DOI: 10.1007/s00267-003-0220-1

# ENVIRONMENTAL ASSESSMENT

# Hot Spots of Perforated Forest in the Eastern United States

**KURT H. RIITTERS***
USDA Forest Service
Southern Research Station
3041 Cornwallis Road
Research Triangle Park, North Carolina 27709, USA

**JOHN W. COULSTON**
Department of Forestry
North Carolina State University
Box 8008
Raleigh, North Carolina 27695, USA

ABSTRACT / National assessments of forest fragmentation satisfy international biodiversity conventions, but they do not identify specific places where ecological impacts are likely. In this article, we identify geographic concentrations (hot spots) of forest located near holes in otherwise intact forest canopies (perforated forest) in the eastern United States, and we describe the proximate causes in terms of the non-forest land-cover types contained in those hot spots. Perforated forest, defined as a 0.09-ha unit of forest that is located at the center of a 7.29-ha neighborhood containing 60–99% forest with relatively low connectivity, was mapped over the eastern United States by using land-cover maps with roads superimposed. Statistically significant ($P < 0.001$) hot spots of high perforation rate (perforated area per unit area of forest) were then located by using a spatial scan statistic. Hot spots were widely distributed and covered 20.4% of the total area of the 10 ecological provinces examined, but 50.1% of the total hot-spot area was concentrated in only two provinces. In the central part of the study area, more than 90% of the forest edge in hot spots was attributed to anthropogenic land-cover types, whereas in the northern and southern parts it was more often associated with seminatural land cover such as herbaceous wetlands.

Forest loss and consequent habitat degradation have led to concerns for the survival of forest-dependent species and maintenance of biodiversity. In tropical forests, where global deforestation rates probably exceed 50,000 km$^2$ per year (Achard and others 2002) and might approach 160,000 km$^2$ per year (Matthews 2001), attention is naturally focused on the direct impacts of forest habitat loss (e.g., Whitmore and Sayer 1992; Brook and others 2003). In contrast, the area of North American temperate and boreal forests has increased slightly in recent decades (Matthews 2001; USDA Forest Service 2001), and more attention is given to changes in habitat quality, including forest composition, ownership, and spatial arrangement (e.g., Montréal Process Liaison Office 2000).

Although the total forest area within the United States is relatively stable, periodic surveys indicate substantial shifts in the spatial distribution of privately owned forest (USDA Natural Resources Conservation Service 2000). Between 1982 and 1997, the change in privately owned forest area exceeded ±3% in 25 states and ±10% in 5 states. Approximately 90,000 km$^2$ of privately owned forest was gained from farmland, mostly in the Midwest, and 40,000 km$^2$ was lost to urban land, primarily along the Atlantic seaboard. All but two states on the urbanizing Atlantic seaboard had net losses of forest, and all but two states in the predominantly agricultural region bordering the Mississippi and Ohio Rivers had net increases.

Those statistics indicate a regional spatial rearrangement of ~5% of the eastern forest during the 15-year period. Depending on local spatial patterns of forest gains and losses, there could also have been substantial changes in local forest fragmentation. For example, if 5% of the forest in a completely forested region is removed in uniformly distributed 0.09-ha parcels, all of the remaining forest will be subject to ''edge effects'' extending up to 100 m from forest edge (Riitters and others 2002). However, land-cover changes are not uniformly distributed in real landscapes. Urbanization often fragments intact forest cover near existing urban areas, and abandoned farmland is often adjacent to existing forest. It is possible that forests on the Atlantic seaboard are now more fragmented than

KEY WORDS: Ecological assessment; Forest fragmentation; Scan statistic; Spatial pattern

Published online April 25, 2005.

*Author to whom correspondence should be addressed; *email:* kriitters@fs.fed.us

 

before, whereas forests in the Midwest are less fragmented, but the survey protocols (USDA Natural Resources Conservation Service 2000) do not enable detailed assessments of spatial pattern.

Other national assessments have examined current spatial patterns in a more detailed way using high-resolution, national land-cover maps derived from satellite imagery (Heilman and others 2002; Riitters and others 2002, 2004). Forest tends to be the dominant land-cover type where it occurs, yet fragmentation is so extensive that most forests are exposed to edge effects that penetrate up to 100 m into the forest interior. The largest remnants of core (i.e., not fragmented) forest are concentrated in only a few places, and fragmentation is highest in regions that are dominated by anthropogenic land uses. Those studies showed that there was substantial geographic variation in overall forest fragmentation in 1992, but they focused on the location of core forest and did not examine geographic variation in the type of fragmentation that is responsible for the absence of core forest.

The objective of this research is to identify geographic concentrations (''hot spots'') of perforated forest in the eastern United States. Perforations create a forest edge near the interior of forest patches and thereby eliminate core forest. Perforations are an ecologically important type of fragmentation because they introduce potential edge effects deeper into intact forests, in comparison to the erosion of forest patch perimeters. Perforations are also important in a land-use context because they might represent a pattern of disturbance with a direct relationship to human activities such as residential development. Finally, perforations are important in a risk assessment framework because human land uses tend to be spatially correlated and current concentrations of perforations might represent incipient foci of future forest loss and fragmentation.

