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Library of Congress Cataloging-in-Publication Data

Managing for healthy ecosystems / edited by David J. Rapport ... [et al.].
p. cm.
Papers from an international congress held at the University of California, Davis, Aug. 1999.
Includes bibliographical references.
ISBN 1-56670-612-2 (alk. paper)
1. Ecosystem management—Congresses. I. Rapport, David.

QH75 .A1 M37 2002
333.95—dc21 2002070099

Direct all inquiries to CRC Press LLC, 2000 N.W. Corporate Blvd., Boca Raton, Florida 33431.

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Lewis Publishers is an imprint of CRC Press LLC

International Standard Book Number 1-56670-612-2
Library of Congress Card Number 2002070099
Printed in the United States of America 1 2 3 4 5 6 7 8 9 0
Printed on acid-free paper

CHAPTER 61

Application of Landscape Models to Alternative Futures Analyses

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INTRODUCTION

Scientists and environmental managers alike are concerned about broadscale changes in land use and landscape pattern and their cumulative impact on environmental and economic end points, such as water quality and quantity, species habitat, productivity, erosion potential, recreational value, and overall ecological health (Rapport et al., 1998). They also are interested in predicting short- and long-term future impacts on ecological goods and services based on current land management policies and decisions (Steinitz, 1996). Because we have the means to adjust land management policies, it is worthwhile to develop approaches that can predict the consequences (alternative futures) of different land management policies for different environmental end points. This type of analysis can, for example, allow decision makers in resource conservation and restoration programs to estimate how they can get the most ecological benefit for the least cost (Steinitz, 1996).

Modeling alternative futures can be a simple or complicated process depending on the method employed and the environmental end point in question. The questions and end points of interest are paramount in developing a valid model. Very different models, for example, are needed to predict terrestrial wildlife habitat vs. aquatic conditions. Even within a broad category such as aquatic conditions, different models may be appropriate for different aspects of aquatic conditions (e.g., macroinvertebrate health, pesticide toxicity, eutrophication potential). While different models may be required for different end points, common landscape composition and pattern metrics may be employed in a number of models for a suite of environmental end points. For example, percent natural cover or road density in a measurement unit (e.g., watershed) is an important factor for many environmental end points, including water quality and quantity, wildlife habitat, erosion potential, and recreational value (Burns, 1972; Harden, 1992; Saunders et al., 1992; Kattan et al., 1994; Koopowitz et al., 1994; Short and Turner, 1994; Jones et al., 2000).

Different methods have been proposed and used to predict future conditions, but the basic premise is the same; they are predicated on (1) what land managers and the public want based on needs and values, and (2) the biophysical constraints of the environment (Steinitz, 1996). Models may be developed by establishing trends based on past and present conditions and projecting those trends into the future. They also may be developed empirically by assessing how conditions of

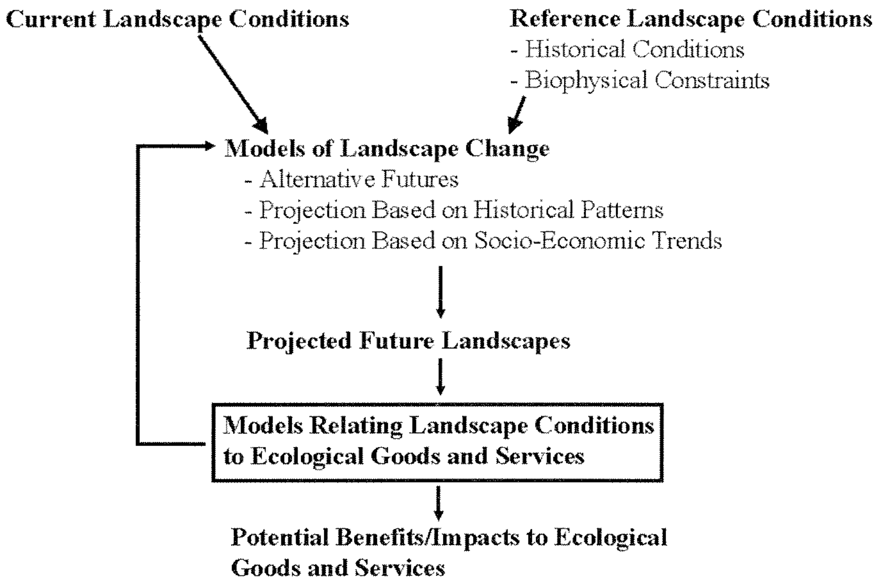

Figure 61.1 The use of landscape models in assessing the consequences of alternative landscape futures.

environmental end points vary with landscape composition and pattern (substituting spatial variability for time) and then manipulating the landscape conditions for different scenarios to project environmental end point conditions based on those different scenarios. Figure 61.1 describes the process used to apply a landscape model to alternative futures analysis.

