Monitoring Environmental Quality at the Landscape Scale

Using landscape indicators to assess biotic diversity, watershed integrity, and landscape stability

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Over the past century, technological advances have greatly improved the standard of living in the United States. But these same advances have caused sweeping environmental changes, often unforeseen and potentially irreparable. Ethical stewardship of the environment requires that society monitor and assess environmental changes at the national scale with a view toward the conservation and wise management of our natural resources.

Some of the most important environmental changes occur at the spatial scale of landscapes. Obvious examples include clearcutting for lumber, urbanization, the loss of wetlands, and the conversion of forest and prairies into crop and grazing systems. Decisions about how to change land cover may be made by individual landowners, but their impacts are seen cumulatively, as a change in spatial pattern on the landscape. The landscape scale is also important because political decisions to manage natural resources are made at broad scales, such as river basins, forest districts, and states.

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Remote imagery, geographic information systems, and principles from landscape ecology can be combined into a powerful approach for monitoring environmental quality

Landscape changes have direct impacts on ecological processes (Forman and Godron 1986). In fact, ecological interactions often produce the spatial pattern on the landscape. For example, Levin (1976, 1978) showed that predator–prey interactions, combined with spatial movement, can result in a patchy spatial pattern of the populations. Paine and Levin (1981) demonstrated that cycles of disturbance and recovery also produce spatial pattern. In turn, spatial pattern influences the ways in which organisms move on the landscape (Wiens and Milne 1989) and use resources (O'Neill et al. 1988b). Dispersal processes interact with spatial pattern to separate competitors in space (Comins and Noble 1985), making their coexistence possible. The relationship between spatial pattern and coexistence has been shown for both animals (Kareiva 1986) and plants (Pacala 1987). Changes in spatial pattern in the form of habitat fragmentation have been implicated in the decline of biological diversity and in the ability of the ecosystem to recover from disturbances (Flather et al. 1992).

Determining status and trends in the pattern of landscapes can, therefore, be useful for understanding the overall condition of ecological resources (Graham et al. 1991, Urban et al. 1987). The potential now exists to monitor landscapes by combining remote satellite imagery of land cover, geographic information systems (GIS), and advances in landscape ecology. Clearly, however, not all environmental changes can be detected through alterations in land cover. Stream pollution or the replacement of native wildlife with introduced species may cause little or no change in remote imagery. To completely assess the condition of ecological resources, landscape monitoring must be integrated with field studies. Nevertheless, society can begin immediately to evaluate some important changes at broad scales (Hunsaker et al. 1990). In this article, we explore landscape approaches to environmental monitoring, focusing on biotic diversity, watershed integrity, and landscape stability.

Biotic integrity and diversity

One smeasure of biotic integrity and diversity is the frequency distribution of patch sizes of natural vegetation. The most important cause of species loss and the subsequent reduction in species diversity is the loss of habitat. The remaining habitat becomes fragmented into patches—distinct stands of natural vegetation surrounded by land subject to human uses, such as agriculture or urban development. The loss of connecting corridors between the stands of natural habitat cause the patches to become isolated (Forman and Godron 1986). As corridors are lost and habitat becomes disconnected, disturbances can cause local extinctions. Patches that are isolated from seed sources and dispersal pathways have difficulty recovering from disturbances (Wiens 1985).

Some spatial arrangements of patches may be particularly vulnerable to fragmentation. Isolated habitat may be configured in a longitudinal pattern, like a string of pearls. Examples include alpine tundra along ridgetops of the Rockies, dune vegetation along beaches, and granite outcrops. Removal of a single patch may split the entire habitat in two, if the gap exceeds the dispersal ability of the populations.

Watershed integrity

Water quality depends on the landscape's ability to collect and purify water. In addition, intact natural vegetation helps to reduce or control floods and retain soil. With a decrease in natural vegetation (e.g., forests, wetlands, and prairies) comes an increased potential for future water quality problems (Hunsaker and Levine 1995). Land uses within a watershed can account for much of the variability in stream water quality (Omernik 1977). Planting crops on slopes greater than 3%, for example, increases the risk of erosion (Wischmeier and Smith 1978). Both empirical studies and models have established the causal relationship between watershed characteristics and nutrient and sediment loads to streams (Levine et al. 1993). For example, a drastic change in vegetation cover, such as clearcutting in the Pacific Northwest, can almost double runoff (Franklin 1992).

