ResearchGate

See discussions, stats, and author profiles for this publication at: https://www.researchgate.net/publication/278678438

Agricultural Land Fragmentation and Biological Integrity: The Impacts of a Rapidly Changing Landscape on Streams in Southeastern Wisconsin

Technical Report · May 2007

CITATIONS
6

READS
198

2 authors:

Richard Ross Shaker
Ryerson University
48 PUBLICATIONS **253** CITATIONS

SEE PROFILE

Timothy J. Ehlinger
University of Wisconsin - Milwaukee
50 PUBLICATIONS **1,151** CITATIONS

SEE PROFILE

Some of the authors of this publication are also working on these related projects:

Project Sustainability indicators, Patterns of Sustainable Development, and Sustainable Landscape Change View project

Project Network for Ecohealth and One Health - NEOH View project

All content following this page was uploaded by Richard Ross Shaker on 18 June 2015.

The user has requested enhancement of the downloaded file.

Fish Ecology Laboratory
University of Wisconsin - Milwaukee, Milwaukee, WI 53201

TECHNICAL REPORT

AGRICULTURAL LAND FRAGMENTATION AND BIOLOGICAL INTEGRITY: THE IMPACTS OF A RAPIDLY CHANGING LANDSCAPE ON STREAMS IN SOUTHEASTERN WISCONSIN

Richard Shaker, MSc.
Research Associate
Timothy J. Ehlinger, Ph.D.
Co-Primary Investigator

Project sponsored by the Grant No. R83-0885-010 to Northeastern University and the University of Wisconsin-Milwaukee from the US EPA/NSF/USDA STAR Watershed Program

Bernice Smith
EPA Project Officer

Iris Goodman
EPA Program Director

Boston, Massachusetts
May, 2007

ABSTRACT

Since the end of World War II, there has been significant population growth in the United States causing development to radiate outward from the traditional urban core. This has resulted in the metabolization of large areas of rural countryside with sprawling suburbs and exurbs. Due to the primary conversion of agricultural lands to residential development, exurbia is considered to be the fastest transitioning landscape in the United States. Even though there is limited knowledge on the effects of exurbanization, it is presumed that exurbia has the same effects on ecosystems and ultimately human health as other types of urban development. Development affects the natural ecosystems through: fragmenting landscapes, isolating habitat patches, simplifying biodiversity, degrading natural habitats, modifying landforms and drainage networks, introducing exotic species, controlling and modifying disturbances, and disrupting energy flow and nutrient cycling.

We present a method for examining this transitioning landscape of exurbia utilizing the theory and practices established within the field of landscape ecology. In this paper, 31 watersheds were used to separate Southeastern Wisconsin into analyzable landscapes. The overall objectives were: (1) to identify a subset of metrics that capture the majority of variation in agriculture land fragmentation in southeastern Wisconsin, and (2) to identify a subset of metrics that capture the relationship between agricultural land fragmentation and a measure of biotic integrity (IBI: an index score based on fish population variables). Seventy-two landscape metrics were calculated and statistically analyzed. In the end, six landscape metrics were identified that explained 84 percent of the variation in aquatic environmental integrity for Southeastern Wisconsin. The parameters that were linked with higher IBI were associated with larger landscape patch area and distance between patches. Parameters contributing to declines in IBI were associated with patch shape complexity and variability. The strength of these relationships indicates that the spatial design of human development in watersheds has a significant impact on aquatic ecological integrity and that principles of landscape design may have direct relevance to efforts at river and stream restoration and protection.

ACKNOWLEDGEMENTS

The modeling research and analysis contained in this report was sponsored by the U.S. Environmental Protection Agency/National Science Foundation/U.S. Department of Agriculture STAR Watershed Program by a Grant No. R83-0885-010 to Northeastern University and the University of Wisconsin-Milwaukee. Additional support for data collection was provided by the UW-Milwaukee American Geographical Society Collection and the Wisconsin Department of Natural Resources. The authors greatly appreciate this support. The findings and conclusions contained in this report are those of the authors and not of the funding agencies, nor the STAR program.

Table of Contents

1. Introduction

The structure, function, and dynamics of contemporary ecosystems are profoundly influenced by human activities (Alberti 2005), and understanding the mechanisms responsible for environmental changes requires the integration of both natural and human processes. Pervasive ecological changes have occurred as a result of human activities (Alberti 2005). Changes in land cover through the appropriation of natural landscapes to provide for human needs in one of the most pervasive alterations resulting from human activity (Vitousek 1994). While the ecological and sociological effects of land conversion for agricultural uses are well studied (Riebsame et al. 1994), the effects of agricultural land conversion for human habitation, or urbanization, are less well understood (Pickett et al. 1997).

A less studied aspect of urbanization is the conversion of agricultural lands to low-density residential land use beyond the urban fringe (exurban). Exurbanization is considered the fastest transitioning form of landscape development in the United States (Crump 2003; Theobald 2002; Daniels 1999). The change in landscape configuration resulting from appropriation of agricultural lands for exurban development can have a variety of ecological effects. Conversion of agricultural lands to residential lands may alter environmental integrity through a range of processes including: fragmenting landscapes, isolating habitat patches, simplifying biodiversity, degrading natural habitats, modifying landforms and drainage networks, introducing exotic species, controlling and modifying disturbances (e.g., floods, forest fires), and disrupting energy flow and nutrient cycling (Alberti 2005; Alberti et al. 2003; Picket et al. 2000).

This study investigates the spatial configuration of agricultural lands in relation to exurban development and ecological integrity in southeastern Wisconsin. Specifically, 31 watershed delineated landscapes were used to investigate the environmental effects of fragmented agricultural lands associated with exurban growth.

2. Background

2.1. Sprawling Suburbs and Exurbs and the Decline of Agricultural Lands

Since the end of World War II, U.S. population has radiated outward from the traditional urban core, with suburbs and exurbs metabolizing large areas of rural countryside. Encouraged by economic growth, improved infrastructure, and the demand for rural living, people continue to move to exurban locations beyond the urban fringe at an accelerated rate (Crump 2003). Economic globalization and technological advances in in information and transportation infrastructure facilitate greater options for US citizens regarding where to live (Daniels 1999). Motivated by the desire to live in a more natural environment, exurban residents have been found to be more interested in landscape factors such as: aesthetically pleasing viewscapes, open space, recreational opportunities, and solitude when compared to suburban residents (Crump 2003; Morrill 1992; Nelson 1992). With more people being capable of moving further away from the city, and the "rural" factors significantly influencing where people are moving to, exurbia is at risk of being "loved to death".

Today more than 70% of the U.S. population lives in urbanized areas; however, the rapid growth of exurbs indicates that many U.S. citizens find rural environments appealing (Crump 2003; Morrill 1992; Nelson 1992). Studies suggest a substantial preference for exurban locations among much of the U.S. population. For example, Blackwood and Carpenter (1978) surveyed 1,400 residents of urban Arizona to find out where they would prefer to live and what factors they liked best about those places. More than 66% of the respondents favored rural counties with populations less than 50,000. Additionally, 41% stated they would prefer to live in a town less with than 10,000 people. When asked to rate which factors were most influential for choosing rural or small town locations, the participants chose population size and environmental quality (Blackwood and Carpenter 1978).

In another example, the *Milwaukee Journal Sentinel* (MJS) (2000) surveyed 600 Wisconsinites with a series of questions pertaining to urban growth and development. One question asked: Where would you predict development to occur in Wisconsin over the next 20 years? In this case, the answers were spread fairly evenly: 23% major cites, 34% suburbs, 23% other cities and villages, and 20% rural areas. The next question asked was: If you could control development where would you prefer it to take place? The answers for this question were more for urban and less for rural with 34% major cities, 30% suburbs, 22% other cities and villages, and 15% rural areas. However, when asked, if you were to move today where would you like to live? Ironically, the survey came back as 6% major cities, 27% suburbs, 23% other cities and villages, and 44% rural areas. Clearly, people of southeastern Wisconsin desire rural environments and thus exurban living.

Exurbia, a term coined in the late 1940s, has been defined several ways. In a research paradigm starting with Myers and Beegle (1947), exurbia was known as "the rural-urban fringe" or "urban-rural interface". By the 1960s exurbia was being called "the middle ground between densely populated urban and suburban regions, and sparsely settled rural towns and countryside" (Marx 1964). In the early 1980s, exurbia was commonly defined as "the transition zone between the city or urban areas and the surrounding countryside" (Lamb 1983). Most recently, Daniels (1999) referred to exurbia as "the metropolitan fringe".

No matter how the geographic area of exurbia has been defined, researchers have assigned similar distances to urban areas and population density. To summarize, Daniels (1999) characterized exurbia to be 10 to 50 miles from urban centers with a population of approximately 500,000, and/or 5-30 miles from cities with a population of approximately 50,000. Additionally, he stated that exurban commuters travel on average 25 minutes or more to work, exurban communities contain a mixture of long-term and new residents, and exurban communities have low density development and a mix of urban and rural land uses.

Exurbia is considered to be the fastest transitioning landscape associated with economic development in the United States (Crump 2003; Daniels 1999). Due to its geographic location, and proximity to urban areas, development in exurbia primarily consumes agricultural lands. In the United States, by the 1950s, agricultural lands surpassed 1,161 million acres nationwide (Theobald 2002). However, soon thereafter, fuelled by the residential development boom of the 1950s, the percentage of lands in agriculture started to steadily decline. From 1950 to 1997 agricultural lands declined at a rate of 5.39 million acres per year (USDA 1999). This fast rate of agricultural land loss can be attributed to the demand for low density housing beyond the urban fringe (Crump 2003; Theobald 2002; Daniels 1999). Specifically, there has been an increase in acres per housing unit for roughly one-third of US counties from 1960 to 2000; in 2000, roughly 38 million acres were settled at urban densities, and nearly ten times that much land was settled at low, exurban densities (Theobald 2002).

2.2. The Effects of Development beyond the Urban Fringe

Several studies have investigated the ramifications of exurbanization. The most notable of those studies investigated the effects of exurbanization, concluding that it let to loss of productive agriculture land, open space, and wildlife habitat (Platt 1985; Diamond and Noonan

1996; Beatley and Manning 1997; Sorensen, Greene, and Russ 1997; Riebsame et al. 1994; Rome 1998; Waldie 2000; Heimlich and Anderson 2001; Theobald 2002). While primarily the economic, ecological, and sociological effects have been studied for agricultural land, open space, and wildlife habitat, the effects of land conversion for human habitation, or urbanization, is less understood (Pickett et al. 1997). Ecologically, exurbanization has similar affects on ecosystems and ultimately human health as other types of urban development. Because humans depend on the earth's ecosystems for food, water, and other goods and services, changes in ecological conditions ultimately affects their own health and well being (Alberti 2005).

Development affects the natural ecosystems by: fragmenting landscapes, isolating habitat patches, simplifying biodiversity, degrading natural habitats, modifying landforms and drainage networks, introducing exotic species, controlling and modifying disturbances, and disrupting energy flow and nutrient cycling (Alberti 2005; Alberti et al. 2003; and Picket et al. 2000). In response to analyzing and understanding the transitioning landscapes of exurbia, theory and practices established within landscape ecology were used.

2.3. Landscape Ecology and Exurbia

A paradigm started by the German biogeographer Carl Troll (1939), landscape ecology combines both the spatial approach of the geographer and the functional approach of the ecologist (Naveh and Lieberman 1984; Forman and Godron 1986). The principles of landscape ecology are relevant when addressing the dynamic makeup of the exurban landscape. Landscape ecology, as defined by Richard T. T. Forman, (1983) incorporates: (1) the spatial relationship among landscape elements, or ecosystems, (2) the flow of energy, minerals, nutrients, and species among the elements, and (3) the ecological dynamics of the landscape mosaic through time. Today, landscape ecology is considered to be an interdisciplinary science drawing from a variety of different disciplines (i.e., anthropology, architecture, biology, ecology, economics, geography, and forestry). A key component of landscape ecology addresses anthropogenic effects on both natural and built landscapes; furthermore, understanding that human activity is a central factor for shaping the environment (Bissonette and Storch 2003; Dramstad et al. 1996; Forman 1995; Turner 2001).

