studies (Jones et al. 1997) but have rarely been so prominent in the research design. In fact, some research efforts have found roads too problematic and have elected to avoid them altogether (Heinz Center 1999). We believe our use of roads is an important contribution and fully warranted by the overwhelming body of scientific literature describing the negative impacts that roads have on natural systems (Trombulak and Frissell 2000). There are other ways to examine roads, but roads are too important to just ignore. There is also an issue of scale, particularly as it applies to roads. The map scale of the roads data used in generating the forest fragmentation database (1:100,000) is reasonable as a first approximation, especially when analyzing such a large geographic extent, but incorporating finer scales (e.g., 1:24,000) is more desirable. We are currently applying the same basic approach described in this article for various subregions around the country using 1:24,000 scale roads data and including additional forest quality information. At this scale, the total length of roads increases roughly 40% for these areas. Furthermore, while there is a fair amount of agreement between scales in terms of roads distribution and concentration, there are examples where the 1:100,000 roads data contained very few roads, but the 1:24,000 scale roads data showed an extensive network.

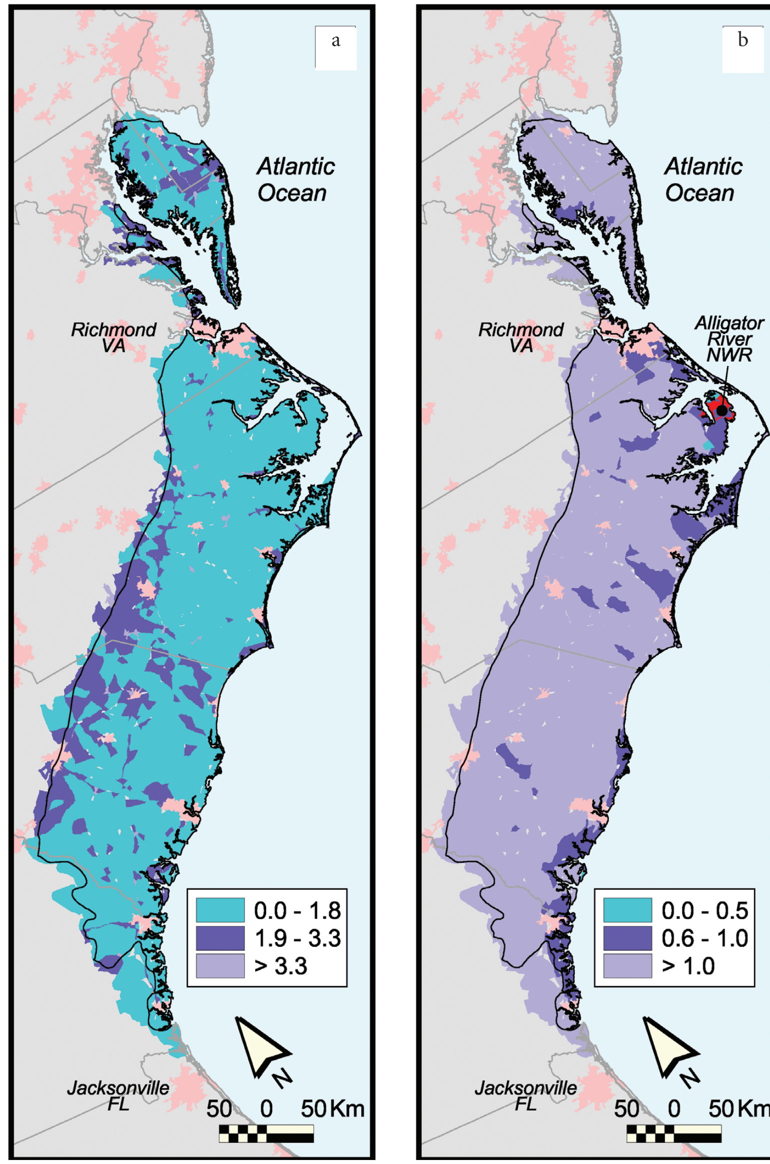

Figure 5. (a) Road density results (km/km^2) for ecoregion 20 (Middle Atlantic Coastal Forest) using natural breaks classification, based on Jenks's optimization method. (b) Road density results (km/km^2) for ecoregion 20 using biologically based classification ranges. Note that for both (a) and (b), the displayed results are actual road density values per land unit and not ordinal score results. For both panels, pale red areas denote cities with a population of at least 50,000 people.