Hydrologically active areas—areas within a watershed that produce surface runoff—are often associated with riparian and wetland habitats. Intact riparian areas are associated with high water quality (Karr and Schlosser 1978, Lowrance et al. 1984, 1985). Riparian habitat functions as a "sponge," greatly reducing nutrient and sediment runoff into streams (Peterjohn and Correll 1984).

Landscape resilience

Landscape resilience refers to the rate at which vegetation on the landscape recovers after a disturbance. As habitat is fragmented, distances increase to source areas that provide seeds and animal migrants needed for recovery. For example, northern hardwoods normally take 60–80 years to replace biomass and nutrients that are lost in harvesting (Likens et al. 1978). However, this recovery time is significantly increased if distances to seed sources are increased or if topsoil is lost through erosion. Therefore, resilience can be related to the distance between patches.

Experience with erosion in the American plains and desertification in the African Sahel demonstrates that critical thresholds exist in landscape pattern. Beyond these thresholds, positive feedbacks can take over that drive the landscape into new, undesirable configurations (Schlesinger et al. 1990). For example, Grover and Musick (1990) have shown that grazing and climate interact to allow shrubs to encroach on natural grasslands. Shrubland encroachment, in turn, causes accelerated wind erosion, which prevents a stable recovery to grasslands even in the absence of grazing pressure.

Indicators of landscape status

To quantify the relationship between spatial pattern and ecological functions, it is necessary to develop simple metrics that quantify landscape pattern. These metrics can then be correlated with specific aspects of ecosystem function. Changes in spatial metrics are, therefore, indicators of changes in the ecological condition of the landscape.

Indices based on information theory (O'Neill et al. 1988a) and fractal dimension (Milne 1992) summarize basic features of the pattern. A variety of such metrics have been applied to landscape monitoring and assessment (Hunsaker et al. 1994, Riitters et al. 1995). For example, the metric of dominance (O'Neill et al. 1988a) indicates the extent to which the landscape is dominated by a single landcover type. That of contagion expresses the probability that land cover is more "clumped" than the random expectation (Li and Reynolds 1993). Finally, the fractal dimension of patches indicates the extent of human reshaping of landscape structure (Krummel et al. 1987), because humans create simple shapes, whereas nature generates complex configurations. A fractal dimension index can be calculated by regressing the log of the patch perimeter against the log of the patch area for each patch on the landscape. The index equals twice the slope of the regression line. In addition to these general measures of pattern, specific indicators can be suggested for each of the landscape properties discussed above.

Biotic integrity and diversity. The simplest indicator of biotic integrity is the total change in land cover. Changes in natural vegetation cover reflect the loss of wildlife habitat (O'Neill et al. 1992b). One method to assess land cover would be to ask: How does the present land cover compare with the cover that would be in a region if humans were not present? Figure 1 compares Kuchler's (1964) map of potential natural vegetation with Loveland et al.'s (1991) estimate of current vegetation, which was taken from Advanced Very High Resolution Radiometer (AVHRR) satellite imagery (1 km^2 resolution) and augmented with data on urban areas (ESRI 1994). Kuchler defined potential natural vegetation as the vegetation that would exist if humans were removed from the scene and plant succession was completed. Figure 1 uses Omernik's (1987) 13 aggregated ecoregions to compare Kuchler's and Loveland's maps. In addition, Kuchler's 117 cover classes and Loveland's 167 classes were aggregated to the same seven classes: rangeland, forest, wetland, barren, cropland, water, and urban. The comparison reveals that mountainous areas have largely retained natural vegetation, whereas the Atlantic and Gulf Coastal areas, the Midwest, and the central valley of California all show the effects of extensive agriculture and urban development.