This chapter will describe a model to predict nitrogen loading, one aspect important to water quality of streams, from a suite of landscape metrics and then will apply this model to a series of alternative future landscapes. This example also will illustrate important issues to consider when developing models for future conditions. Although we will describe only the process for modeling nitrogen loading, the methods presented could easily be applied to other environmental end points. A similar approach, using virtually the same suite of landscape pattern metrics, was used in an assessment relating landscape metrics to breeding bird richness in the Middle Atlantic region (Jones et al., 2000).

BACKGROUND INFORMATION FOR EXAMPLE MODEL

Key to model development was establishing a quantitative relationship between landscape pattern metrics and nitrogen loading to streams. Nitrogen is of particular interest to the U.S. Environmental Protection Agency (EPA) because, as a nutrient, it is essential to the health and continued functioning of natural ecosystems. However, when nutrient inputs exceed the assimilative capacity of a water body system, the system progresses toward hypereutrophic conditions. Excessive nutrient loadings can result in excessive growth of macrophytes or phytoplankton and potentially harmful algal blooms (HAB), leading to oxygen declines, imbalances between prey and predator species, public health concerns, and a general decline of the aquatic resource (U.S. Environmental Protection Agency, 1998).

A number of studies have shown strong relationships between water quality and landscape characteristics. A decrease in natural vegetation indicates a potential for future water quality problems (Likens et al., 1977; Franklin, 1992; Walker et al., 1993; Hunsaker and Levine, 1995; Smith et al., 1997). Many studies have shown that land use within a watershed can account for much of the variability in stream and estuary water quality (Omernik et al., 1981; Omernik, 1987;

Hunsaker et al., 1992; Charbonneau and Kondolf, 1993; Roth et al., 1996; Herlihy et al., 1998; Jones et al., 2001). Changes in landscape conditions in the riparian zone and in areas surrounding water quality sample sites may have a greater influence on water quality than broader-scale watershed conditions (Lowrance et al., 1984). The relationship between intact riparian areas and high water quality is well established, especially in the eastern U.S. (Karr and Schlosser, 1978; Yates and Sheridan, 1983; Lowrance et al., 1984; Cooper et al., 1987). Riparian habitat functions as a sponge, greatly reducing nutrient and sediment runoff into streams (Peterjohn and Correll. 1984; Cooper et al. 1987). Wetlands also play an important role in reducing nutrient loads to surface waters (Weller et al., 1996). High amounts of impervious surface and roads on watersheds also may result in high loadings of nutrients and sediment to streams (Burns, 1972; Harden, 1992; Arnold and Gibbons, 1996), and atmospheric deposition may be a significant source of nitrogen in surface waters (Stensland et al., 1986). Degraded water quality and quantity can, in turn, affect many other environmental end points such as species habitat, productivity, recreational value, and overall ecological health.

In 1996, a regional-scale land-cover database was developed for the five-state area of the U.S. Middle Atlantic region, and this database, along with other regional landscape coverages (e.g., topography, soils, road networks, stream networks, and human population density) were used to assess landscape conditions across the entire region down to a scale of 30 m (Jones et al., 1997). The assessment used a set of landscape metrics (O'Neill et al., 1988, 1997) to evaluate the spatial patterns of human-induced stresses and the spatial arrangement of forest, forest-edge, and riparian habitats. Advances in computer technology and geographic information systems (GISs) have made it possible to calculate landscape metrics over large areas (e.g., regions) at relatively fine scales (e.g., down to 30 m).

Using landscape metric data generated from Jones et al. (1997), and nitrogen loading data provided by the U.S. Geological Survey (USGS) (Langland et al., 1995), we developed a preliminary model predicting nitrogen loading to streams from landscape metrics for a subset of Middle Atlantic watersheds found in the Chesapeake Bay basin (Jones et al., 2001). The analyses presented in this chapter are demonstrative only and should not be construed as an ultimate model for predicting future nitrogen loading. The research to develop predictive nitrogen loading models based on landscape metrics is still ongoing.

