URBAN RIPARIAN AREAS: ECOLOGICAL AND STREAMSIDE-ORDINANCE ASSESSMENTS

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DOCTOR OF PHILOSOPHY

By

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STREAMSIDE-ORDINANCE ASSESSMENTS

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ABSTRACT

Streamside protection ordinances have been established in many urbanizing areas; however, there has been a paucity of assessments of the effectiveness of such ordinances. A quantitative assessment of the ecological impairments in an urban river system can provide a key component in a basin-scale management plan. A study was conducted in the Blue River Basin, Kansas and Missouri, to determine the effectiveness of streamside ordinances and assess the temporal and spatial changes in the ecological health of the river. Study objectives included the determination of the vegetation change within ordinance protected and non-ordinance protected areas within the study area, and a spatial and temporal assessment of ecological impairment through the development of a quantitative index—the Ecological Index of Urban Stream Health (EUISH).

SPOT imagery was used to classify landscape changes over time (1992 through 2012), across multiple jurisdictions, and pre- and post-ordinance implementation periods. The GIS-based EIUSH included eight spatial data layers representing five environmental categories including physical habitat, hydrology, water quality, land use/land cover, and aquatic communities.

Results of the overall effectiveness of streamside ordinance protection indicated tree cover declined 12.5%, grass cover declined 9.7%, and developed land increased 22.9% during the 20-year analysis period. These results indicate that streamside ordinances along

the Blue River need modifications in order to be more effective at limiting land use and land cover change.

The mean EIUSH score was about 45 (0-100 scale) and ranged from 25 to 82. Index scores differed substantially by river reach and with time, with the lowest scores determined for the lower part of the basin and a major tributary, Indian Creek, and the highest scores determined for the upstream part of the basin. Temporal updates of EIUSH scores indicated the greatest index decline was in the upper Blue River—the area of highest ecological integrity in the basin. The index can be used to spatially target and maintain the riparian areas of highest ecological integrity. Alternatively, the index can be used to spatially identify areas of highest ecological impairment, and the likely causes of the impairment, so that the conditions can be addressed.

APPROVAL PAGE

The faculty listed below, appointed by the Dean of the School of Graduate Studies, have examined a dissertation titled "Urban Riparian Areas: Ecological And Streamside-Ordinance Assessments," presented by Trina Elaine Weilert, candidate for the Doctor of Philosophy degree and hereby certify that in their opinion it is worthy of acceptance.

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ABBREVIATIONS

E.coli	*Escherichia coli*
EIUSH	Ecological Index of Urban Stream Health
ERDAS	Earth Resources Data Analysis System
ESRI	Environmental Systems Research Institute
FIB	Fecal indicator bacteria
GIS	Geographic Information Systems
IHA	Indicators of Hydrologic Alteration software
LULC	Land Use Land Cover
MDC	Missouri Department of Conservation
NLCD	National Land Cover Dataset
NPDES	National Pollutant Discharge Elimination System
RMSE	Root Mean Square Error
SPOT	System Pour l'Observation de la Terre
TWPE	Toxic Weighted Pounds Equivalent
USACE	United States Army Corps of Engineers
USCB	United States Census Bureau
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Services
USGS	United States Geological Survey
USNRCS	United States Natural Resources Conservation Services
UTM	Universal Transverse Mercator

WWTP	Waste Water Treatment plant

CONVERSION FACTORS

International System of Units to Inch/Pound

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	0.0002471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
liter (L)	33.82	ounce, fluid (fl. oz)
liter (L)	0.2642	gallon (gal)
cubic meter (m^3)	35.31	cubic foot (ft^3)
	Flow rate	
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)

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All the SPOT images used in this study were acquired through a previous remote sensing study of urban wetland changes in the study area that was supported by US EPA Grant CD 97701501 and by the grant of the Friends of Library of University of Missouri – Kansas City (Ji, 2006).

CHAPTER 1

INTRODUCTION

Background

Historically, waterways were considered a commons area, owned by none and used by all. Ward (1997) considered the history of the water commons and the value that was placed on the preservation of these areas. He emphasized that since the waterways were a limited and important resource serving the good of many, users would have been extra careful to protect and cherish them. The modern distribution of potable water resources has resulted in the alienation of humans from their water sources (Ward, 1997). In advanced or developed countries, water is provided to its users by being processed and piped into individual residences. Ward (1997) described the disconnection from the commons because in developed countries, most people do not have to go out and collect their water, or worry about its quality, or a lack of supply, and, as such, they feel less concerned about its protection. This perception has changed as a result of the shift from a rural towards an urban-centered society. In 1860, less than 20% of the total population of the United States lived in urban areas (Gibson & Jung, 2005), whereas in 2010, the urban population represented over 80% of the total population (USCB, 2010). The increase in urbanized areas and corresponding loss of natural conditions has produced environmental impairment that continues to worsen in older urban areas with aging and failing infrastructure. Additionally, urban areas commonly have been established near or alongside rivers to satisfy transportation and water resource needs, which has resulted in increased anthropogenic impacts on urban rivers. The most common causes of non-point source pollution in the United States waterways include agriculture, as well as urban

development impairments—hydrologic modifications, habitat modifications, an urban runoff (USEPA, 2005).

The importance of waterways has changed with time in the United States. The economy has shifted from an agricultural-based economy in the 1700s and 1800s that required a water source in proximity to residences for human and animal consumption, to an industrial economy that used waterways for transportation purposes. Individuals largely are separated from a direct dependency on waterways in our modern economy, but even today, the full benefits of a river system are immeasurable to society (Karr, 1999). These benefits frequently go unnoticed by individuals until the resources are limited or unfit for use. This is a result of competition for resources as clearly a struggle exists between the societal desire for stream and environmental protection, and the economics and opportunity costs of that protection when it conflicts with development (Cropper & Oates, 1992).

River management is a complex process with the various components provided under the jurisdiction of entities at a variety of scales, political boundaries, and levels of oversight or authority. The protections needed to ensure the ecological health of a river system are most effective when they include the whole system, from headwaters to the mouth of the river, and management is a relative scale issue, starting at the watershed (river system) level and going down to a river reach level issue. Geographic Information System (GIS) analysis can provide a means of quantifying the spatial and temporal effects of historical and recent management outcomes in stream systems across multiple jurisdictions and scales. A need exists for a basin-wide assessment that conveys the geographical and spatial distribution of issues that can utilize multiple sources of data and indicators of stream-system health. A system-wide or large-scale assessment is particularly important in river systems in urban areas because of the array of

stressors. These stressors include both point and non-point sources of pollution, highly varied land use changes that generally results in altered vegetation, and extensive physical alteration of the stream channel and riparian corridor. Such an effort can identify, map, and rank areas that are least disturbed (most natural) to most disturbed thus showing hotspots and areas of concern. Spatially-weighted indexes are a good solution that have been used as a means of assessing a number of variables of ecological integrity of river systems, including biological and water quality factors (Karr, 1999; Reif, 2002; Vander Laan & Hawkins, 2014) and addressing the leading sources of water-quality impairment.

Urbanized areas have high levels of impermeable surfaces that limit the infiltration of rainwater into the soil, alter the quantity and timing of runoff, and restrict contributions to the underlying water table. One factor limiting the effectiveness of ecosystem services provided by the riparian zone occurs when cities pipe the rainfall runoff directly to the river, bypassing the riparian buffer system. A second limiting factor in the effectiveness of the riparian zone is human alteration of the natural vegetation, the loss of which can reduce the ability of these areas to filter and retain pollutants. Despite the proven benefits, vegetation losses and degradation of riparian areas continue (Meyer & Paul, 2001; Jones et al., 2010) with alterations in natural land cover and the development of urban areas (Figure 1.1). The use and preservation of riparian areas is presented as a best management practices in managing non-point source pollutants in runoff (USEPA, 2014). The EPA also presents proposed model aquatic ordinances for administrative areas to adopt as their best management practices (USEPA, 2016). The model guidance includes varying buffer widths that are dependent on the drainage basin acreage, the slope of the river banks, and various factors that administrative

jurisdictions have to consider. These proposed ordinances are designed as a starting point for the jurisdictions to take and adopt.

Figure 1.1. Schematic showing a natural riparian zone and an urban riparian zone.

The unaltered drainage areas of rivers, streams, and lakes generally act as natural filtration systems removing silt and chemical pollutants. Primary filtration takes place within the riparian areas or stream buffers (strips of streamside vegetation used in conservation practices). Secondary filtration takes place in the channel banks and subsoils as water travels

to the outflow point by interflow. Riparian areas in urbanizing settings have been shown to experience vegetation loss and degradation as a result of transportation needs (Booth et al., 2004), flood management efforts (Arnold & Gibbons, 1996), efforts to increase efficiency in runoff (Groffman et al., 2003; Bettez & Groffman, 2012), the effects of increased overland flow (Booth & Jackson, 1997; Walsh et al., 2005b), and sediment and pollutants from alterations in the surrounding landscape (Hatt, Fletcher, Walsh, & Taylor, 2004).

Riparian areas are difficult to protect and regulate because they are frequently located in relatively flat landscapes that are desirable for development with a diversity of land covers and features within their lateral and longitudinal extents. One measure put in place to protect riparian areas in urban areas is the passage of streamside protection ordinances (hereafter referred to as streamside ordinances), but little information is available as to the effectiveness of these voluntary measures to protect land cover from human alterations. Rivers cross jurisdictional boundaries and this can lead to varying protection of the adjacent riparian areas as a result of inconsistent levels of implementation of streamside ordinances, or a lack of implementation in some jurisdictions. Water that arrives already impaired into a protected area is afforded little opportunity for remediation. Streamside ordinances can help to prevent the degradation of water quality, but they have little effect if the water is already impaired, if the area is already heavily impacted by development, or if protected areas are below thresholds in size needed to improve water quality.

Streamside ordinances have been designed and implemented across the United States at municipal levels as a method of protecting riparian areas and the connected water body. Ordinances are laws that are implemented at local levels to govern matters not addressed at the state or federal level. Limited research has been conducted to determine the effectiveness of the

streamside ordinances, and, more specifically, how effective urban streamside ordinances are at protecting urbanized riparian areas following the ordinance implementation. Yeakley, Ozawa, and Hook (2006) and Ozawa and Yeakley (2007) are two of the few studies addressing the performance of streamside ordinances. The consideration of the rate of land cover change prior to the ordinance implementation and after the ordinances were implemented has not been fully addressed and continues to be a shortcoming in this area of research.

The implementation of streamside ordinances alone does not provide protection of riparian vegetation without effective oversight (USEPA, 2002). To assess the effectiveness of streamside ordinances on maintaining riparian vegetation during a 20-year period, a study was undertaken in an urbanized basin within the Kansas City Metropolitan area. Streamside vegetation is but one of several controlling factors in the ecological integrity of stream systems. A second objective of this study was to conduct an assessment of the ecological integrity of an urbanized stream system that could aid in a holistic approach to the management of riparian and stream systems within this urban area.

Research Objectives

The primary objectives of this study included:

1. To determine the effectiveness of streamside ordinances at protecting riparian vegetation along the urban stream system, pre- and post-ordinance periods were compared for potential land cover changes. Specific objectives included:

    a) Analyze the riparian areas for temporal and spatial land-use changes within the study corridor.
    b) Analyze the riparian buffer vegetation for temporal and compositional changes per municipality.
    c) Determine the rate of change of riparian land-use over time.

2. Develop a spatial assessment of the ecological health of an urban river system that included quantifying multiple ecological components to describe spatial and temporal characteristics of the health of the river system. Specific objectives included:

a) Compile existing ecological data to determine and demonstrate an effective means of assessing and displaying the major components that contribute to the ecological integrity of the river system.
b) Develop a weighted ecological index based on major stream-system components to produce an assessment of the overall health of the river system. This index will be developed for a period coinciding with the ordinance assessment period (version I), and the index will be developed for a second period in time using available updated data (version II).

The assessment of ecological impairments and the determination of the health of an urban river system can provide a starting point and key component of decision making processes (Figure 1.2) within a possible basin management plan. Study objective 1 includes the determination of the vegetation change within ordinance protected and non-ordinance protected areas within the study area with consideration of the time frames before and after streamside ordinances were implemented. The maintenance of natural riparian vegetation directly or indirectly affects the overall ecological assessment of the study area as determined-in objective 2.

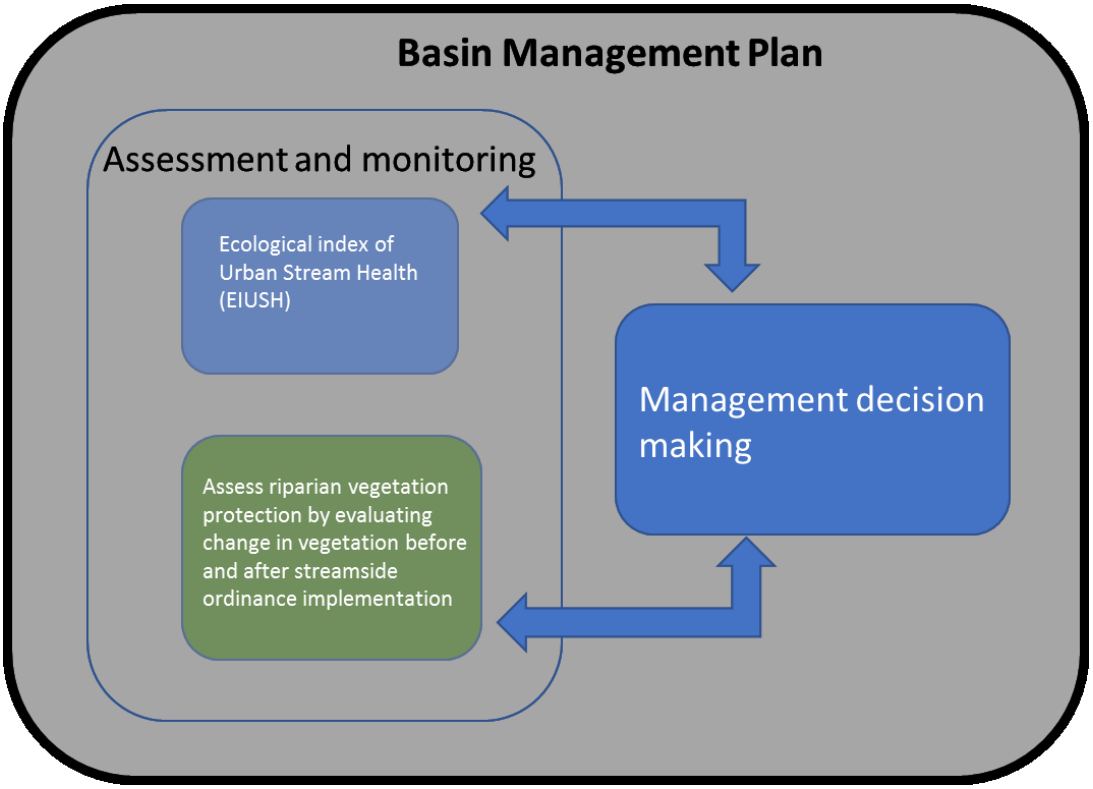


Figure 1.2. Project Flowchart showing potential use of study components in a basin management plan.

Study Area

The study area consisted of a portion of the Blue River Basin (725 km^2) located within the Kansas City metropolitan area (Figure 1.3). The Blue River Basin covers parts of two states (Missouri and Kansas); five counties, including Johnson, Miami, and Wyandotte Counties in Kansas and Jackson, and Cass Counties in Missouri; and 22 municipalities (Mid-America Regional Council, 2017). The geographic scope of the study was limited to the part of the Blue River main stem that is located within Jackson County, Missouri, and Johnson County, Kansas. The Blue River headwaters are in Johnson County, Kansas, and the river flows for seven miles in Kansas in a primarily northeastern direction before entering Jackson County, Missouri,

where it continues for another 35 miles before reaching the confluence with the Missouri River (Figure 1.3; MDC, 2016).