Exurbia, a result of urbanization, has traditionally been researched by geographers and economist, has recently received increasing interest from ecologists who treat the urbanization process as a transformation of landscape patterns and functions (Bessey 2002; Huang 1998). Landscape ecologists have started to document the impacts that various arrangements of patch structure have on ecosystem function (Godron and Forman 1982; Turner 1989; Forman 1995; Collinge 1996).

In landscape ecology, the patch is the fundamental element of the landscape. The patch, defined by Richard Forman (1995), is an area of specific type (e.g., agricultural field, woodlot, lake) that is different than its surrounding types in a landscape. The size and shape of the patch, its proximity to other patches, and its edges are particularly important patch characteristics that have significant ecological and environmental impacts (Forman 1995; Turner et al. 2001; Alberti 2005). The patch is the primary component in landscape ecology used for developing the analytical metrics in a land cover or land use analysis. Exurbia provides ecologists with an opportunity to examine the urbanization process as a transformation of landscape patterns and functions (Bessey 2002; Huang 1998). One approach is to characterize the relationships between various arrangements of patch structure and ecosystem functions (Godron and Forman 1982; Turner 1989; Forman 1995; Collinge 1996).

The use of landscape metrics for analyzing spatial patterns has become quite popular, and some effort has been made to examine the behavior and limitations of landscape metrics for better understanding urbanization (Turner 1987; Turner 1988; Wu 1998; Jennerette and Wu 2001). However, the use of landscape metrics specifically for monitoring agricultural land fragmentation in regards to exurban development is virtually nonexistent. With literally hundreds of landscape ecology metrics available, it is imperative to address several questions when using landscape metrics in assessment efforts: (1) What are the objectives of the study; (2) What is the behavior of the metrics over a range of landscape configurations; (3) What are the effects of scale on the metrics; and (4) are the metrics correlated or redundant (Turner et al. 2001)? In some instances, efforts to study landscape fragmentation have used artificial landscapes in their analysis (Gustafson and Parker 1992; Hargis et al. 1998). In this analysis, a set of real landscapes are used to synthesize independent metrics into an overall measure of agriculture fragmentation, an explanatory model of exurban development, and a predictive model of environmental quality.

2.4. Land use, Watersheds, and Biological Integrity

The catchment or watershed paradigm started in the mid 1970s changed the way stream ecologists look at the landscape. “In every respect, the valley rules the stream” (Hynes 1975). “Rivers and streams serve as a continent’s circulatory system, and the study of those rivers, like the study of blood, can diagnose the health not only of the rivers themselves but of their landscapes” (Sioli 1975). Since then, rivers have been studied from a landscape perspective, both as individual landscapes (Robinson et al. 2002, Ward 1998, Wiens 1989), and as ecosystems that are strongly influenced by their surroundings at multiple scales (Townsend et al. 2003; Fausch et al. 2002; Allan et al. 1997; Schlosser 1991). Increased attention to the landscape perspective of rivers continues to evolve with the growth of landscape ecology as a field of study (Turner et al. 2001; Wiens 1989) and because of an increased focus on catchment-scale studies by freshwater ecologists (Allan 2004).

Researchers have investigated the effects of land use or land cover on biological processes, leading to significant work exploring ecological regions (Heilman et al. 2002), buffer areas (Wang et al. 2001), Landsat image boundaries (Tinker et al. 1998), hexagonal units (Griffith et al. 2000), and watersheds boundaries (Potter et al. 2005; Cain et al. 1997) for dividing the landscape. In all cases, the physical characteristics of streams that shape biotic communities are influenced by a variety of landscape features, including geology, catchment area, and land use (Richards et al. 1996). Based on the demonstrated ability of watersheds explaining a greater amount of variability in aquatic ecosystems (Potter et al. 2005; Sliva and Williams 2001; Wang et al. 2000; Weigel 2000; Roth et al. 1996; Allan 1995); this study uses watersheds to separate the study area of southeastern Wisconsin into 31 individual landscapes.

There have been many terms used to describe or capture the status of river system, such as ecological integrity, stream condition, and river health. Typically, the ideas behind these terms were motivated by a desire to characterize a stream’s response to human influences (Allan 2004). When assessing river health, several indicators, such as the number of intolerant species and taxa richness [Index of Biologic Integrity (IBI), see Karr 1991] have been used. The number of observed taxa related to the expected can be used [Rivpacs, see Wright 1995; Ausrivas, see Norris & Hawkins 2000). Additional measures include: taxa richness of sensitive species; body size and shape, life history, and behavioral traits (Usseglio-Polatera et al. 2000; Corkum 1999; Pan et al. 1999; Townsend & Hildrew 1994); pollution tolerance (Hilsenhoff 1988); and ecosystem processes, such as photosynthesis and respiration (Bunn et al. 1999). Habitat and water quality measures using individual variables or combined metrics are also available (Barbour et al. 1999). Thus, a plethora of methods are available for assessing the response of stream condition to land use or land cover.

Investigating agricultural fragmentation as an indication of development and linking agricultural fragmentation to an Index of Biotic Integrity (IBI) will be very valuable to the science community and planners alike. A primary objective of this analysis is to examine the relationship between agricultural fragmentation metrics and fish IBI, and to evaluate fragmentation as an indicator of environmental quality in warm water streams for Southeastern Wisconsin.

2.5 Coupling Agricultural Landscape Metrics and Biotic Integrity

Ecologists have a long history of incorporating the design and complexity of landscapes into their analyses of freshwater ecosystems long before the field of landscape ecology was coined (Wiens 2002). Examples of their work include the early appreciation that lake trophic status was related to agricultural productivity, longitudinal river studies recognized the importance of slope and vegetation cover, and linking the complexity of ecosystems with their particular environmental settings (Allan & Johnson 1997).

With the advancement of numerous methods for evaluating ecosystems, combined with technological increase in geographic information systems and spatial analysis, a plethora of works linking land use/land cover to river condition has developed. Specifically, when investigating agricultural effects, a decline in water quality, habitat, and biological assemblages occurred as the extent of agricultural lands increase within the catchments (Richards et al. 1996; Roth et al. 1996; Wang et al. 1997; Skinner et al. 1997; Sponseller et al. 2001). Further, researchers commonly report that streams draining agricultural lands support fewer species of sensitive insect and fish taxa than other forms of land cover (Cooper 1993; Lenat & Crawford 1994; Wang et al. 1997; Genito et al. 2002). With the advancement of this type of research, the ability to improve science-based conservation and management of rivers also improves. It has been stated by Allen (2004) that the catchment approach to the management of river ecosystems can be conceived of in four steps: (1) identify the land-water unit, (2) asses the status or “health” of the river, (3) identify the stressors that influence the river status, and (4) develop management and restoration plans, grounded in good ecological science, to reveres or mitigate impacts.

Even with the availability of calculating landscape metrics through available software such as FRAGSTATS (McGarigal & Marks 1995) linkages to ecological process and function remains largely untested (Allen 2004; Alberti et al. 2003; Picket et al. 2001; Grimm et al. 2000). Additionally, studies associated to the urban – rural gradient are often simple transects and miss the complexities of landscape patterns emerging by the distribution of land use and land cover (Alberti, 2005). In order to combat the paradox of limited landscape ecology research on ecological function and process, the fragmentation of agricultural lands in Southeastern Wisconsin will be compared to the Wisconsin Index of Biotic Integrity (IBI). By studying agricultural land fragmentation in Southeastern Wisconsin, the interactions between human processes and biological complexities in exurbia are investigated. Specifically, this research links agricultural land fragmentation to the fastest transitioning landscape of exurbia, with a measure of environmental quality, that can be used for watershed management and planning.

3. Goals and Objectives

Although there has been considerable work done in the theory of landscape ecology, far less has been accomplished in applying these theories to practice (Bissonette and Storch 2003). Bissonette (1997) presented some conceptual frameworks for applying landscape ecology theory to the practice of wildlife management. Bissonette and Storch (2003) furthered this effort by linking landscape ecology theory with applications useful to wildlife and natural resource managers. Turner et al. (2001) presented the practical application of the many methods and techniques from landscape ecology to forest management. Forman (1995) discussed many basic concepts of landscape ecology and their applicability to resource management. To help quantify spatial patterns of landscape structure McGarigal and Marks's (1995) work serves as a hallmark detailing the various metrics developed for landscape ecology. This study will help in the growth of landscape ecology as a discipline by using real data and real landscapes to answer tangible questions. By applying the metrics and concepts of landscape ecology, a greater understanding of exurbia and its ecological effects can be found.

By focusing on exurbia, the most rapidly transitioning landscape, urbanization can be further understood. Using an improved method for quantifying ecosystem integrity, and linking it to agricultural fragmentation, knowledge will be gained on the effects of differently structured landscapes. This research will help gain understanding about the loss of agricultural lands and the repercussions of urbanization and sprawl (Theobald 2002). Lastly, this research will help gain information against the paradox: that few studies explicitly address how urban patterns affect ecosystem function (Grimm et al. 2000; Picket et al. 2001; Alberti et al. 2003).

In this paper, we first investigate the correlations among agriculture landscape metrics for southeastern Wisconsin from Landsat derived data and then characterize the effects of agricultural land fragmentation on a measure of stream biotic integrity. Our overall objectives are: (1) to identify a subset of metrics that best captures the majority of variation in agriculture land fragmentation in southeastern Wisconsin; (2) to identify a subset of metrics that capture the relationship between agricultural land fragmentation and a measure of biotic integrity (IBI: an index score based on fish population variables).

To accomplish this, a subset of independent 72 landscape metrics was computed for 31 watersheds is southeastern Wisconsin. Pearson correlation and stepwise multiple regression were used to create a model for agricultural fragmentation predicting the average IBI for Southeastern Wisconsin.

4. Study Area Description

Upon settlement, most of southeastern Wisconsin's native prairies were transformed into agricultural lands. Those agricultural lands remained the hallmark of southeastern Wisconsin until shortly after WWII. Soon after WWII population growth outside of urban centers began to increase rapidly. From post WWII to the present, agricultural lands have continued to decline as residential development boomed and populations increase beyond the metropolitan fringe. Today, southeastern Wisconsin is vital to Wisconsin due to its large population, urban centers, and remaining agricultural lands. This analysis of agricultural land fragmentation examined 31 watersheds crossing 15 counties of southeastern Wisconsin (Figure 1 and Figure 2). Those counties (Green Lake, Fond du Lac, Sheboygan, Columbia, Dodge, Washington, Ozaukee, Dane, Jefferson, Waukesha, Milwaukee, Rock, Walworth, Racine, and Kenosha) had a population of 2,547,635 in 1970 which grew to 2,953,174 by 2000; an increase of 14 percent (405,539 people) over a span of 30 years (United States Census 2000). Much of this growth has occurred in counties primarily dominated by agricultural lands surrounding Madison and Milwaukee. The

15 counties used for this analysis had 4,250,000 acres of farmland in 1970 which decreased to 3,261,000 acres of farmland by 2000, a loss of 23 percent (989,000 acres) over a span of 30 years (NASS 2000). Population increase and agricultural land decrease between 1970 and 2000 are listed in Table 1 and Table 2, respectively, for the counties in the study area. The State of Wisconsin has made efforts (e.g., smart growth initiatives) to control rural population growth related to urban sprawl, but it is likely that further fragmentation of the state's agricultural lands will occur. Mapping the fragmentation of southeastern Wisconsin's agriculture lands is critical to protecting terrestrial and aquatic ecosystems and controlling the environmental affects of human population growth.

5. Data

5.1 Land Cover and Watershed Data

The land cover data set used in this analysis is titled: WISCLAND Land Cover (WLCGW930). It was developed for the Wisconsin Department of Natural Resources (WIDNR) as part of a larger project for the Upper Midwest Gap Analysis Program (UMGAP) Image Processing Protocol (1998). The dataset was published for use in Wisconsin in 1998, and is available online in Geographic Information System (GIS) compatible format from the WiDNR at: http://dnr.wi.gov/maps/gis/datalandcover.html. The WISCLAND Land Cover data set is a raster representation of vegetation and land cover for the entire state of Wisconsin that was acquired from the larger national Multi-Resolution Land Characteristics Consortium (MRLC) data set.