Riitters and others (2002) attributed about half of the national fragmentation of core forest to small (less than 7.29 ha) perforations but did not examine geographic variation in perforated forest. Coulston and Riitters (2003) found geographic concentrations of perforated forest that were associated with major transportation corridors, but they examined only the southeastern United States and used administrative units that were not well suited for detecting or interpreting the apparent concentrations. Our goal here is to identify specific locations with relatively high risk of habitat degradation from edge effects resulting from perforations and to describe the proximate causes of perforations in terms of the land-cover types associated with forest edges in those locations.

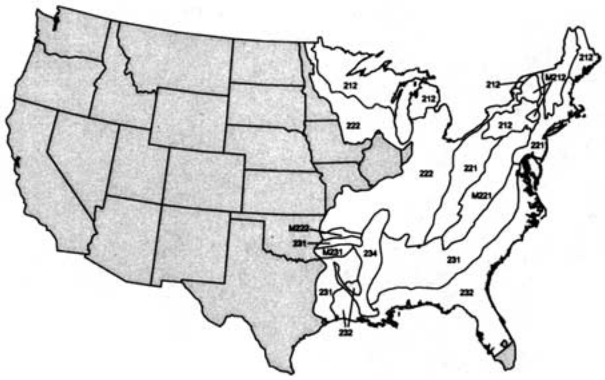

**Figure 1.** The study area includes 10 ecological provinces (Bailey 1995) with a substantial amount of forest. The labels refer to province names as shown in Table 1.

## Methods

### Data

We performed all of the measurements on a land-cover map with roads superimposed. The base map was a forest–nonforest map of the conterminous states from the National Land-Cover Database (NLCD). The NLCD project used Landsat Thematic Mapper (TM) data (circa 1992) to map 21 classes of land cover at a spatial resolution of 0.09 ha/pixel (Vogelmann and others 1998, 2001). We aggregated the 4 NLCD forest classes (coniferous, deciduous, mixed, and wetland forest) into one forest class, and 15 of the remaining NLCD classes into one nonforest class. The water (including ice and snow) and bare rock (including bedrock, talus, and desert, but excluding quarries and mines) NLCD classes were treated as missing values that did not fragment forests. We superimposed a detailed road and street map (Geographic Data Technology 2002) upon the forest map and converted all forest pixels that contained at least one road segment to nonforest pixels (Riitters and Wickham 2003). No distinctions were drawn among type of road, traffic volume, or other factors. The analysis was conducted in the 10 eastern ecological provinces (Bailey 1995) with significant amounts of forest (Figure 1).

### Classification Model

Spatial pattern is important in structuring ecological communities (Levin 1976), and individual species perceive habitat patterns in different terms and at different spatial scales (Wiens 1989). Habitat spatial pattern is conceptually separate from the amount of habitat available (Fahrig 1997), and measures of both are simultaneously needed to accurately characterize suitable habitat (Fortin and others 2003). Although

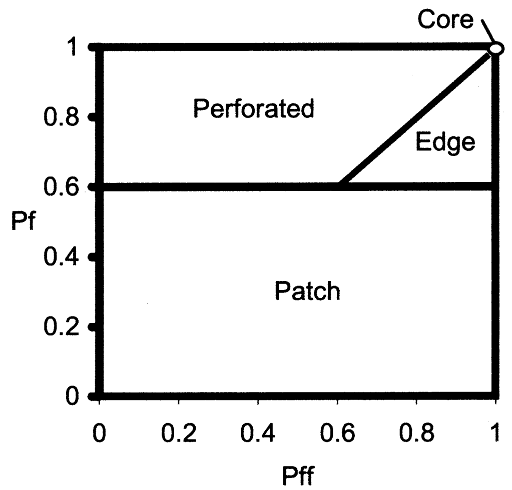


**Figure 2.** The classification model segments the parameter space defined by forest amount (Pf) and forest connectivity (Pff) into four fragmentation categories. [Adapted from Riitters and others (2002); see text for explanation.]

many measures of pattern are available (Gustafson 1998), the choice of any one is difficult (Bogaert 2003) without a framework for conceptualizing human impacts on landscape patterns (e.g., McIntyre and Hobbs 1999).

The model (Figure 2) classifies each forest pixel according to the type of forest fragmentation in the neighborhood surrounding that pixel (Riitters and others 2000). The classification is determined by measurements of forest amount (Pf) and connectivity (Pff) within the neighborhood that is centered on a subject forest pixel. Pf is the probability that a pixel in the neighborhood is forest, and Pff is the probability that a pixel next to a forest pixel is also forest (Riitters and others 2000). Other partitions of the parameter space are possible, and more categories can better represent the underlying gradients of pattern (e.g., Civco and others 2002). The classification of a given forest pixel changes with neighborhood size (Riitters and others 2000) and the changes are meaningful when evaluating habitat for species with different life-history requirements (e.g., Riitters and others 1997). For this analysis, we used a square, 9-pixel × 9-pixel (7.29-ha) neighborhood because that size best differentiated among types of fragmentation in an earlier assessment (Riitters and others 2002) and because it approximates a realistic parcel size when humans decide land uses.

