GEOGRAPHIC TECHNIQUES AND RECENT APPLICATIONS OF REMOTE SENSING TO LANDSCAPE-WATER QUALITY STUDIES

JERRY A. GRIFFITH
Department of Geography, University of Southern Mississippi, Hattiesburg, MS, U.S.A.
(Current address: U.S. Geological Survey, EROS Data Center, Sioux Falls, SD 57198, U.S.A.,
e-mail: griffith@usgs.gov, fax: +1 605 594 6529)

(Received 28 August 2000; accepted 1 August 2001)

Abstract. This article overviews recent advances in studies of landscape-water quality relationships using remote sensing techniques. With the increasing feasibility of using remotely-sensed data, landscape-water quality studies can now be more easily performed on regional, multi-state scales. The traditional method of relating land use and land cover to water quality has been extended to include landscape pattern and other landscape information derived from satellite data. Three items are focused on in this article: 1) the increasing recognition of the importance of larger-scale studies of regional water quality that require a landscape perspective; 2) the increasing importance of remotely sensed data, such as the imagery-derived normalized difference vegetation index (NDVI) and vegetation phenological metrics derived from time-series NDVI data; and 3) landscape pattern. In some studies, using landscape pattern metrics explained some of the variation in water quality not explained by land use/cover. However, in some other studies, the NDVI metrics were even more highly correlated to certain water quality parameters than either landscape pattern metrics or land use/cover proportions. Although studies relating landscape pattern metrics to water quality have had mixed results, this recent body of work applying these landscape measures and satellite-derived metrics to water quality analysis has demonstrated their potential usefulness in monitoring watershed conditions across large regions.

Keywords: ecological monitoring, landscape ecology, landscape-water quality studies, NDVI, remote sensing/GIS, stream water quality

1. Overview and Problem Statement

An estimated $70 billion is spent annually in the U.S.A. on environmental regulatory programs to protect ecological resources (Whittier and Paulsen, 1992). In particular, the quality and condition of the nation's water resources have become a prime concern over the past few decades (Carpenter *et al.*, 1998; Loague, 1998; O'Neill *et al.*, 1997). Yet, documentation of regional stream conditions has been problematic. While aquatic regulatory programs have historically used a site-specific approach to protecting and improving water quality, difficulties in assessing less well-defined impacts and effects from non-point source pollution, habitat degradation, and cumulative effects of long-term sublethal levels of contaminants still remain unresolved (Whittier and Paulsen, 1992). In fact, about 80% of stream

Water, Air, and Soil Pollution **138:** 181–197, 2002.

miles go unassessed in the U.S. with respect to the requirements for Section 305(b) of the Clean Water Act (GAO, 2000).

A major factor in the degradation of water quality in streams and rivers has been the heavy application of agricultural chemicals and fertilizers commonly used to increase crop yields (Carpenter *et al.*, 1998; Matson *et al.*, 1997; Kolpin, 1997). Because water quality is so strongly affected by people's use of the land, and because human health is so dependent upon clean water, assessing the condition of aquatic resources is perhaps one of the most important areas of interdisciplinary environmental research (Wear *et al.*, 1998). In addition, since land use and land cover (LULC) are such strong determinants of water quality, methods are needed to efficiently and cost-effectively monitor regional landscape conditions. Despite this need, there is a paucity of programs that monitor LULC-water quality at the regional scale (Herlihy *et al.*, 1998). Remote sensing technology can be of great utility for this type of monitoring.

The usefulness of remote sensing techniques in stream condition/water-quality analyses relates to the geographically widespread nature of non-point source pollution (NPSP) problems, in particular. This characteristic, and the resultant effects on streams of NPSP, necessitate the assessment of watershed condition and water quality on a regional scale and increases the importance of developing landscape-scale studies (He *et al.*, 2000) For example, impacts from non-point source pollution are of special interest in a predominantly agricultural region such as the U.S. Midwest and Great Plains. Reasons for this derive from the physical changes to streams that occur when native grassland is changed to pasture or cropland, such as: 1) the pattern of discharge; 2) the physical nature of the channel; 3) the bed disturbance regime; 4) temperature and light regimes; 5) chemistry; and 6) input of dissolved and particulate terrestrial organic matter (Townsend and Riley, 1999). These physical changes in turn can impact stream biota (Wichert and Rapport, 1998; Townsend *et al.*, 1997; Richards *et al.*, 1996). Due to these impacts from agriculture as well as other land use activities, nutrients, sediment and pesticides reportedly affect 55% of impaired stream miles in the U.S. (Wells, 1992).