The use of the Kansas City metropolitan area offered the opportunity to include an urbanized setting with diverse socioeconomic makeup, and a multi-jurisdictional setting along a river system representing similar ecological and environmental conditions. The Kansas City metropolitan area has more freeway miles per car than any other city in the United States (Vault.sierraclub.org, 1998). Long-term population trends in Kansas City, Missouri, and Overland Park, Kansas, (Figure 1.4) indicate that Kansas City experienced the greatest population growth between 1860 and 1970, with a decline in population between 1970 until the late 1980s, followed by a secondary increase in population from 1990 to 2010 (U.S. Census Bureau, 2010). The city of Overland Park has experienced steady population increases from 1960 through the latest census in 2010. The rate of population growth has been much greater over the last 50 years in the later developing suburb of Overland Park, Kansas, compared to Kansas City, Missouri, and Kansas City has had only minor changes in its population over the last 50 years (Gibson, 1998; Gibson & Jung, 2005).

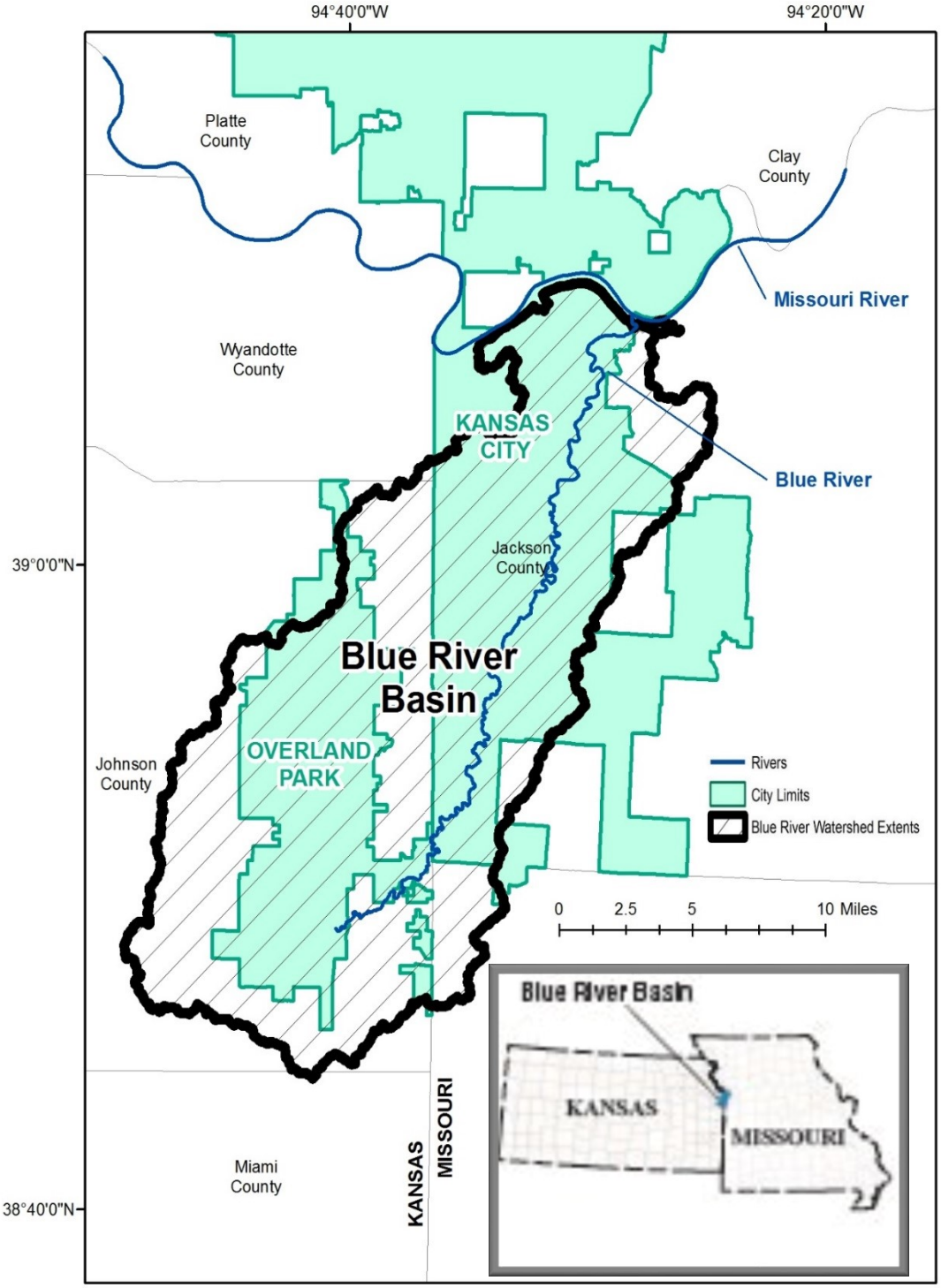


Figure 1.3. Blue River Basin, Kansas City, Missouri, and Overland Park, Kansas.

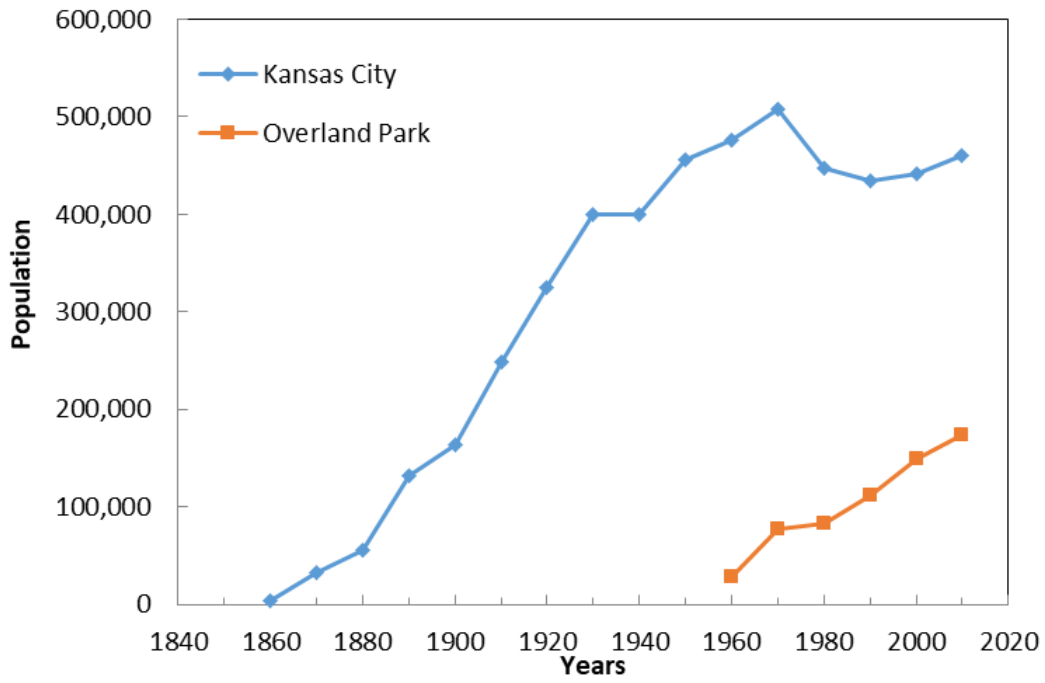
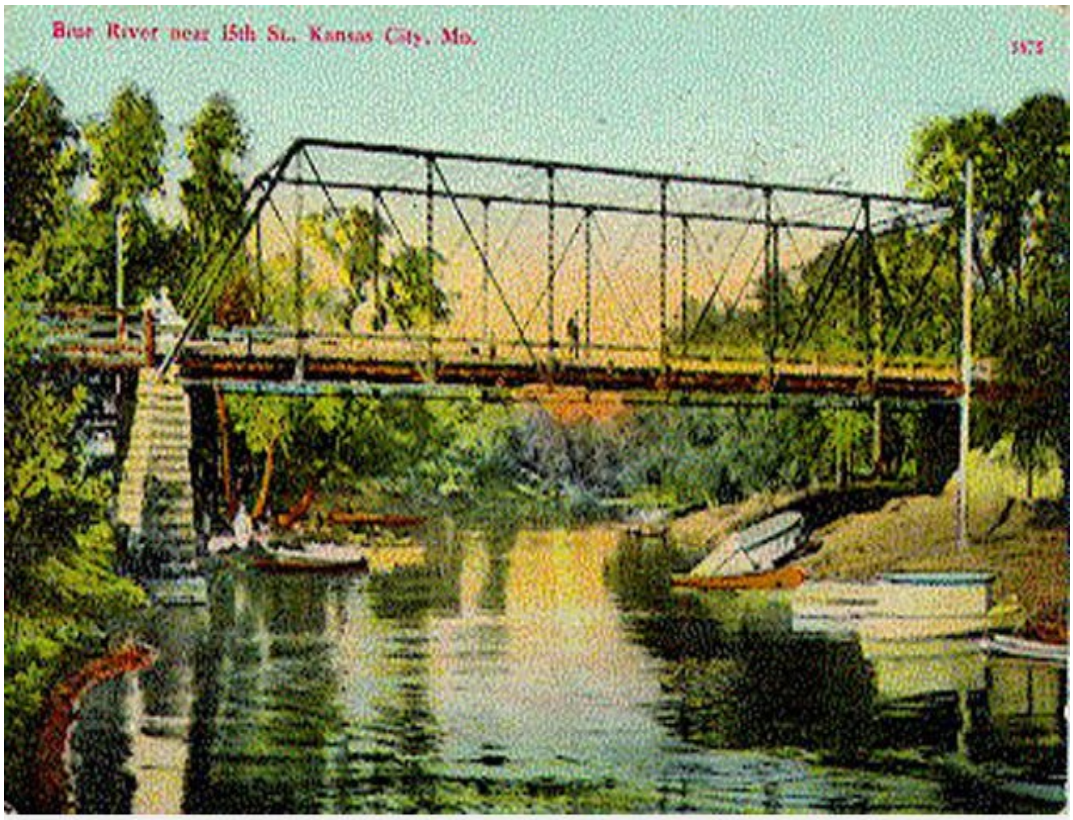


Figure 1.4. Population trends of the city of Kansas City, Missouri, and city of Overland Park, Kansas, 1860-2015. (Source: Gibson & Jung 2005; U. S. Census Bureau, 2015).

The land within the Blue River Basin was developed earlier in Missouri than in Kansas because of the earlier population growth in the city of Kansas City compared to that of the outlying suburbs including Overland Park, Kansas (Figure 1.3; Figure 1.4). Additionally, the downstream portion of the Blue River in Missouri historically played a large role in recreation and entertainment in the Kansas City area (Figure 1.5). The riverfront of the Blue River also provided economic benefits in jobs and income for the working class to rent boats and provide services to the elite during the summers (Schirmer & McKinzie, 1982).



MVSC, Kansas City Public Library

Figure 1.5. Historical setting of the lower Blue River, 1912. (Source: Kansas City Public Library, 1912)

Despite the later formation and development, the city of Overland Park, Kansas, implemented streamside ordinances prior to the city of Kansas City. The city of Overland Park implemented ordinances in 2003 (City of Overland Park, 2002) and Kansas City, Missouri, implemented streamside ordinances in 2008 (City of Kansas City, 2008). The temporal scope of the analysis of streamside ordinances in this study (1992 – 2012) includes pre- and post-streamside ordinance implementation periods within the Blue River riparian corridor in order to determine the effects of the ordinances on land use. An Ecological Index of Urban Stream Health (EIUSH) was developed for a period coinciding with the ordinance evaluation period

(data primarily from or near 2006) and then reassessed and available data layers were updated for a later period (through 2016).

CHAPTER 2

LITERATURE REVIEW

Riparian Areas

A major component of the ecological integrity of urban stream systems is the riparian area. The following discussion will focus on the definitions of the riparian area, the many functions and services of these areas, and the use of vegetation buffers in place of intact riparian areas in disturbed areas.

Definition

There are many definitions of the riparian area depending on agency, management goals, and geographic setting (Verry Dolloff, & Manning, 2004). Given the many definitions and that the boundaries and extents of riparian areas are difficult to define through time (Naiman & Decamps, 1997; Ferreira, Aguiar, & Nogueira, 2005), it also is hard to create legislation that fully protects these areas. At the simplest level, riparian areas are defined as a protection zone that serves to buffer the adjacent waterbody from, for example, nutrients and sediment during the runoff process. Riparian zones are referred to as "an area of direct interaction between terrestrial and aquatic ecosystems" (Gregory, Swanson, McKee, & Cummins, 1991). More formally, the following definition of riparian areas is presented by Lowrance, Leonard, and Sheridan (1985):

"a complex assemblage of plants and other organisms in an environment adjacent to water. Without definitive boundaries, it may include stream banks, floodplain, and wetlands, as well as sub-irrigated sites forming a transitional zone between upland and aquatic habitat. Mainly linear in shape and extent, they are characterized by laterally flowing water that rises and falls at least once within a growing season".

Ecosystem Functions and Services

Riparian areas provide many ecological functions including serving as a natural filters of runoff (Johnston & Naiman, 1990; Li et al., 2014) and nonpoint sources of pollution (Lowrance et al., 1985), controlling runoff and flooding (Turner-Gillespie, Smith, & Bates, 2003; Mitsch, 1992), providing shade and temperature control (Mitchell, 1999; Moore, Spittlehouse, & Story, 2005), and providing habitat for wildlife (National Research Council, 1995; Lowrance et al., 1985).

The riparian corridor provides ecological services that produce value for humans, and the corridor improves the ecological functioning in the surrounding stream ecosystem. The type and magnitude of the ecological functions can vary in relation to the vegetation composition and areal extent (Wenger, 1999). Riparian areas can exist in a remnant or natural unaltered state, in various levels of altered states, or as manmade, and restored riparian systems. The vegetation composition can vary between forested, herbaceous, shrub cover, or any combination of the three vegetation types (Nilsson & Berggren, 2000). Additionally, the vegetation can be made up of invasive, introduced, or non-native species, which also contribute to the overall health of the riparian area and will relate to its effectiveness as a riparian buffer.

Riparian zones can provide ecosystem services for humans as a result of the benefits of flood protection, chemical runoff uptake, and associated improvements to water quality, and by providing recreation, shade, and green space in urbanized areas. The variation of the riparian zone can alter the values and degree of values provided for humans. A mature, densely forested, riparian zone can uptake more water and chemical runoff than can be taken up by younger forests, sparsely treed forests, shrubs, or grasslands (Lowrance & Sheridan, 2005). Shade benefits would be greatest from more densely forested riparian zones compared to those

with sparse cover, whereas the recreational use of green space likely would be greater in a grassy riparian zone (Center for Environmental Policy, 2000). A riparian area comprised primarily of shrubs and vines would impede runoff, reduce flash flooding and trap sediments and pollutants better than a grassy riparian zone, but not as well as a forested zone.

In a natural or remnant state, the riparian area enhances the stream ecosystem by providing continuous linear terrestrial habitat for wildlife, and the riparian vegetation also stabilizes stream banks and limits erosion and transport of sediment into the waterbody. The intact riparian vegetation corridor also can improve the stream ecosystem by providing organic matter for ecological production, canopy protection, and reducing the fluctuation in water temperature and light levels, which also can help prevent algae blooms (Meyer & Paul, 2001). The remnant riparian zone also can support a more diverse wildlife and ensure a healthier ecosystem than a modified riparian zone that has exotic species overtaking the habitat. Severe weather such as droughts or heavy periods of rainfall will likely be tolerated more by the native species compared to the exotic species as a result of climatic adaptation, which would result in a healthier riparian zone (McDowell et al., 2008).

Riparian Vegetation Buffers

In urban and agricultural systems across the United States, the natural riparian vegetation generally is removed or modified leaving designated narrow strips of vegetation adjacent to the waterbody. These vegetation strips are designed to maintain some level of stream protection and beneficial ecological functions, and still allow for the maximization of the desired land use modifications. These vegetation strips—termed buffer strips, greenways, conservation buffers, windbreaks, filter strips, and streamside ordinance zones—provide

benefits including protection of water quality, terrestrial and aquatic habitat, erosion protection, aesthetics, and recreational uses (Figure 2.1) (Bentrup, 2008).

Structural characteristics of a buffer such as size and shape and the vegetation type largely determine how well a buffer is capable of functioning at a given location. Planners can manipulate these variables to achieve the desired objectives (Bentrup, 2008), although in most cases the primary design factor is the buffer width, either a static or variable width, based on stream size or drainage area.