In the model, the ''core'' category indicates a forest pixel that is surrounded by 7.29 ha of forest that is not fragmented, and the ''patch'' category indicates a forest pixel residing in a ''patchy'' neighborhood containing less than 60% forest. Pixels in the ''perforated'' and ''edge'' categories are both surrounded by 60–99% forest, but they differ in the arrangement of the nonforest pixels (Riitters and others 2000). Edge pixels reside in neighborhoods with large (relative to neighborhood size) nonforest patches, whereas perforated pixels are in neighborhoods characterized by relatively small (<7.29 ha) nonforest patches. The term ''perforation'' is more descriptive of the small nonforest patches but is used here to label forest pixels that are near the nonforest patches.

Figure 3 illustrates the model with four simulations of forest removal from a mostly forested initial condition. The four scenarios are as follows: (1) ''random,'' in which individual forest pixels are randomly selected for removal; (2) ''contagious,'' in which individual forest pixels are randomly selected but removed only if they are adjacent to a nonforest pixel; (3) ''line/contagious,'' in which the first 10% of removed pixels are located by placement of random lines on the map and subsequent removals are by the ''contagious'' rule; and; (4) ''random/contagious,'' in which the first 10% of removed pixels are located by the ''random'' rule and subsequent removals are by the ''contagious'' rule. All but 52 of the 62,500 pixels are originally forest pixels, and the original nonforest pixels are ''seeds'' for the contagious scenario. The residual forest pixels are shaded in Figure 3 according to the classification model after removing 10%, 40%, and 90% of the original forest pixels.

Random removals quickly eliminate core forest and perforate all of the residual forest, and edge is a minor component of total fragmentation. Contagious removals preserve core forest and maintain edge forest, and perforated forest occurs in relatively small amounts. Individual patches become more discernable after removing 40% of the forest by any scenario, and the residual forest is easier to describe in terms of patch characteristics. To better understand the model, it is helpful to also examine the movement of the ''cloud'' of pixel values in the (Pf, Pff) parameter space for the random and contagious scenarios (Figure 4). A random removal scenario tends to minimize the connectivity (Pff) of the remaining forest pixels, whereas a contagious removal scenario tends to maximize their connectivity. More types of pattern are visually apparent when more of the total parameter space is occupied.

Real landscapes (e.g., Figure 5) are composites of different types of fragmentation that depend on the spatial patterns of forest gain and loss over time. To identify geographic concentrations, it is convenient to compare the rate of perforation within analysis units

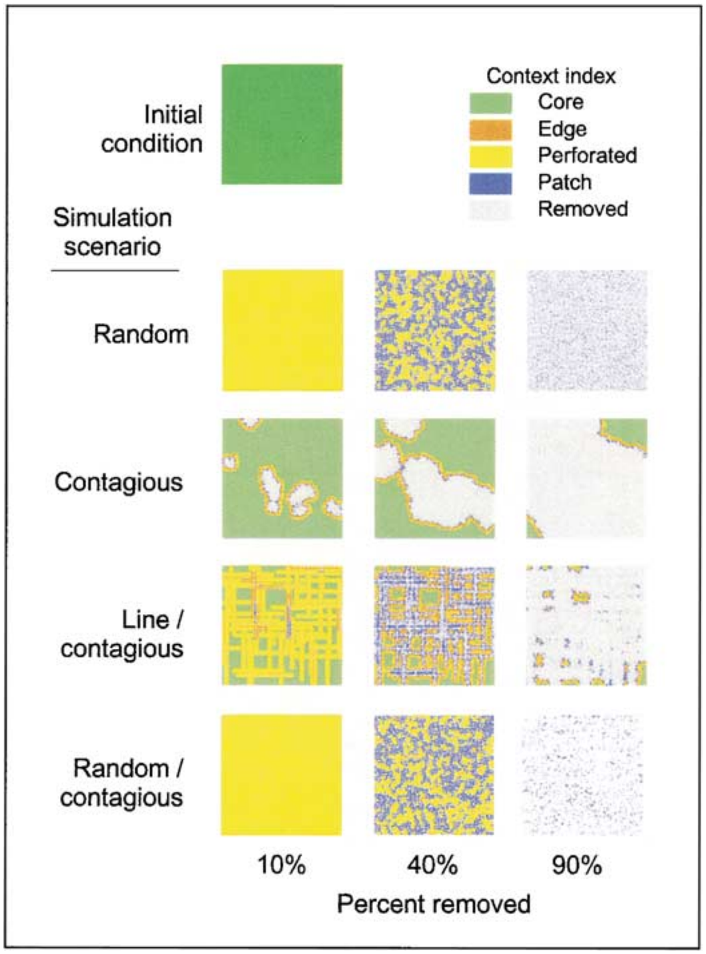


**Figure 3.** Forest fragmentation maps showing the application of the classification model in simulations. Forest was removed from the map of the initial condition (top) according to the four scenarios described in the text. The classification of the remaining forest is shown after removing 10%, 40%, and 90% of the original forest.

that contain large numbers of forest pixels. We calculated rates within 48,797 nonoverlapping 56.25-km$^2$ square analysis units (Figure 5) as the ratio of forest area in the perforated category divided by the total area of forest. For subsequent analysis, the geographic location of each analysis unit was established by the location of its center point. The analysis units were large enough (62,500 pixels each) to provide reliable rate estimates, and small enough to provide a sufficient number of analysis units for the hot-spot analysis.