2. Background

The following section reviews important aspects of regional landscape and water quality monitoring including: 1) the increased recognition of the importance of landscape-scale approaches to water quality studies; 2) the increased recognition of the role of landscape ecology and landscape pattern in studying human-environment relationships 3) the application of remote sensing technologies to the study of aquatic ecosystems; and 4) landscape-water quality studies in the context of ecological monitoring programs and indicators of ecological condition.

2.1. Landscape-scale approaches to water quality assessment

> 'The character of streams and rivers reflects an integration of physical and biological processes occurring in the catchment, yet ecological studies and management of natural resources have traditionally occurred at the scale of stream reach, forest stand or vegetation plot' (Johnson and Gage, 1997, p. 113).

Hydrologists and aquatic ecologists have long known that the surface across which water travels to a stream or lake has a major effect on water quality. Accordingly, the relative amounts of particular LULC types in a watershed will affect water quality as well. Because water quality integrates geomorphic, hydrologic, and biological processes, it is a fundamental component of a healthy watershed (Hem, 1985). Previous research has documented significant relationships between LULC and water quality (e.g., Basnyat *et al*., 1999; Roth *et al*., 1996; Osborne and Wiley, 1988; Omernik *et al*., 1981; Karr and Schlosser, 1978). In fact, there is a long history of such studies on LULC – water quality relationships (Johnson *et al*., 1997; Allan *et al*., 1997; Roth *et al*., 1996; Basnyat *et al*., 1999; Osborne and Wiley, 1988; Omernik, 1976). Johnson *et al*. (1997) provides a summary of studies which examine the impact of LULC on water quality. In particular, strong relationships have been found between LULC and phosphorus and nitrogen (Bolstad and Swank, 1997; Hall and Schreier, 1996; Collins and Jenkins, 1996; Keeney and DeLuca, 1993; Lowrance *et al*., 1985; Peterjohn and Correll, 1984).

The importance of these interrelationships between LULC and water quality is reflected by the increased recognition over the past two decades of NPSP as a major environmental concern (Loague, 1998; Sharpley and Meyer, 1994). In addition to affecting physical and chemical aspects of water quality, LULC has been shown to affect stream habitat, which, when in concert with water quality, affects stream biological community composition, including fish and macroinvertebrate communities (Schlosser, 1991). Schlosser (1991) contends that land use activities strongly impact fish population and community dynamics. Agriculture, deforestation, and grazing can all affect stream structural relationships by reducing the amount of woody debris entering the stream which in turn affects stream depth and substrate, or by directly removing stream spatial complexity through channelization (Schlosser, 1991). Hence, new techniques should also be appropriate for investigating stream biological quality (e.g., He *et al*., 2000; Hughes *et al*., 2000) as well as for stream physical and chemical indicators (e.g., Jolly *et al*., 1996).

While many studies address the general relationship of LULC to water quality, another avenue of research compares the relative influence of entire catchments versus riparian buffers or other smaller portions of the catchment in determining water quality. Johnson *et al*. (1997) summarizes these studies; some investigators maintain the importance of riparian vegetation in protecting water quality (Carpenter *et al*., 1998; Schlosser and Karr, 1981), while others conclude that entire catchments have greater impact (Roth *et al*., 1996; Hunsaker and Levine, 1995; Osborne and Wiley, 1988; Omernik *et al*., 1981). An increasing number of studies, it appears, have stressed the importance of landscape-level views over more localized stream reach studies (Allan *et al*., 1997; Schlosser, 1991; Johnson and Gage, 1997; Johnson *et al*., 1997; Wiley *et al*., 1997; Roth *et al*., 1996; Richards *et al*., 1996). Allan *et al*. (1997) and Roth *et al*. (1996) found that the area of agriculture at the catchment scale was the most important predictor of local stream conditions, whereas local riparian vegetation was a weak secondary predictor of habitat quality and the Index of Biotic Integrity. In their studies, local riparian vegetation was not even correlated with overall land use. Based on these studies, Allan *et al*. (1997) question the wisdom of using riparian buffer width guidelines alone as a basis for management because of the message it sends that LULC throughout a catchment can be ignored, or at least is less important, than LULC in riparian buffer zones.