METHODS

The obvious first step in developing alternative future models is to collect the appropriate input data for the models. In some cases, the researchers may have the luxury of designing and collecting their own data. In many cases, however, we are limited to existing data, especially if we want to include historical data in our analyses.

In this project, data were compiled from two independent sources. The nitrogen yield data were acquired from the USGS (U.S. Geological Survey, 1995) and the landscape metric data were derived from the data used by Jones et al. (1997).

The USGS calculated annual nutrient and suspended-sediment loads and yields for 148 nontidal streams within the Chesapeake Bay basin. The Chesapeake Bay basin contains more than 150,000 stream miles in the District of Columbia and parts of New York, Pennsylvania, Maryland, Virginia, West Virginia, and Delaware. The basin comprises six major river systems: the Susquehanna, Patuxent, Potomac, Rappahannock, York, and James Rivers (Langland et. al., 1998). The USGS annual nitrogen yield estimates were based on the USGS water year which is October 1 through September 30. We calculated a median annual nitrogen yield based on the yields for the years 1989 through 1996. The inputs for the USGS model were measured concentration of nitrogen in milligrams per liter, measured discharge in cubic feet per second, and time measured in decimal years. USGS model methods and results are described in detail in Langland et al. (1998). The USGS annual loads and yields are based on two different sampling regimes: flow-driven and fixed-interval sampling. In the flow-driven (or total stream flow) sampling regime, samples are collected on the basis of stream flow conditions. Fixed-interval sampling programs collect samples on a regular schedule, usually monthly or quarterly (Langland et al., 1995). Loads are reported by USGS in tons per year. Yields, which have been normalized by watershed area, are reported in pounds per acre. Our analyses were conducted with the yield data only and we combined data for flow-driven and fixed-interval sampling.

Watershed support areas were delineated using Arc/Info GIS software (ESRI, 1996) for each of the USGS water quality monitoring locations so our support areas consisted of only that part of the watershed actually contributing to the water quality monitoring point. For the landscape metrics, we acquired digital coverages of landscape metrics generated by Jones et al. (1997) and then calculated landscape metrics for each of the delineated watersheds. The source of the land cover map from which many of the landscape metrics were derived was the Multi-Resolution Land Characteristics (MRLC) project (Vogelmann et al., 1998). The MRLC data were derived from Landsat Thematic Mapper and had a resolution of 30 m and 15 land cover classes (Vogelmann et al., 1998). However, before calculating the landscape metrics, we used an Arc/Info routine to aggregate the 15 land cover classes into six classes: urban, agriculture, wetland, forest, barren, and water. The landscape metrics used in this analysis are listed in Table 61.1.

Figure 61.2 shows the delineated watersheds used in this analysis in the context of the entire Middle Atlantic region. Although it cannot be detected in Figure 61.2, several of the watersheds are actually nested within larger watersheds. The points on the figure represent water quality data collection sites. Any delineated watershed that did not overlap the Middle Atlantic study area by at least 75%, such as those in the northern portion of the study area, were deleted from the analysis.

Table 61.1. List of Landscape Metrics Compared to Nitrogen Loads[a]

Name of Metric	Explanation
Riparian agriculture	Percentage of watershed with agricultural land cover adjacent to stream edge; 1 pixel wide
Riparian forest	Percentage of watersheds with forest land cover adjacent to stream edge; 1 pixel wide
Forest fragmentation	Forest fragmentation index for watershed; of all edges in the watershed involving at least 1 forested pixel, the percent that joins a forest pixel to a nonforest pixel; higher values indicate higher fragmentation
Road density	Road density for watershed expressed as an average number of kilometers of roads per square kilometer of watershed; normalized to approximate scale of land-cover metrics
Forest land cover	Percentage of watershed with forest land cover
Agricultural land cover	Percentage of watershed with agricultural land cover (pasture/crops)
Agricultural land cover on steep slopes	Percentage of watershed with agriculture occurring on slopes greater than 3%
Nitrate deposition	Estimated average annual wet deposition of nitrate
Potential soil loss	Proportion of watershed with the potential for soil losses greater than 1 ton per acre per year
Roads near streams	Proportion of total stream length that has roads within 30 m; normalized to approximate scale of land-cover metrics
Slope gradient	Average percent slope gradient for watershed
Slope gradient range	Percentage slope gradient range (maximum–minimum) for watershed
Slope gradient variance	Percentage slope gradient variance for watershed; normalized to approximate scale of land-cover metrics
Urban land cover	Percentage of watershed with urban land cover
Wetland land cover	Percentage of watershed with wetland land cover
Barren land cover	Percentage of watershed with barren land cover; includes quarry areas, coal mines, and transitional areas, such as clear-cut areas

[a] Calculation methods and details of each indicator can be found in Jones et al. (1997). Metrics were calculated for each watershed support area.