Issue and Objectives	Buffer Functions
Water Quality	
Reduce erosion and runoff of sediment, nutrients, and other potential pollutants Remove pollutants from water runoff and wind	Slow water runoff and enhance infiltration Trap pollutants in surface runoff Trap pollutants in subsurface flow Stabilize soil Reduce bank erosion
Biodiversity	
Enhance terrestrial habitat Enhance aquatic habitat	Increase habitat area Protect sensitive habitats Restore connectivity Increase access to resources Shade stream to maintain temperature
Productive Soils	
Reduce soil erosion Increase soil productivity	Reduce water runoff energy Reduce wind energy Stabilize soil Improve soil quality Remove soil pollutants
Economic Opportunities	
Provide income sources Increase economic diversity Increase economic value	Produce marketable products Reduce energy consumption Increase property values Provide alternative energy sources Provide ecosystem services
Protection and Safety	
Protect from wind or snow Increase biological control of pests Protect from flood waters Create a safe enviroment	Reduce wind energy Modify microclimate Enhance habitat for predators of pests Reduce flood water levels and erosion Reduce hazards
Aesthetics and Visual Quality	
Enhance visual quality Control noise levels Control air pollutants and odor	Enhance visual interest Screen undesirable views Screen undesirable noise Filter air pollutants and odors Separate human activities
Outdoor Recreation	
Promote nature-based recreation Use buffers as recreational trails	Increase natural area Protect natural areas Protect soil and plant resources Provide a corridor for movement Enhance recreational experience

(Source: Bentrup, 2008)

Figure 2.1. Functions of vegetation buffers related to - issues and objectives of buffer development.

Ecological Effects of Urbanization

As of 2001, over 75% of the United States population lived in urban areas (Meyer & Paul, 2001). Urban expansion in the United States is expected to continue over the next 25 years with a projected increase of 79%, raising the portion of the total land base that is developed from 5.2 to 9.2%. The expected growth is projected to affect areas that are already stressed from anthropogenic impacts, including river systems, wildlife habitat, sensitive watersheds, and riparian areas (Alig, Kline, & Lichtenstein, 2004).

Urbanization has been linked to the impairment of river systems as noted in the term, "urban stream syndrome", coined by Meyer, Paul, and Taulbee (2005), which describes the ecological degradation of urban streams. Urbanization affects the physical, chemical, and biological characteristics of streams and understanding these effects is important for managing aquatic resources. The effects of urbanization on streams varies widely, depending on the geographic area studied, the initial ecosystem conditions, and stage of urbanization. Results of an investigation of the effects of urbanization on stream ecosystems (USGS, 2014) found that no single environmental factor was universally important in explaining why the health of streams decline as levels of urban development increase. Urbanization results in a number of alterations in the land cover, associated ecological processes, and biological communities in the affected areas as discussed in the following sections.

Aquatic/Biological Communities

Increasing levels of urban land use have been connected to a decrease in water quality and a decrease in sensitive aquatic biological communities (Meyer & Paul, 2001; Riley et al., 2005). In a study of multiple urban areas throughout the United States, USGS (2017a) found that biological communities were sensitive to even low levels (5% in some cases) of urban

development, as sensitive species were affected in even relatively undisturbed watersheds. Even with moderate to high levels of urban development, however, the biological community was not degraded to the point that it only included the most tolerant species, and they hypothesize that stream restoration efforts could still have a positive rehabilitating effect on the biological community despite high levels of development.

Wilkison, Armstrong, and Hampton (2009) found a statistically significant relation between aquatic-life metric scores and percent urbanization—defined as the roadway surface area plus the commercial, industrial, and residential land use—at sites in the Blue River Basin. The results indicated that aquatic community health and diversity declined as the percent of urbanization increased in the basin.

Hydrology

The effects of urbanization on stream hydrology have been shown to include an increase in peak flows (Leopold, 1968; Rose & Peters, 2001), and an increase in the rate of change in flow (Hirsch, Walker, Day, & Kallio, 1990; Poff et al., 1997; Arnold & Gibbons, 1996; Rose & Peters, 2001) as a result of greater impervious area. The focus of watershed management in urbanized systems generally has been on flood protection (Walsh, Fletcher, & Ladson, 2005a) including more effective flood abatement designs in streams, channelization, and large networks of storm drains through the city. A decrease in groundwater recharge (Barringer et al., 1994; Rose & Peters, 2001) also has been found to be related to urbanization as a result of the increase in impervious surfaces (Rose & Peters, 2001), loss of tree cover (Hough, 1995), and corresponding increase in direct runoff and reduced infiltration. Roy et al (2006) found that the influence of hydrologic alteration had a greater effect on the richness and abundance of fish assemblages in an urban catchment than that of the riparian forests. They

state that a consistency in flow regime in streams is necessary to maintain species richness as alterations to the natural flow regime can cause biological populations to decline and change because they do not have time to adapt to the new habitat.

Land Use

Urbanization can result in the degradation of urban riparian areas and a substantial change in riparian vegetation composition (Burton & Samuelson, 2007; Atkinson, Hunter, & English, 2010). Roy et al (2006) expressed concern over the lack of knowledge and understanding of the functionality of riparian forests within urban systems, as such, presenting concern that "applying riparian buffers as management tools based on false assumptions of performance could lead to ineffective protection". Reduction in tree cover in urban areas has been shown to increase the rate of runoff (Hough, 1995). Research also has shown that a higher Index of Biotic Integrity is associated with greater riparian forest cover (Meyer & Paul, 2001). Riparian deforestation also has been shown to cause a reduction in food availability within the stream corridor, an increase in stream temperature, and as a result of the increased runoff has led to alteration of sediment and nutrient and toxin uptake (Meyer & Paul, 2001). Land use and land cover upstream of sampled sites were determined to be highly correlated with the quality of sampled aquatic macroinvertebrate communities in the Blue River Basin in Kansas and Missouri (Rassmussen, Lee, & Ziegler, 2008; Wilkison et al., 2009; Poulton, Graham, Rasmussen, & Stone, 2015).

Physical Habitat

Stream habitats can be severely degraded in areas of urban development resulting from vegetation loss and associated erosion and altered sediment transport, stream channelization, or other man-made channel alterations. Urban development affects stream hydrology and

hydraulics; sediment input, transport, and deposition; and can thereby alter channel form, aquatic habitat, and the aquatic organisms (Garie & McIntosh, 1986; Yoder & Rankin, 1996; Kennen, 1999; Meyer & Paul, 2001; Akay, Sivrikaya, & Gulci, 2013). The effects of channelization or increased magnitude of streamflows associated with urbanization often results in incised stream channels or an increase in the stream-channel cross-sectional area. The available sediment for erosion and transport may increase due to clearing of vegetation and construction. Channel and flood-plain processes of sediment erosion, transport, and deposition also may change as a result of alterations in streamflow (Wolman & Schick, 1967; Graf, 1975; Gregory, Davis, & Downs, 1992). Channel alteration and loss of riparian vegetation accounted for almost two-thirds of the difference in stream physical habitat assessment scores that were determined at sites along the Blue River (Wilkison et al., 2009) in Missouri and Kansas. The magnitude of responses to alterations depends on natural environmental factors, including characteristics of remaining riparian vegetation, geology, and soils. The sediment transport and channel response in altered urban systems may take many decades to stabilize (Finkenbine, Atwater, & Mavinic, 2000; Henshaw & Booth, 2000; Bledsoe & Watson, 2001).

Water Quality

Concentrations of contaminants including nutrients, chloride, pesticides, and fecal-indicator bacteria (FIB), which includes *Escherichia coli* (*E. coli*), have been shown to increase with urban development and originate from both point and non-point sources. Five reaches of the Blue River in Missouri are listed in the State 303d list of impaired waters as the result of elevated *E. coli* levels (MODNR, 2018). Lee, Mau, and Rasmussen (2005) and Rasmusssen et al (2008) studied the relation between the occurrence of selected contaminants and degree of urbanization in Johnson County, Kansas, including sites in the Blue River Basin. Results

indicated that concentrations of these constituents (which included nutrients, suspended sediment, and FIB) generally were larger in more urban watersheds (including the Blue River and Indian Creek) than in non-urban basins.

Waste water treatment plants (WWTPs) are associated with high populated urban areas and have been shown to have substantial effects on water quality including increases in dissolved ammonia, (USGS, 1999) total phosphorous (Winter & Duthie, 2000; Pope & Putnam, 1997), and fecal coliform bacteria (Pope & Putnam, 1997). Wilkison et al (2006) determined that sampling sites in the Blue River Basin downstream from WWTPs had the largest concentrations of nutrients whereas FIB in the basin primarily were associated with non-point sources.

In urban areas with substantial snow fall, the variation in chloride concentrations throughout the year was particularly strongly related to urban development, likely a result of road salt applications during the winter. Wilkison et al (2009) determined that chloride concentrations were positively correlated with impervious cover and increased urban density in the Blue River Basin, Missouri and Kansas. Over 80 percent (%) of chloride samples collected at Blue River sampling sites in the month of January between 1998 and 2007 had concentrations greater than the U.S. Environmental Protection Agency national recommended aquatic life criteria of 230 mg/L for chronic aquatic life effects (USEPA, 2017a). Christensen and Krempa (2013) determined that chloride concentrations from Blue River and adjacent Little Blue River sample locations in the Kansas City metropolitan area had the highest correlation with a simple urban intensity index based on percent of impervious cover, population density, and forest cover in a 30-m buffer zone.

Ecological Models – Urban Ecological Indexes

Ecological health assessments are one of the key tools in river management and in the identification of major factors contributing to impaired systems. Quantified assessments of the ecological integrity of aquatic systems can be determined using one or more environmental indicator factors, or indexes, that serve to simplify and represent multiple and complex interactions among major components of system health. Selected metrics included in such an index should be applicable to the type of pollutants and impairments that are present in the system and represent the appropriate temporal and spatial scales (Boulton, 1999). Early studies on river health and early indexes focused on water quality metrics (McClellund, 1974; Yoder, 1991) due to the ease of monitoring, and water quality indexes are of value in areas in which water quality represents the primary impairment (Cude, 2001; Hallock, 2002; Finotti, Finkler, Susin, & Schneider, 2015; de Rosemond, Duro, & Dube, 2009; Sanches et al., 2006). Other ecological indexes have focused on aquatic communities including fish (Karr, 1981; Karr, Fausch, Angermeier, & Yant, 1986) and macroinvertebrates (Kerans & Karr, 1994; Lang & Reymond, 1995). The focus of other ecological indexes has been on physical habitat characteristics (Brooks et al., 2009; Munne, Prat, Sola, Bonada, & Rieradevall, 2003), whereas some indexes include multiple environmental factors (Ladson et al., 1999; Lee et al., 2011; Lee & An, 2014; Kim & An, 2015).

A variety of approaches have been used in creating disturbance indices and several basic decisions are required including which variables should be included, how should the included variables be weighed, and how should data values be translated into an index score. Urbanization will affect the ecological integrity of a stream system in several ways as described in the previous sections. In urban settings, ecological impairment largely is the

result of chemical pollutants and physical habitat degradation (Kim & An, 2015), but can include accompanying alterations in land use, hydrology, and aquatic communities. Water-quality monitoring alone, therefore, may not be sufficient for determining ecological health and further biological and physical assessments are needed. A water quality index, for example, could be insensitive to a decline in physical habitat structure, land cover changes, or altered hydrology.

In an urban setting, therefore, the variables to be included in an index should reflect the common impairments associated with development including physical habitat and channel reach changes, altered hydrology, altered land use and land cover, point source and non-point sources of contaminants, and the cumulative effects on the biological community of the stream.

Remote Sensing and GIS Applications in Urban River and Riparian Systems

Remote sensing has been used extensively for land use change studies in urban areas. Studies have been using remotely sensed imagery since the 1980s to detect change in the land cover type as well as the fragmentation of land cover over space which has been found to be important to species diversity (Congalton, 1991; Tso & Mather, 2001; Campbell & Wynne, 2011). Remotely sensed imagery offers many advantages compared to field studies including access, reduced costs and consistent updates. Such imagery allows for the analysis of large areas and for the determination of temporal changes as a result of land use land cover (LULC) changes resulting from development, disease, and natural disasters. The introduction of Landsat and SPOT satellite systems provides opportunities for reasonably priced repetitive captures of high spatial and spectral resolution satellite imagery. The imagery then is processed and analyzed in remote sensing software packages such as ERDAS Imagine (Intergraph Corporation, 2012). Previous studies have classified land use within riparian buffers (Goetz,

2006) finding that the use of Landsat can be challenging at the 30-m. resolution, but is doable with the combination of ancillary datasets to improve the classification. Gu and Liu (2010) also performed a study on riparian buffers focusing on using remote sensing and GIS to analyze and define riparian buffers, also incorporating Landsat imagery.

Adaptive Management and Scenario Planning

Addressing the potential impairment of multiple environmental factors in an urban setting crossing multiple jurisdictions requires a conservation management approach involving multiple and diverse agencies, information to make decisions, and the flexibility to vary tactics based on results. One management strategy—adaptive management—provides a method for making informed decisions about strategies, testing the effectiveness of strategies used, and learning and adapting to improve strategies. Termed a structured process of "learning by doing" this approach begins with an effort to turn interdisciplinary experience and scientific information into models used to predict the effects of alternate policies (Walters, 1997). Generally applied to complex situations, the adaptive management approach may be applicable to the Blue River Basin study area considering the multiple municipal, county, and state jurisdictions included in the basin and the complexities in attaining the associated economic, environmental, and social objectives needed to be addressed in the management of this system.

The adaptive management decision-making approach began in the 1970s (Holling, 1978) and differs from traditional approaches in that it addresses uncertainty by using management as a tool to gain knowledge on which decisions are based (Johnson, 1999). Monitoring is a critical part of any adaptive management application to provide knowledge and define data gaps. Learning from existing knowledge and observed or modeled responses is a critical component of the process to respond to changing conditions and improve management

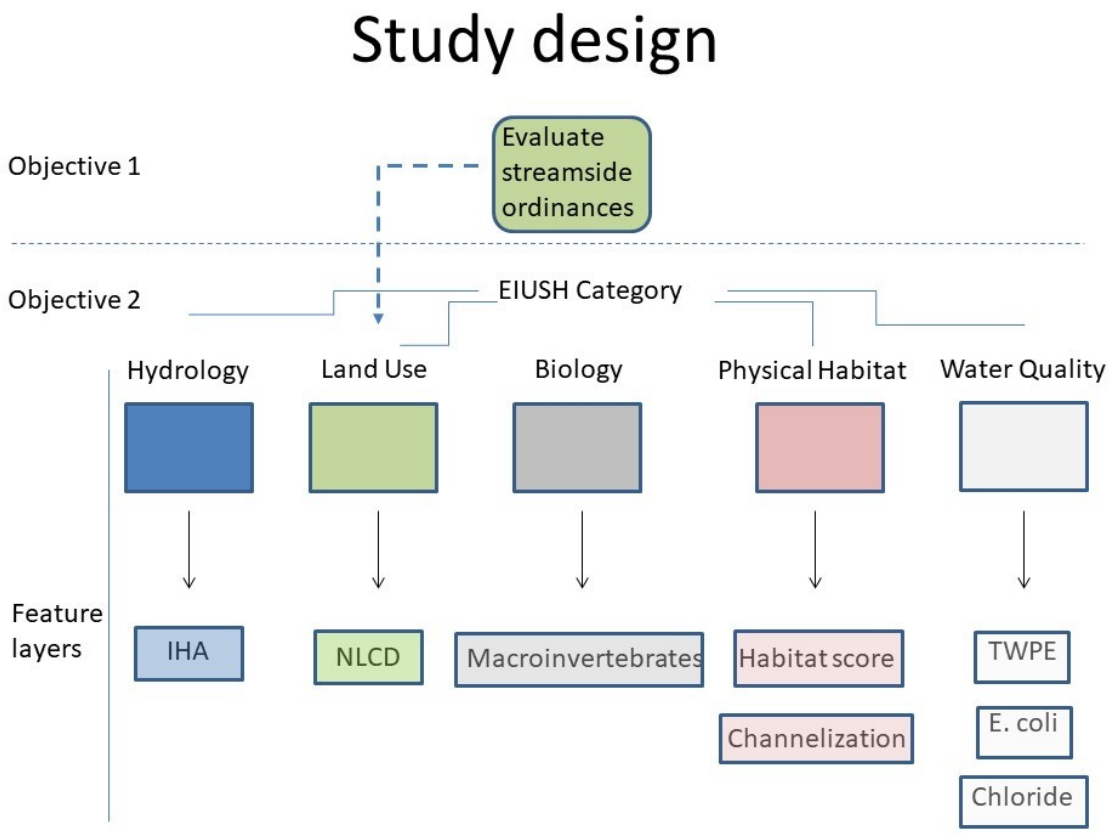
success. This approach provides a unique tool for management of difficult multi-layered problems by bringing diverging interests together to design an approach to the problem with each group having a vested interest and incorporating knowledge to the problem in a circular fashion (Johnson, 1999).