Herlihy *et al*. (1998) used whole catchments as study units along with stream chemistry data from a national-scale monitoring program that is based on a random sampling design. For five U.S. Mid-Atlantic states, they used GIS data derived from aerial photography and classified Landsat Thematic Mapper data to analyze landscape-water quality relationships. They found that Cl^- concentrations, nutrients, acid neutralizing capacity and base cations were the analytes most strongly related to watershed land cover.

2.2. Landscape ecology

More recently, studies of LULC impacts on water quality have extended to include analysis of the spatial arrangement of land cover (Griffith, 2000; Johnson and Gage, 1997; Hunsaker and Levine, 1995; Hunsaker *et al*., 1992). Among other benefits of such an approach, ecotoxicologists have suggested that a landscape approach which incorporates pattern may be useful in explaining differences in the effects of more moderate chemical stresses on aquatic systems (Cairns and Niederlehner, 1996).

Landscape ecology deals with the biological, physical, and societal causes and consequences of spatial variation in landscapes (Moss, 1999). Its unifying concept is analysis of the effect of landscape pattern on ecological processes. In the late 1980's and early 1990's, the re-emergence of landscape ecology, as a distinct cross-disciplinary field of study, accelerated with numerous studies employing landscape metrics as a means to understand biophysical patterns (O'Neill *et al*., 1988; Turner, 1989, 1990). New spatial tools such as geographic information systems (GIS) and remote sensing have given geographers and ecologists unprecedented capacity to quantify land cover patterns and understand spatial heterogeneity and landscape structure (Turner and Carpenter, 1998). These technologies have enabled more efficient and comprehensive characterization of landscape structure through measures referred to as landscape pattern metrics (LPMs). See Gustafson (1998) for a state-of-the-art review on quantifying landscape pattern.

Landscape pattern metrics are measurements designed to quantify and capture aspects of landscape pattern and include forest fragmentation indices or the per-

TABLE I

Some examples of landscape pattern metrics from. Full descriptions of these metrics, and equations for their calculations are provided in McGarigal and Marks (1995)

Metric name (units)	Description
Area-Weighted Mean Shape Index	Mean patch shape complexity, weighted by patch area; equals 1 when all patches are circular and increases as patches become non-circular
Contagion (%)	Approaches 100 when the distribution of adjacencies of individual cells among unique patch types becomes increasingly uneven. Equals 0 when all patch types are equally adjacent to each other. Larger values denote a landscape composed of larger, more clumpy patches. Smaller values denote a landscape composed of many, small patches
Edge Density (m ha^{-1})	Sum of length of all edge segments divided by total area
Interspersion and Juxtaposition Index	Approaches 0 when distribution of adjacencies among patch types becomes increasingly uneven; IJI equals 100 when all patch types are equally adjacent to all other patch types
Mean Patch Size (ha)	Total landscape area divided by the total number of patches (contiguous units of land cover)
Modified Simpson's Diversity Index	Diversity measure; increases with number of patch types and as the proportional distribution of area among patch types becomes more equitable
Patch Density (no. 100 ha^{-1})	Number of patches divided by total landscape area
Patch Richness	The number of land use/land cover types
Shannon Diversity Index	Diversity measure; equals negative of the sum, across all patch types, of the proportional abundance of each patch type, multiplied by that proportion

Source: Modified from McGarigal and Marks (1995).

centage of an area occupied by the largest contiguous patch of grassland. Other metrics include simple measurements such as LULC proportions, amount of wetlands loss, and percent intensive-human land use in an area, to more complex metrics that quantify patch shape, patch isolation, and patch interspersion and juxtaposition. These measures capture important information because differences in shape, size, and distribution of land cover patches play a major role in modifying the configuration of landscape at a regional scale (EPA, 1994). Examples of some of these metrics are shown in Table I.