Beyond simple change in cover, much of the influence of landscape pattern on ecological processes is due to the spatial configuration of patches (Franklin and Forman 1987, Kareiva 1986). For example, fragmentation of a landscape into many isolated patches has been shown to reduce native biodiversity (Saunders et al. 1991, Wiens 1985). As the distribution of patch sizes changes, the landscape becomes more hospitable to some species and less hospitable to others (Wiens and Milne 1989). The mean, variance, and skewness of the patch size distribution become potential indicators of species change.

The frequency distribution of distances between patches is another indicator of biotic integrity. Nearest-neighbor distances are related to risks incurred by wildlife moving across open areas. Another indicator of change through time would be the number of miles of new roads. Roads fragment the landscape and have an immediate impact on wildlife mortality. Another metric of biotic integrity is the loss of corridors between patches of natural habitat. Wildlife use these corridors to move among resource patches (Mwalyosi 1991).

The length of forest edge on a landscape is also an important indicator of the integrity of wildlife habitat (Gardner et al. 1991). The forest edge forms a unique habitat that is favored by many species. In addition, the ratio of patch size to edge length can be significant. For example, cowbirds on forest edges are brood parasites on warblers and other birds that nest in the forest interior (Harris 1988, Terborgh 1992). Forest patches must be sufficiently large so that warbler nest sites are far enough from edges that cowbirds cannot find them.

Status and trends in landscape potential for specific wildlife can also be quantified (Danielson 1992). Consider a "window" the size of an organism's home range. Within the window are found a variety of habitat requirements, such as vegetation mixture, edge, and available water. By placing the window over a corner of the landscape map, it is possible to determine whether the land covers that are within the window meet all habitat requirements. The window could then be moved systematically over the map, yielding an overall indicator of the status of the landscape for this organism. A suite of windows for individual species, guilds, or populations could be designed by adjusting the resolution of the data, the size of the home range window, and the habitat requirements. This approach provides a simple indicator of the impact on wildlife of a change in landscape pattern.

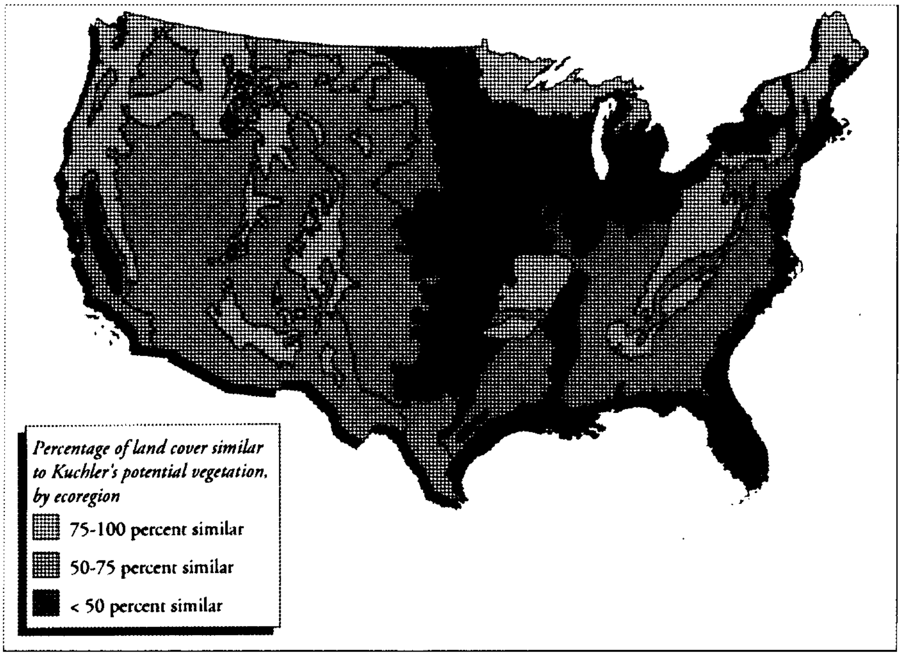

Figure 1. Potential loss of native biodiversity in ecoregions of the United States (Omernik 1987) due to land use conversion and habitat loss. The map compares Kuchler's (1964) potential natural vegetation with Loveland et al.'s (1991) current vegetation analysis. See text for details.