Like adaptive management, scenario planning is another method that can be used for making management decisions about future outcomes of complex ecological systems (Petersen, Cumming, & Carpenter, 2002). Both methods consider many alternative future models rather than focus on a single possible outcome. Scenario planning is most useful when there is a high degree of uncertainty about the managed system and manipulations are difficult. In such cases, a range of imaginative scenarios are considered that are not limited to the current trends or a few limited variables.

CHAPTER 3

METHODS

To address the objectives of this study, a Geographic Information Systems (GIS)-based analysis approach was used in the development of two products for the defined study area. The first was a small-scale assessment focused on the riparian vegetation and LULC change associated with the establishment of streamside ordinances. The second was a holistic, or broad-scale, ecological assessment of the Blue River study area. The ecological assessment utilized a GIS modeling tool to determine and quantify the spatial and temporal ecological integrity of the Blue River—an Ecological Index of Urban Stream Health (EIUSH). Figure 3.1 illustrates the components of the study design including the small-scale (riparian vegetation and streamside ordinance) and large-scale (EIUSH) assessments. The methods used in the determination of these objectives- are described in the following sections.



IHA, Indicators of Hydrologic Alteration; NLCD, National Land Cover Database; TWPE, Total weighted pounds equivalent; E. coli, *Escherichia coli*

Figure 3.1. Study design overview.

Effects of Streamside Ordinances on Riparian Vegetation

Defined Riparian Area

For the purpose of this study, the riparian area limits were defined by multiple digital data sources reflecting the hydrologic interaction of the river and its floodplain. Determination of the area limits was based on a functional approach to riparian area extents, similar to the functional approach presented in Holmes and Goebel (2011). The riparian area perimeter included the extent of the estimated 1-percent annual exceedance probability streamflow (100-year recurrence interval flood) (Heimann, Weilert, Kelly, & Studley, 2014), the hydric soils

layer (U.S. Department of Agriculture, 2014), a 33-m river bank offset, a wetlands layer (U.S. Fish Wildlife Service, 2014), and the defined city of Kansas City streamside buffer zones 1 – 3 (City of Kansas City, 2014). The final area boundary of the multiple composite features was defined by the lateral extent of the merged data sets. Using ArcGIS version 10.3 (Environmental Systems Research Institute, 2014), the boundary layer was smoothed and interpolated to remove map feature islands that were smaller than 3-square meters (m^2) in the select areas for which datasets were incomplete. The resulting riparian buffer areas, corresponding to the streamside ordinance requirements, were produced for the study extents and the various administrative areas (Figure 3.2).

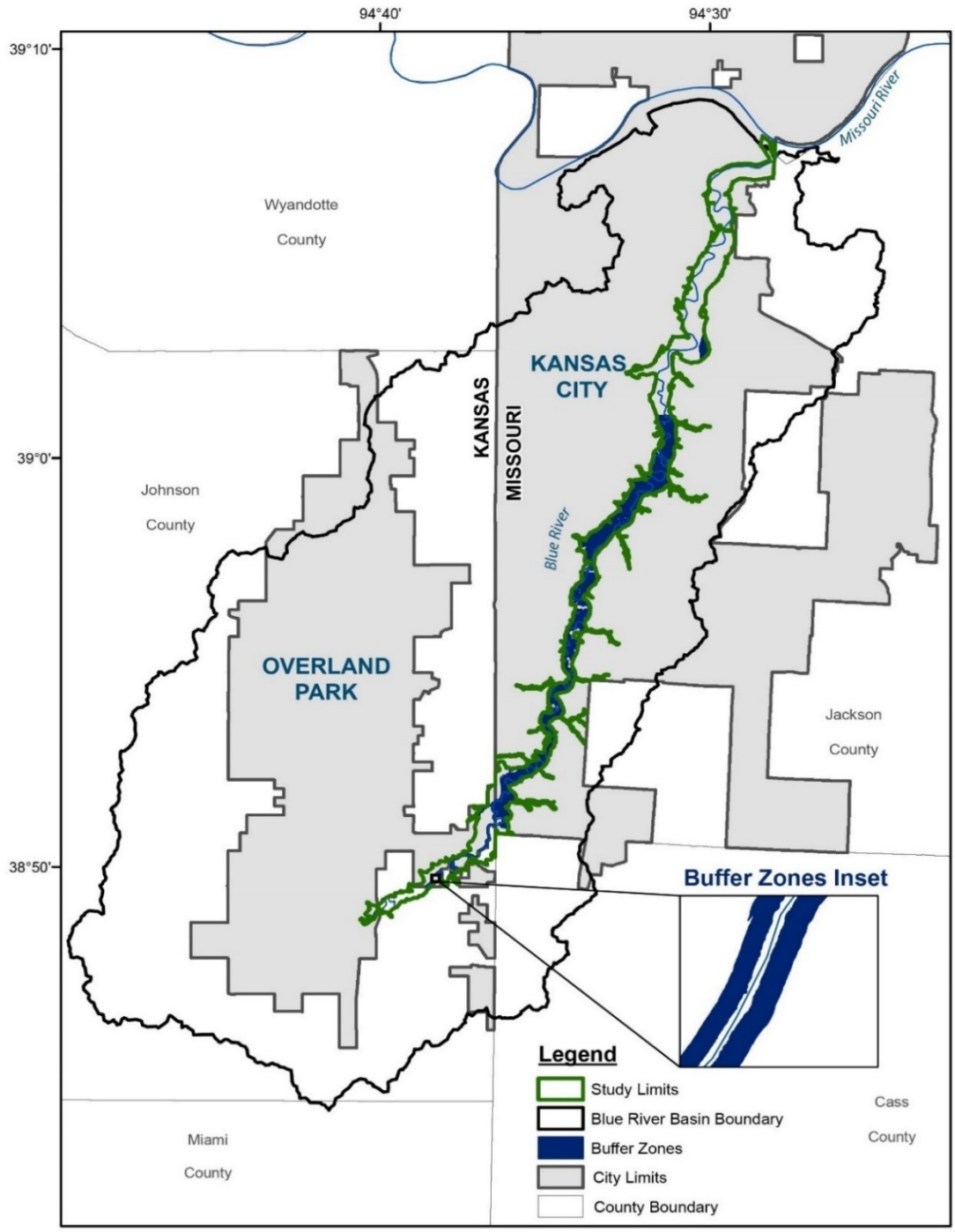


Figure 3.2. Kansas City metropolitan area and Blue River Basin buffer zones.

Streamside Ordinances

The temporal scope of the study was the 20-year period from 1992–2012 and included a pre-ordinance (1992-2009) and post-ordinance (2009-2012) period for Kansas City, Missouri, and a pre-ordinance (1992-2002) and post-ordinance (2003-2012) period for Overland Park, Kansas. The city of Kansas City, Missouri, passed a stream buffer ordinance in 2008 and implemented the ordinance on February 14, 2009 (City of Kansas City, Missouri, 2008). The streamside ordinances were written with the purposes of helping to protect life and property, prevent flooding, preserve water quality, and conserve wildlife habitat (City of Kansas City, 2008; 88-415). The ordinance created three buffer protection zones (Zones 1-3). Zone 1 extends from the edge of the stream to a consistent 25-foot (ft), (7.6-m) lateral buffer. Zone 2 is a middle zone that starts at the edge of the streamside zone (i.e. Zone 1) and extends landward to include the U.S. Federal Emergency Management Agency or city designated 100-year recurrence interval floodplain. Zone 3 starts at the edge of zone 2 and extends 75 ft, (22.9 m) landward. If there was no zone 2 specified, then zone 3 started at the edge of the streamside zone (Zone 1). The ordinance does not limit any flood control activities. The ordinance included a "grandfather clause" that exempted from regulation any previously submitted development plans or plats, as long as they met the requirements set forth in 88-415-02-D.1, 88-415-02-D.2 and 88-415-02-D.3 (KCMO, 88-415). The regulation in 88-415-02-D.1outlines and calls for city commission approval of the final plat to be requested and approved within one year of the landowner being notified by the city through certified correspondence, as long as the plat or plan was approved before January 1, 2003. The regulation in 88-415-02-D.2 outlines that streamside ordinance regulations for the next phases of the plat will not apply as long as city commission approval was granted by February 14, 2009, and that the city

commission approval has been granted by February 14, 2014. The last requirements are in 88-415-02-D.3, which outlines the exclusion for plats and plans approved after January 1, 2003 and before February 14, 2009, stating that these permits will be exempt from the streamside ordinance if the city commission grants final plat approval. The remaining sections on these exemptions go on to state that as long as the next plat is approved within 3 years of the previous plat, the exemptions will continue.

The city of Overland Park, Kansas, passed a streamside ordinance in 2002 and implemented it on October 7, 2002 (City of Overland Park, 2002). Overland Park implemented a one-zone, variable-buffer area based on the acreage of the tributary area of the stream. The maximum buffer zone offset is 120 ft. (36.6 m) for drainage areas of 5,000 acres or greater, which is the category that includes the Blue River in Overland Park. The buffer starts at the top of bank and extends the specified distance on each side of the stream to create a protection zone. The ordinance restricts the building of permanent structures but allows for exceptions when changes made are a result of flood mitigation efforts (City of Overland Park, 2002). The ordinance also permits agricultural uses, recreational uses, limited golf course uses, and permits tree trimming but restricts vegetation destruction (City of Overland Park, 2002).

The Blue River riparian corridor within the unincorporated area of Johnson County, Kansas, was not within a designated streamside ordinance zone throughout the study period. This area provided an additional portion of the Blue River outside of any ordinance zone to use as a control for a non-streamside ordinance condition.

Imagery Processing

The use of the SPOT imagery with a 20-m resolution provided a better solution than Landsat (30-m), due to the increase in spatial resolution. SPOT images acquired in 1992, 2003,

2009, and 2012 (The Centre National D'études Spatiales, 1992, 2003, 2009, 2012) were used in this study to determine LULC change within the riparian zone over the 20-year period (Figures. 3.3, 3.4, 3.5, 3.6). The earliest image (1992) was selected based on availability and the objective to define a pre-ordinance reference condition. Table 3.1 summarizes the characteristics of the SPOT Imagery. The XS and XI spectral modes correspond to 20-m pixel spatial resolution, and the J spectral mode corresponds to 10-m. pixel resolution.

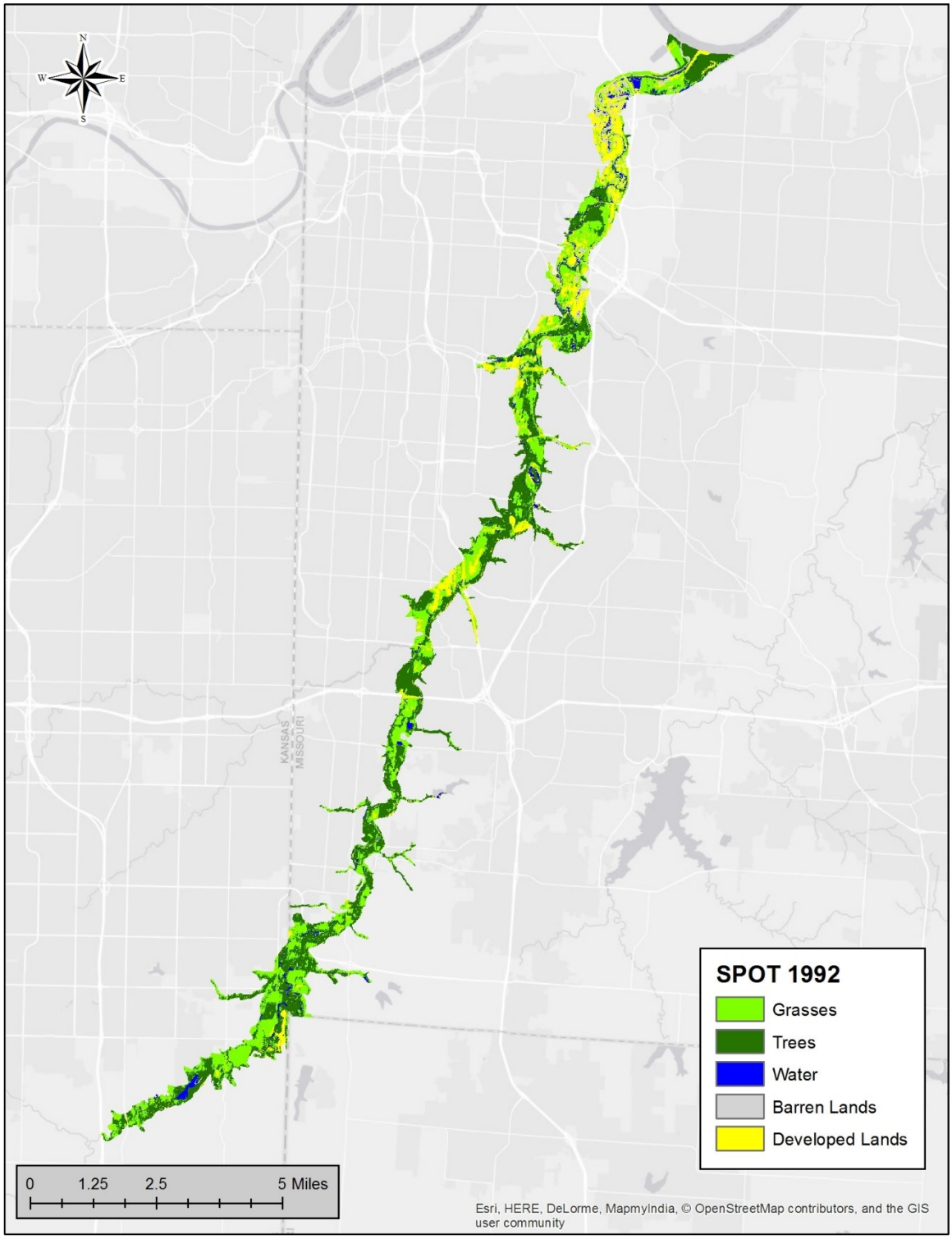


Figure 3.3. SPOT 1992 classified imagery.