The MRCC created the data set for UMGAP using dual-date Landsat Thematic Mapper (TM) imagery data primarily from 1992. The original pixel size of the TM source data is 30 meters; however, excluding urban areas, data was generalized to an area no smaller than four contiguous pixels (approximately one acre). The results of the smoothing process will allow any feature five acres or larger to be resolved in the data, giving a Minimum Mapping Unit (MMU) of five acres. During the generalization process the data set was transformed from its original raster format into a more user-friendly vector format. With this MMU the data set is designed to be used between scales of 1:40,000 to 1:500,000 for a wide variety of resource management and planning applications.

The land cover classification scheme was designed to be compatible with the UNESCO and Anderson's classifications. The WISCLAND Land Cover data set has two different land cover classification descriptions associated with it. Description One has six land cover classes associated with it (agricultural land, barren land, forest land, urban/built-up land, water, wetland) while Description Two has 24 land cover classes associated with it (bays and estuaries, beaches, commercial and services, confined feeding operations, cropland and pasture, deciduous forest land, evergreen forest land, forested wetlands, industrial, industrial and commercial complexes, lakes, mixed forested lands, mixed urban or built-up lands, nonforested wetlands, orchards, groves, vineyards, and nurseries, other agricultural land, other built up land, reservoirs, residential, sandy areas other and beaches, streams and canals, strip mines, quarries, and gravel pits, transitional areas, transportation, communications, and utilities). Because this analysis focuses on agricultural land fragmentation, Description One was employed. Further, by using the Description One classification scheme, it can be presumed that there is lower misclassification error. Description one land cover classification data for the state of Wisconsin can be seen in Figure 3.

The watershed geographic data used to divide the land cover data into workable landscapes are the WIDNR level 5, 10-digit Hydrologic Unit Hierarchy (HUC). This data set is considered to be at the watershed scale with 334 units contiguously covering Wisconsin. The primary purpose of this data set is to prepare reference base maps for the DNR NPS Water Pollution Abatement Program (a.k.a., NPS Priority Watershed Program). This geographic data set can be downloaded from the WiDNR in GIS compatible format at: http://dnr.wi.gov/maps/gis/datahydro.html.

Both the agricultural land fragmentation data derived from the WISCLAND Land Cover (WLCGW930) and the 10-digit Hydrological Unit Hierarchy (HUC) were utilized in investigation the relationship between agricultural land fragmentation and measure of environmental quality. Additionally, Wisconsin fish IBI data collected by the Wisconsin Department of Natural Resources (WDNR) was used.

5.2 Fish Biological Data

The idea of an Index of Biological Integrity (IBI) originated during the late 1970s and early 1980s to assess river quality in Indiana and Illinois (Karr et al. 1986). There are many types of IBIs (e.g. birds, macroinvertebrates, fish), but in this analysis fish are the indicator species examined. Due to the effects of environmental determinism, different states eventually developed their own IBI. After three years of research by John Lyons and his colleagues an IBI based on fish communities was developed for the state of Wisconsin (Lyons 1992). The Wisconsin version of fish IBI consists of 12 metrics that can be simplified into three categories (Table 3). The index was created to capture variation of species in a community in relation to variation in the environmental quality in the watershed (Lyons 1992). Following the Wisconsin method for wadeable streams, fish samples are collected from a segment of stream with a length equal to thirty five times the mean stream width, thus including on average different habitats. The fish IBI for Wisconsin is calculated for an individual sample from each stream segment and calibrated by comparing the observed values of each metric with values expected in comparable streams of high environmental quality (Karr 1999; Karr et al. 1986). The overall IBI score is calculated by summing the scores for the first 10 metrics and the 2 correction factors. The comprised score is normalized to equal 100 % of the environmental integrity for a geographic ecoregion, thus giving a score range of 0-10 for each of the first 10 metrics found in table 3. For the last two metrics (correction factors) a possible 10 points could be subtracted from the overall total IBI score, which decrease the score if there is a low abundance of fish or a high percentage of fish with eroded fins, deformities, lesions or tumors. The Wisconsin IBI scales between 0 and 100, with increasing scores correlating with higher the environmental quality (Karr 1999; Karr et al. 1986).

The IBI data used in this analysis were collected by and obtained from the Wisconsin Department of Natural Resources. The samples were collected over a span of four years (2001-2005) and were used to provide an average score per sample site. In the study area, 152 fish IBI sites were used to investigate the effects agricultural fragmentation is having on a measure of environmental quality for Southeastern Wisconsin (Figure 4).

6. Methods

6.1 Determining the Appropriate Scale

Various methods have been use to create boundaries between ecological regions (Heilman et al. 2002), using watershed boundaries (Cain et al. 1997), simple Landsat image boundaries (Tinker et al. 1998) and hexagonal units of set or arbitrary size (Griffith et al. 2000; Hunsaker et al. 1994; White et al. 1992) when doing landscape ecology research. However, we used watersheds as the boundaries by which landscapes are defined in this analysis of their greater relevance in explaining variability in aquatic ecosystems (Potter 2005; Sliva and Williams 2001; Wang et al. 2000; Weigel 2000; Wang et al. 1998; Richards et al. 1997; Roth et al. 1996; Allan 1995),

As determined by Pfister (2004) the values for nearly all landscape metrics level out between 4000-6000 ha. Since the level 5, 10-digit Hydrologic Unit Hierarchy (HUC) provides this level of scale (Table 4) we decided to use it as our region definitions for sampling the landscape regions of southeastern Wisconsin.

6.2. Calculation of Landscape Metrics

Geographic Information System (ESRI ArcGIS 9.1), we separated the 31 watersheds into their individual shapefiles from the original 334 watersheds. The vector land cover for each watershed was clipped along its border, and the vector data were converted into the raster ArcGrid format. The raster format was used for further analysis in the landscape ecology software FRAGSTATS 3.3 (McGarigal & Marks 1995, available at http://www.umass.edu/landeco/research/fragstats/fragstats.html). Because the original land cover data were collected at 30-meter resolution, the pixel size for the conversion process was kept at 30 meters by 30 meters.

In order to determine the metrics that best characterize the arrangement of agricultural lands for southeastern Wisconsin, 72 FRAGSTAT metrics were calculated using a sample of 31 individual watershed landscapes of roughly 3,525 ha. Class metrics in FRAGSTATS are computed for every patch type or land cover class in the landscape. There are two basic types of metrics at the class level: (1) indices of the amount and spatial configuration of the class, which can be referred to as primary metrics, and (2) distributional statistics that provide central tendency (e.g., mean and area weighted mean) and variance (e.g., standard deviation and coefficient of variation) statistical summaries of the patch metrics for the focal class (McGarigal and Marks 1995). Metrics were normalized by either log10 or arcsine transformation. Pearson correlation coefficients test to determine and eliminate highly correlated ($|r| > 0.90$) metrics using SPSS 13 (SPSS 2003). Results representing primary metrics or central tendency metrics were selected first, because they are considered to represent to high or low agricultural land fragmentation. Fifty of the original 72 landscape ecology metrics remained after running the Pearson correlation coefficients test. A summary of the class level metrics and methodology is found in (Table 5).

6. 3 Creation of a Environmental Quality Model

The database of normalized agricultural metrics per watershed was joined with the database of the 152 fish sites and an average IBI score was calculated for each watershed (Figure 6). Forward-stepping stepwise multiple regression (SYSTAT 12.0) was used to select the best set of the 50 normalized agricultural land fragmentation metrics that predicted the average Fish IBI score per watershed. This resulted in the selection of 6 agricultural fragmentation metrics.

Finally, for each of the landscape regions (i.e. watersheds), the 6 remaining metrics were weighted by their respective contribution to the change in R-square change and summed to generate an index of agriculture fragmentation that predicted average fish IBI in a watershed.

7. Results

7.1. Stream Environmental Quality Model

Using the matrix of Pearson correlation coefficients ($|r| > .90$) 22 of the original 72 metrics were eliminated (See Table 5 for listing of metric). Of the 22 metrics eliminated 5 (23%) were primary metrics, 8 (36%) were central tendency metrics and 9 (41%) were variance metrics. Of the 50 retained metrics, 20 (40%) were primary metrics, 16 (32%) were central tendency metrics (i.e., mean or area weighted mean), and 14 (28%) were variance metrics (i.e., standard deviation or coefficient of variation).

Stepwise multiple regression eliminated 44 of the remaining 50 agricultural metrics. Of the remaining agricultural land fragmentation metrics three were primary metrics, two were central tendency metrics, and one was a variance metric (Table 6). The three primary metrics of Area/Edge Density were included: *Largest Patch Index* (LPI), *Total Class Area* (CA), and Normalize Landscape Shape Index (NSLI). One central tendency metric for Shape, *Area Weighted Mean Shape Index* (SHAPE_AM) and one variance metric for shape, *Standard Deviation of Mean Fractal Dimension Index* (FRAC_SD), were included. One primary metric for Proximity/Isolation, *Mean Euclidian Nearest Neighbor Index* (ENN_MN) was included. *Largest Patch Index* (LPI) is equal the area of the largest patch of agricultural land divided by the total landscape area, multiplied by 100 to create a percentage (McGarigal and Marks 1995). LPI at the class level quantifies the percentage of total landscape area comprised by the largest patch; LPI can be considered as a simple measure of dominance. *Class Area (*CA) equals the sum of the areas of all the agricultural land patches, divided by 10,000 to create hectares (McGarigal and Marks 1995). CA is a measure of landscape composition; specifically, how much of the landscape is comprised of the agricultural land type. *Area Weighted Mean Shape Index* (SHAPE_AM) equals agricultural patch perimeter divided by the minimum patch perimeter in the landscape; further the shape scores are divided by the sum of all shape scores in the landscape (McGarigal and Marks 1995). CORE equals the area within the patch that is further than the specified depth-of-edge distance from the patch perimeter, divided by 10,000 (to convert to hectares). *Euclidean nearest-neighbor distance* is perhaps the simplest measure of patch context and has been used extensively to quantify patch isolation. Here, nearest neighbor distance is calcuated using simple Euclidean geometry as the shortest straight-line distance between the focal patch and its nearest neighbor of the same class. *Fractal dimension index* indicates a departure from Euclidean geometry (i.e., an increase in shape complexity). FRAC approaches 1 for shapes with very simple perimeters such as squares, and approaches 2 for shapes with highly convoluted, plane-filling perimeters.

The relationship between IBI Score and the six metrics: LPI, CA, ENN_MN, FRAC_SD, NLSI and SHAPE_AM explained 84 percent of the variation in IBI among watershed (Table 9). Positive effects on IBI included measures of fragment size and isolation. The strongest positive influence of an individual agricultural land fragmentation metric in predicting the IBI score was *Largest Patch Index* (LPI, std. coeff. = 0.74, $p < .001$, Table 6, Figure 5). Other positive influences on IBI included *Total Core Area* (CA, Figure 6) and *Mean Euclidean Nearest Neighbor* (ENN_MN). The combined effect of these three factors indicates that larger patches of contiguous landscape further apart contribute more to environmental quality than smaller patches closer together. The three metrics that had a negative contribution to IBI were measures of variability in landscape patch shape and size. The strongest negative contribution was from *Weighted Mean Shape Index* (SHAPE_AM, std. coeff. = 0. 55, $p < 0.001$, Table 6, Figure 7), followed closely by Normalized Landscape Shape Index (NLSI) and *Standard Deviation of Mean Fractal Dimension Index* (FRAC_SD). This combination of negative effects indicates that increased complexity and variability in patch shape within watersheds contributes lower aquatic biological integrity.

A plot of empirically measured IBI versus that predicted from the landscape metrics is presented together with a plot of IBI versus percent-developed land in Figure 8. The landscape fragmentation model is significantly better at predicting aquatic biological integrity in exurban environments compared to the more often-used metric of urbanization.