Another potential indicator uses an imaginary organism moving randomly across the landscape, one map unit at a time. The organism steps freely (probability = 1.0) onto natural vegetation, and less freely (probability << 1.0) across clearings, agriculture, or other land uses. By releasing many organisms in a computer simulation, allowing each to take a large number of steps, and recording the number of times a site is visited, it is possible to evaluate how organisms will use a landscape configuration. This approach is particularly valuable when remote imagery indicates a change in landscape pattern. The modeling results then allow one to hypothesize what populations of wildlife might be affected by the change.

Humans themselves can be affected by changes in landscape configurations. For example, human recreation is an important use of the natural vegetation areas on the landscape. Changes in land cover, particularly in the vicinity of urban centers, can mean a tangible loss of environmental quality to the human population. Figure 2 illustrates how remotely sensed land cover could be applied to assess the utility of the landscape for recreation. Circles of 150 km diameter were drawn around the 25 largest metropolitan areas in the United States, and an estimate of recreation potential was obtained by dividing the number of people who live within each census area by the total area of natural land cover (AVHRR data). As Figure 2 shows, urban communities differ by a factor of five or more in their opportunity to experience and enjoy natural areas.

Another indicator of biotic integrity can be developed by weighting individual landcover changes. One

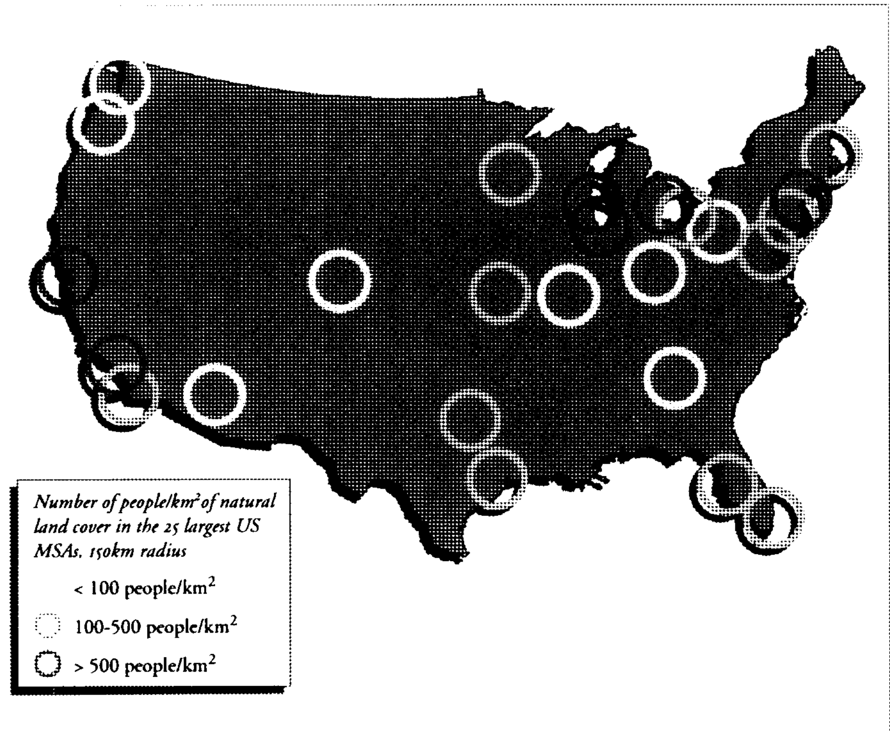

Figure 2. Potential for recreation in natural areas near urban centers in the United States. Circles with 150 km radii are drawn around the 25 largest metropolitan areas, and the number of people per km^2 of natural vegetation is given. Natural land cover includes forest, rangeland, wetland, and water.