By using highways, we offer a different approach to dissecting landscapes into ecologically meaningful analytical units. This technique worked particularly well in much of the eastern United States, where the highway network is extensive, by dissecting the landscape into smaller units of analysis. In regions where the road network is less dense, use of highway-defined land units resulted in units of analysis that encompassed areas substantially different than the ecoregion being studied. Addressing this problem in the future may call for using different criteria to define land units, depending on the type of ecoregion being analyzed.

Natural versus anthropogenic disturbance. Another difficult analytical issue pertains to natural forest patchiness. Fragmentation is not always an ecological negative. Natural patchiness is important to many forest types, whether disturbance is caused by large-scale fires or localized wind-

Box 1. General items, ordinal score items, and fragmentation indices for the land units database.

Item	Level	Brief description
AREA	n.a.	Area in square meters
PERIMETER	n.a.	Perimeter length in meters
LANDUNITS#	n.a.	Internal identification number
LANDUNITS-ID	n.a.	User assigned unique identification number
CBILABEL	n.a.	Textual identification
CBICODE	n.a.	Identification (1 = land unit, 2 = non–land unit)
ROAD-DENS-S1	n.a.	Original road density ordinal score
CA-S1	n.a.	Original class area ordinal score
PCT-LAND-S1	n.a.	Original percentage of landscape ordinal score
TCAI-S1	n.a.	Original total core area index ordinal score
MNN-S1	n.a.	Original mean nearest neighbor ordinal score
SUM-S1	n.a.	Sum of all used original ordinal scores
ROAD-DENS-S2	n.a.	Expanded road density ordinal score
CA-S2	n.a.	Expanded class area ordinal score
PCT-LAND-S2	n.a.	Expanded percentage of landscape ordinal score
TCAI-S2	n.a.	Expanded total core area index ordinal score
MNN-S2	n.a.	Expanded mean nearest neighbor ordinal score
SUM-S2	n.a.	Sum of all used expanded ordinal scores
ROAD-LENGTH	n.a.	Total USGS road length in meters
ROAD-LENGTH-KM	n.a.	Total USGS road length in kilometers
TOTAL-SQKM	n.a.	Total land unit area in square kilometers
ROAD-DENS	n.a.	Land unit road density in km/km^2
TYPE	Class	Patch type
CA	Class	Class area
TA	Class	Total landscape area
PCT-LAND	Class	Percentage of landscape
LPI	Class	Largest patch index
NP	Class	Number of patches
PD	Class	Patch density
MPS	Class	Mean patch size
PSSD	Class	Patch size standard deviation
PSCV	Class	Patch size coefficient of variation
TE	Class	Total edge
ED	Class	Edge density
LSI	Class	Landscape shape index
MSI	Class	Mean shape index
AWMSI	Class	Area weighted mean shape index
DLFD	Class	Double log fractal dimension
MPFD	Class	Mean patch fractal dimension
AWMPFD	Class	Area weighted mean patch fractal dimension
C-PCT-LAND	Class	Core area percentage of landscape
TCA	Class	Total core area
NCA	Class	Number of core areas
CAD	Class	Core area density
MCA1	Class	Mean core area per patch
CASD1	Class	Patch core area standard deviation
CACV1	Class	Patch core area coefficient of variation
MCA2	Class	Mean area per disjunct core
CASD2	Class	Disjunct core area standard deviation
CACV2	Class	Disjunct core area coefficient of variation
TCAI	Class	Total core area index
MCAI	Class	Mean core area index
MNN	Class	Mean nearest neighbor distance
NNSD	Class	Nearest neighbor standard deviation
NNCV	Class	Nearest neighbor coefficient of variation
L-TA	Landscape	Total area
L-LPI	Landscape	Largest patch index
L-NP	Landscape	Number of patches
L-PD	Landscape	Patch density
L-MPS	Landscape	Mean patch size
L-PSSD	Landscape	Patch size standard deviation
L-PSCV	Landscape	Patch size coefficient of variation
L-TE	Landscape	Total edge
L-ED	Landscape	Edge density
L-LSI	Landscape	Landscape shape index
L-MSI	Landscape	Mean shape index
L-AWMSI	Landscape	Area weighted mean shape index

throw. In some forest types, such as ponderosa pine *(Pinus ponderosa),* natural fragmentation is a sign of higher ecological integrity. Intensively managed ponderosa pine forests often display greater tree densities than unmanaged, native stands.