8. Discussion

The past decade has witnessed an increased awareness that human activities in watersheds have and are continuing to contribute to the decline of biotic integrity of the nations waters (Novotny et al. 2005). But at the same time, few articles in the literature have established strong predictive models that go beyond simplistic relationships of the IBIs to one or a few parameters. It is critical for the sake of our ability to manage watershed development that we are able to identify proximal causes of impairment. Percent of imperviousness is a surrogate for many adverse stresses caused by urbanization and development (Wang et al 2000). The results of this study indicate that a strong relationship exists between biotic integrity and the spatial arrangement and shapes of development in exurban watersheds that goes beyond simply the amount of a particular type of development. It is less important the amount of development that impacts biological integrity, but rather its arrangement on the landscape.

There is limited knowledge of an ideal method of how to measure patterns of fragmentation. It has been stated that the application and interpretation of landscape metrics remains difficult and there is not enough data to truly understand the relationship between metrics and ecological processes (Turner et al. 2001). It has been suggested that factors involved in fragmentation are often so complex that the use of one single measure or technique is not adequate (Davidson 1998). Furthermore, Bissonette and Storch (2003) have suggested that the use and appropriateness of fragmentation metrics is a necessary step in order to draw significant conclusions from empirical research. Turner, (1989) goes on to states that the ability to understand spatial patterns of land cover may be crucial to understanding the affects of land cover change. Our research selected a suite of four metrics from an initial 72 metrics that best represent patterns of agricultural land fragmentation and produced a viable method for determining agricultural land fragmentation patterns and creating an overall index for southeastern Wisconsin.

Much of landscape structure and pattern analysis is based upon research that has focused on landscape scale (Griffith et al. 2000; Cain et al. 1997; Haines-Young and Chopping 1996; Ritters et al. 1995; O'Neill et al. 1988). Studies examining class and landscape level metrics of land use and land cover have suggested an importance of using class level indices (Heilman et al. 2002; Tischendorf 2001; Griffith et al. 2000; Tinker et al. 1998; Gustafson 1998). Agricultural land fragmentation is a class-level process, and to our knowledge there has been no studies attempting to systematically determine an appropriate suite of indices to capture this phenomenon. Furthermore, to our knowledge, there have been no studies attempting to model agricultural land fragmentation in association to exurban development.

Despite a plethora of research on habitat fragmentation, researchers consistently identify key metrics despite differences in scale, region, and overall methods. At the landscape scale, the factors of greatest importance appear to be measures of landscape diversity and texture first, with measures of shape and size being of lesser importance (Griffith et al. 2000; Cain et al. 1997). Specifically, at the class level, measures of patch area, core area, patch shape, and patch isolation appeared consistently (Cumming and Vernier 2002; Griffith et al. 2000; Tinker et al. 1998). In an analysis by Tinker et al. (1998), the class: forest was examined; fragmentation among 12 watersheds found mean patch area, core area, and edge density metrics as the key factors among watersheds highly fragmented by clear cutting and roads. Other class based research by Cumming and Vernier (2002), found that up to 67% of the variation was related to patch shape, core area, and patch isolation.

Our findings, even though a different class type was utilized, have major consistencies with the literature. Measures of core area (LPI and CA) represent the first two factors in the model. These two measures, representing patch size and dominance, accounted for the largest coefficients for positive effects on IBI. Measures of patch shape and complexity (SHAPE_AM and FRAC_SD) represented greatest negative impacts on biological integrity. Finally, a measure of proximity/isolation (ENN_MN) suggested that the distance between landscape patches is a significant factor impacting aquatic ecosytems.

In Summary, this approach of creating an agricultural land fragmentation index and exurban development model is a practical method that can be replicated in other regions. The results of doing such research can be useful to ecologist, natural resource managers, and planners alike. Agricultural land fragmentation information has been typically underestimated because ground-based measurements of land-use change are difficult (Riebsame, Gosnell, and Theobald

1996; Theobald, Gosnell, and Riebsame 1996). The fragmentation of agricultural lands has many negative and often irreversible effects such as the change in water chemistry, biodiversity, and increased flooding (Alberti 2005; Theobald 2002; Daniels 1999). The relationships identified in this study provide an effective and efficiently tool for measuring and monitoring agricultural land fragmentation and may lead to informed recommendations for future planning and conservation efforts.

Studies such as this, coupled with remote sensing and GIS techniques, make it possible to monitor current conditions and predict changes. This would be fundamental to understanding the spatial pattern affects of land use change (Turner 1987). Measurement of agricultural fragmentation within landscape regions is a key step to understanding impacts of differences and change, and ultimately making wise planning decisions. By calibrating landscape metrics to a measure of environmental quality, in this case fish IBI, a surrogate or proxy method of measuring environmental quality can be further developed refined and replicated in other regions. This proxy technique for evaluating environmental quality could be used in concert with traditional sampling techniques; furthermore, improving the speed of analysis and prioritization for restoration. Creation of maps that identify areas highly fragmented can be useful for prioritizing land parcels for purchase by natural resource manages. Furthermore, an improved understanding of the mechanisms that link landscapes to rivers is vital to the design of conservation plans, management and restoration, and improved methods of bioassessment (Allen 2004). Studies such as this make it possible to monitor current conditions and predict changes- a crucial step in understanding the affects of spatial patterns on ecological processes (Grimm et al. 2000; Picket et al. 2001; Alberti et al. 2003; Alberti 2005).

Since the freshwater paradigm of the 1970s many researchers have tried to link terrestrial variables to aquatic variables (Karr and Chu 1999). Even with the advances in remote sensing technology and computerized geographic and environmental data, there is still a lack of knowledge linking landscape metrics to ecological process and function (Allen 2004; Alberti et al. 2003; Picket et al. 2001; Grimm et al. 2000). Turner (2001) goes further by explaining that the difficulties are due to a lack of data for application and interpretation of landscape metrics and ecological measures. Bissonette and Storch (2002) suggested that determining a use of fragmentation metrics will ultimately help determine their use and appropriateness. Recently, the rapidly expanding research investigating streams in the context of their catchments and landscapes indicate that stream ecosystems are strongly affected by human actions (Allen 2004). "Not only does the valley rule the stream, as Hynes (1975) so aptly put it, but increasingly, human activities rule the valley" (Allen 2004). Because humans are part of the ecosystem, depending on the earth for food, water, and other goods and services, changes in ecological condition ultimately affects their own health and well being (Alberti 2005).

9. Conclusion

It has become apparent that the region known as exurbia, in Southeastern Wisconsin, is consuming agricultural lands at an astounding rate; metabolizing the landscape at an uncontrolled pace due to limited policy and planning in that geographic region. In an attempt to objectively study exurbanization and its effects, 72 landscape metrics were statistically analyzed identifying six metrics that when combined created an explanatory model of environmental quality for Southeastern Wisconsin that explained 84% of the variance in biotic integrity of fish.

By statistically reducing landscape metrics, weighting by their relevance, and combining these metrics mathematically it was possible to create an index of agricultural fragmentation, an exurban development model, and a model of environmental quality for Southeastern Wisconsin. The parameters that were linked with higher IBI included patch size area and proximity, and those linked with declines in IBI were associated with shape complexity and variability.

This research contributes to bridge the gap between theories and practice for the science of landscape ecology. By mapping the fragmentation of agricultural lands and the environmental quality model in Southeastern Wisconsin insight may be gained with regard to protecting terrestrial and aquatic ecosystems, and thus ultimately human heath. It is possible that the metrics for exurban development and the environmental quality models could be mechanistic drivers of ecological change, but further research will be required. The techniques and maps found within this thesis could be used in further research to prioritize lands for conservation and predict the affects of uncontrolled exurbanization.

10. Literature Cited

Alberti, M. 2005. The effects of urban patterns on ecosystem function. *International Regional Science Review* 28 (2): 168-192.

Alberti, M., J. Marzluff, E. Shulenberger, G. Bradley, C. Ryan, and C. Sumbrunnen. 2003. Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. *Bioscience* 53 (12): 1169-1179.

Allan, J. D. 1995. Stream ecology: structure and function of running waters. *Chapman & Hall, London.*

Allan, J. D. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37: 149-161.

Allan, J. D., L. B. Johnson. 1997. Catchment scale analysis of aquatic ecosystems. *Freshwater Biology* 37: 107-111.

Allan, J. D. 2004. Landscape and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Syst. 35: 257-284.

Barbour M. T., J. Gerritsen, B. D. Snyder, J. B. Stribling. 1999. Rapid bioassessment Protocols for use in streams and wadeable rivers: peripheryton, benthic macroinvertebrates and fish. *Second addition. Rep.* EPA/841-B-99-002, US EPA, Off. Water, Washington, DC.

Bessey, K. M. 2002. Structure and dynamics in an urban landscape: toward a multi-scaled view. *Ecosystems* 5: 360-375.

Beatley, T., and K. Manning. 1997. The ecology of place: planning for environment, economy, and community. Washington, D. C.: Island Press.

Bissonette, J. A. 1997. Wildlife and landscape ecology: effects of pattern and scale. Springer-Verlag, New York.

Bissonette, J. A. and I. Storch. 2003. Landscape ecology and resource management: linking theory with practice. Island Press, Washington, D.C., USA.

Blackwood, L. G., & E. H. Carpenter (1978). The importance of anti-urbanism in Determining residential preferences and migration patterns. *Rural Sociology*: 43: 31-47.

Bunn, S. E., P. M. Davies, T. D. Mosisch. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biology* 41: 333-346.

Cain, D. H., K. Riiters, K. Orvis. 1997. A multi-scale analysis of landscape statistics. *Landscape Ecology* 12: 199-212.

Collinge, S. 1996. Ecological consequences of habitat fragmentation: implications for landscape architecture and planning. *Landscape and Urban Planning* 36: 50-77.

Cooper, C. M. 1993. Biological effects of agriculturally derived surface pollutants on aquatic systems- a review. *Journal of Environmental Quality* 22: 402-408.

Corkum, L. D. 1999. Conservation of running waters: beyond riparian vegetation and species richness. *Aquatic Conservation* 9: 559-564.

Crump, J. 2003. Finding a place in the country: exurban and suburban development in Sonoma County, California. *Environment and Behavior* 35 (2) 187-202.

Cumming, S. G., P. Vernier. 2002. Statistical models of landscape pattern metrics, with applications to regional scale dynamic forest simulations. *Landscape Ecology* 17: (5): 433-444.

Daniels, T. 1999. When city and country collide. Island Press, Washington, D.C., USA.

Davidson, C. 1998. Issues in measuring landscape fragmentation. *Wildlife Society Bulletin* 26: 32-37.

Diamond, H. L., and P. F. Noonan, eds. 1996. Land use in America. Washington, D. C.: Island Press.

Dramstad, W., J. Olson, and R. Forman. 1996. Landscape ecology principles in landscape architecture and land-use planning. Island Press, Washington, D. C., USA.

Fausch K. D., C. E. Torgersen, C. V. Baxter, H. W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience* 52: 483-498.

Forman, R. T. T. 1983. An ecology of the landscape. *BioScience* 33:535.

Forman, R. T. T., M. Godron. 1986. *Landscape ecology*. New York: John Wiley.

Forman, R. T. T. 1995. Land Mosaics: the ecology of landscapes and regions. Cambridge, University Press, New York, NY, USA.

Genito, D., W. J. Gburek, A. N. Sharpley. 2002. Response of stream macro Invertebrates to agricultural land cover in small watersheds. *Journal of Freshwater Ecology* 17: 109-119.

Godron, M., and R. T. T. Forman. 1982. Landscape modification and changing ecological characteristics. In *Disturbance and ecosystems: Components of response,* ed. H. Mooney and M. Godron. New York: Springer-Verlag.

Griffith, J., E. Martinko, K. Price. 2000. Landscape structure analysis of Kansas at Three scales. *Landscape and Urban Planning* 52 (1): 45-61.

Grimm, N. B., J. M. Grove, S. T. A. Pickett, C. L. Redman. 2000. Integrated approaches To long-term studies of urban ecological systems. *BioScience* 50: 571-584.

Gustafson, E. 1998. Quantifying landscape spatial patterns: What is the state of the art? *Ecosystems* 1: 143-156.

Gustafson, E. J., and G. R. Parker. 1992. Relationships between landcover proportion and indices of landscape spatial patterns. *Landscape Ecology* 7: 101-110.