The structure of landscapes that is described by these metrics includes both composition and configuration. Composition refers to features related to the presence or amount of land cover types without being spatially explicit, whereas landscape configuration refers to the physical distribution or spatial arrangement of cover types within the landscape and includes measures describing the shape of patches or the placement of cover types relative to one another (McGarigal and Marks, 1995). It is important to be able to quantify spatial pattern in order to test hypotheses concerning the relationship of landscape pattern to human and ecological processes. For example, metrics that capture aspects of landscape pattern are needed to correlate landscape spatial pattern with important environmental attributes or processes such as water quality, avian population dynamics, large mammal movements, or nutrient flows.

If landscape pattern metrics indicate significant change in LULC patterns over time, it is likely that many ecological processes will be affected (Swanson *et al.*, 1988; Risser *et al.*, 1984; Urban *et al.*, 1987; Hunsaker and Carpenter, 1990; Turner, 1989). Therefore, the relationship of these indices to empirical data is an important research topic in both landscape ecology and remote sensing (Stoms and Estes, 1993). At the same time, others have encouraged use of these metrics in water quality studies (Johnson and Gage, 1997; O'Neill *et al.*, 1997; Jones *et al.*, 1996; Hunsaker and Levine, 1995). Heggem *et al.* (2000) used a landscape ecology approach in a Louisiana (U.S.A.) watershed to reveal its increasingly distressed condition over a 20 yr period, and noted changes in % forest, largest and average patch sizes, length of new roads, number of road/stream crossings, etc.

Several issues related to the derivation of the more complex landscape pattern metrics, however, need to be examined and fully understood before implementation of these measures in environmental monitoring programs. One thread of research in the remote sensing/landscape ecology literature concerns the effect of different spatial resolutions on the behavior of landscape metrics. For example, in a given geographic area, are LPMs derived from Landsat Thematic Mapper (TM) data different from those derived from the coarser resolution NOAA Advanced Very High Resolution Radiometer (AVHRR) data? Furthermore, would the relationship between landscape pattern and biophysical variables be significantly different using land cover maps derived from different sensors? Cain *et al.* (1997) and Griffith *et al.* (2000) addressed these issues and showed, in general, the robustness across scales of the most important aspects of landscape structure as quantified by landscape pattern metrics.

Although LPMs have frequently been suggested as tools to study water quality (e.g., Jones *et al.*, 1996) the relatively few studies that have examined them have had mixed results. Wear *et al.* (1998) examined land cover and landscape pattern along an urban-rural gradient and its implications for water quality. They suggested that the most remote portion of a watershed and the outer edge of urban development may hold disproportionate influence over future water quality. Furthermore, these areas have a typical structure of landscape pattern, or a 'landscape signature', that can be associated with certain trends in water quality. In another study, Hunsaker *et al*. (1992) found that contagion, a metric describing how dispersed or clumped land cover patches are, explained 20% of conductivity levels in southern Illinois watersheds. In a later study of the same area, however, Hunsaker and Levine (1995) found some evidence of LPMs impacting water quality, but determined that land cover proportions explained more of the variance in phosphorus and conductivity levels. Johnson *et al*. (1997) and Richards *et al*. (1996) used patch density and found it explained variation in water quality in Michigan in some seasons, however, other landscape factors (e.g., geology, LULC, slope) generally were more important. Sharpe (1994) found no correlation between LPMs and water quality using a nutrient runoff model.

Griffith (2000) and Griffith *et al*. (2002a) offered further direction to LPM-water quality studies by providing an account of the relationships between LPMs and empirical stream data across a multi-state region in the U.S. Central Plains. They noted that despite the limitations of LPMs demonstrated in their study, there were a few significant relationships found that may be helpful in monitoring watershed conditions. In general, LPMs and their relationship with water quality were more understandable in more 'natural' or simpler landscapes or where a strong urban-rural gradient existed. Examples of this situation they cited included the Interspersion and Juxtaposition Index in the Ozark Highlands, and landscape diversity indices in the Nebraska Sand Hills (see Table I for descriptions of these landscape pattern metrics). In the Mississippi River Lowlands of Missouri, higher patch density was strongly correlated with higher habitat index scores ($r^2 = 0.85$). They recommended that, if analyzing relatively small watersheds (about <50 km^2), LULC data resolution should be at least 30 m to allow a high probability that a full range of land cover types would occur, and so that class-level metrics can be used. Their research also demonstrated the need to further refine the use of LPMs with respect to water quality applications.