Because of the limits of the input data, it was not possible to differentiate in this study between natural and anthropogenic disturbance. For many forest types, the combination of 30 m resolution satellite imagery and a minimum mapping unit of 1 ha eliminated the majority of smaller natural openings. With regard to natural patchiness, we intentionally avoided the most problematic ecoregions, such as those characterized by open forest or savannas. This problem, however, could not be avoided entirely. For example, Jeffrey pine *(Pinus jeffreyi)* forests, which are naturally patchy forests that grow in very harsh serpentine soils on a small percentage of the Klamath–Siskiyou ecoregion, showed up in the land cover database as quite patchy. Differentiating between Jeffrey pine natural openings and neighboring clearcut blocks was not possible without exhaustive effort. Expanding this effort for the other open forest types scattered throughout the country was untenable. This problem would have been far more serious had the data scale been more detailed, thereby resulting in the delineation of small openings. More detailed investigations will need to address this problem by using disturbance data.

Fragmentation index redundancy and applicability. It has been stated that many fragmentation indices are redundant over a range of spatial and attribute scales, making it important to choose the most relevant indicators (Cain et al. 1997). In addition, indices should be carefully chosen and interpreted to provide ecologically relevant information specific to each research question. We included all of the class- and landscape-level fragmentation results in the land units database to allow for the widest possible utility. We believe that a national forest fragmentation database should be as inclusive as possible, because we are still in the early stages of interpreting spatial pattern. It is still unknown which index (or suite of indices) tells us the most about forest fragmentation, and until we learn more about the mechanism and impact of forest fragmentation, we believe it is better to provide too much

Box 1, continued

L-DFLD	Landscape	Double log fractal dimension
L-MPFD	Landscape	Mean patch fractal dimension
L-AWMPFD	Landscape	Area weighted mean patch fractal dimension
L-TCA	Landscape	Total core area
L-NCA	Landscape	Number of core areas
L-CAD	Landscape	Core area density
L-MCA1	Landscape	Mean core area per patch
L-CASD1	Landscape	Patch core area standard deviation
L-CACV1	Landscape	Patch core area coefficient of variation
L-MCA2	Landscape	Mean area per disjunct core
L-CASD2	Landscape	Disjunct core area standard deviation
L-CACV2	Landscape	Disjunct core area coefficient of variation
L-TCAI	Landscape	Total core area index
L-MCAI	Landscape	Mean core area index
L-MNN	Landscape	Mean nearest neighbor distance
L-NNSD	Landscape	Nearest neighbor standard deviation
L-NNCV	Landscape	Nearest neighbor coefficient of variation
L-SHDI	Landscape	Shannon's diversity index
L-SIDI	Landscape	Simpson's diversity index
L-MSIDI	Landscape	Modified Simpson's diversity index
L-PR	Landscape	Patch richness
L-PRD	Landscape	Patch richness density
L-RPR	Landscape	Relative patch richness
L-SHEI	Landscape	Shannon's evenness index
L-SIEI	Landscape	Simpson's evenness index
L-MSIEI	Landscape	Modified Simpson's evenness index
L-CONTAG	Landscape	Contagion index

n.a. Not applicable.

data rather than not enough. With this database, it may be useful to employ principal component-based factor analysis (Johnston 1980), a multivariate procedure designed to identify the most important factors driving variability, as demonstrated by Cain and colleagues (1997). It would also be advantageous to incorporate promising new indices, such as patch cohesion (Shumaker 1996).

Spatial filtering techniques using discrete units, such as watersheds (Riitters et al. 1997), have been used to analyze and map regional spatial patterns. This technique has been applied using multiple window sizes (9 x 9 pixels, 27 x 27 pixels, and 81 x 81 pixels) that sense the landscape at different scales to model habitat suitability for species. Hybridizing our approach with spatial filtering algorithms may prove very fruitful.