To improve the interpretation of results with respect to the proximate causes of perforations, we used the original land-cover map to partition the total forest–nonforest edge within each analysis unit into ‘‘anthropogenic edge’’ and ‘‘natural edge.’’ Anthropogenic edge was defined as a forest pixel adjacent to a pixel of developed, agriculture, or barren/disturbed land cover, whereas natural edge was forest adjacent to herbaceous wetland, shrubland, or grassland land cover (Vogelmann and others 2001).

### Spatial Scan Statistic

Spatial scan statistics (Kulldorff 1997, 1999) are designed to detect geographic clusters with significantly high rates of an event in a population and to rank those clusters according to the statistical likelihood that the observed rate is higher than the background population rate. The scan statistic was first developed for and used in human epidemiological studies (Kulldorff 1999). Patil and others (2001) suggested ecological applications and compared the scan statistic with other approaches. Coulston and Riitters (2003) used scan statistics to identify counties comprising hot spots of perforated forest in the southeastern United States and persistent clusters of insect outbreaks in the Pacific Northwest.

One advantage of the scan statistic approach is that it avoids preselection bias by searching for clusters without specifying their size or location, thereby reducing the zoning aspect of the modifiable area unit problem (Jelinski and Wu 1996; Wu 2004). A disad-

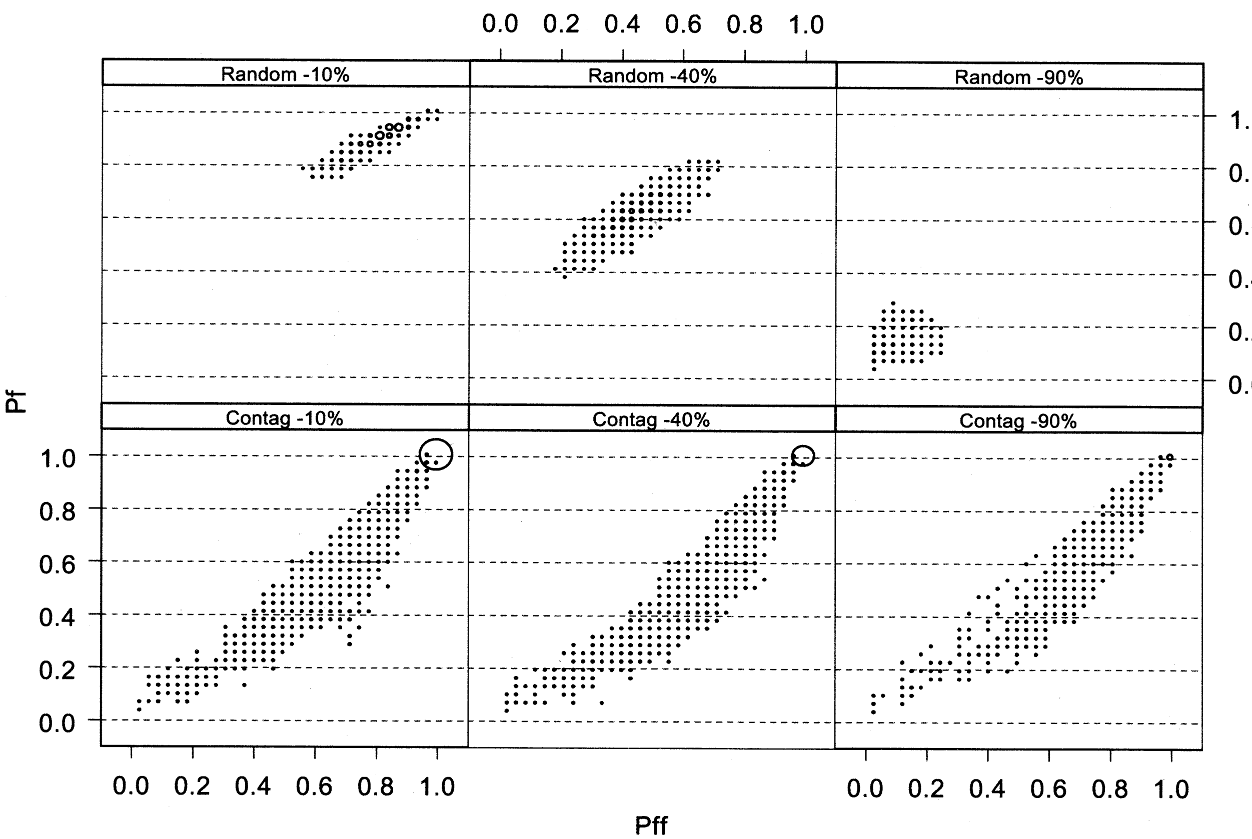


**Figure 4.** Bubble charts showing the frequencies of pixel values of forest amount (Pf) and forest connectivity (Pff) after removing 10%, 40%, and 90% of the forest by two of the four simulation scenarios. Symbol size is proportional to pixel frequency, and "contag" refers to the contagious scenario.