Several problems with using landscape pattern metrics were noted in the above study, including: small watersheds having only one or two patches, which prevented calculation of some LPMs; collinearity with LULC data; and counterintuitive or inconsistent results that resulted from basic differences in land use/cover patterns among ecoregions or from other important factors that determine water quality (Griffith, 2000; Griffith *et al*., 2002a). Basic differences in landscape structure likely caused some of the inconsistency, in that different landscape metrics were related to different parameters in different ecoregions. The same metric will likely not work for every ecoregion or for every water quality parameter. Due to this result, using a suite of metrics to evaluate conditions is appropriate (Jones *et al*., 1996; Qi and Wu, 1996). When using LPMs, it may be useful to stratify watersheds into size classes to reduce the effect that size of the watershed or other unit has on patch shape variables (O'Neill *et al*., 1996; Turner, 1989). There is also a need to be aware of site-specific factors when interpreting LPMs. In the above studies, other factors influenced water quality besides LPMs and caused counterintuitive relationships in some cases.

2.3. Remote sensing and water quality

A variety of applications of remotely-sensed imagery to aquatic systems exist, and Johnson and Gage (1997) provide a summary of many of them. Examples of such applications that they cite include mapping LULC and terrestrial vegetation types (Jensen *et al*., 1986), mapping aquatic macrophyte distribution (Gross and Klemas, 1986; Ackleson and Klemas, 1987; Jensen *et al*., 1987), mapping chlorophyll *a* distribution and primary production (Rundquist *et al*., 1996; Harding *et al*., 1995; Ruiz-Azuara, 1995), and detecting water quality elements such as turbidity, temperature, and pollutant plumes (Bolgrien *et al*., 1995; Liedtke *et al*., 1995; Jupp *et al*., 1994; Goodin *et al*., 1993; Lillesand *et al*., 1983; Lathrop and Lillisand, 1989, 1986; Kirk, 1988). While these applications have generally involved direct examination of spectral reflectance measurements, indices derived from reflectance measures have also been shown to be useful.

2.3.1. *The Normalized Difference Vegetation Index (NDVI)*

The NDVI is a commonly used vegetation index in remote sensing studies because it is roughly correlated with green plant biomass. The NDVI is based on the relative spectral (i.e., light) reflectance values in the red and near infrared (NIR) wavelengths:

$$\text{NDVI} = (\text{NIR} - \text{Red})/(\text{NIR} + \text{Red})$$

Vegetation indices are commonly used to reduce effects of atmospheric conditions or different soil backgrounds on spectral reflectance values. The amount of red solar energy reflected by vegetation cover depends primarily on chlorophyll content, whereas the amount of near infrared energy reflected by vegetation is affected by the amount and condition of green biomass, leaf tissue structure and water content (Jensen, 1996).

Evaluating the potential of NDVI and NDVI-derived metrics for watershed monitoring and water quality studies is important in gaining an increased understanding of landscape-water quality relationships. NDVI has been shown to be sensitive to biophysical characteristics of vegetation such as leaf area, net primary production and levels of photosynthetic activity (Rundquist *et al*., 2000; Stoms and Hargrove, 2000; Paruelo and Lauenroth, 1995; Spanner *et al*., 1990; Box *et al*., 1989; Goward *et al*., 1985; Tucker *et al*., 1983), as well as LULC (Loveland *et al*., 1995). Because of its ability to integrate both land cover and biophysical conditions, NDVI can be helpful in assessing regional watershed conditions that affect water quality and stream condition. Jones *et al*. (1996) evaluated the theoretical potential of NDVI to assess watershed health and hypothesized that it could indicate losses in productivity, increased erosion, and losses of the buffer capacity along riparian corridors. They suggested examining NDVI patterns and changes in NDVI values, as well as comparing observed versus expected NDVI based on soils, topography, vegetation and climate. Whistler (1996) explored NDVI values derived from Landsat Multi-Spectral Scanner (MSS) imagery as a surrogate for biomass, and hypothesized that NDVI values would have stronger relationships with water chemistry parameters than with land cover proportions derived from the same imagery. He found significant relationships between NDVI and selected water quality parameters which, in fact, were stronger than relationships to LULC in most cases.