Haines-Young, R., and M. Chopping. 1996. Quantifying landscape structure: a review of landscape indices and their application to forested landscapes. *Progress in Physical Geography*: 20 (4): 418-445.

Hargis, C. D., J. A. Bissonette, J. L. David. 1998. Understanding measures of landscape pattern. In J. A. Bissonette (ed.). Wildlife and landscape ecology: effects of pattern and scale. Springer-Verlag, New York, USA: 231-261.

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

Heimlich, R. E., and W. D. Anderson. 2001. Development at the urban fringe and Beyond: impacts on agriculture and rural land. Agricultural Economic Report No. 803. Washington, D. C.: U.S. Department of Agriculture, Economic Research Service.

Hilsenhoff, W. L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. *Journal of North American Benthol. Soc.* 7:65-68.

Huang, S. –L. 1998. Ecological energetics, hierarchy, and urban form: A system modeling approach to the evolution of urban zonation. Environment and Planning B: *Planning and Design* 25: 391-410.

Hunsaker, C. T., R. V. O'Neill, B. L. Jackson, S. P. Timmins, D. A. Levine, D. J. Norton. 1994. Sampling to characterize landscape pattern. *Landscape Ecology* 9: 207-226.

Hynes, H. B. N. 1975. The stream and its valley. *Verh. Int. Ver. Theor. Ang. Limnol.*

19: 1-15.

Jenerette, G. Darrel, Jianguo Wu. 2001. Analysis and simulation of land-use change in the central Arizona-Phoenix region, USA. *Landscape Ecology*. 16: 611-626.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant. 1986. Assessing biological integrity in running waters: a method and its rationale. *Special Publication 5. Champaign, IL: Illinois Natural History Survey*: 5.

Karr, J. R. 1991. Biological integrity- a long neglected aspect of water-resource management. *Ecol. Appl.* 1:66-84.

Karr, J. R., E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press: Washington DC.

Lamb, R. F. 1983. The extent and form of exurban sprawl. *Growth and Change* 14 (1): 40-47.

Lyons, J. (1992). Using the Index of Biotic Integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. United States Department of agriculture. *General Technical Report*: NC-149.

Lenat D. R., J. K. Crawford. 1994. Effects of land-use on water-quality and aquatic Biota of three North Carolina Piedmont streams. *Hydrobiologia* 294: 185-199.

Marx, L. 1964. The machine in the garden. Oxford University Press, New York, NY, USA.

McGarigal, K., B. J. Marks. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. General Technical Report PNW-GTR-351, USDA Forest Service, Pacific Northwest Research Station, Portland, OR.

Milwaukee Journal Sentinel (MJS). Dreams of green. 2000. http://www.jsonline.com/news/state/jan00/dreams013000.asp

Morrill, R. 1992. Population redistribution within metropolitan regions in the 1980s: core, satellite, and exurban growth. *Growth and Change* 23: 277-302.

Myers, R. R., and J. A. Beegle. 1947. Delineation and analysis of the rural-urban fringe. *Applied Anthropology* 6: 14-22.

National Agricultural Statistics Service. 2000. Retrieved on 6 January 2007 from http://www.nass.usda.gov/Statistics_by_State/Wisconsin/index.asp.

Naveh, Z., and S. Lieberman. 1984. Landscape ecology: theory and application.

Ecology 66: (2) 641.

Nelson, A. C. 1992. Characterizing exurbia. *Journal of Planning Literature* 6 (4): 350-368.

Norris, R. H., C. P. Hawkins. 2000. Monitoring river health. *Hydrobiologia* 435: 5-17.

Novotny, V., Bartošová, O'Reilly, N. and Ehlinger, T. 2005. Unlocking the Relationship of Biotic Integrity of Impaired Waters to Anthropogenic Stresses. Journal of Water Research. 34: 189-198.

O'Neill, R. V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D. L. DeAngelis, B. T. Milne, M. G. Turner, B. Zygmunt, S. W. Christenson, V. H. Dale, R. L. Graham. 1988. Indices of landscape pattern. *Landscape Ecology* 1: 153-162.

Pan, Y. D., R. J. Stevenson, B. H. Hill, P. R. Kaufmann, A. T. Herlihy. 1999. Spatial patterns and ecological determinants of benthic algal assemblages in Mid-Atlantic streams, USA. *J. Phycol*. 35:460-468.

Platt, R. H. 1985. The farmland conversion debate: NALs and beyond. *Professional Geographer* 37 (4): 433-442.

Pfister, J. L. 2004. Using landscape metrics to create an index of forest fragmentation For the sate of Maryland. Thesis: Towson University.

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, C. H. Nilon, R. V. Pouyat, W. C. Zipperer, and R. Costanza. 2001. Urban ecological systems: linking terrestrial ecology, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32: 127-157.

Pickett, S. T. A., M. L. Cadenasso, and C. G. Jones. 2000. Generation of heterogeneity by organism: creation, maintenance, and transformation. In *Ecological consequences of habitat heterogeneity,* ed. M. Hutchings, L. John, and A. Stewart, 33-52. New York: Blackwell.

Pickett, S. T. A., W. R. Jr. Burch, S. E. Dalton, T. W. Foresman, J. M. Grove, and R. Rowntree. 1997. A conceptual framework for the study of human ecosystems in urban areas. *Urban Ecosystem* 1: 185-199.

Potter, K. M., F. W. Cubbage, R. H. Schaaberg. 2005. Multiple-scale landscape Predictors of benthic macroinvertebrate community structure in North Carolina. *Landscape and Urban Planning* 71: 77-90.

Richards, C, R. J. Haro, L. B. Johnson, G. E. Host. 1997. Catchment- and reach-scale properties as indicators of macroinvertebrate species traits. *Freshwater Biology* 37: 219-230.

Richards, C., L. B. Johnson, G. E. Host. 1996. Landscape-scale influences on stream habitats And biota. *Canada Journal of Fish and Aquatic Sciences*: 53 (Suppl. 1) 295-311.

Riebsame, W. E., H. Gosnell, and D. M. Theobald. 1996. Land use and landscape change in the Colorado mountains, I: theory, scale, and pattern. *Mountain Research and Development* 16 (4): 395-405.

Riebsame, W. E., W. J. Parton, K. A. Galvin, I. C. Burke, L. Bohren, R. Young, and E. Knop. 1994. Integrated modeling of land use and cover change. *Bioscience* 44: 350-356.

Riiters, K. H., R. V. O'Neill, C. T. Hunsaker, J. D. Wichham, D. H. Yankee, S. P. Timmins, K. B. Jones, B. L. Jackson. 1995. A factor analysis of landscape pattern and structural metrics. *Landscape Ecology* 10: 23-39.

Rome, A. W. 1998. William Whyte, open space, and environmental activism. *Geographic Review* 88 (2): 259-274.

Robinson, C. T., K. Tockner, J. V. Ward. 2002. The fauna of dynamic riverine Landscapes. *Freshwater Biology* 47: 661-677.

Roth, N. E., J. D. Allan, D. L. Erickson. 1996. Landscape influences on stream biotic Integrity assessed at multiple scales. *Landscape ecology* 11 (3) 141-156.

Schlosser, I. J. 1991. Stream fish ecology: a landscape perspective. *BioScience* 41: 704-712.

Silva, L., and D. D. Williams. 2001. Buffer zone versus whole catchment approaches To studying land use impact on river water quality. *Water Resources* 35 (16) 3462-3472.

Sioli, H., 1975. Tropical rivers as an expression of their terrestrial environment. *Tropical Ecological Systems*: 275-288.

Skinner, J. A., K. A. Lewis, K. S. Bardon, P. Tucker, J. A. Catt, and B. J. Chambers. 1997. An overview of the environmental impact of agriculture in the U. K. *Journal of Environmental Management* 50: 111-128.

Sorensen, A. A., R. P. Greene, and K. Russ. 1997. Farming on the edge. Washington, D.C.: American Farmland Trust: DeKalb: Northern Illinois University, Center for Agriculture in the Environment.

Sponseller, R. A., E. F. Benfield, H. M. Valett. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. *Freshwater Biology* 46:

1409-1424.

Theobald, D. M. 2002. Land-use dynamics beyond the American urban fringe. *The Geographical Review* 91 (3): 544-564.

Theobald, D. M., H. Gosnell, and W. E. Riebsame. 1996. Land use and landscape change in the Colorado mountains, II: a case study of the east river valley. *Mountain Research and Development* 16 (4): 407-418.

Tinker, D. B., C. A. C. Resor, G. P. Beauvais, K. F. Kipfmueller, C. I. Fernandes, W. L. Baker. 1998. Watershed analysis of forest fragmentation by clearcuts and Roads in Wyoming forest. *Landscape Ecology* 13: 149-165.

Tischendorf, L. 2001. Can landscape indices predict ecological processes consistently? *Landscape Ecology* 16:: 235-254.

Townsend C. R., S. Doledec, R. Norris, K. Peacock, and C. Arbuckle. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. *Freshwater Biology* 48: 768-785.

Townsend, C. R., and A. G. Hildrew. 1994. Species traits in relation to a habitat templet for river systems. *Freshwater Biology* 31: 265-276.

Troll, C. 1939. Luftbildplan und okologische bodenforschung. Zeitschrift der Gesellschaft fur Erdkund, Berlin, Germany. 241-298.

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* 20: 171-197.

Turner, M. G. 1987. Spatial simulation of landscape changes in Georgia: a comparison of 3 transition models. *Landscape Ecology* 1:29-36.

Turner, M., R. Gardner, and R. O'Neill. 2001. Landscape ecology in theory and practice: pattern and process. Springer-Verlag, New York, USA.

United States Census Bureau. 2000. Retrieved on 20 December 2006 from http: www.census.gov/census2000/states/md.html.

USDA [U.S. Department of Agriculture]. 1999. *1997 Census of Agriculture*. Vols. 1, 1A, 1B, 1C, *Geographic Area Series*. Washington, D.C.: National Agricultural Statistics Service.

Usseglio-Polatera P, M. Bournaud, P. Richoux, H. Tachet. 2000. Biomonitoring through biological traits of benthic macroinvertebrates: how to use species trait databases? *Hydrobiologia* 422: 153-162.

Vitousek, P. M. 1994. Beyond global warming: ecology and global change. *Ecology* 75: 1861-1876.

Waldie, D. J. 2000. Do the voters really hate sprawl? New York Times (Late edition, Final), 3 march, A, 21.

Wang, L., J. Lyons, P. Kanehl. 1998. Development and evaltuation of a habitat rating system for low-gradient Wisconsin streams. *N. Am. Fish Management* 18 (4): 477-485.

Wang, L., J. Lyons, P. Kanehl. 2001. Impacts of urbanization on stream habitat and Fish across multiple spatial scales. *Environmental Manag.* 28: 255-266.

Wang, L., J. Lyons, P. Kanehl, R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. *Fisheries* 22 (6): 6-12.

Wang, L., J. Lyons, P. Kanehl, R. Bannerman, E. Emmons. 2000. Watershed urbanization and changes in fish communities in southeastern Wisconsin streams. *Journal of the American Water Resources Association* 36 (5): 1173-1189.

Ward, J. V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biol. Conservation* 83: 269-278.

Weigel, B.M., J. Lyons, L. K. Paine, S. I. Dodson, D. J. Undersander. 2000. Using stream macroinvertebrates to compare riparian land use practices on cattle farms in southwestern Wisconsin. *Journal of Freshwater Ecology* 15 (1) 93-106.

Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology* 47: 501-515.

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

White, D., A. J. Kimerling, W. S. Overton. 1992. Cartographic and geometric components of a global sampling design for environmental monitoring. *Cartography and Geographic Information Systems* 19 (1) 5-22.

Wright, J. F. 1995. Development and use of a system for predicting macroinvertebrates in flowing waters. *Aust. J. Ecology* 20: 181-197.

Wu, F. 1998. Simulating urban encroachment on rural land with fuzzy-logic-controlled Cellular automata in geographic information system. *J. Environ. Management* 53: 293-308.