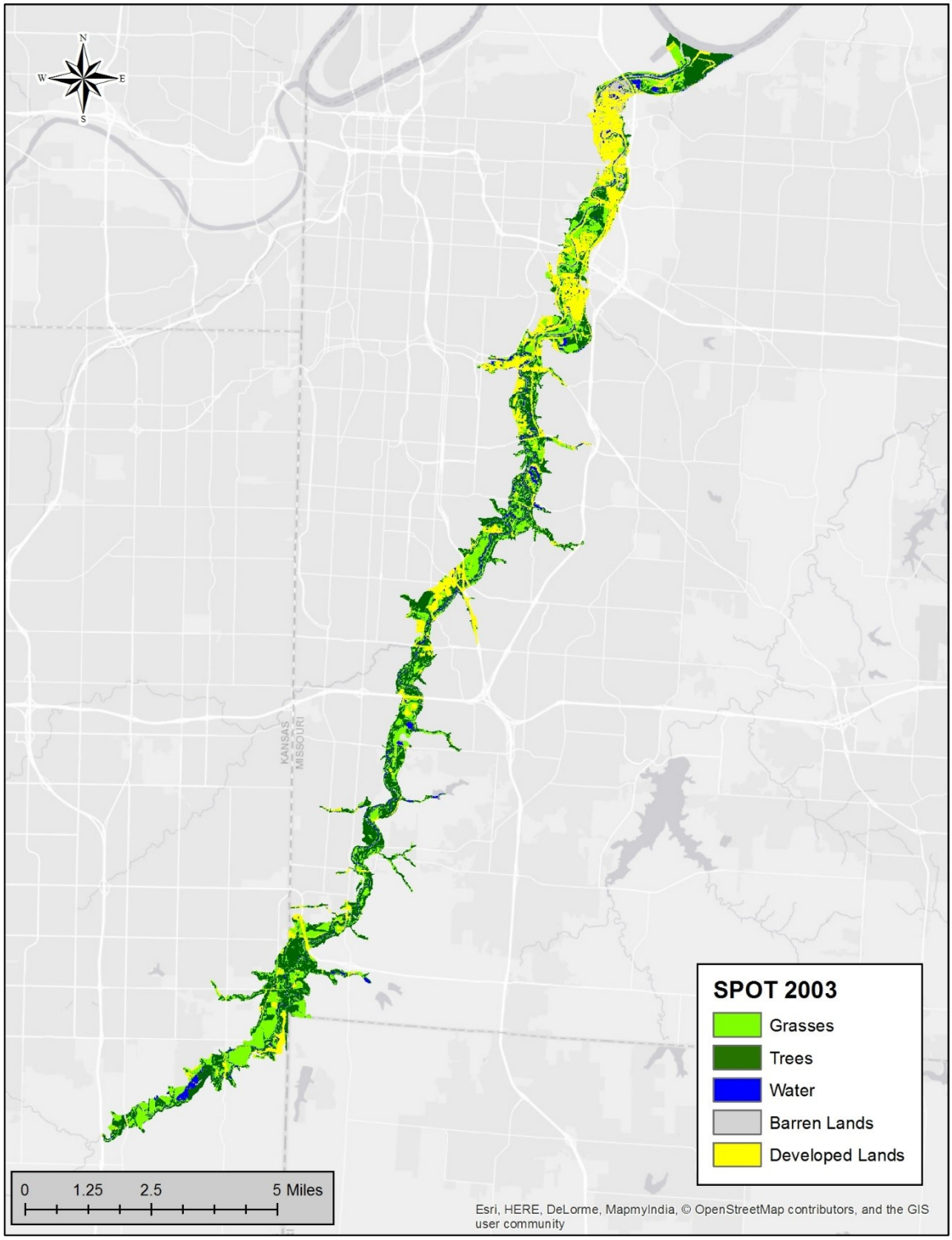


Figure 3.4. SPOT 2003 classified imagery.

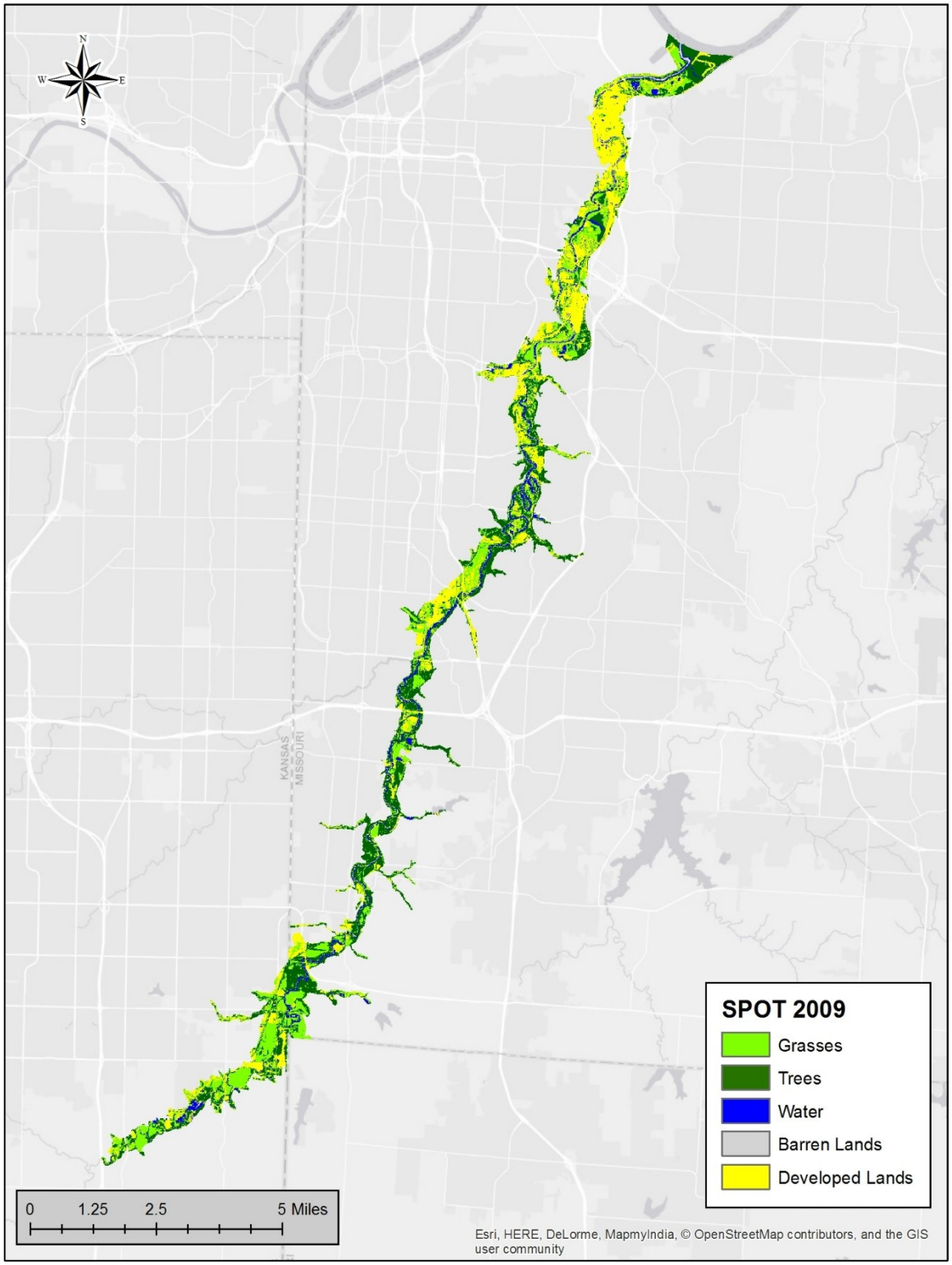


Figure 3.5. SPOT 2009 classified imagery.

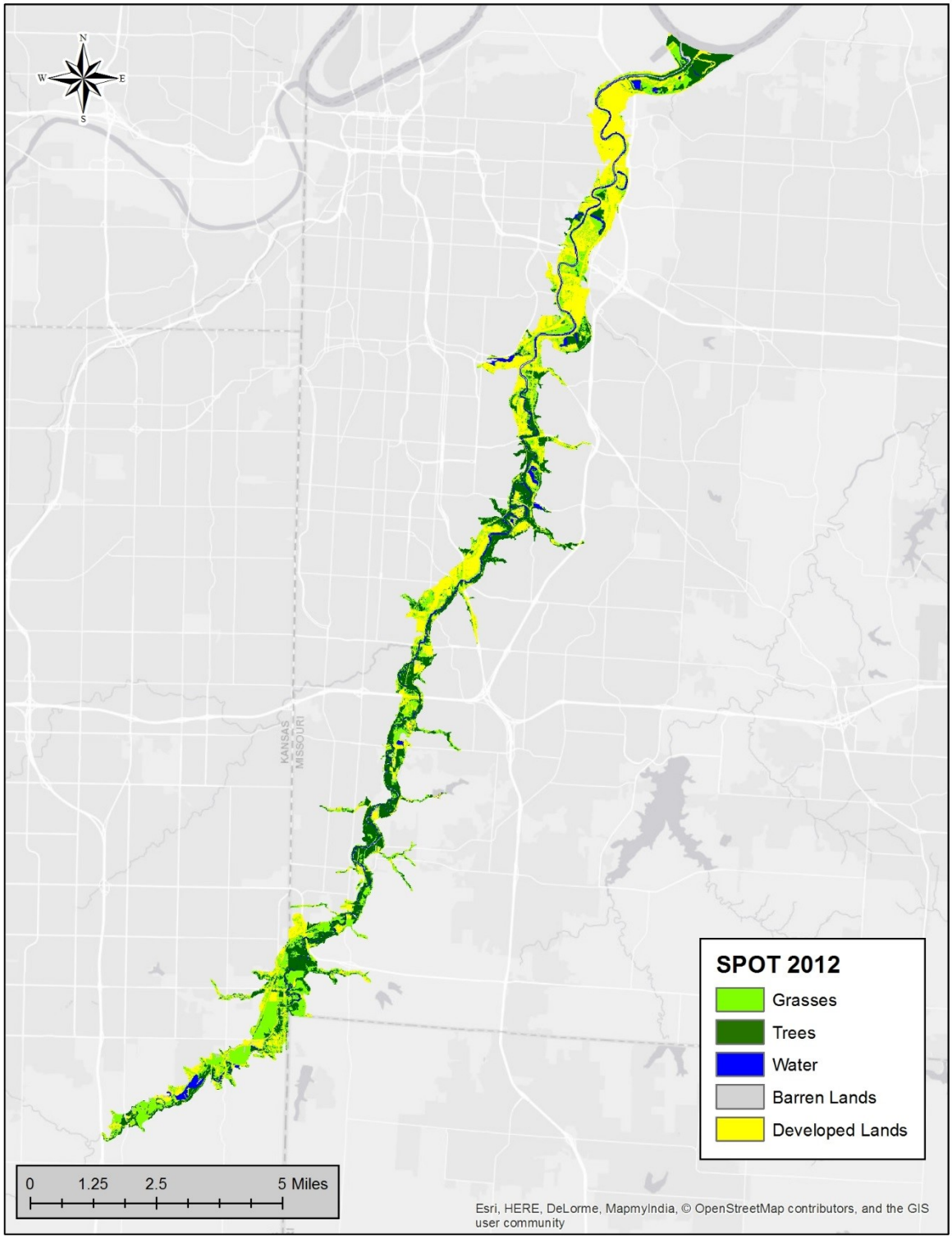


Figure 3.6. SPOT 2012 classified imagery.

Table 3.1

Characteristics of SPOT imagery included in the study.

Sensor	Orbit	Date	Spectral mode
SPOT 2	587-272	29-Jan-92	XS
SPOT 2	587-271	29-Jan-92	XS
SPOT 4	587-271	15-Sep-03	XI
SPOT 5	587-271	23-Apr-09	J
SPOT 5	587-271	06-Feb-12	J

The 1992 imagery was made up of two scenes that were mosaicked together to form one scene. The metropolitan area was covered in one scene in 2003, 2009 and 2012 imagery. ERDAS Imagine 2013 (Intergraph Corporation, 2012) was used to pre-process, classify, and assess the accuracy of the classifications. The 2003 and 2012 SPOT imagery was geo-rectified and the Root Mean Square Error (RMSE) measurement was calculated to ensure that it was within the 0.5 acceptable range of error. The users and producers accuracy also was calculated for each classified imagery. The imagery from 1992 and 2009 were rectified in a previous study (Murambadoro, Xu, & Ji, 2015). The two geo-rectified SPOT images also required the calculation of the RMSE to measure the accuracy of the imagery rectification, and to ensure

the images were rectified within an acceptable level of error. The 2003 georectified SPOT imagery had a root mean square error (RMSE) of 0.36 and the 2012 SPOT imagery resulted in an RMSE of 0.42. The 1992 and 2009 imagery analyses resulted in a RMSE less than 0.5.

The spatial resolution of the 1992 and 2003 imagery was 20 m, and that of the 2009 and 2012 imagery was 10 m, therefore, image resampling was conducted on the 2009 and 2012 imagery to obtain a consistent 20-m resolution for all imagery. Supervised maximum likelihood classification (Tso & Mather, 2001) was conducted on the four images based on a modified USGS land cover classification system (Anderson, Hardy, Roach, & Witmer, 1976). The maximum likelihood classification method assigns equal prior probability of a pixel belonging to a class, and then the pixel was assigned to the class with the highest probability (Jensen, 2005). The images were classified into five land-cover classes, including barren land, developed land, grasses, trees, and water. Barren land was defined as disturbed or bare-earth lands. Developed land included any type of urban development, roads, or pavement. The grasses category also included agricultural lands (pasture, rangeland) or cropland. The trees classification represented a class that predominately contained woody vegetation. The water class included all rivers, lakes, streams, ditches, or manmade reservoirs that were large enough to be represented at the 20-m scale. This classification system was used to simplify LULC in a complex urban landscape while retaining the land cover of primary interest for the study.

Imagery Classification

The classified SPOT images were checked for accuracy by assigning 250 randomly stratified ground-truth points to each classified image and matching these points against corresponding high-resolution imagery acquired in or near the year of the SPOT imagery. The 1992 classification used USGS National Aerial Photography Program (NAPP) aerial

photograph imagery from 1991 (USGS, 1991). The later classifications used National Agriculture Imagery Program (NAIP) images (U.S. Department of Agriculture, Farm Service Agency, various), which have 1-m spatial resolution and were available for ground truthing in 2003, 2009, and 2012.

The User's and Producer's accuracies (Jensen, 2005; Campbell & Wynne, 2011) as well as the overall accuracy for each classified image were assessed (table 3.2). The errors of commission results related to the probability of the pixels on the map accurately placing into the class what was being represented on the ground. The errors of omission or producer's accuracy was calculated on the pixels that were omitted from that class. The overall accuracy of the images was calculated to provide the general agreement of the accuracy of the classification based on the total number of pixels that were correctly classified.

The image classification accuracies were considered satisfactory for this study, which identifies the patterns and trends of related land cover changes. The accuracy assessment was performed on the classified images and ranged from 87.89% to 92.58% overall classification accuracy (Table 3.2). The overall accuracy was 88% for the 1992 imagery, 90% for 2003, 92% for 2009, and 91% for the 2012 imagery. The user's and producer's accuracies were in the 80-90 percentile for the trees and grasses land cover.

Table 3.2

Assessment results of the user's and producer's accuracies of the SPOT imagery.

1992 Accuracy Assessment			2009 Accuracy Assessment		
Class Name	Producer's Accuracy	User's Accuracy	Class Name	Producer's Accuracy	User's Accuracy
Barren Land	71.40%	83.33%	Barren Land	87.50%	82.35%
Grasses	90.91%	87.91%	Grasses	92.16%	92.16%
Trees	89.19%	88.39%	Trees	96.15%	93.46%
Urban Land	84.62%	86.84%	Urban Land	92.00%	94.52%
Water	72.73%	88.89%	Water	70.00%	87.50%
Overall Classification Accuracy	87.89%		Overall Classification Accuracy	92.58%	
2003 Accuracy Assessment			2012 Accuracy Assessment		
Class Name	Producer's Accuracy	User's Accuracy	Class Name	Producer's Accuracy	User's Accuracy
Barren Land	72.73%	88.89%	Barren Land	80.00%	88.89%
Grasses	85.07%	90.48%	Grasses	88.00%	93.62%
Trees	93.00%	90.29%	Trees	94.68%	93.69%
Urban Land	92.19%	93.65%	Urban Land	94.19%	91.11%
Water	100.00%	77.78%	Water	81.25%	86.66%
Overall Classification Accuracy	90.23%		Overall Classification Accuracy	91.80%	

Data Analysis

The riparian boundary file was subdivided by state and municipal boundaries (Mid-America Regional Council, 2010) as well as by ordinance protection areas using ArcGIS version 10.3 (Environmental Systems Research Institute, 2014). The Missouri portion of the study area, entirely within the city of Kansas City, Missouri, was separated into two files. The first file was clipped to the city of Kansas City stream buffer zones and included both pre- and post-ordinance periods, and the second file represented the area that was outside of the stream buffer zones representing a non-ordinance protected area, or control. The city of Kansas City

stream buffer zones file was further divided into three files representing buffer protection zones 1 through 3 and the varying lateral extents of protection. The study area in Kansas was divided into two separate files including one containing the ordinance protected areas located entirely within the city of Overland Park, and the other covering the area outside of Overland Park—the non-ordinance area referred to as un-incorporated Johnson County. The area within Overland Park was further divided into two files—the area with stream ordinance protection including both a pre-ordinance and post-ordinance period, and the non-ordinance protected area.