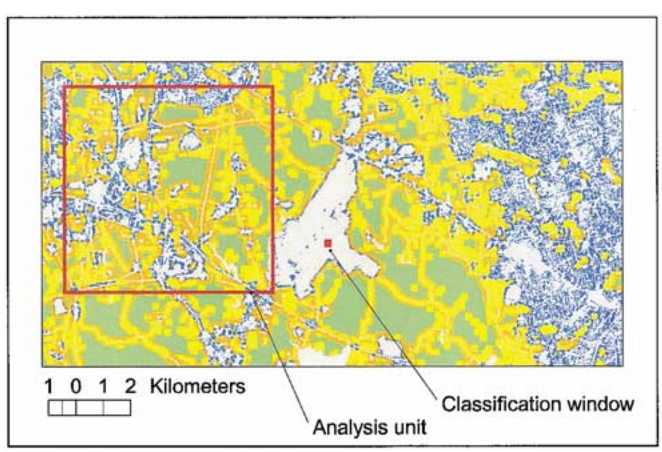


**Figure 5.** Forest fragmentation near the Raleigh–Durham (North Carolina) airport. The smaller square at the airport illustrates the size of the neighborhood that was used to classify the type of fragmentation for each forest pixel. The larger square shows the size of the analysis unit that was used to summarize the pixel values prior to applying the scan statistic. The legend is the same as in Figure 3.

vantage is that the approach uses circular scanning windows and, as a result, the hot spots that it detects are circular in shape. Patil and Taillie (2004) suggest modifications of the procedure to identify arbitrarily shaped geographic clusters. We approximated irregular shapes by specifying a small maximum circle size, such that large irregular hot spots would be approximated by several small hot spots.

We implemented the scan statistic by using version 3.1.2 of the SaTScan software (Kulldorff and Information Management Services Inc. 2002). The SaTScan algorithm places a circular window on the center point of each analysis unit and tests circular windows of increasing size. The analysis units contained within each circle constitute a potential geographic cluster, and the test statistic is based on the likelihood of obtaining the observed excess of events in a larger window. Using the SaTScan terminology, the ''events'' were the number of hectares of forest that were perforated, and the ''controls'' were the number of hectares of forest that were not perforated.

We used the likelihood function for a specific window under the Bernoulli model (Kulldorff and Nagarwalla 1995). SaTScan maximized the likelihood function over all window locations and sizes, and the most likely cluster comprised the analysis units contained in the window with the maximum likelihood [see Coulston and Riitters (2003) for additional discussion]. The likelihood ratio for this window constitutes the maximum likelihood ratio test statistic. Under the null hypothesis, the rate of perforated forest is the same everywhere. The distribution of the test statistic under the null hypothesis was obtained by repeating the analysis on 999 random replications of the dataset (Kulldorff and Information Management Services Inc. 2002). If the observed test statistic was among the 0.1% highest, then the test was declared significant ($P < 0.001$). SaTScan also identified secondary clusters and ranked them according to their likelihood ratio, for which the same test is conservative (Kulldorff 1999).

We specified other SaTScan parameters (Kulldorff and Information Management Services Inc. 2002) so as to improve the interpretation of results within and among ecological provinces. To get small and reasonably homogeneous clusters, we specified a maximum scan circle radius of 0.25 decimal degrees (~20 km depending on latitude). We also specified that the clusters not overlap one another so that each analysis unit appeared in at most one cluster. These choices produced a large number of significant ($P < 0.001$) and small (<30 analysis units each) clusters and we retained the 500 most significant clusters.

## Results

Figure 6 shows the geographic locations of the 500 statistically significant hot spots of perforated forest.

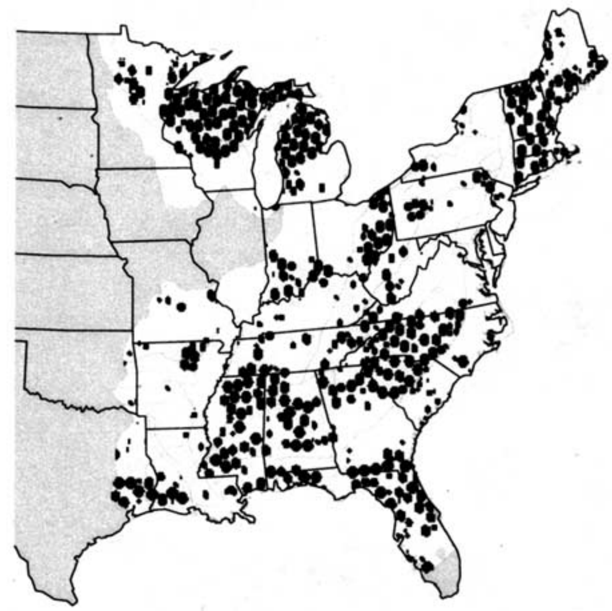

**Figure 6.** The locations of 500 statistically significant ($P <$ 0.001) hot spots of perforated forest identified by the spatial scan statistic in the eastern United States. State boundaries are shown for reference.