11. Tables and Figures

11.1 Tables

Table 1. Population increase between 1970 and 2000 for the 15 counties associated with this analysis.

County	Population (1970)	Population (2000)	Population Gain	Percent Population Gain
Green Lake	18878	19105	227	1
Fon du Lac	84567	97296	12729	13
Sheboygan	96660	112646	15986	14
Columbia	40150	52468	12318	23
Dodge	69004	85897	16893	20
Washington	63830	117493	53663	46
Ozaukee	54461	82317	27856	34
Jefferson	60060	74021	13961	19
Waukesha	231335	360767	129432	36
Milwaukee	1054249	940164	-114085	-12
Dane	290272	426526	136254	32
Rock	131970	152307	20337	13
Walworth	63444	93759	30315	32
Racine	170838	188831	17993	10
Kenosha	117917	149577	31660	21
Total	2547635	2953174	405539	14

Table 2. Agricultural land decrease between 1970 and 2000 for the 15 counties associated with this analysis.

County	Farm Acres (1970)	Farm Acres (2000)	Acrage Decline	Farmland Percent Loss
Green Lake	181000	148000	33000	18
Fon du Lac	414000	359000	55000	13
Sheboygan	267000	201000	66000	25
Columbia	407000	356000	51000	13
Dodge	490000	426000	64000	13
Washington	212000	135000	77000	36
Ozaukee	110000	75000	35000	32
Jefferson	303000	263000	40000	13
Waukesha	190000	111000	79000	42
Milwaukee	18000	7000	11000	61
Dane	668000	356000	312000	47
Rock	420000	382000	38000	9
Walworth	299000	226000	73000	24
Racine	151000	127000	24000	16
Kenosha	120000	89000	31000	26
Total	4250000	3261000	989000	23

Table 3. List of 12 fish IBI metrics by group. Full description and explanation can be found in Lyon (1992).

Group	IBI Metric
SPECIES RICHNESS AND COMPOSITION	
	1. Total number of native species
	2. Number of darter species
	3. Number of sucker species
	4. Number of sunfish species
	5. Number of intolerant species
	6. Percent (by number of individuals) that are tolerant species
TROPHIC AND REPRODUCTIVE FUNCTION	
	7. Percent that are omnivores
	8. Percent that are insectivores
	9. Percent that are top carnivores
	10. Percent that are simple lithophilous spawners
FISH ABUNDANCE AND CONDITION (Correction Factors)	
	11. Number of individuals (excluding tolerant species) per 300 m sample
	12. Percent with deformities, eroded fins, lesions, or tumors (DELT)

(Note: The last two metrics are not normally included in the calculation of IBI, but can lower the overall IBI score if they have extreme values.

Table 4. 10 digit HUC statistics based on 31 watersheds used in analysis.

Area	km^2	ha
Min	48327	483
Max	751742	7517
Mean	352460	3525
SD	205703	2057

Table 5. List of class level metrics used in fragmentation analysis. Full descriptions can be found in McGarigal and Marks (1995).

Metric	Abbreviation	Name
AREA/EDGE/DENSITY		
	CA[2,5]	Total Class Area
	PLAND[2,5]	Percentage of Landscape
	NP[2,5]	Number of Patches
	PD[2,5]	Patch Density
	LPI[2,5,6]	Largest Patch Index
	TE[2,5]	Total Edge
	ED[2,3,5]	Edge Density
	LSI[2,5]	Landscape Shape Index
	AREA_MN[1]	Mean Patch Area
	AREA_AM[1]	Area Weighted Mean Patch Area
	AREA_SD[1]	Standard Deviation of Mean Patch Area
	AREA_CV[2,5]	Standard Deviation of Mean Patch Area
	GYRATE_MN[2,5]	Mean Radius of Gyration Distribution
	GYRATE_AM[1]	Area Weighted mean of Radius of Gyration Distribution
	GYRATE_SD[2,5]	Standard Deviation of Radius of Gyration Distribution
	GYRATE_CV[1]	Coefficient of Variation of Radius of Gyration Distribution
	NLSI[2,3,5]	Normalized Landscape Shape Index
SHAPE		
	SHAPE_MN[2,5]	Mean Shape Index
	SHAPE_AM[2,5]	Area Weighted Mean Shape Index
	SHAPE_SD[2,4,5]	Standard Deviation of Mean Shape Index
	SHAPE_CV[2,5]	Coefficient of Variation of Mean Shape Index
	FRAC_MN[1]	Mean Fractal Dimension Index
	FRAC_AM[2,5]	Area Weighted Mean Fractal Dimension Index
	FRAC_SD[2,5]	Standard Deviation of Mean Fractal Dimension Index
	FRAC_CV[1]	Coefficient of Variation of Mean Fractal Dimension Index
	PARA_MN[2,5]	Mean Perimeter Area Ratio
	PARA_AM[2,5]	Area Weighted Mean Perimeter Area Ratio
	PARA_SD[1]	Standard Deviation of Mean Perimeter Area Ratio
	PARA_CV[2,5,6]	Coefficient of Variation of Mean Perimeter Area Ratio
	CIRCLE_MN[2,3,5]	Mean Related Circumscribing Circle
	CIRCLE_AM[2,5]	Area Weighted Mean Related Circumscribing Circle
	CIRCLE_SD[2,4,5]	Standard Deviation of Mean Related Circumscribing Circle
	CIRCLE_CV[2,5]	Coefficient of Variation of Mean Related Circumscribing Circle
	CONTIG_MN[2,5]	Mean Contiguity Index
	CONTIG_AM[2,5]	Area Weighted Mean Contiguity Index
	CONTIG_SD[1]	Standard Deviation of Contiguity Index
	CONTIG_CV[1]	Coefficient of Variation of Contiguity Index
	PAFRAC[2,3,5]	Parameter Area Fractal Dimension
CORE AREA		
	TCA[1] *	Total Core Area

CPLAND[1] *	Core Percentage of Landscape
NDCA[2,4,5] *	Number of Disjunct Core Areas
DCAD[2,5] *	Disjunct Core Area Density
CORE_MN[1] *	Mean Core Area
CORE_AM[2,4,5] *	Area Weighted Mean Core Area
CORE_SD[1] *	Standard Deviation of Core Area
CORE_CV[2,5] *	Coefficient of Variation of Core Area
DCORE_MN[1] *	Mean Disjunct Core Area Distribution
DCORE_AM[2,5] *	Area Weighted Mean Disjunct Core Area Distribution
DCORE_SD[2,5] *	Standard Deviation of Disjunct Core Area Distribution
DCORE_CV[2,5] *	Coefficient of Variation of Disjunct Core Area Distribution
CAI_MN[2,5] *	Mean Core Area Index
CAI_AM[1] *	Area Weighted Mean Core Area Index
CAI_SD[1] *	Standard Deviation of Core Area Index
CAI_CV[1] *	Coefficient of Variation of Core Area Index

PROXIMITY/ISOLATION

PROX_MN[1] **	Mean Proximity Index
PROX_AM[2,3,5] **	Area Weighted Mean Proximity Index
PROX_SD[2,5] **	Standard Deviation of Proximity Index
PROX_CV[2,5] **	Coefficient of Variation of Proximity Index
ENN_MN[2,5]	Mean Euclidian Nearest Neighbor Index
ENN_AM[2,5]	Area Weighted Mean Euclidian Nearest Neighbor Index
ENN_SD[2,5]	Standard Deviation of Euclidian Nearest Neighbor Index
ENN_CV[2,3,5]	Coefficient of Variation of Euclidian Nearest Neighbor Index

CONTATION/INTERSPERSION

CLUMPY[1]	Clumpy Index
PLADJ[1]	Percentage of Like Adjacencies
IJI[2,5,6]	Interspersion Juxtaposition Index
DIVISION[2,5]	Landscape Division Index
MESH[1]	Effective Mesh Size
SPLIT[2,5]	Splitting Index
AI[2,5]	Aggregation Index

CONNECTIVITY

CONNECT[2,5]	Connectance Index
COHESION[2,5]	Patch Cohesion Index

---

[1] Metric was removed based on Pearson Correlation Matrix (|r| > 0.9).
[2] Metric was used in Principle Component Analysis (PCA).
[3] Metric was chosen to represent one of the ten axes but was removed from final weighted index.
[4] Metric was used in final weighted index of agricultural land fragmentation.
[5] Metric was used in Stepwise Multiple Regression.

[6] Metric was chosen to model percent developed in Stepwise Multiple Regression analysis.
* Calculated using a fixed edge depth of 50 meters.
** Calculated using a search radius of 100 meters.

Table 6. Results of stepwise multiple regression average steam biological integrity in a watershed (AVBG_FISHIBI) as a function of agricultural fragmentation landscape metrics. (A) Final regression model showing standardized coefficients, (B) Analysis of variance for overall significance of final model.

A. Standardized regression model

Effect	Standard Coefficient	t	p-value
CONSTANT	0.00	-5.03	0.00
LPI	0.74	4.89	0.00
CA	0.57	5.11	0.00
ENN_MN	0.24	2.18	0.05
FRAC_SD	-0.27	-2.59	0.02
NLSI	-0.34	-2.18	0.05
SHAPE_AM	-0.55	-4.68	0.00

B. Analysis of Variance

Source	SS	df	Mean Squares	F-ratio	p-value
Regression	652.01	6.00	108.67	13.34	0.00
Residual	122.24	15.00	8.15		

Dependent Variable	AVG_FISHIB
N	22
Multiple R	0.92
Squared Multiple R	0.84
Standard Error of Estimate	2.86

11.2 Figures

Figure 1. Map of watersheds used in analysis.

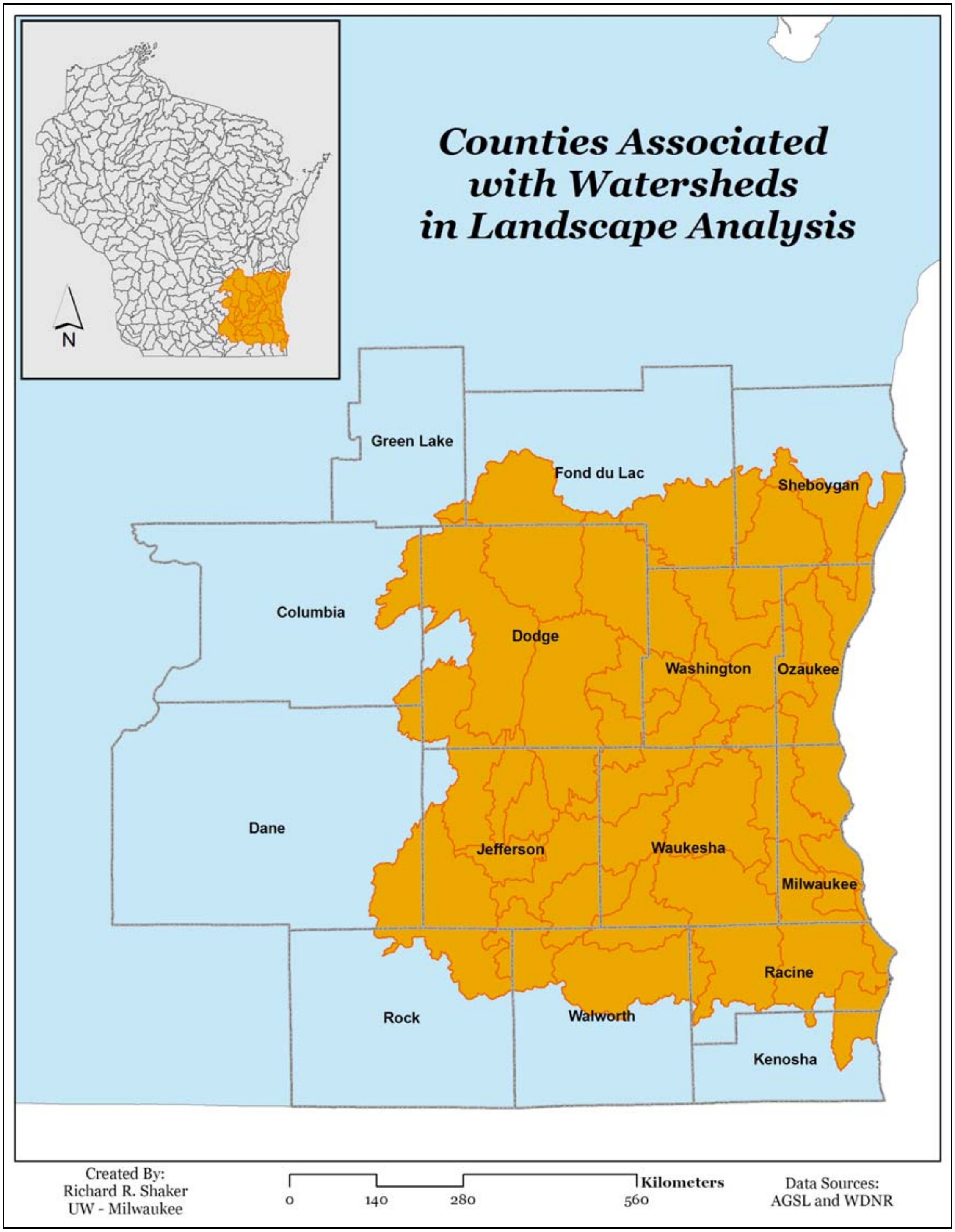

Figure 2. Map of counties associated with this analysis.