The city of Overland Park designated the ordinance zone by means of drainage area and a corresponding buffer protection zone. A streams data layer (Mid-America Regional Council, 2009) was used to select the Blue River and Negro Creek Basins within the study boundary area within Overland Park. These two streams were determined to have stream tributary areas greater than or equal to 5,000 acres, corresponding to the maximum ordinance protection of 120 ft. (36.6 m) (City of Overland Park, 2002). The elevation contour lines derived from a Blue River Basin terrain model (Heimann et al., 2014) were used to delineate the high-water mark extents. The high-water marks were buffered by the ordinance buffer width (120 ft, 36.6 m) to create the extents of the protected area. The extents were used to define the buffered protection layer and the remaining unprotected study areas in Overland Park. The resulting seven state-municipal-ordinance defined study area extents then were converted to raster files.

The total and annualized rate of change for each classification category were determined by municipality, pre-post-ordinance periods, and by ordinance zones. The study extent raster files were used to extract the land use-land cover categories for each of the four

classified images. The land cover values, extracted by state-municipal-ordinance boundaries, provided discrete land cover sampling points corresponding to the year of imagery. The land cover classes within the study limits were determined for the pre- and post-ordinance periods defined using the available SPOT images (1992, 2003, 2009, and 2012). The quantitative changes in the land cover classes were divided by the years between image dates to determine land cover values per study boundary type, per year.

Ecological Index of Urban Stream Health

A general assessment of the ecological health of the Blue River was conducted using a GIS-based index—Ecological Index of Urban Stream Health (EIUSH). The development of the EIUSH included five primary ecological categories associated with the health of aquatic systems (Karr, Toth, & Garman, 1983), each consisting of one or more spatial data layers. The ecological categories included biological community, hydrology, land use and land cover, physical habitat, and water quality. The composite index values were determined from a spatial and equal numerical weighting of the values from the five defined categories and their associated layers, as shown in Figure 3.7. The equal weighting across the five categories is a reasonable starting point, providing equal importance to each of the primary environmental factors in the stream system, and could be modified in the future based on research-based justification.

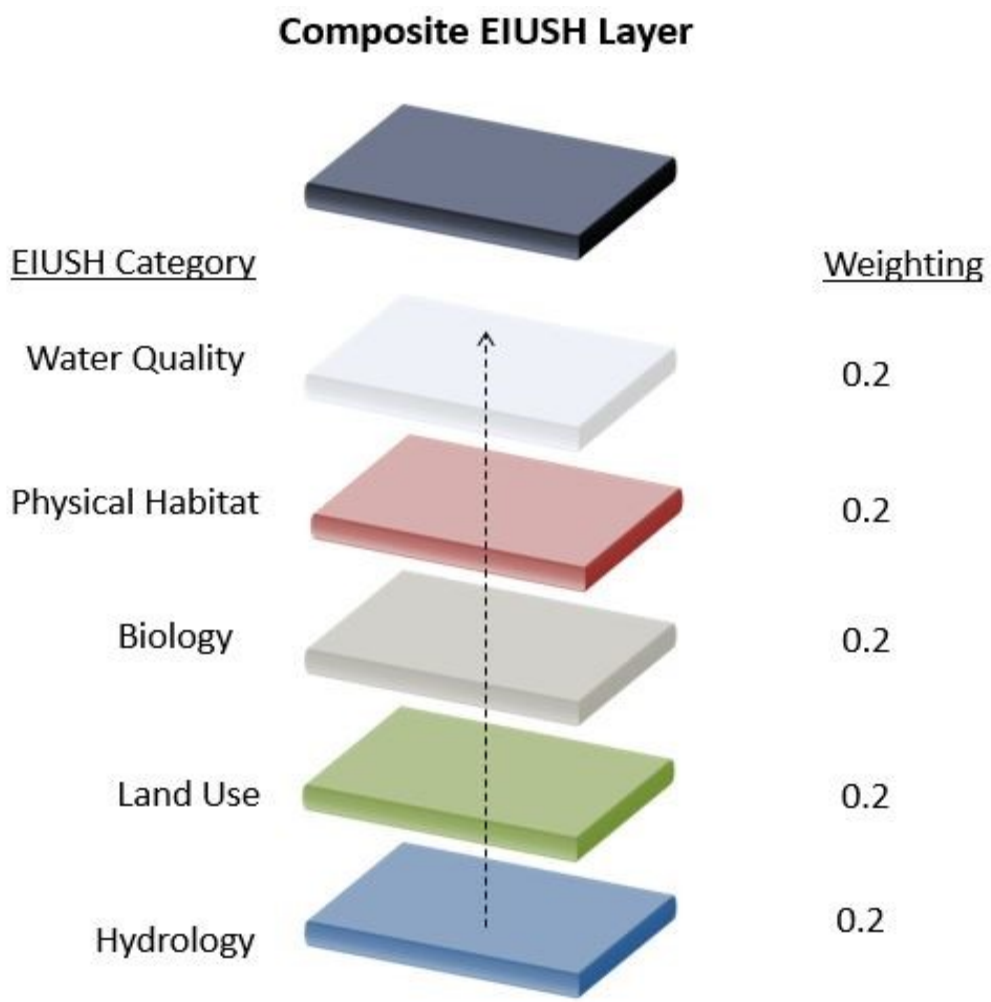


Figure 3.7. EIUSH weighted overview.

Sources of Data

Data from five major environmental categories, and eight separate data layers, were included in the development of EIUSH. Tables 3.3 and 3.4 provide the datasets and sources that were incorporated into two temporally different EIUSH models of the Blue River study area. The EIUSH version I included data from 1999 through 2013, and version II demonstrates a temporal progression of EIUSH with data from the same data layers used in version I updated through 2016, where available.

Table 3.3

Table of data sets used in the development of the Ecological Index of Urban Stream Health (EIUSH) version I.

Dataset	Datasource	Time Frame	Model Parameters	Derived Data	Derived Data
Macroinvertebrate metrics samples	Wilkison et al, 2009; Rasmussen et al, 2008	Kansas 2007	Biological	Raster Layer of Macroinverts	EIUSH Model
Macroinvertebrate metrics samples	Wilkison et al, 2009; Rasmussen et al, 2008	Missouri 2006	Biological	Raster Layer of Macroinverts	
Daily Water Flows	USGS, 2017b	1940-2006	Hydrologic	Raster Layer of Hydrologic Alteration	
Land use and Land Cover of the Study	Fry et al, 2011	2006	Land Use Land Cover	Raster Layer of Land Use Land Cover	
Physical Habitat Assessment	Wilkison et al, 2009	2006	Physical Habitat	Raster Layer of Physical Habitat	
Channelization	Developed for this study, see methods	2012	Physical Habitat	Raster Layer of Channelization Scores	
Toxic Weighted Pounds Equivalent	U.S. EPA, 2017b	2007-2009	Water Quality	Raster Layer of TWPE	
Chloride	USGS, 2017b, and City of Kansas City, Water Services Dept., written commun., 2017	1999 - 2013	Water Quality	Raster Layer of Chloride	
E Coli	USGS, 2017b, and City of Kansas City, Water Services Dept., written commun., 2017	1999 - 2013	Water Quality	Raster Layer of E Coli Levels	

Table 3.4

Table of data sets used in the development of the Ecological Index of Urban Stream Health (EIUSH) version II.

Dataset	Datasource	Time Frame	Model Parameters	Derived Data	Derived Data
Macroinvertebrate metrics samples	Wilkison et al, 2009; Rasmussen et al, 2008	Kansas 2007	Biological	Raster Layer of Macroinverts	EIUSH II Model
Macroinvertebrate metrics samples	Wilkison et al, 2009; Rasmussen et al, 2008	Missouri 2006	Biological	Raster Layer of Macroinverts	
Daily Water Flows	USGS, 2017b	1940 - 2016	Hydrologic	Raster Layer of Hydrologic Alteration	
Land use and Land Cover of the Study	Homer et al, 2015	2011	Land Use Land Cover	Raster Layer of Land Use Land Cover	
Physical Habitat Assessment	Wilkison et al, 2009	2006	Physical Habitat	Raster Layer of Physical Habitat	
Channelization	Developed for this study, see methods	2012	Physical Habitat	Raster Layer of Channelization Scores	
Toxic Weighted Pounds Equivalent	U.S. EPA, 2017b	2014-2016	Water Quality	Raster Layer of TWPE	
Chloride	USGS, 2017b, and City of Kansas City, Water Services Dept., written commun., 2017	1999 - 2013	Water Quality	Raster Layer of Chloride	
E Coli	USGS, 2017b, and City of Kansas City, Water Services Dept., written commun., 2017	1999 - 2013	Water Quality	Raster Layer of E Coli Levels	

Each of the five major environmental categories in EIUSH were represented by data layers selected based on the spatial and temporal availability of data, and the applicability to represent that environmental component in an urban environment. A description of specific datasets within each category is provided in the following sections.

Spatial Depiction of Data Layers

With the exception of the NLCD layer, the digital data layers within EIUSH are depicted as polygons, with the downstream extent of the polygon determined by the sampling location, and the upstream extent of the polygon corresponded to the next upstream sampling location. The resolution of the NLCD layer is continuous and determined by the developed 30-m resolution raster layer. The resolution of the remaining seven data layers is determined by

the number of sampling/analysis points associated with the data layer. The lateral extent of all layers within EIUSH is determined by the riparian area boundary as described in the previous "Defined Riparian Area" section of the Methods. The point samples included in the index data layers reflect, local and basin contributions, and are measured within the active river channel, and attributed to that location. The representation of the data layer samples to the defined riparian area extents allows for a lateral dimension in the spatial expression of the point samples.

Biological Community

The biological community category of the EIUSH was represented by a macroinvertebrate features layer. The data used in this layer were collected by the U.S. Geological Survey in 2006 in Missouri (Wilkison et al., 2009) and 2007 in Kansas (Rassmussen et al., 2008). The macroinvertebrate index values were assigned to reach polygons defined by the nine sample locations (Figure 3.8). This was the most current and complete macroinvertebrate dataset for the Blue River. This macroinvertebrate layer was compared to developed aquatic life use status (ALUS) standards in each state (Sarver, Harlan, Rabeni, & Sowa, 2002; Kansas Department of Health and Environment, 2000). The ALUS was based on composite macroinvertebrate metrics and included three life support categories of: "fully supporting", "partially supporting", and "non-supporting" of aquatic life. These categories were developed from a stream condition index score (SCI) that used total taxa richness, the taxa diversity as measured in the Shannon Diversity Index (SDI) and the Biotic Index (Wilkison et al., 2009). The life support categories, which were common to results from both states, were assigned numerical index values in order to represent the results between states in a consistent manner. The category of "fully supporting" of aquatic communities was

assigned an index scale value of 100, the middle category of “partially supporting” of aquatic communities was assigned an index scale value of 50, and the “non-supporting” category of aquatic communities was assigned an index scale value of 10 (Figure 3.9). Each raster cell value was then multiplied by a 0.2 category weight and was used to populate the macroinvertebrate raster layer.

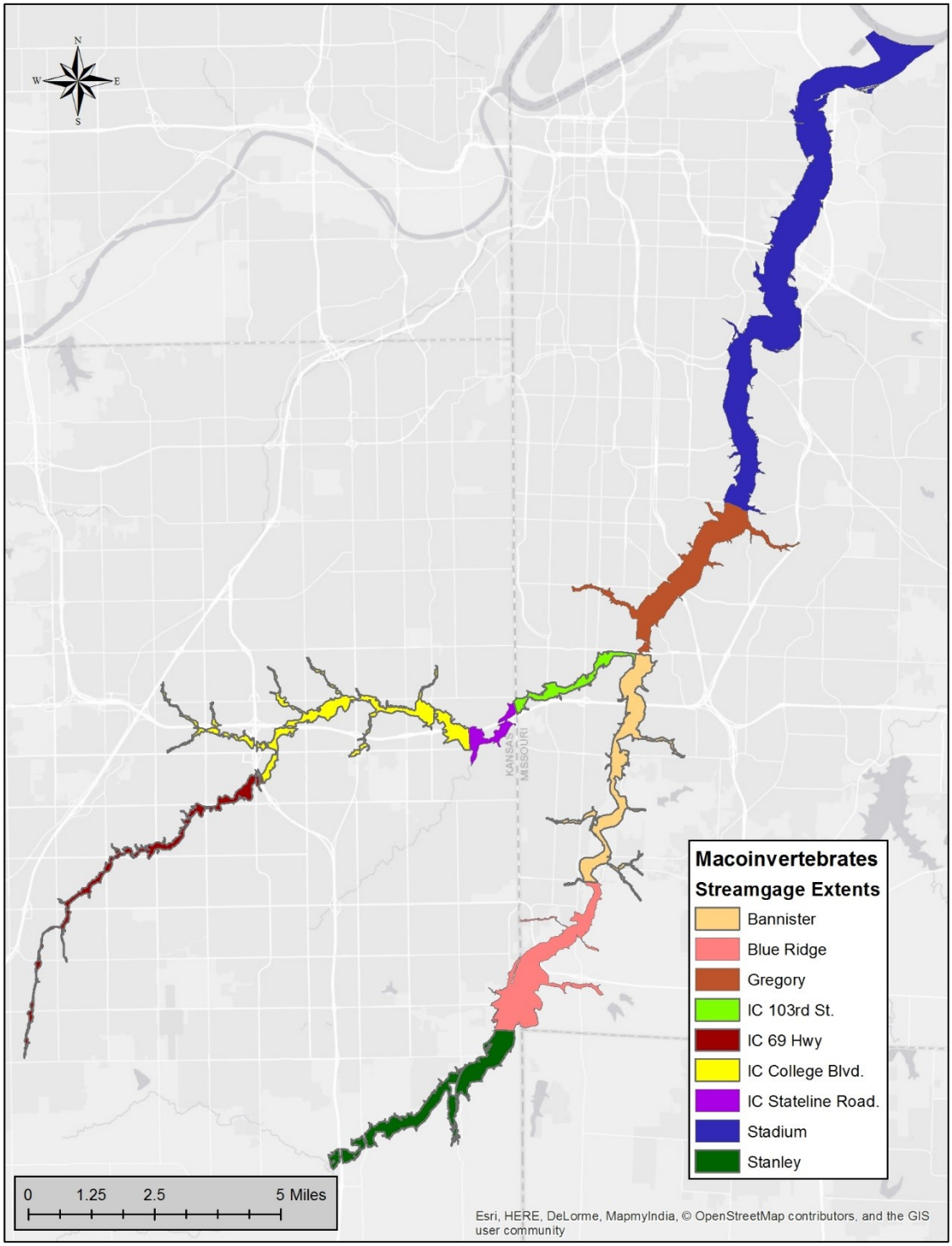


Figure 3.8. Biological (macroinvertebrate) reach extents used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