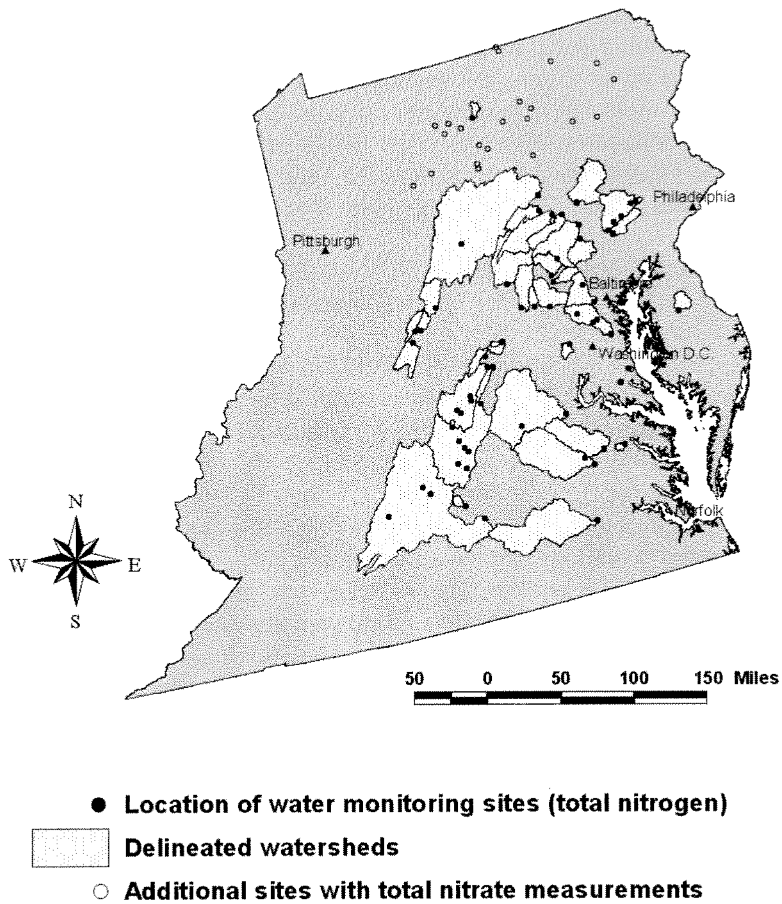

Figure 61.2 Location of water sampling points and their associated delineated watersheds.

We first examined the individual relationships between nitrogen yield (dependent variable) and landscape metrics (independent variables) using individual scatter plots and preliminary regression analyses (SAS Institute, 1990). From our preliminary analyses, we concluded that a log transformation of the nitrogen yield data was necessary. We also performed a square-root transformation of the nitrate deposition landscape variable to linearize the relationship between it and the dependent variables.

We then ran several different regression analyses (SAS Institute, 1990) for nitrogen with the suite of landscape metrics listed in Table 61.1 to help us understand the importance of each variable. We selected a model based on four requirements:

1. The model had to be valid (i.e., basic principles of regression analysis, such as normality, constancy, and independency of error terms were not violated).
2. The model explained a high proportion of the variance in the nitrogen yield.
3. The model included variables over which we may have more control (e.g., selecting riparian forest as a predictor variable over agricultural land cover; agricultural land cover may have explained more overall variance in the model had it been included, but there is a low likelihood of changing the amounts of agriculture in a watershed significantly).
4. The model did not include variables that were highly collinear, as it is difficult to separate the effects of different variables when multicollinearity is an issue.