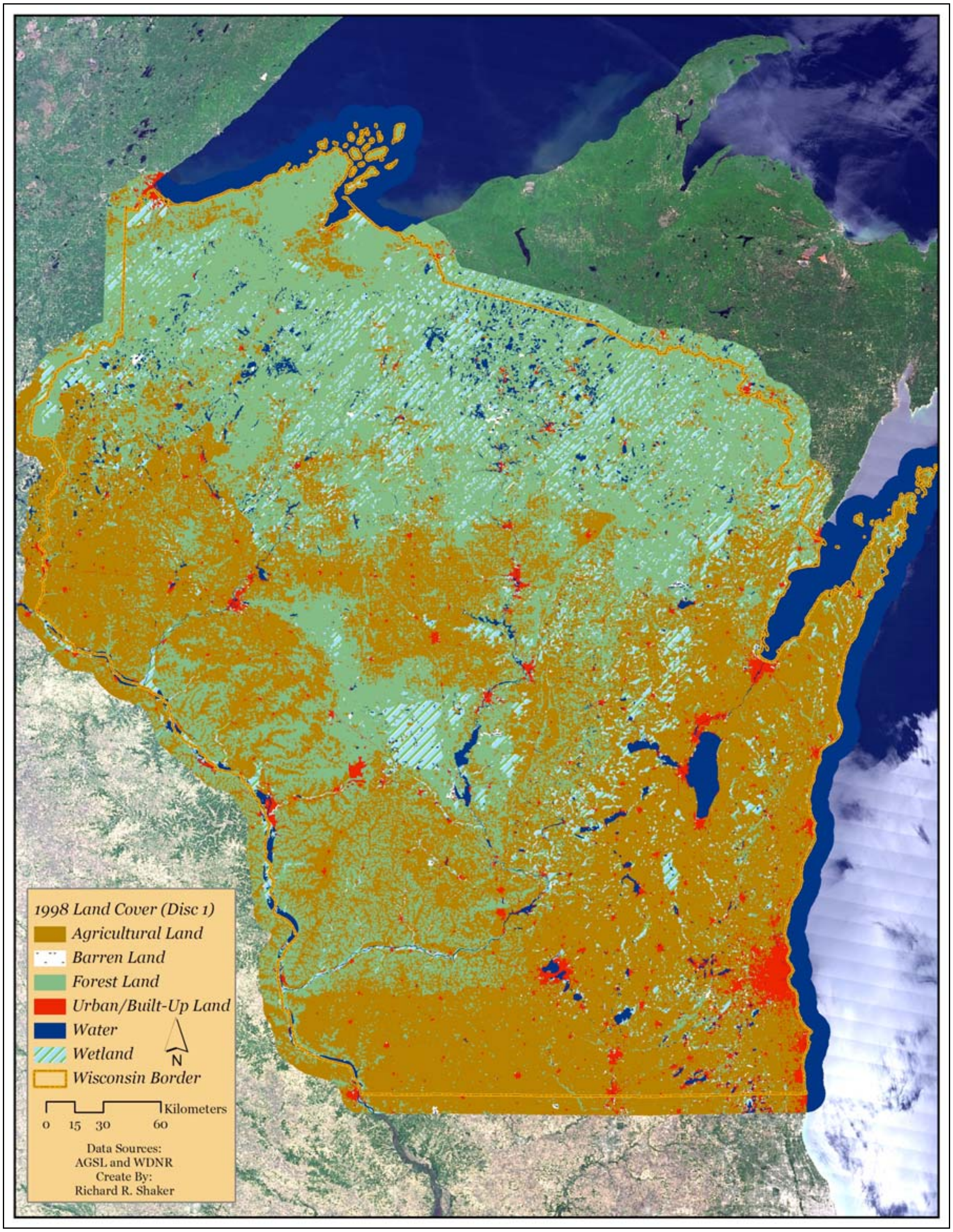

Figure 3. Map of WISCLAND land cover data for Wisconsin (Description 1).

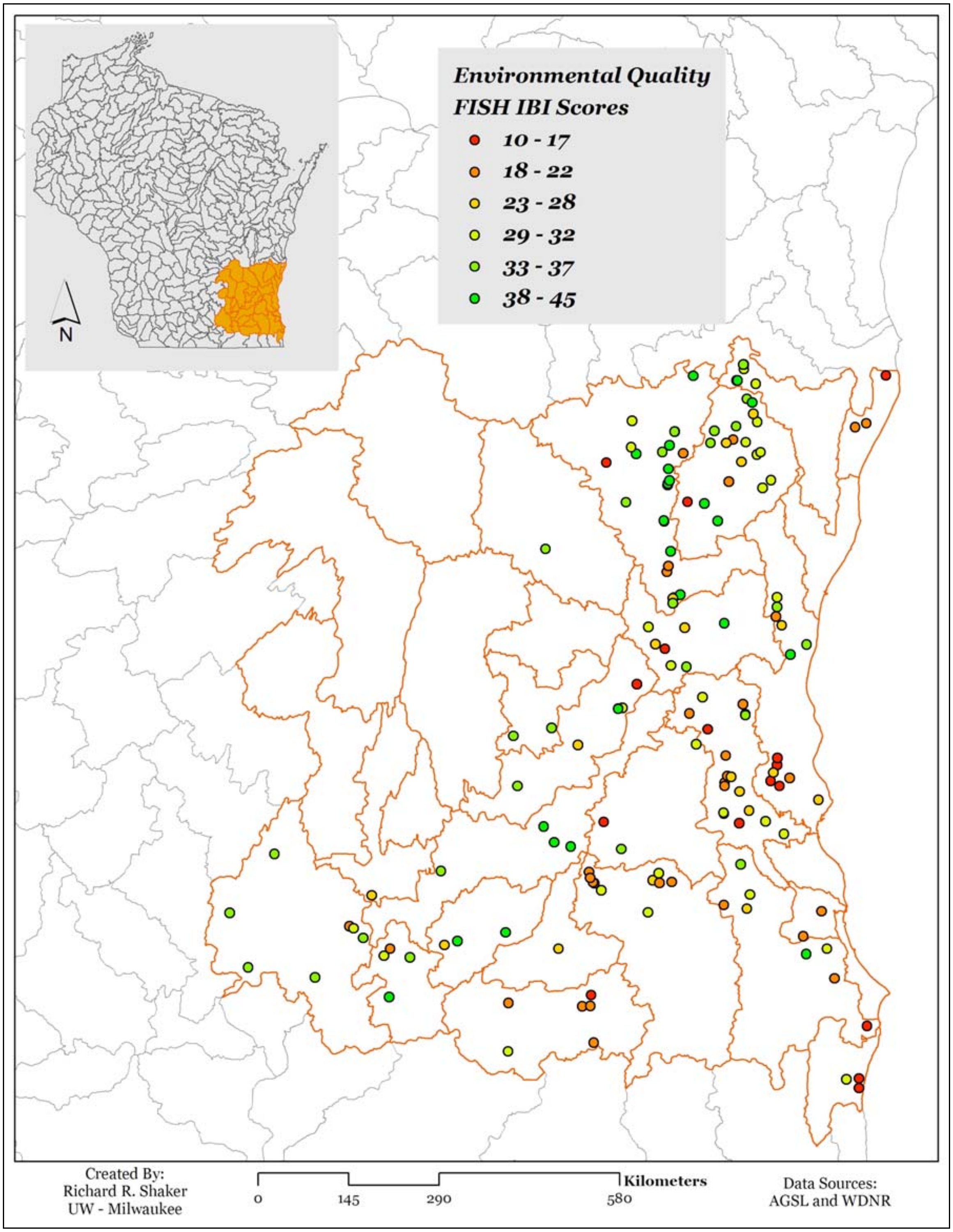

Figure 4. 152 fish sample sites associated with the measure of environmental quality (IBI scores) for the 31 watersheds in study area.

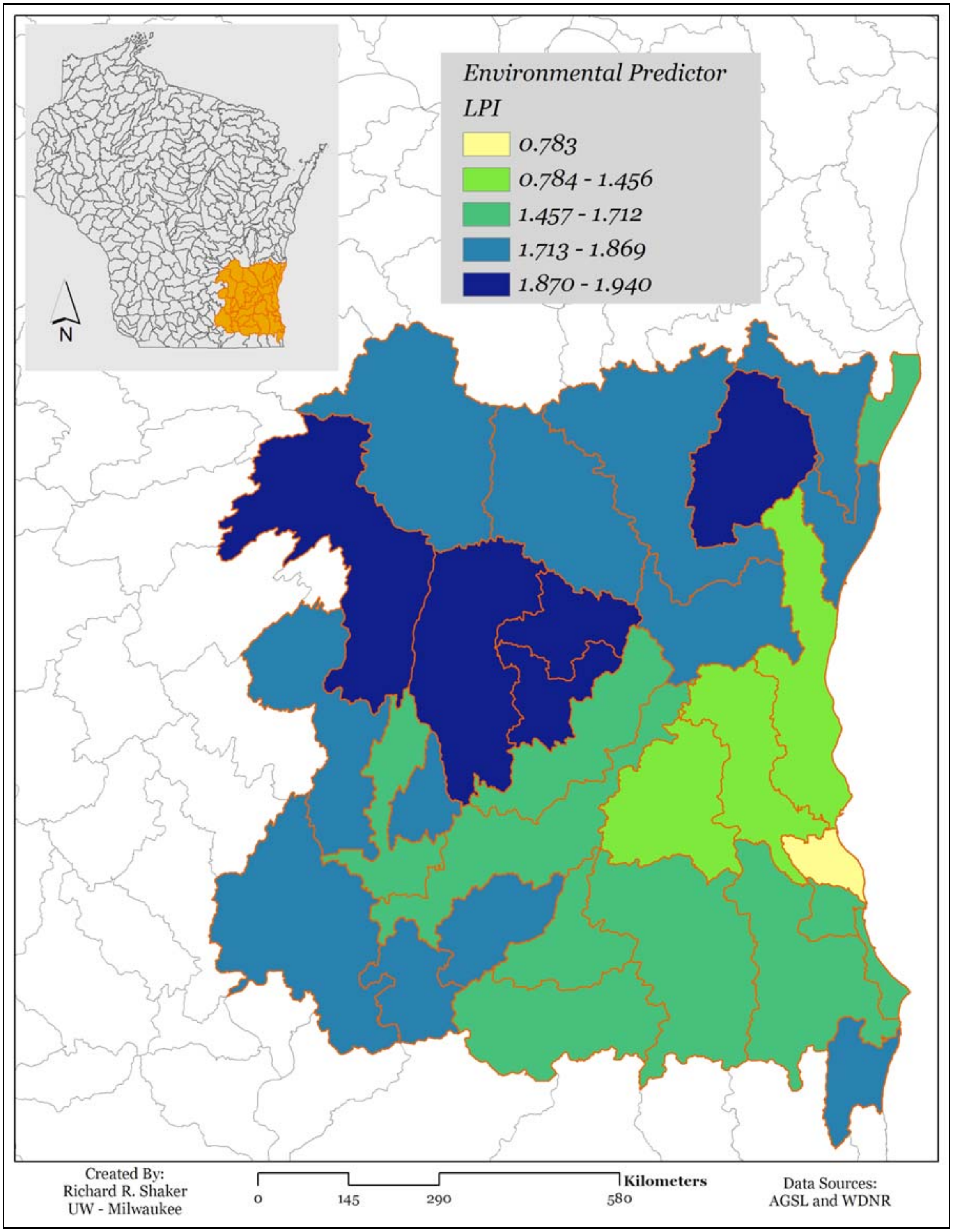

Figure 5. Map of largest patch index (LPI).