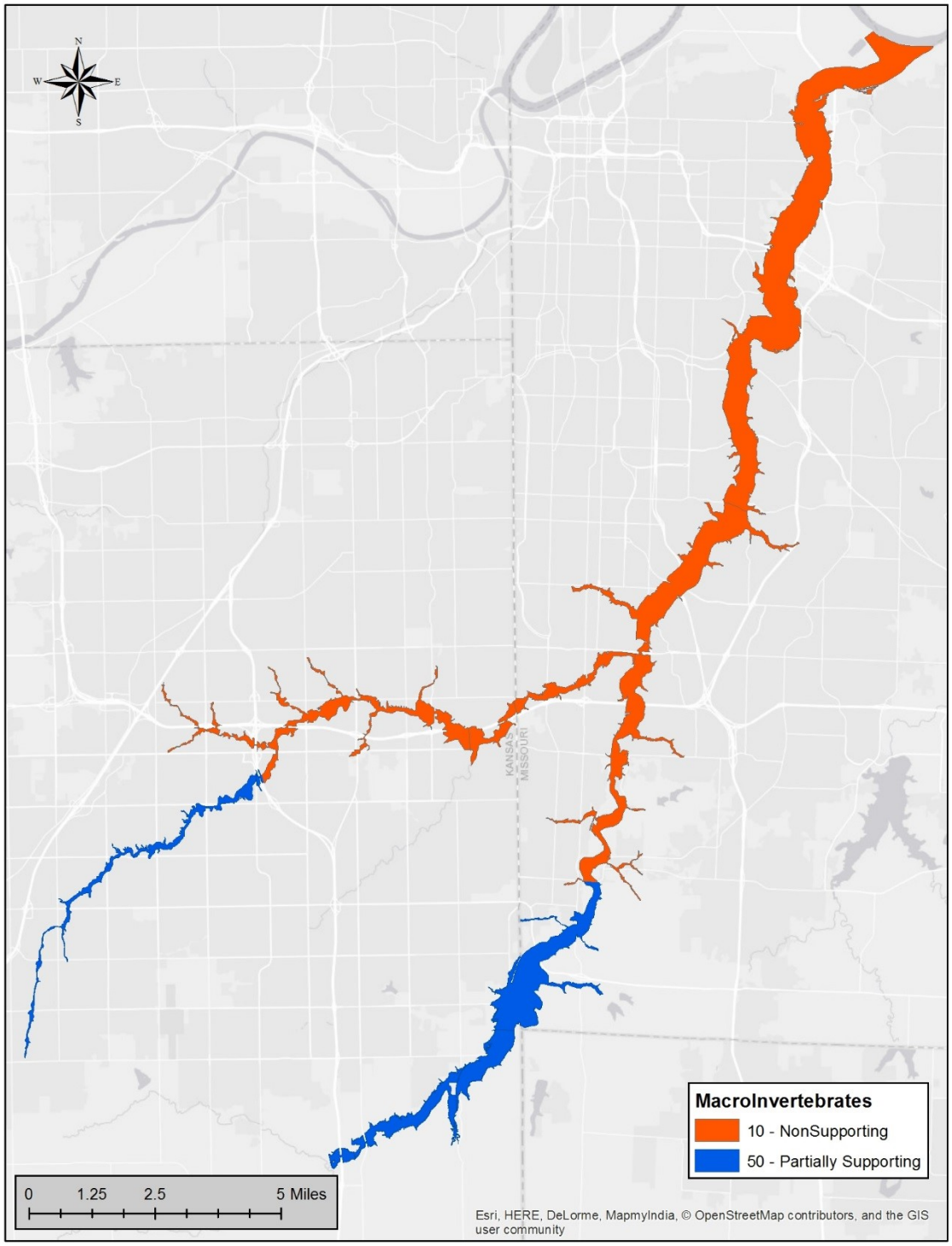


Figure 3.9. Biological (macroinvertebrate) data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

Hydrologic Component

The hydrological component of EIUSH consisted of one feature layer determined from the hydrologic analyses of daily streamflow data at three long-term U.S. Geological Survey streamgages within the Blue River Basin. The daily streamflow data were analyzed using the software Indicators of Hydrologic Alteration version 7.1 (Nature Conservancy, 2017). The streamgages included the Blue River near Stanley, Kansas (USGS station no. 06893080; record period 1974 - 2017); Indian Creek at Overland Park, Kansas, (station no. 06893300; record period 1963 - 2017); and the Blue River at Kansas City, Missouri, (USGS station no. 06893500; record period 1939-2017; USGS, 2017b). Temporal trends in 33 hydrologic parameters and 34 environmental flow components (67 total parameters) (Nature Conservancy, 2009) were assessed within IHA for temporal changes using a least-squares fit regression and the corresponding *p*-value. The 67 hydrologic parameters included median monthly streamflows, 1-, 3-, 7-, 30-, 90-day minimum and maximum streamflows, monthly low flows, and flow characteristics with high flow, small flood, and large flood streamflow categories. The EIUSH version I included the hydrologic alterations based on streamflow records though 2006 (Table 3.5, Figure 3.10) and the EIUSH version II included hydrologic alterations based on streamflow records through 2016 (Table 3.5, Figure 3.11). The IHA results were summarized and incorporated into the index as the proportion of ecological hydrologic parameters that showed a statistically significant change (p-value < 0.05) for the analysis period. The summary value incorporated into the index was determined by the formula

1 - (number of statistically significant hydrologic parameters / 67) *100.

The value presented in the index, therefore, was the percent of the 67 hydrologic parameters at a station not experiencing a statistically significant hydrologic alteration. The

layer values were then multiplied by a 0.2 category weight to account for the hydrological component.

Table 3.5

Degree of hydrologic alteration as determined by the Indicators of Hydrologic Alteration software (Nature Conservancy, 2017) at selected U.S. Geological Survey streamgaging stations along the Blue River.

	Hydrologic Alteration			
	EIUSH I (through 2006)		EIUSH II (through 2016)	
Stations	Percent of IHA parameters altered	Percent of IHA parameters unaltered used in index	Percent of IHA parameters altered	Percent of IHA parameters unaltered used in index
Blue River at Stanley, KS	4.48	95.5	13.4	86.6
Indian Creek at Overland Park, KS	61.2	38.8	64.2	35.8
Blue River at Kansas City, MO	38.8	61.2	46.3	53.7

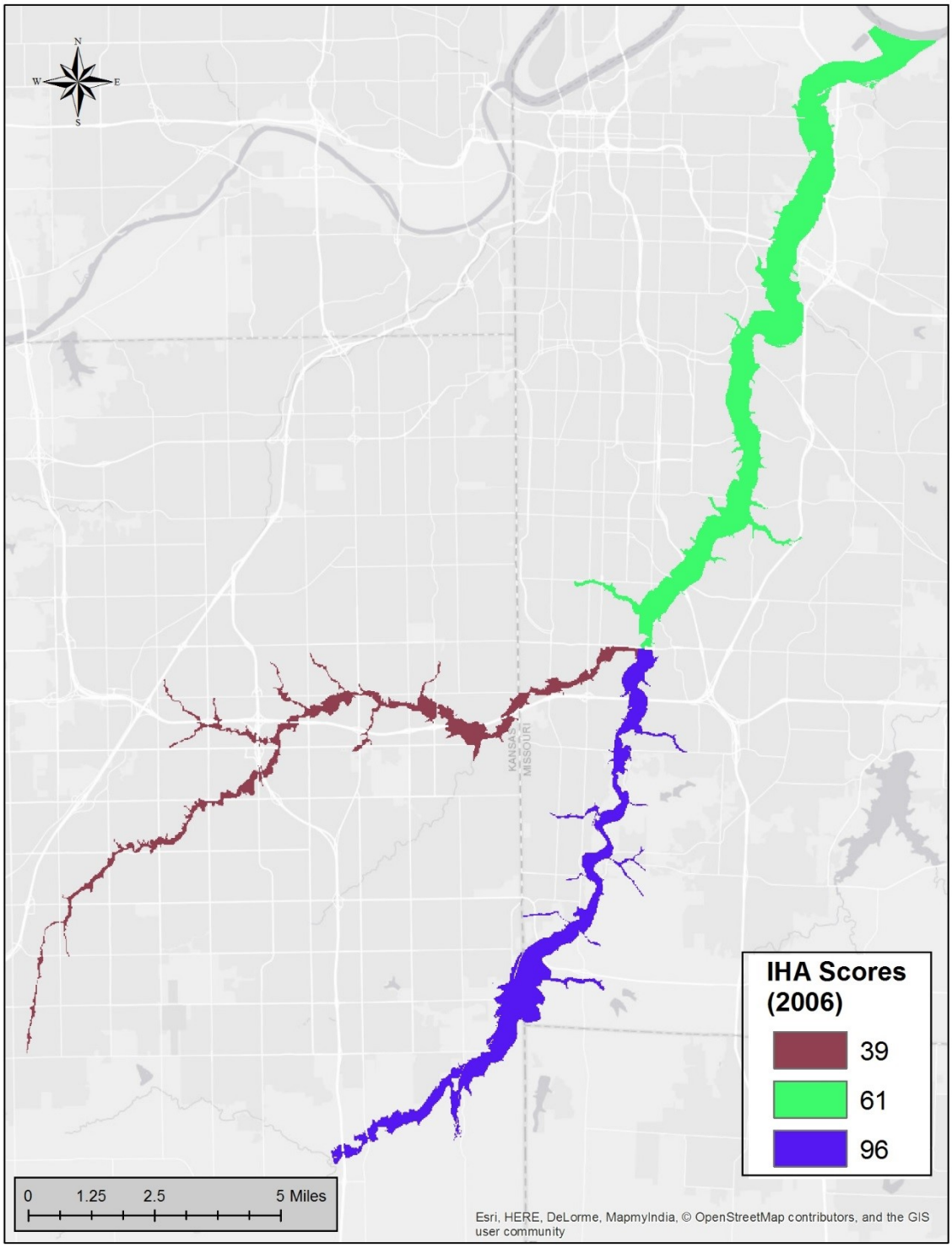


Figure 3.10. Indicators of Hydrologic Alteration (Hydrology) used in the development of the Ecological Index of Urban Stream Health version I, Kansas City, Missouri, and vicinity.

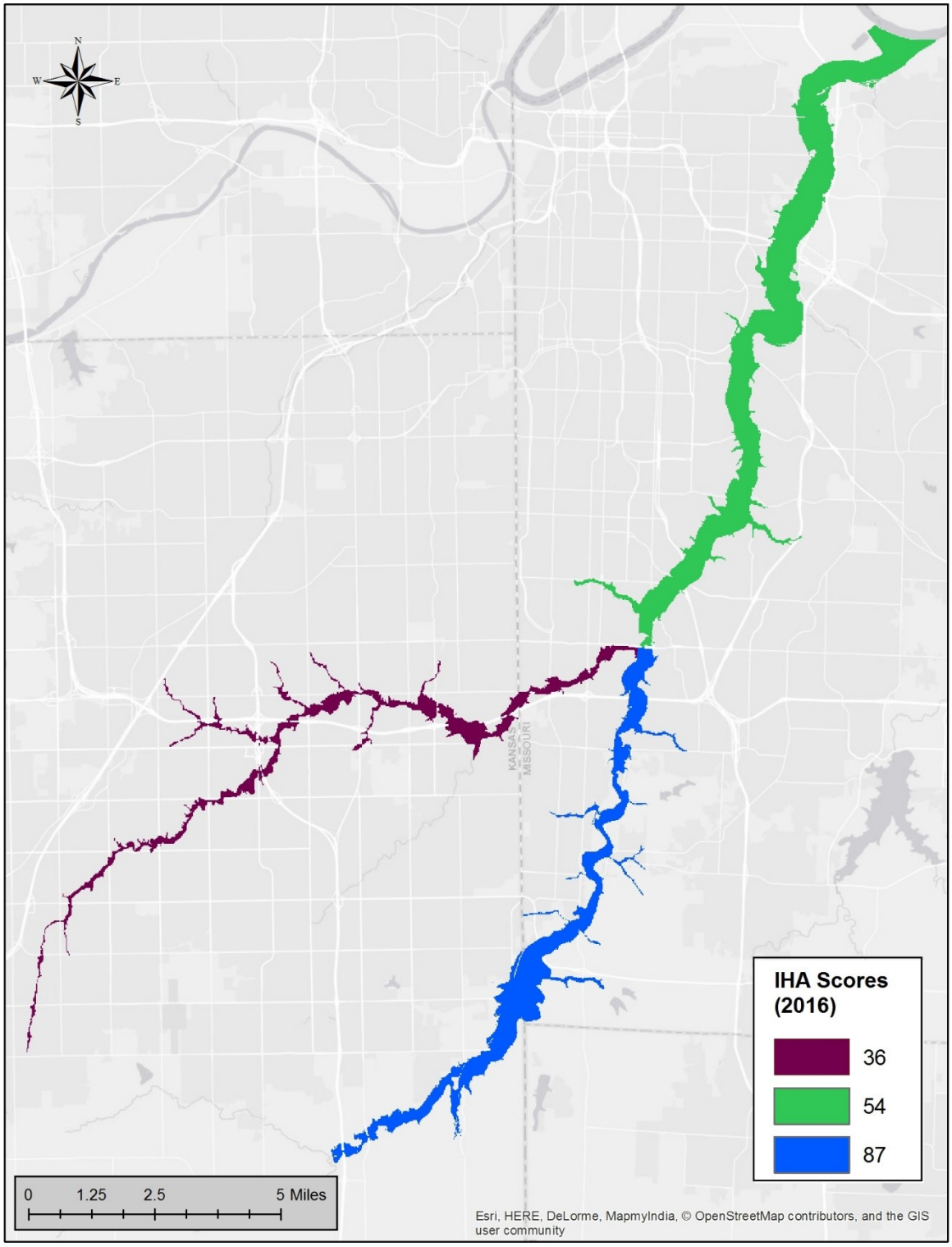


Figure 3.11. IHA data layer used in the development of the Ecological Index of Urban Stream Health version II, Kansas City, Missouri, and vicinity.

Land Use and Land Cover

The LULC component used in EIUSH was derived from the 30-meter (m) resolution National Land Cover Dataset (NLCD). The 2006 NLCD was used in EIUSH I (Fry et al., 2011) (Figure 3.12). This land-use land-cover component was weighted at 0.20 in the index model with each class being scaled from 0 to 100 (Table 3.4). The land use and land cover dataset was updated to the 2011 National Land Cover Dataset (Homer et al., 2015) for the EIUSH II (Figure 3.13). The 16 land use-land cover classes stayed the same for the two different years of imagery.

Table 3.6

Land use and Land Cover layer categories and model weighting used in Ecological Index of Urban Stream Health version I and II.

Model Category	Classes	Class ID	Scale Value	Weight
Land use/land cover(LULC) (source:NLCD 2006/2011)	Open Water	11	80	0.2
	Developed Open Space	21	20	
	Developed Low Intensity	22	20	
	Developed Medium Intensity	23	10	
	Developed High Intensity	24	0	
	Barren Land	31	40	
	Deciduous Forest	41	70	
	Evergreen Forest	42	70	
	Mixed Forest	43	70	
	Shrub/Scrub	52	60	
	Grassland/Herbaceous	71	80	
	Pasture/Hay	81	40	
	Cultivated Crops	82	50	
	Woody Wetlands	90	100	
	Emergent Herbaceous Wetlands	95	100	

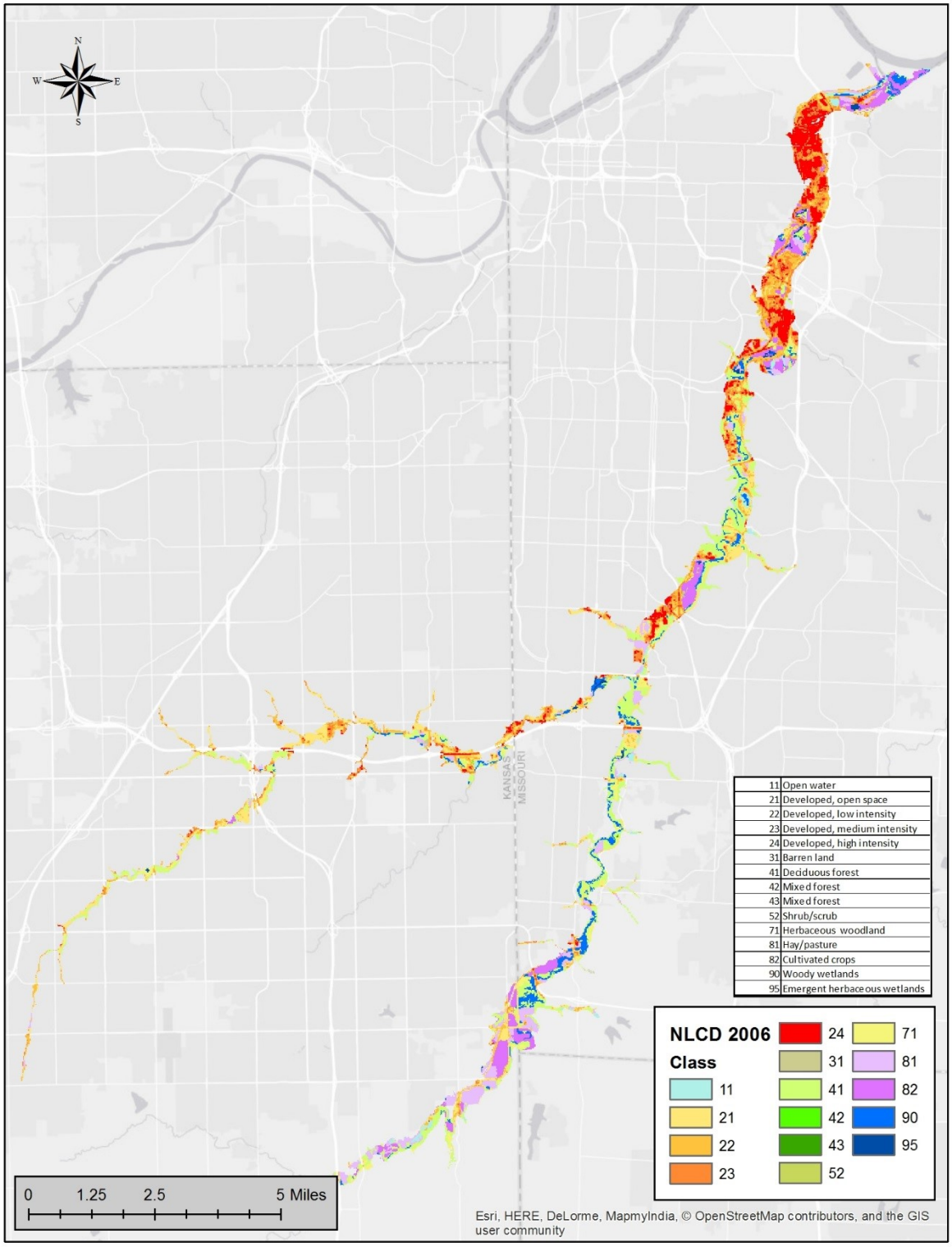


Figure 3.12. National Land Cover Dataset (2006) layer used in the development of the Ecological Index of Urban Stream Health version I, Kansas City, Missouri, and vicinity.