After selecting the model, we applied it to three future scenarios for all watersheds (defined by USGS hydrologic accounting units) in the Middle Atlantic region. The variables selected for this regression model, based on the criteria discussed above, were amount of riparian forest cover and air nitrate deposition. Amount of riparian forest cover is defined as the percentage of watershed with forest land cover adjacent to the stream edge with a buffer size of 30 m (i.e., one 30 × 30 m picture element). Air nitrate deposition is a modeled value that represents the estimated average annual wet deposition of nitrate. The resulting model from this analysis was:

$$\text{total nitrogen yield} = \exp[0.9056 - 0.02769(\text{riparian forest cover}) + 0.00168\,(\text{nitrate deposition})]$$

This model, based on 69 observations, explained 87% of the variation in nitrogen loading values with 68% explained by riparian forest and 19% explained by nitrate deposition. We next applied this model to the landscape metrics (riparian forest and nitrate deposition) from Jones et al. (1997) for the USGS hydrologic accounting units and manipulated the amount of riparian forest to create three future scenarios for nitrogen loading.

We subsequently applied this model to all the hydrologic accounting units in the Middle Atlantic region based on current conditions of riparian forest and nitrate deposition. We then projected a future scenario if the current amount of riparian forest were increased by an additional 10% to a maximum value of 100%. We also projected a future scenario if the amount of riparian forest was decreased by 10% to a minimum value of 0%. These predictions assume that the amount of nitrate deposition remains constant. This is a simple model and does not take into account several potentially important variables, such as loadings from point sources, hydrogeologic conditions, groundwater contribution, and other highly correlated but important variables, such as total amount of agriculture on the watershed. It also does not take into account other things that could affect future scenarios (e.g., probable population growth patterns).

RESULTS

Figure 61.3 shows nitrogen loading conditions as predicted by current conditions. Figure 61.4 shows a future scenario if the amount of riparian forest were increased by 10%. Figure 61.5 shows a future scenario if the amount of riparian forest were decreased by 10%. These figures demonstrate that by adjusting the riparian forest percentages, we could have a significant impact on nitrogen loading to streams. The pattern of transition from lower to higher nitrogen loadings to streams is clear in these three figures. The cutoff values for each of the five classes shown in Figures 61.3, 61.4, and 61.5 are based on quantiles and were defined from current condition data (Figure 61.3). Cutoff values perhaps would be more meaningful if they were based on some known water quality criteria for nitrogen such as those developed on an ecogregional basis by the EPA (U.S. Environmental Protection Agency, 2000).

A potentially serious limitation of this analysis is extrapolating the model derived from limited spatial coverage (Figure 61.1) to a larger spatial extent. The farther away we get from those watersheds shown in Figure 61.1 (i.e., to the north and the west), the less reliable our model may be. We may be able to justify extrapolating to the north by examining the relationship between total nitrate and the same landscape metrics. We had total nitrate data for 79 sites, including 20 sampling sites to the north of those shown in Figure 61.1. The resulting model, if we regress total nitrate on the same two landscape variables (riparian forest and nitrate deposition), is as follows:

$$\text{total nitrate load} = \exp[0.9621 - 0.0366(\text{riparian forest cover}) + 0.00163\,(\text{nitrate deposition})]$$

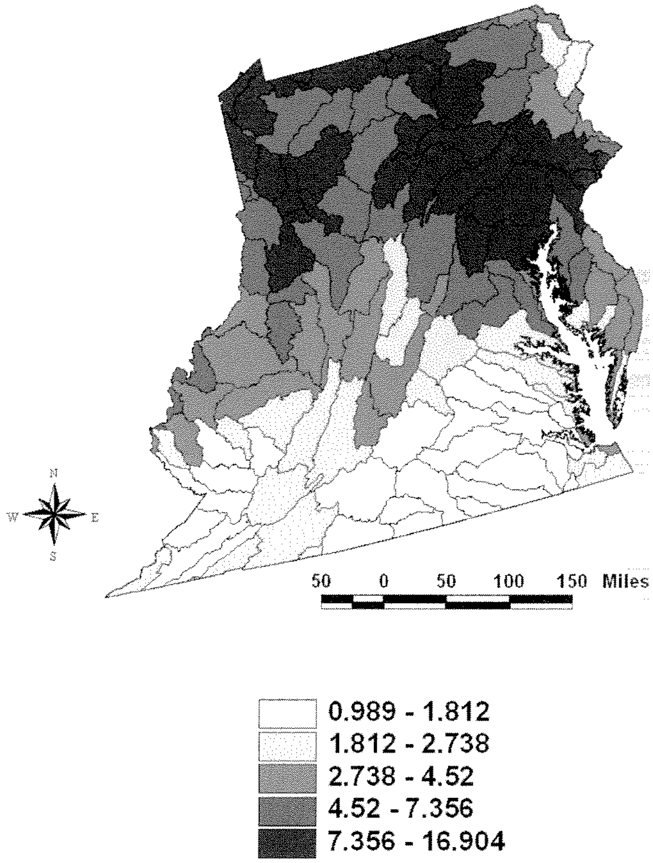

Figure 61.3 Nitrogen loadings (lbs/acre/year) as predicted by present conditions.