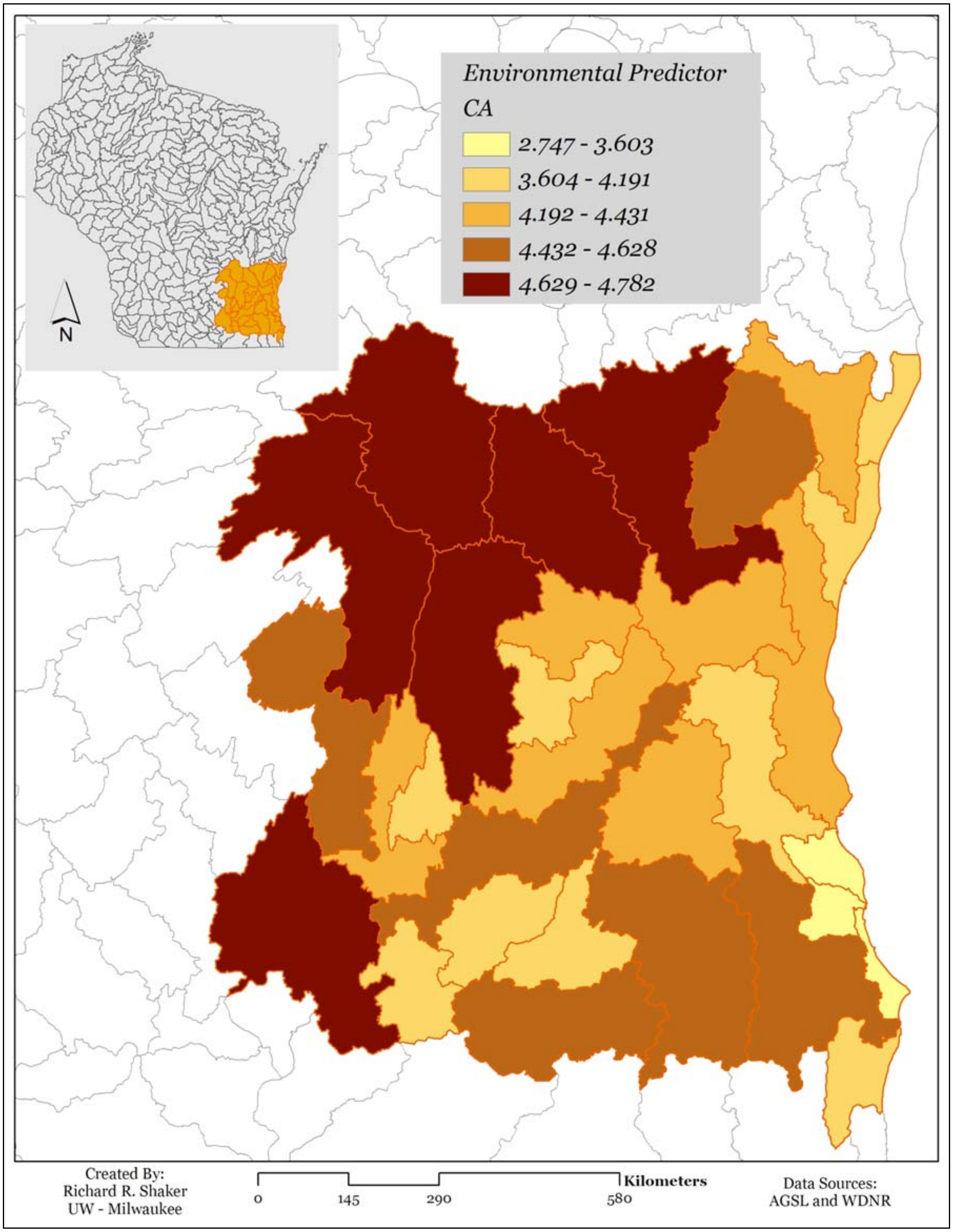

Figure 6. Map of total class area (CA).

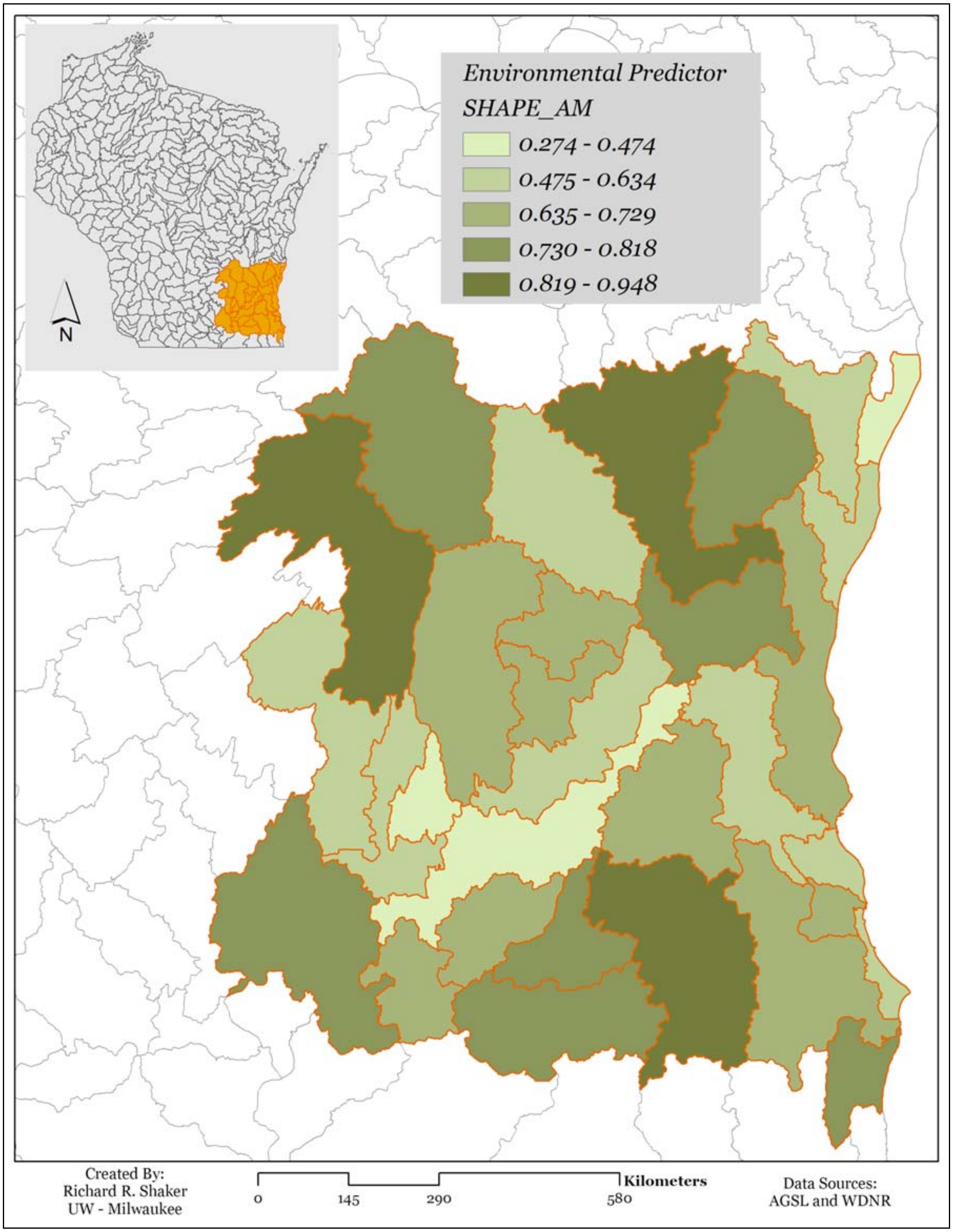

Figure 7. Map of area weighted mean shape index (SHAPE_AM).

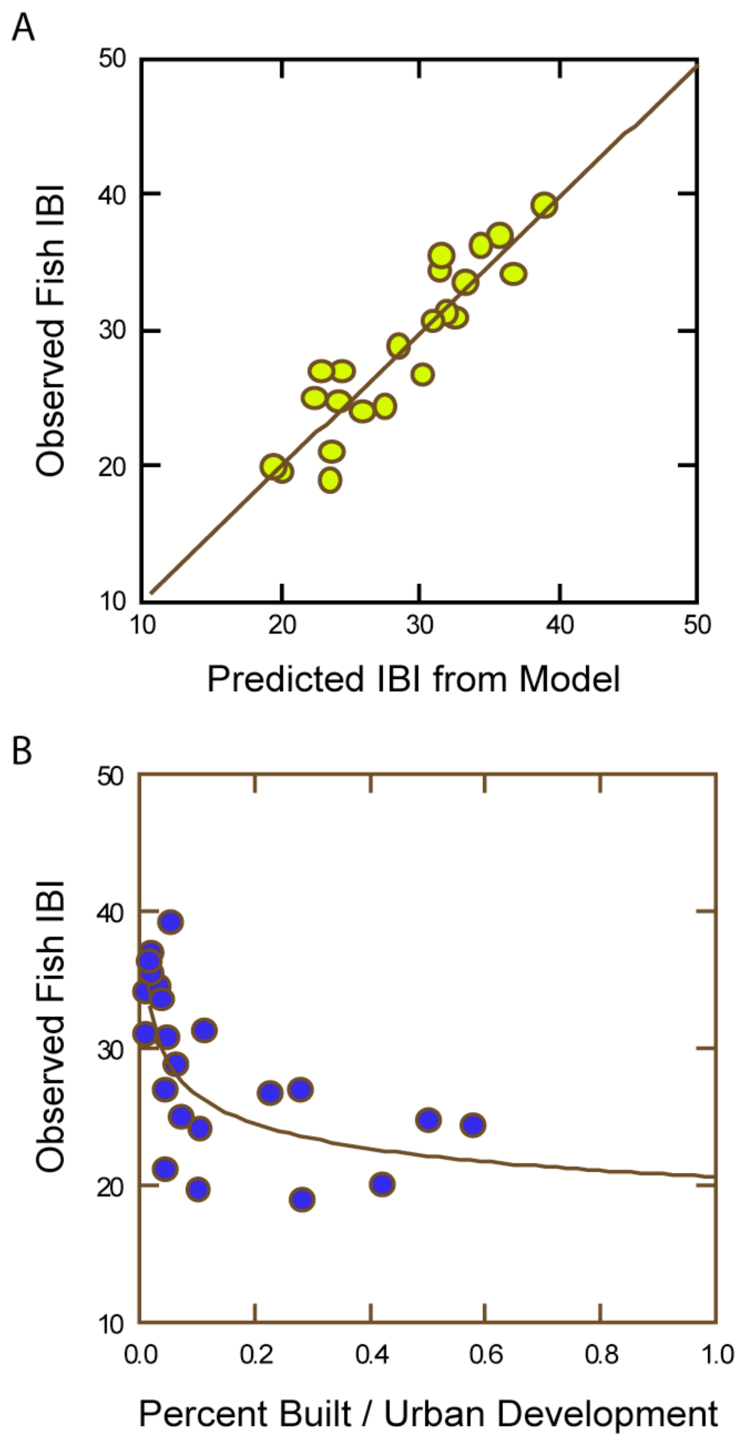

Figure 8. Scatter plot of Observed Fish IBI in the study watersheds versus (A) predicted IBI from multiple regression of landscape fragmentation metrics, and (B) percentage of built-up / urban development in the watersheds.

View publication stats