Conclusions

Land cover data derived from satellite imagery offers outstanding potential for analyzing forest fragmentation (Riitters et al. 2000b). In this article we outline a methodology for assessing forest fragmentation and offer a comprehensive data set for further investigation by researchers. Repeated use of our methodology could become part of a national forest monitoring protocol. Emerging spatial analysis techniques, along with computer mapping advances, have the potential to promote meaningful planning for biodiversity conservation at multiple spatial and temporal scales. Although we are making advances in planning at multiple spatial scales (Poiani et al. 2000), we are still at the early experimental stages of handling the topic analytically in the GIS environment. Despite the numerous technical advances, we see little value in computer mapping technologies unless they can work in close concert with field biology. Without a strong commitment to field surveys and evaluations, we will lose a tremendous opportunity to effectively address the many conservation issues of our time. In the meantime, it is premature to conclude that any region's forests have recovered (Moffat 1998) until one of the most important measures of biodiversity decline, habitat fragmentation, is properly assessed.

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References cited

Ambuel B, Temple SA. 1983. Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. Ecology 64: 1057–1068.

Anderson M, Comer P, Grossman D, Groves C, Poiani K, Reid M, Schneider R, Vickery B, Weakley A. 1999. Guidelines for Representing Ecological Communities in Ecoregional Conservation Plans. Arlington (VA): The Nature Conservancy.

Andrén H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A review. Oikos 71: 355–366.

Burgess RL, Sharpe DM. 1981. Forest Island Dynamics in Man-Dominated Landscapes. New York: Springer-Verlag.

Cain DH, Riitters K, Orvis K. 1997. A multi-scale analysis of landscape statistics. Landscape Ecology 12: 199–212.

Chen J, Franklin JF, Spies TA. 1995. Growing-season microclimate gradients from clearcut edges into old-growth Douglas-fir forests. Ecological Applications 5: 74–86.

Curtis JT. 1956. The modification of mid-latitude grasslands and forests by man. Pages 721–736 in Thomas WL Jr, ed. Man's Role in Changing the Face of the Earth. Chicago: University of Chicago Press.

DellaSala DA, Staus NL, Strittholt JR, Hackman A, Iacobelli A. 2001. An updated protected areas database for the United States and Canada. Natural Areas Journal 21: 124–135.

Forman RTT, Godron M. 1986. Landscape Ecology. New York: John Wiley and Sons.

Franklin JA. 1993. Preserving biodiversity: Species, ecosystems, or landscapes? Ecological Applications 3: 202–205.

Heinz Center. 1999. Designing a Report on the State of the Nation's Ecosystems: Selected Measurements for Croplands, Forests and Coasts & Oceans. Washington (DC): H. John Heinz III Center.

Henderson MT, Merriam G, Wegner J. 1985. Patchy environments and species survival: Chipmunks in agricultural mosaic. Biological Conservation 31: 95–105.

Jenks GF, Caspall FG. 1971. Error on choropleth maps: Definition, measurement, reduction. Annals of the Association of American Geographers 61: 217–244.

Johnston RJ. 1980. Multivariate Statistical Analysis in Geography. New York: John Wiley and Sons.

Jones KB, Riitters KH, Wickham JD, Tankersley RD Jr, O'Neill RV, Chaloud DJ, Smith ER, Neale AC. 1997. An ecological assessment of the United States mid-Atlantic region: A landscape atlas. Washington (DC): US Environmental Protection Agency. USEPA/600/R-97/130.

Krummel JR, Gardner RH, Sugihara G, O'Neill RV, Coleman PR. 1987. Landscape patterns in a disturbed environment. Oikos 48: 321–324.

Loveland TR, Reed BC, Brown JF, Ohlen DO, Zhu Z, Yang L, Merchant JW. 2000. Development of a global land cover characteristics database and IGBP DISCover from 1km AVHRR data. International Journal of Remote Sensing 21: 1303–1330.

Mader HJ. 1984. Animal habitat isolation by roads and agricultural fields. Biological Conservation 29: 81–96.

McGarigal K, Marks BJ. 1995. FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure. Portland (OR): US Department of Agriculture, Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-351.

Mladenoff DJ, Sickley TA, Haight RG, Wydeven AP. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the northern Great Lakes region. Conservation Biology 9: 279–294.

Moffat AS. 1998. Temperate forests gain ground. Science 282: 1253.

Montréal Process. 1996. Progress on implementation of the Montreal Process on criteria and indicators for the conservation and sustainable management of temperate and boreal forests. Ottawa: Montreal Process, Natural Resources Canada, Canadian Forest Service.