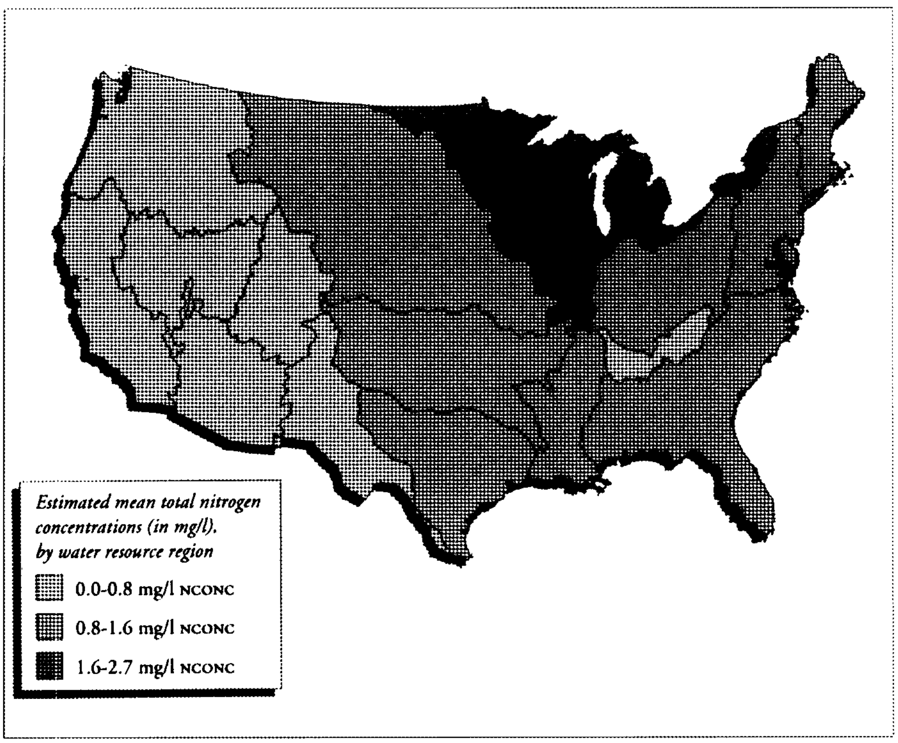

Figure 3. Watershed integrity in the United States as indicated by total nitrogen concentration (NCONC) in surface waters. Estimates are based on the relationship between land cover and nitrogen concentrations established by Omernik (1977) and land cover from Loveland et al. (1991). Data are from the US Geological Survey's Water Resource Regions (Seaber et al. 1984).

might, for example, apply a greater weight to a change that fragments a large patch. Similarly, a change could be multiplied by the probability of forming a barrier to animal movement or distrupting a corridor. It would be important to distinguish between 100 map units scattered randomly and 100 map units in a line, forming a new barrier to animal movement. Individual transitions can also be weighted by characteristics of the entire landscape. In an area with little wetland (or riparian or critical habitat), loss of a habitat site is more important than in a region where this land cover is abundant. Weighting the original data introduces a bias into the analysis, however. Caution must be used with such biased indicators to prevent the weightings, rather than the original cover data, from dominating the analysis.

Watershed integrity and water quality. Nitrogen, phosphorus, turbidity, temperature, and intragravel dissolved oxygen are all indicators of lotic condition (MacDonald et al. 1991). The first four correlate closely with landscape properties (e.g., land cover, topography, and soils). A significant proportion of the nutrient and sediment load in streams enters through runoff from the surrounding landscape. The correlations between landscape properties and lotic condition suggest indicators that relate spatial pattern to water quality (Hunsaker and Levine 1995, Omernik et al. 1981). That is, across a region, increases in agriculture and urban land cover or decreases in natural vegetation indicate a potential for water quality problems.

Figure 3 demonstrates that total nitrogen concentration in surface waters can be estimated from the proportion of agriculture and urban lands on a watershed. Estimates of nitrogen concentration are summarized by US Geological Survey's Water Resource Regions (Seaber et al. 1984) and are based on empirical studies by Omernik (1977) applied to current land cover (Loveland et al. 1991). Figure 3 show that the Tennessee valley and western water resource regions have low nitrogen concentrations (0.0–0.8 mg/l), indicating intact watershed vegetation. The

Great Lakes and upper Midwest have the poorest watershed integrity (nitrogen concentrations are 1.6–2.7 mg/l). For comparison, nitrogen concentrations of 0.01–1.2 mg/l have been reported for undisturbed headwater streams in Oregon (Brown et al. 1973), and of 0.002–0.018 mg/l for an undisturbed hardwood watershed in North Carolina (Swank 1987).