This model, based on 79 observations, explained 78% of the variation in nitrate loading values with 55% explained by riparian forest and 23% explained by nitrate deposition. While this model is somewhat similar to the model for total nitrogen, riparian forest explains less of the variation but has a more severe impact on nitrogen loading (steeper regression slope).

Another potential limitation of the model concerns the important issue of scale. The model was developed based on data from multiple scales ranging from very small to very large watersheds; some of these were larger than the USGS hydrologic accounting units and many of them were smaller. The model parameter estimates could be very different if, for example, all the watersheds were based on either very small or very large watersheds.

DISCUSSION

Wall-to-wall landscape data of relatively fine spatial scale (e.g., 30 m), and field sample measurements of a variety of stream chemistry parameters, permit the development of spatially distributed empirical models that can be used to assess potential loadings to streams across an entire region. As demonstrated in this chapter, the development of these models is critical to assess how future landscape scenarios might affect stream water quality across a region. In our study, riparian extent along streams was a strong predictor of nitrogen yield. Because of this relationship, we were able to assess how two future scenarios of riparian habitat condition would affect nitrogen loadings to streams at a watershed or catchment scale. We used two simple

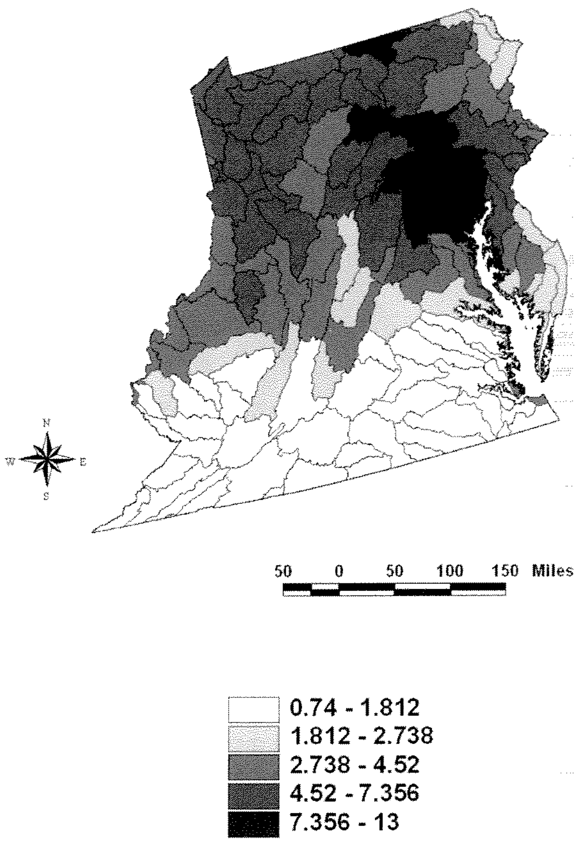

Figure 61.4 Nitrogen loadings (lbs/acre/year) as predicted with a 10% increase of riparian forest.

future scenarios — one with a 10% increase in the amount of riparian habitat at the catchment scale and one with a 10% decrease in riparian habitat — to demonstrate how the approach can be used to target those watersheds that would benefit the most from an increase in riparian habitat (restoration) and those that would be at the greatest risk due to decreases in riparian habitat (catchments needing protection). The decreased riparian habitat scenario is similar to a treadmill stress test in humans (after the ecohealth concept of Rapport et al., 1998). It finds those catchments where streams are most vulnerable to increases in nitrogen load and, therefore, a potential dramatic effect on human (waterborne diseases and drinking water quality) and ecological health (stream biota).