Olson DM, et al. 2001. Terrestrial ecoregions of the world: A new map of life on Earth. BioScience 51: 933–938.

O'Neill RV, et al. 1988. Indices of landscape pattern. Landscape Ecology 1: 153–162.

O'Neill RV, Hunsaker CT, Timmins SP, Jackson BL, Jones KB, Riitters KH, Wickham JD. 1996. Scale problems in reporting landscape pattern at the regional scale. Landscape Ecology 11: 169–180.

O'Neill RV, Hunsaker CT, Jones KB, Riitters KH, Wickham JD, Schwartz PM, Goodman IA, Jackson BL, Baillargeon WS. 1997. Monitoring environmental quality at the landscape scale. BioScience 47: 513–519.

Poiani KA, Richter BD, Anderson MG, Richter HE. 2000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. BioScience 50: 133–146.

Rejmánek M. 1989. Invasibility of plant communities. Pages 369–388 in Drake JA, et al., eds. Biological invasions: A Global Perspective. New York: John Wiley and Sons.

Ricketts TH, et al. 1999. Terrestrial Ecoregions of North America: A Conservation Assessment. Washington (DC): Island Press.

Riitters KH, O'Neill RV, Jones KB. 1997. Assessing habitat suitability at multiple scales: A landscape-level approach. Biological Conservation 81: 191–202.

Riitters KH, Wickham JD, Vogelmann JE, Jones KB. 2000a. National landcover pattern data. Ecology 81: 604.

Riitters KH, Wickham JD, O'Neill RV, Jones KB, Smith E. 2000b. Global-scale patterns of forest fragmentation. Conservation Ecology 4: 3. (05 April 2002; *www.consecol.org/vol4/iss2/art3)*

Saunders DA, Hobbs RJ, Margules CR. 1991. Biological consequences of ecosystem fragmentation: A review. Conservation Biology 5: 18–32.

Schumaker NH. 1996. Using landscape indices to predict habitat connectivity. Ecology 77: 1210–1225.

Scott JM, et al. 1994. Gap analysis: A geographic approach to protection of biological diversity. Wildlife Monographs 123: 1–41.

Staus NL, Strittholt JR, Robinson R. 2001. Conservation planning for aquatic biological integrity in the Klamath–Siskiyou ecoregion using multiple spatial scales. (05 April 2002; *www.consbio.org/cbi/what/ks_aquatic.htm)*

Thiel RP. 1985. Relationship between road densities and wolf habitat suitability in Wisconsin. American Midland Naturalist 113: 404.

Trombulak SC, Frissell CA. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14: 18–30.

Turner MG. 1989. Landscape ecology: The effect of pattern on process. Annual Review of Ecology and Systematics 20: 171–197.

Urban DL, O'Neill RV, Shugart HH. 1987. Landscape ecology. BioScience 37: 119–127.

Verboom B, Apeldoorn R. 1990. Effects of habitat fragmentation on the red squirrel, *Sciurus vulgaris*. Landscape Ecology 4: 171–176.

Vermeulen JJW. 1993. The composition of the carabid fauna on poor sandy road-side verges in relation to comparable open areas. Biodiversity and Conservation 2: 331–350.

Vogelmann JE, Sohl TL, Campbell PV, Shaw DM. 1998. Regional land cover characterization using Landsat Thematic Mapper data and ancillary data sources. Environmental Monitoring and Assessment 51: 415–428.

Whitcomb RF, Robbins CS, Lynch JF, Whitcomb BL, Klimkiewicz MK, Bystrak D. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. Pages 125–205 in Burgess RL, Sharpe DM, eds. Forest Island Dynamics in Man-Dominated Landscapes. New York: Springer-Verlag.

Whitney GG. 1994. From Coastal Wilderness to Fruited Plain: A History of Environmental Change in Temperate North America from 1500 to the Present. New York: Cambridge University Press.

Wilcove DS. 1987. From fragmentation to extinction. Natural Areas Journal 7: 23–29.

Wilcox BA, Murphy DD. 1985. Conservation strategy: The effects of fragmentation on extinction. American Naturalist 125: 879–887.

Woodroffe R, Ginsberg JR. 1998. Edge effects and the extinction of populations inside protected areas. Science 280: 2126–2128.