A more refined indicator of watershed integrity might weight land cover by distance to streams, soil type, and slope calculated from digital elevation models. Such an indicator could also take into account the loss of riparian zones, which are important for maintaining water quality in streams (Naiman and Décamps 1990). Possible indicators include changes in width of riparian zones weighted by slope or miles of riparian zone that are narrower than desirable. Similar indicators might be loss of wetlands or formation of contiguous agriculture adjacent to a stream or lake. For example, a landcover change that increases contiguous agriculture along a stream could be weighted more heavily. Once again, however, great care must be taken in using weighted indicators to prevent inherent bias from overwhelming the analysis.

A second type of watershed indicator might focus on the potential for undesirable hydrologic events. A flood indicator could include vegetation cover, slope, and surficial geology. Because hydrologic pathways are altered by road surfaces (Franklin 1992, Swift 1987), a change in miles of road, types of road (width, surface type, and intensity of use), and number of intersections between roads and streams could be used as indicators of flood potential.

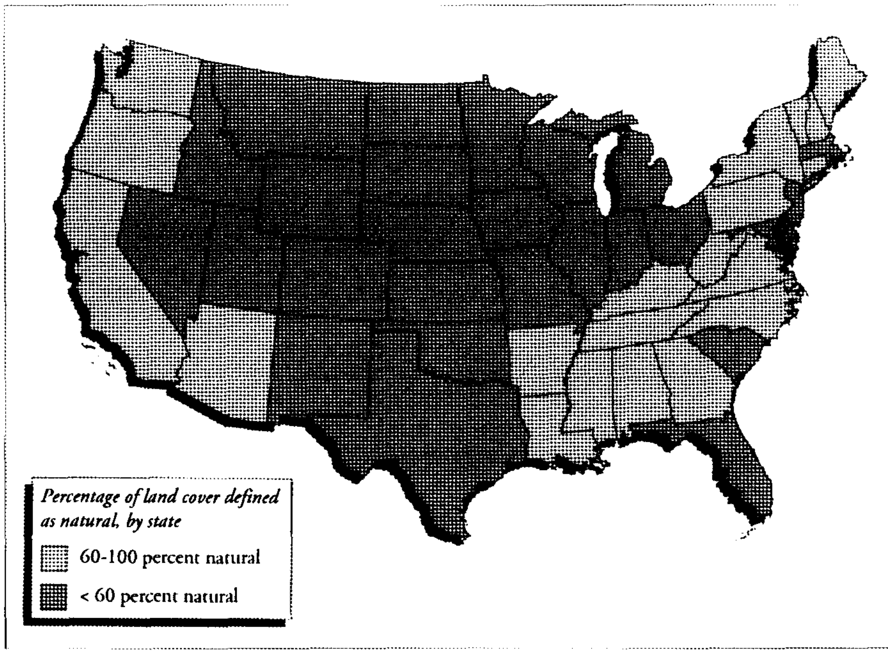

Figure 4. Estimate of landscape sustainability for each of the states in the United States using natural landscape connectedness and the theoretical percolation threshold of 60%. Natural land cover includes forest, rangeland, wetlands, and water. Landcover data are from Loveland et al. (1991).

Landscape fragmentation. Percolation theory (Gardner et al. 1987, Stauffer 1985) provides a framework for assessing landscape resilience by defining thresholds of habitat coverage (Gardner et al. 1992). Simulation studies have shown that on a random map, portrayed as an array of square pixels, the critical value for percentage cover is 59.28%. At this value, there is an abrupt increase in the probability of finding a continuous habitat corridor across the landscape. If percentage cover is reduced below this value, the landscape becomes dissected into isolated patches. The resource utilization scale is derived from percolation theory and measures the scale at which an organism must operate to use the resources on a fragmented landscape (O'Neill et al. 1988b). Percolation theory also permits estimates of diffusion rates and of a percolation backbone, which is defined as the fewest steps needed to traverse the landscape.

Percolation theory is also useful for monitoring the potential for disturbances to spread across the landscape (Turner 1987). Specifically, if disturbance-prone land cover is higher than the threshold value (approximately 60%), a disturbance may be able to spread throughout the landscape (Gardner et al. 1989, 1992). By combining epidemiology theory with percolation theory, it is possible to calculate the probability that a disturbance or pest will spread or become endemic (O'Neill et al. 1992a).