Hot spots of perforation are widespread in the sense that every ecological province contains at least one of the hot spots. However, the hot spots are not uniformly distributed and there is variation in hot-spot occurrence within and among provinces (Table 1). Of the total hot-spot area, 50.1% is contained in the Laurentian mixed forest and Southeastern mixed forest provinces. The other eight provinces each contain less than 15% of the total hot-spot area. These comparisons are partly confounded by province size because larger provinces naturally have more hot-spot area even if hot spots are uniformly distributed. For that reason, it is also informative to compare the percentage of total province area that is contained in a hot spot. For example, the relatively small Adirondack–New England mixed forest–coniferous forest–meadow province contains only 5.3% of the total hot-spot area, but a relatively high proportion (26.1%) of that province is contained in a hot spot. The Ouachita mixed forest–meadow province contains the lowest proportion of perforated forest (3.7%) and the Laurentian mixed forest province contains the highest (36.6%).

The proximate causes of perforations also showed substantial geographic variation (Figure 7). Anthropogenic land-cover types created more than 90% of forest edge in almost all hot-spot areas in the central part of the study area, as well as in some hot spots in the north and south. Seminatural land-cover types

Table 1. Distribution of the area contained in 500 hot spots of perforated forest within and among ecological provinces

| Province name and code | All analysis units | | Analysis units contained in 500 hot spots | |
|---|---|---|---|---|
| | No | Percent of total | Percent of province area | Percent of hot-spot area |
| Laurentian mixed forest, 212 | 6,771 | 13.9 | 36.6 | 24.9 |
| Eastern broadleaf forest (oceanic), 221 | 4,812 | 9.9 | 22.9 | 11.1 |
| Eastern broadleaf forest (continental), 222 | 12,438 | 25.5 | 9.9 | 12.4 |
| Southeastern mixed forest, 231 | 8,905 | 18.2 | 28.1 | 25.2 |
| Outer coastal plain mixed forest, 232 | 7,978 | 16.3 | 18.4 | 14.7 |
| Lower Mississippi riverine forest, 234 | 2,040 | 4.2 | 5.3 | 1.1 |
| Adirondack–New England mixed forest–coniferous forest–meadow, M212 | 2,010 | 4.1 | 26.1 | 5.3 |
| Central Appalachian broadleaf forest–coniferous forest–meadow, M221 | 3,136 | 6.4 | 15.3 | 4.8 |
| Ozark broadleaf forest–meadow, M222 | 298 | 0.6 | 13.4 | 0.4 |
| Ouachita mixed forest–meadow, M231 | 409 | 0.8 | 3.7 | 0.2 |
| All provinces | 48,797 | 100.0 | 20.4 | 100.0 |

*Note*: The distribution of all analysis units is shown for comparison. Refer to Figure 1 for province location.

**Figure 7.** The analysis units comprising the 500 hot spots in Figure 6 contain different proportions of anthropogenic and seminatural forest edges. The right-hand map shows the analysis units where <50% of total edge is anthropogenic, the middle map shows units with 50–90% anthropogenic edge, and the left-hand map shows units with >90% anthropogenic edge.

created the majority (>50%) of forest edge in only a small number of hot spots that were located mainly in the Laurentian mixed-forest province and Outer coastal plain mixed-forest province. Mixtures of anthropogenic and seminatural forest edge with 50–90% of total forest edge attributable to anthropogenic land-cover types were also concentrated in the extreme northern and southern parts of the study area.

## Discussion

To the extent that the quality of forest habitat for obligate interior species is adversely affected by the presence of small perforations in otherwise intact forest canopies, the results of our study indicate that such impacts are probably widespread. However, the rate of impacts per unit area of forest varies among ecological provinces and the highest rates are concentrated in only a few provinces. Furthermore, if anthropogenic land uses are more likely to cause adverse impacts in comparison to seminatural land uses, then higher rates of adverse impacts are more likely in the central part of the eastern United States and less likely in the northern and southern parts.

National assessments pursuant to international biodiversity conventions are not detailed enough for local land management decisions, and the techniques we used are a way to localize or ''downscale'' the available national statistics. The approach taken here can be modified to consider other types of fragmentation that might influence habitat quality and biodiversity. Although not shown here, we conducted separate hot-spot analyses of the core, edge, and patch categories and found that each category exhibits a pattern of geographic variation that is different from perforated forest. Hot spots of core forest are in the same locations that were identified by Heilman and others (2002) and by Riitters and others (2002) (i.e., in the four mountainous ecological provinces that are listed last in Table 1). Hot spots of patch forest are concentrated in the Eastern broadleaf forest (continental) province, and hot spots of the edge forest are concentrated in the Outer coastal plain mixed-forest province. If habitat quality is affected in different ways depending on type of fragmentation, then potential impacts are more or less likely to be found in different ecological provinces.