2.3.2. *Vegetation Phenological Metrics (VPMs)*

Vegetation phenological events such as emergence, maturity, and senescence are important for assessing the condition of agricultural and natural vegetation (Senay and Elliott, 2000; Lee, 1999; Samson, 1993). Therefore, in addition to raw NDVI values, phenological metrics derived from NDVI measurements may also have potential for watershed assessment. Reed *et al*. (1994) defined 12 metrics using AVHRR NDVI biweekly composites that can be categorized into three groups: 1) temporal (based on the timing of a phenological event), 2) NDVI-based (the NDVI value at the time of a phenological event), and 3) metrics derived from time-series characteristics. These vegetation phenological metrics (VPMs) have been successfully used to assess crop condition and potential yield (Lee, 1999), characterize crop phenological variability (Reed *et al*., 1994), separate grasslands by photosynthetic pathway (C_3 or C_4) (Tieszen *et al*., 1997) or different canopy densities (Senay and Elliott, 2000), and map LULC (Loveland *et al*., 1995). The utility of VPMs for applications in water quality monitoring, however, has only recently been explored.

Griffith (2000) and Griffith *et al*. (2002b) examined empirical relationships between both NDVI and vegetation phenological metrics for ecoregions within Nebraska, Kansas, and Missouri. They found statistically significant relationships between selected NDVI values and vegetation phenological metrics, and water quality parameters or habitat/biotic-integrity indices. In most cases, the VPMs or NDVI were more highly correlated to water quality than simple land cover proportions in most cases. General knowledge about the dominance of LULC within the watersheds as well as regional crop types, however, was important to interpreting relationships of the VPMs and NDVI to the stream condition parameters. Spring correlations were highest between NDVI and nitrogen in the Western Corn Belt Plains ecoregion in Whistler's (1996) study. In Griffith's (2000) study as well, early growing season NDVI (early May-early June) or the mean date of onset of greenness was most highly correlated to water quality samples, some of which were collected later in the summer. Griffith (2000) postulated that an important potential benefit from NDVI or VPMs was that, because early growing season NDVI values were most often correlated with stream parameters, the potential exists for estimating summer water quality conditions with springtime AVHRR NDVI data.

Thus, NDVI or VPMs show potential to serve as an early-warning signals of stress (Kelly and Harwell, 1990; Munn, 1988) to aquatic systems. Griffith (2000) suggested that the reason NDVI, and the derivative VPMs, may have certain advantages over simple land cover proportions is that they are biophysical integrators of conditions throughout the watershed. Using NDVI and derived metrics can also capture temporal changes, as opposed to static LULC maps, which do not capture within-class variation and which typically are not updated annually. Moreover, because NDVI values are interval data as opposed to the nominal categories of LULC, they can capture within-class variability of a land cover type.

2.3.3. *Digital Elevation Models (DEMs) and Digital Terrain Models (DTMs)*
Innovative advances in digital representations of the earth's surface elevation and topography/geomorphology will also aid in understanding land cover/water quality relationships. The U.S. Geological Survey (USGS) (1999) has developed a new seamless National Elevation Dataset (NED) for the conterminous U.S. Details about it can be found at the website, http://edcnts12.cr.usgs.gov/ned/. The USGS NED has been developed by merging the highest-resolution, best quality elevation data across the United States into a seamless raster format, at 30 m and sometimes even 10 m spatial resolution. DEMs have typically been produced from stereopairs of aerial photographs or existing maps. However, satellite data are also being used to produce DEMs and DTMs, as stereopairs can be made from both SPOT satellite images and radar interferometry. Jones *et al*. (2000), Jones *et al*. (1997) and Wickham *et al*. (1999) ranked watersheds by vulnerability to environmental degradation in the mid-Atlantic U.S. by highlighting where agriculture was being performed on lands >3% slope, and near headwater reaches. They determined change in agriculture over a time period by examining gain or loss in NDVI values. For other examples of applications using DEMs and DTMs, see Gesch *et al*. (1999), Malleswara *et al*. (1996), Raggam and Almer (1996) Jensen (1993), Moore *et al*. (1993) and Polidori (1991). Another dataset produced by the USGS is NED-H (National Elevation Dataset-Hydrologic Derivatives), which is an interagency effort with its goal the development of a hydrologically correct version of the National Elevation Dataset (http://edcnts12.cr.usgs.gov/ned-h/). With these datasets and the tools derived from them, one will be able to delineate watersheds and subcatchments across large areas of the U.S. with an accuracy and ease heretofore unknown. These tools will doubtless aid in hydrologic models for large regions, especially when combined with the land cover data or vegetation index data.