Figure 4 illustrates how the concept of percolation threshold can be applied to broad-scale monitoring of the environment. AVHRR cover data (Loveland et al. 1991) was used to determine for each state whether total natural cover is above or below the 60% threshold. Although a state may seem a strange unit for reporting ecological data, we used a political unit to emphasize how broad-scale assessments might influence political decisions. This politically oriented map shows that along the East Coast and in the central United States, a highly connected natural landscape has been fragmented by agriculture and urban areas. Such an assessment might motivate political action in these regions.

In addition to percolation thresholds, scale theory may provide additional tools for landscape monitoring. For example, empirical studies (O'Neill et al. 1991a, 1991b) have confirmed the prediction from hierarchy theory (O'Neill 1988, 1989, O'Neill et al. 1986, 1989) that landscapes should show pattern at distinct scales (Turner et al. 1991). Disruption of this scaled structure, that is, the loss of pattern at one scale, means that ecological processes that determined that scale of pattern have been disrupted. For example, the process of plant competition might determine the spacing of individual trees. The regular spacing of the trees then appears as a distinct scale of pattern on the landscape. If the process of competition is disrupted, perhaps by an introduced species, the regular spacing disappears and so does the distinct scale of pattern. By analyzing the number of scales from remote imagery, therefore, it should be possible to determine whether the underlying ecological processes have been disrupted.

The relationship between landscape scales and ecological function has been demonstrated by Holling (1992), who took advantage of the close relationship between vertebrate body size and home range to establish that body sizes can be related directly to landscape scales. Animals with large body sizes utilize resources over a large home range and respond to coarse scales of pattern on the landscape. Small animals have small home ranges and respond only to fine scales of pattern. Holling's work makes it possible to relate the loss of a landscape scale to the risk of losing a guild of vertebrates that depend on that particular scale of resource distribution.

Landscape approaches: limitations and potential

In this article, we have illustrated how remote imagery, GIS, and principles from landscape ecology can be combined into a powerful approach for monitoring environmental quality over large regions. This approach supplements, rather than replaces, finer-scaled monitoring. For example, detailed monitoring in specific areas will remain critical to assess and control point-source pollution. But assessing and controlling non–point source pollution, which often results from landcover changes, will require novel, broad-scaled approaches.

Figures 1–4 illustrate what can be accomplished by a landscape monitoring approach. The figures are based on coarse (1 km^2 resolution) AVHRR imagery; finer scales of remote imagery will be needed to implement many of the pattern indicators discussed in this article. The figures are also based on imagery for a single point in time, whereas the real power of the landscape approach lies in quantifying changes and trends in large-scale patterns through time. The analysis of finer-scaled remote imagery at successive points in time will permit a more complete assessment of environmental quality. The Environmental Protection Agency's Environmental Monitoring and Assessment Program is currently focused on acquiring, classifying, and making available the additional fine-scaled remote imagery that can fulfill the potential for landscape monitoring that is only hinted at in our figures.

Considerable research remains to refine and test the landscape monitoring approach. As we have demonstrated, many potential indicators can be proposed; however, multivariate analysis of available indicators (Riitters et al. 1995) show that many of these are highly correlated. In addition to finding a small number of statistically independent metrics, it will be necessary to test the sensitivity of the indicators to measurement and classification errors before they can be considered to be reliable measures of change.

Research is also needed to identify ecological systems that are particularly sensitive to spatial disturbances. Even a casual observer can observe how small alterations in natural land form result in major changes in aridland vegetation. The basic research need is to establish the sensitivity of landscapes to landcover change so that the impact of a measured change in spatial pattern can be evaluated in terms of a potential change in environmental quality.

Despite the many research questions that remain, the potential for a landscape monitoring approach remains exciting. Despite its limitations, the landscape approach is practical within current technologies and less expensive than approaches using only ground-based surveys. Moreover, it focuses directly on the habitat loss that is a critical component of society's impact on the environment. With continued research and advances in technology, landscape monitoring can reach the same levels of efficiency and accessibility that we have come to expect from routine monitoring of the weather.

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