We evaluated the rate of perforated forest, but it is also possible to identify hot spots based on the absolute amount of perforated forest. To accomplish that, the ''controls'' for the spatial scan statistic would be the total land area (not just forest area) that is not perforated forest. Evaluating the rate of perforation (as in this study) makes it easier to compare regions with different absolute amounts of forest. Examining rates is appropriate for identifying locations where an investment in a habitat management strategy will yield a higher rate of return per unit of forest, whereas evaluating the total amount of perforation is appropriate for identifying locations to yield the highest total return on the same investment.

Our interest centered on small perforations, and for that purpose a 7.29-ha neighborhood was appropriate. Larger neighborhood sizes could be used to study larger perforations (Riitters and others 2002), or the neighborhood size could be selected by analogy to home range size to identify hot spots with reference to particular species. A range of neighborhood sizes could be tested to quantify the persistence of hot spots across neighborhood size, substituting ''neighborhood size'' for ''time'' in the spatial–temporal version of the scan statistic (Kuldorff 1999).

We used a relatively simple technique to quantify anthropogenic versus seminatural forest edge as the proximate cause of perforated forest. This technique was applied at the level of aggregation represented by large analysis units. For more detailed analysis, a method developed by Wade and others (2003) partitions the quantity 1–Pff for each forest pixel into anthropogenic and seminatural components. With that approach, it is possible to describe individual perforations in terms of specific proximate causes and, thus, to identify hot spots caused by particular land-cover types.

National assessments might satisfy international biodiversity conventions, but they do not identify specific places where ecological impacts from fragmentation are likely or where habitat management should be targeted. Statistical approaches such as hot-spot analyses are useful starting points for prioritizing regions for more detailed investigations. This top-down approach based on spatial scan statistics is potentially useful for assessing the actual occurrence and abundance of different species, as well as spatial patterns of forest habitat. Coarse-scale analyses can identify hot spots based on broad categories of species, or habitat, and finer-scale analyses can be used to interpret the implications of the hot spots in particular circumstances. Although statistical significance of hot spots is an important consideration, ecological significance ultimately should be the deciding factor in managing forest habitat to maintain forest biodiversity.

Knowledge of the dominant types of fragmentation in an area could help to choose an appropriate management regime to achieve a certain management goal. For example, if the goal is to maximize the production efficiency of intact forest, then hot spots of perforated forest could be targeted by a management plan designed to fill in the holes in forest patches. The most efficient management regimes for maintaining intact forest might be suggested by looking at the places where there is now a low rate of forest perforations. All types of fragmentation are generally widespread, but the existence of geographic hot spots means that a management strategy targeted at one type of fragmentation will not have uniform benefits in all ecological provinces. At the same time, substantial within-province variation means that ecological provinces are probably not ideal for regional planning. Hot spots of fragmentation are generally smaller than ecological provinces, and land management strategies might be better informed by the hot-spot boundaries than by the boundaries of ecological provinces that contain the hot spots.

## Acknowledgments

Funding was provided by the Strategic Planning and Resource Assessment Staff and Forest Health Monitoring Program of the US Forest Service, and by the Landscape Ecology Branch of the US Environmental Protection Agency through an Interagency Agreement (DW12939283-01-0). The MRLC Consortium supplied the land-cover data, and computing facilities were provided by the Center for Landscape Pattern Analysis. Mention of tradenames does not constitute endorsement or recommendation for use.

## Literature Cited

Achard, F., H. D. Eva, H.-J. Stibig, P. Mayaux, J. Gallego, T. Richards, and J.-P. Malingreau. 2002. Determination of deforestation rates of the world's humid tropical forests. *Science* 297:999–1002.

Bailey, R. G. 1995. Descriptions of the ecoregions of the United States, 2nd ed. Miscellaneous Publication No. 1391, Map scale 1:7,500,000, US Department of Agriculture, Forest Service, Washington, DC, 108 pp.

Bogaert, J. 2003. Lack of agreement on fragmentation metrics blurs correspondence between fragmentation experiments and predicted effects. *Conservation Ecology* 7(1): r6 [online] http://www.consecol.org/vol7/iss1/resp6.

Brook, B. W., N. S. Sodhi, and P. K. L. Ng. 2003. Catastrophic extinctions follow deforestation in Singapore. *Nature* 424:420–426.

Civco, D. L., J. D. Hurd, E. H. Wilson, C. L. Arnold, and M. P. Prisloe Jr. 2002. Quantifying and describing urbanizing landscapes in the northeast United States. *Photogrammetric Engineering & Remote Sensing* 68:1083–1090.

Coulston, J. W., and K. H. Riitters. 2003. Geographic analysis of forest health indicators using spatial scan statistics. *Environmental Management* 31:764–773.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. *Journal of Wildlife Management* 61:603–610.

Fortin, M.-J., B. Boots, F. Csillag, and T. K. Remmel. 2003. On the role of spatial stochastic models in understanding landscape indices in ecology. *Oikos* 102:203–212.