3. Ecological Monitoring and Ecological Indicators

Currently, there are many national monitoring programs designed to assess the state of the environment (Griffith, 1998; Hunsaker and Carpenter, 1990), or assess the ecological health of large regions (Messer *et al*., 1991; Rapport, 1992), watersheds

(Wichert and Rapport, 1998), or stream and river systems (Hughes *et al.*, 2000; Karr, 1999; Whittier and Paulsen, 1992). These monitoring programs are based on the use of indicators (environmental measurements) that serve as surrogates of ecological condition (Fairweather, 1999). For environmental monitoring purposes, it is important for river ecologists to identify and measure these indicators of watershed health (Boulton, 1999). Many landscape factors not discussed in this review are also important determinants of water quality (e.g., geology, slope, soils, etc.). Nevertheless, because sets of rapidly collected indicators may be particularly useful in assessing regional water quality (Hughes *et al.*, 2000; Fairweather, 1999; Harris and Silveira, 1999), it is important to explore the potential of remotely sensed data to identify broad-scale screening indicators for use in such watershed monitoring programs.

4. Summary

In summary, the need to address NPSP problems and to analyze mid- to large-sized watersheds across wide regions in an efficient manner has been the impetus behind a landscape approach to water quality studies. This approach involves examining the entire catchment of streams rather than, or in addition to, near-stream portions (although the riparian area is still considered important to stream conditions). Aerial photography has made the analysis of landscape classification of the watershed easier and continues to be useful in creating DEMs for hydrologic modeling, with imagery from radar sensors and the U.S. Space Shuttle presenting new and exciting opportunities to better characterize geomorphology and watersheds of regions and of the planet. With the advent of satellite remote sensing in the 1970's, the ability to examine larger areas than was previously capable was made possible, and this facility increases along with the increasing computing power of microcomputers. At the present time, new satellite sensors with moderate and high resolution data and hyper-spectral data are furthering the capability to analyze land cover-watershed relationships (Jones *et al.*, 2000).

Limitations to some of these approaches are that some landscape pattern metrics and their connections with water quality are difficult to understand. Moreover, there will always be a need for complementary field investigations and ground work for habitat, stream chemistry, and restoration assessments. Additionally, using remote sensing technology requires a large investment of time and money to for hardware, software and staff training. Although finer resolution imagery (down to 1 m resolution) is now commercially available, using it for analysis of large areas will likely be cost-prohibitive.

Nonetheless, findings from the literature show that the landscape approach complements field-based approaches well, and that remote sensing and GIS techniques will continue to be strong components of future landscape-water quality studies. While landscape ecology has produced some useful indicators from the simpler metrics (i.e. patch size, length of roads, number of stream road crossings, percentage of agriculture on steeper slopes, etc.), it has not yet produced strong and consistent relationships between the more complex pattern metrics (such as metrics describing spatial arrangement of land cover) and water quality. This does not mean, however, that advancement in this field will not produce some robust indicators. Remote sensing will make all such studies more efficient, especially with the advantages provided by information on slope and aspect available from national-scale elevation data sets. In conclusion, the new or advanced applications described herein will further aid the understanding of landscape-scale (i.e., entire catchments) relationships to stream water quality.

Acknowledgements

I thank two anonymous reviewers for recommending improvements to the manuscript. I also thank Drs. Edward Martinko, Kevin Price, Stephen Egbert, Donald Huggins, and F. Jerry DeNoyelles, all from the University of Kansas, for their helpful comments on an earlier draft of the manuscript.

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