Our modeling approach differs from other watershed or catchment models that predict nutrient loadings in that it considers the spatial pattern and distribution of key landscape features; most existing models only consider the percentage of land cover at the catchment scale (U.S. Department of Agriculture, 1972; Liang et al., 1994). Additionally, most existing models lack riparian and atmospheric deposition parameters. Riparian habitats can be significant filters of nutrient inputs to streams (Lowrance et al., 1984; Peterjohn and Correll, 1984; Cooper et al., 1987) and atmospheric nitrogen deposition can be a significant source of nitrogen input at the catchment scale (Stensland et al., 1986; Appleton, 1995).

Our modeling approach is a significant improvement over the comparative watershed approach used by Jones et al. (1997) and Wickham et al. (1999a) because landscape metrics were quantitatively linked to stream conditions. Jones et al. (1997) and Wickham et al. (1999a) ranked the relative vulnerability of watersheds but did not link their ranking to observed water quality.

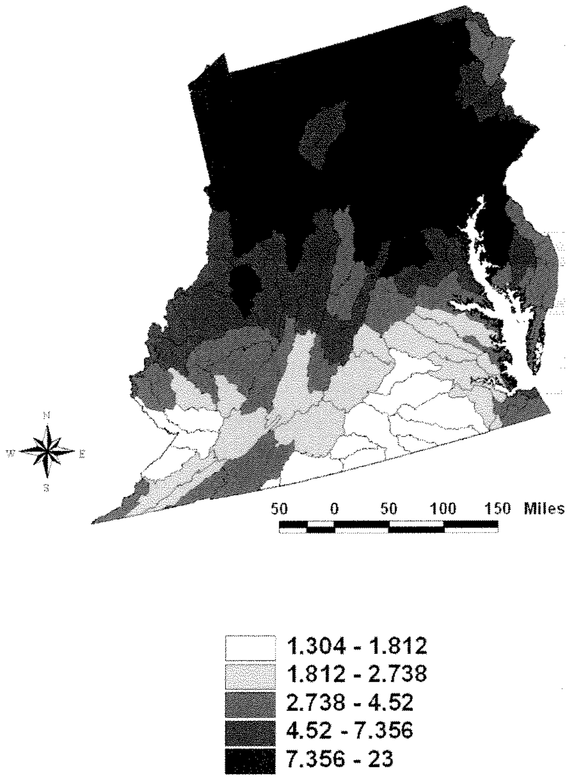

Figure 61.5 Nitrogen loadings (lbs/acre/year) as predicted with a 10% loss of riparian forest.

Because our model includes the spatial pattern of key landscape features, it is possible to find those areas that would benefit most from riparian restoration and riparian protection (those catchments at risk under the reduced riparian habitat scenario). Such information can be used by land managers and private landowners to decide where to restore and protect riparian habitats. This is important because budgets for restoration and protection are often limited. Wickham et al. (1999b) used a similar approach to identify those catchments that had the greatest potential to restore forest connectivity. Such targeting approaches are needed to identify restoration and protection opportunities for other ecological goods and services (e.g., flood abatement, wildlife habitat, and forest, rangeland, and agricultural productivity).

The spatially distributed model demonstrated in this chapter can be used to assess the consequences of other landscape scenarios. For example, the model could be applied to assess the potential change in nitrogen loading to streams associated with different land management scenarios generated from public workshops (Steinitz, 1996). It also could be used in combination with socioeconomic, land-cover change models (Wickham et al., 2000) to assess the consequences of socioeconomic futures on nitrogen loadings to streams.

Our modeling approach requires regionally consistent landscape data and a fairly extensive network of water quality samples. Therefore, availability of regional-scale, land-cover data similar to that being developed by the MRLC Consortium in the U.S. (Vogelmann et al., 1998), and stream samples similar to those being collected by the Environmental Monitoring and Assessment Program program in the U.S. and the Waterwatch program in Australia (Waterwatch, 1997), is critical to the use of our modeling approach. Additionally, since quantitative relationships between landscape pattern and stream water quality vary between different biophysical settings (Omernik et al., 1981; Clarke et al., 1991; Jones et al., 2001), water quality sampling must be sufficient to represent different biophysical settings.

ACKNOWLEDGMENTS

We thank Scott Phillips and Judy Denver of the USGS for providing information on nutrient parameters used in this study. The U.S. Environmental Protection Agency (EPA), through its Office of Research and Development, partially funded and collaborated in the research described here. It has not been peer-reviewed by the EPA. Mention of trade names or commercial products does not constitute endorsement or recommendation by the EPA for use.

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