Geographic Data Technology. 2002. Dynamap/2000 user manual. Geographic Data Technology, Inc., Lebanon, New Hampshire.

Gustafson, E. J. 1998. Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems* 1:143–156.

Heilman, G. E., Jr., J. R. Strittholt, N. C. Slosser, and D. A. Dellasala. 2002. Forest fragmentation of the conterminous United States: assessing forest intactness through road density and spatial characteristics. *BioScience* 52:411–422.

Jelinski, D. E., and J. Wu. 1996. The modifiable areal unit problem and implications for landscape ecology. *Landscape Ecology* 11:129–140.

Kulldorff, M. 1997. A spatial scan statistic. *Communications in Statistics: Theory and Methods* 26:1481–1496.

Kulldorff, M. 1999. Spatial scan statistics: Models, calculations and applications. Pages 303–322 *in* N. Balakrishnan and J. Glaz (eds.), Recent advances on scan statistics and applications. Birkhauser, Boston.

Kulldorff, M., and N. Nagarwalla. 1995. Spatial disease clusters: detection and inference. *Statistics in Medicine* 14:799–810.

Kulldorff, M., and Information Management Services, Inc. 2002. SaTScan v. 3.1: Software for the spatial and space-time scan statistics. http://www.satscan.org/.

Levin, S. A. 1976. Spatial patterning and the structure of ecological communities. *Lecture on Mathematics in the Life Sciences* 81:1–35.

Matthews, E. 2001. Understanding the FRA 2000. Forest Briefing No. 1, World Resources Institute, Washington, DC.

McIntyre, S., and R. Hobbs. 1999. A framework for conceptualizing human effects on landscapes and its relevance to management and research models. *Conservation Biology* 13:1282–1292.

Montréal Process Liaison Office. 2000. Montréal process year 2000 progress report: Progress and innovation in implementing criteria and indicators for the conservation and sustainable management of temperate and boreal forests. The Montréal Process Liaison Office, Canadian Forest Service, Ottawa, Canada.

Patil, G. P., R. P. Brooks, W. L. Myers, D. Rapport, and C. Taillie. 2001. Ecosystem health and its measurement at landscape scale: Toward the next generation of quantitative assessments. *Ecosystem Health* 7:307–316.

Patil, G. P., and C. Taillie. 2004. Upper level set scan statistic for detecting arbitrarily shaped *hotspots*. *Environmental and Ecological Statistics* 11:183–197.

Riitters, K. H., R. V. O'Neill, and K. B. Jones. 1997. Assessing habitat suitability at multiple scales: A landscape-level approach. *Biological Conservation* 81:191–202.

Riitters, K. H., J. D. Wickham, R. V. O'Neill, K. B. Jones, and E. R. Smith. 2000. Global-scale patterns of forest fragmentation. *Conservation Ecology* 4(2):3 [online] URL: http://www.consecol.org/vol4/iss2/art3.

Riitters, K. H., J. D. Wickham, R. V. O'Neill, K. B. Jones, E. R. Smith, J. W. Coulston, T. G. Wade, and J. H. Smith. 2002. Fragmentation of continental United States forests. *Ecosystems* 5:815–822.

Riitters, K. H., and J. D. Wickham. 2003. How far to the nearest road? *Frontiers in Ecology and the Environment* 1:125–129.

Riitters, K. H., J. D. Wickham, and J. W. Coulston. 2004. A preliminary assessment of Montréal Process indicators of forest fragmentation for the United States. *Environmental Monitoring and Assessment* 91:257–276.

USDA Forest Service. 2001. 2000 RPA assessment of forest and range lands. US Department of Agriculture, Forest Service, Washington, DC.

USDA Natural Resources Conservation Service. 2000. Summary report 1997 national resources inventory. US Department of Agriculture, Natural Resources Conservation Service, Washington, DC.

Vogelmann, J. E., T. Sohl, and S. M. Howard. 1998. Regional characterization of land cover using multiple sources of data. *Photogrammetric Engineering & Remote Sensing* 64:45–57.

Vogelmann, J. E., S. M. Howard, L. Yang, C.R. Larson, B. K. Wylie, and N. Driel. 2001. Completion of the 1990s national land cover data set for the conterminous United

States from Landsat Thematic Mapper data and ancillary data sources. *Photogrammetric Engineering & Remote Sensing* 67:650–662.

Wade, T. G., K. H. Riitters, J. D. Wickham, and K. B. Jones. 2003. Distribution and causes of global forest fragmentation. *Conservation Ecology* 7(2):7 [online] URL: http://www.consecol.org/vol7/iss2/art7.

Whitmore T.C., and J. A. Sayer (eds.). 1992. Tropical Deforestation and Species Extinction. Kluwer Academic Publishers, Dordrecht.

Wiens, J. A. 1989. Spatial scaling in ecology. *Functional Ecology* 3:385–397.

Wu, J. 2004. Effects of changing scale on landscape pattern analysis: scaling relations. *Landscape Ecology* 19:125–138.