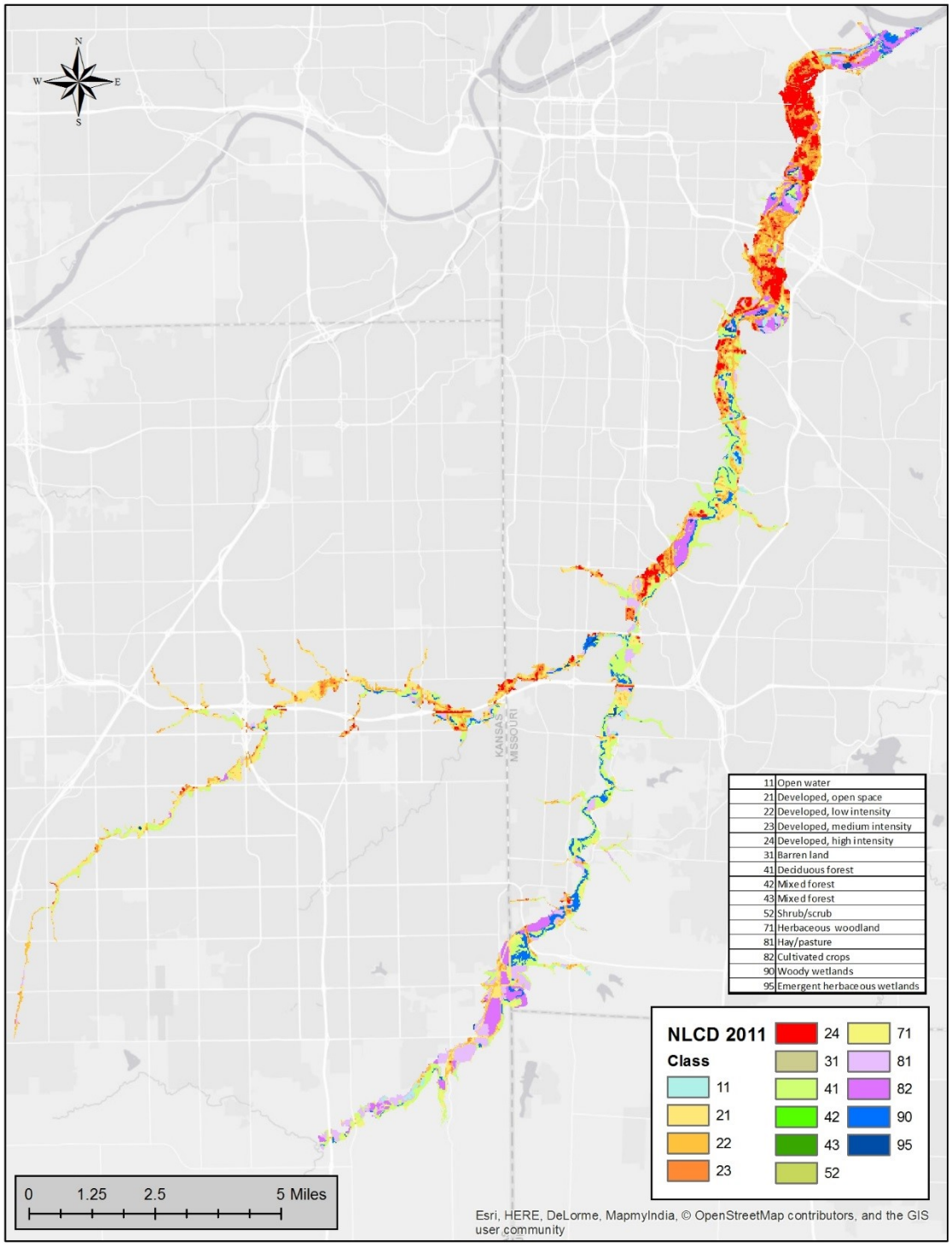


Figure 3.13. National Land Cover Dataset (2011) layer used in the development of the Ecological Index of Urban Stream Health version II, Kansas City, Missouri, and vicinity.

Physical Habitat Component

The physical habitat component of EIUSH I and EIUSH II consisted of two feature layers. The first layer was a physical habitat layer that was based on 2006 Blue River Basin habitat data collected by Wilkison et al (2009) at 11 sites. Physical habitat data were collected as part of a rapid assessment procedure described by Sarver (2003) and included reach scale bank stability, stream sinuosity/channelization, riffle quality, riparian vegetation, impervious surface drainage upstream and buffer length incorporated into a habitat score. The sample locations corresponded to U.S. Geological Survey stations along the Blue River in Missouri and Kansas. The results were incorporated into the index as a percent of the total possible physical habitat score (Figure. 3.14). The habitat scores were designed to measure the local reach habitat rather than the overall land use upstream from evaluation sites. The second physical habitat layer in EIUSH was a channelization condition layer that depicted larger scale areas of channelization and channel modification within the Blue River Basin. The channelized layer was weighted based on four scores indicating varying degrees of channel alteration including substantial changes in channel sinuosity and substrate assessed using digitized stream channel center line from 1956 and 2011 (USGS, 2016) and 2012 NAIP aerial imagery (U.S. Department of Agriculture, Farm Service Agency, various dates). Substantial differences between the 1956 and 2011 channel centerlines, along with obvious artificial channel substrate (uniform boulders, bank armoring) derived from the 2012 imagery, were used to define stream reaches of similar channel conditions. Channel reaches along the Blue River and Indian Creek then were assigned an index score of 20, 40, 60, or 80 (Figure 3.15). A score of 0 indicated a complete alteration of the channel, for example, an extended concrete box culvert or channel.

A score of 20 was assigned to an area with substantial (>50%) substrate changes and channelization present. A score of 40 was assigned to areas of substantial substrate alteration with minimal or no channelization (this would include grade control structures and bank armoring) or substantial channelization with minimal substrate changes. The category score of 60 represented areas of moderate (>10 - <50%) substrate changes or channelization, and the highest category score of 80 was assigned to areas with minimal (<10%) substrate change or presence of channelization. A maximum score in the study area was limited to 80 rather than 100 (no alteration of substrate or channel) based on the assumption that no reach within a stream that had undergone substantial channelization for decades would be entirely unaffected.

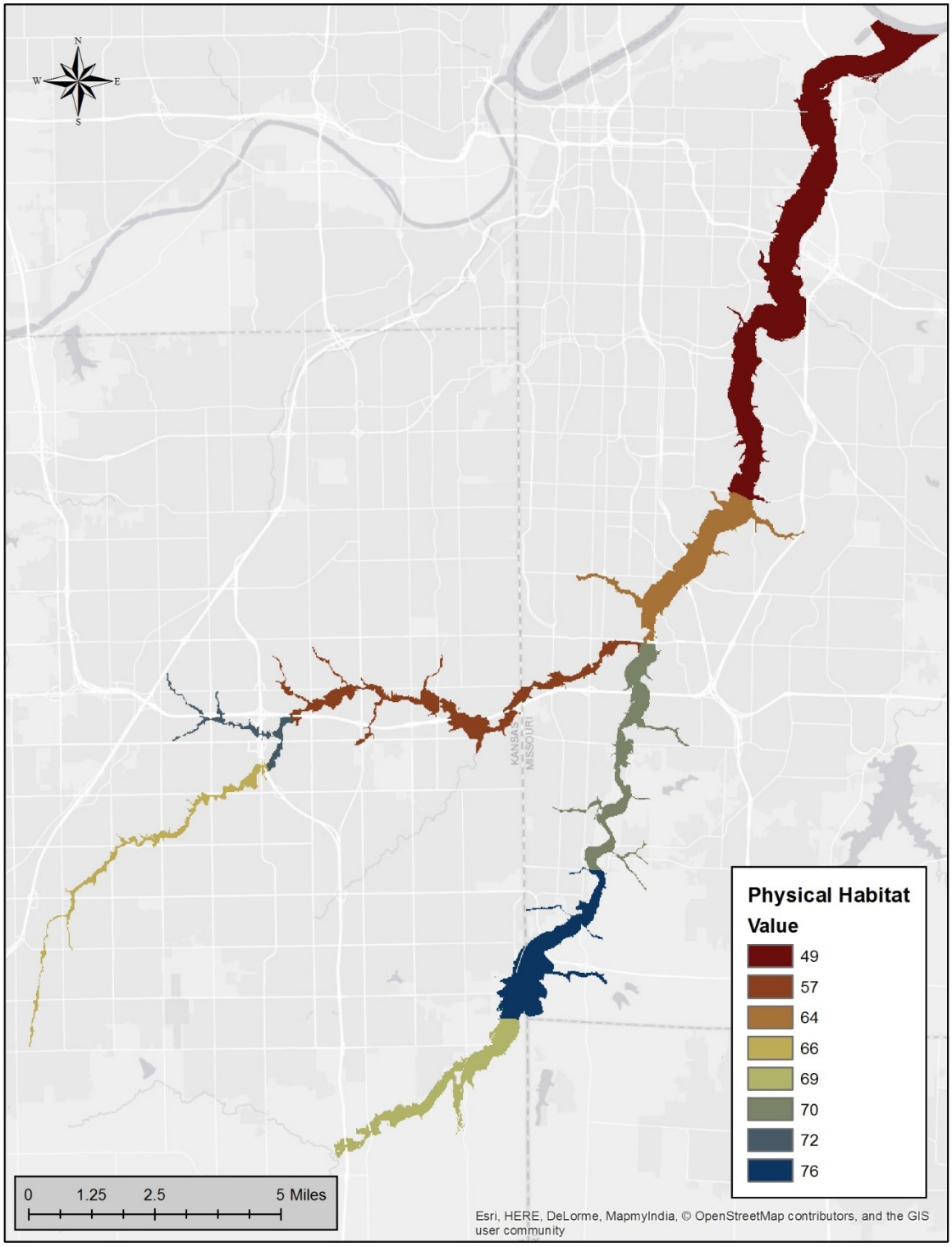


Figure 3.14. Physical habitat data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

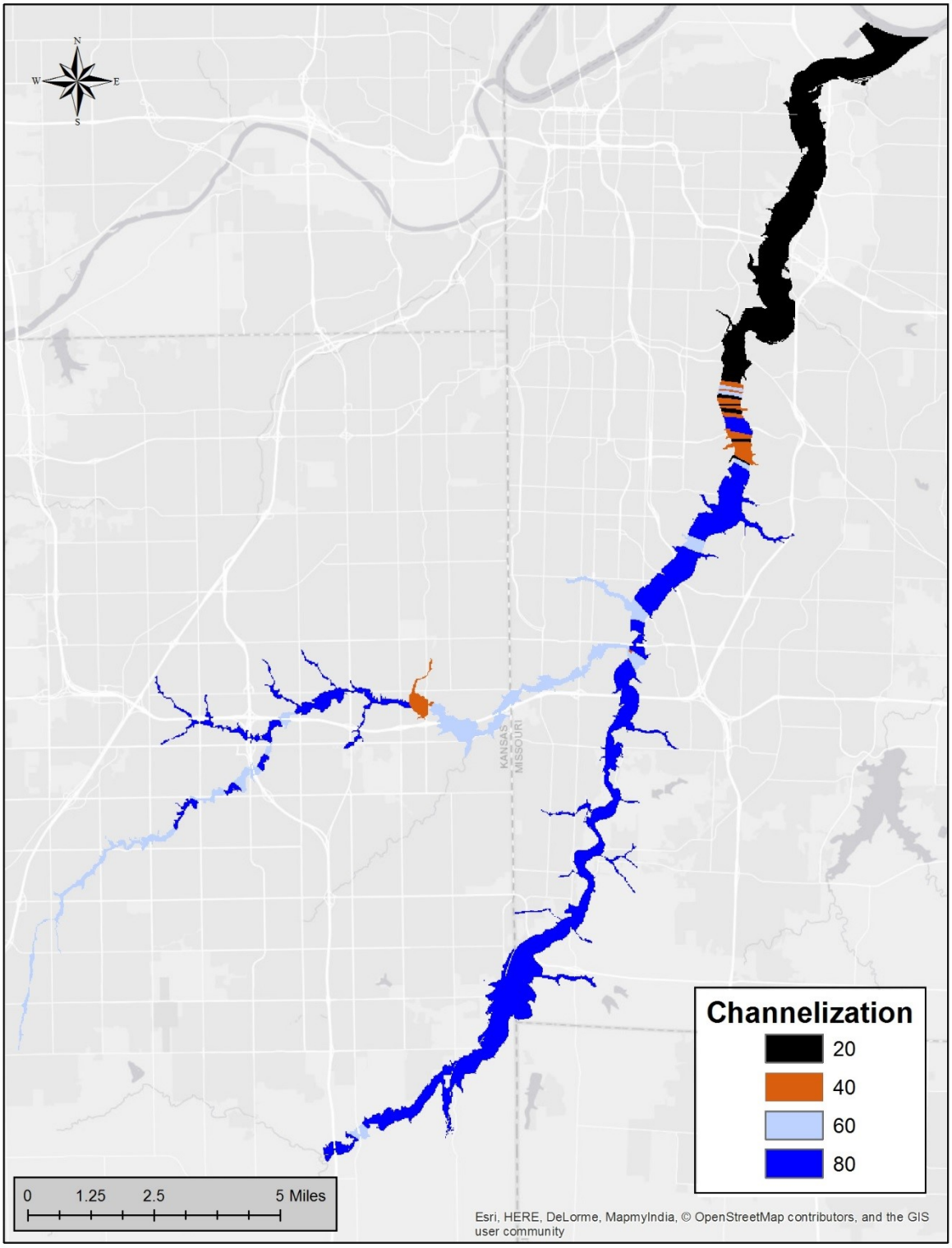


Figure 3.15. Channelization data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

Water Quality Component

The water quality component of the EIUSH was comprised of three feature layers. Contributions from nonpoint sources of contaminants were represented by two common contaminants in urban settings and specifically the Blue River (Wilkison et al., 2009), and included chloride concentrations, and *E. coli* density layers. Point source contaminants were represented by National Pollutant Discharge Elimination System (NPDES) permitted discharges in the study area expressed as the Toxic Weighted Pound Equivalent (TWPE) layer, as these data present the discharge permit sites that have point-source pollutant loadings expressed by their mass and relative toxicity (USEPA, 2017b). The water quality weight of 0.20 was divided equally between point and nonpoint source contaminants. The resulting weight factors were with nonpoint source chloride and *E. coli* layer each receiving an equal 0.05 weight, and the point source of TWPE layer received a 0.10 weight (Table 3.7).

The chloride concentration layer in EIUSH was developed using the mean annual chloride concentrations collected within selected reaches between 1999 and 2013 and the EPA 304(a) recommendations for protection of aquatic life from the Ambient Water Quality Criteria for chloride (USEPA, 2017a). Sampling frequency and total sample sizes at each sampled location varied but ranged from 22 to 84 total samples during the analysis period. The index was divided into three categories; 1), an index value of 100 indicated no chronic exposure as determined by a mean chloride concentration of less than 230 mg/L; 2), an index value of 50 was used for a mean concentrations of chloride values greater than 230 but less than 860 mg/L; and 3), and an index value of 0 indicated acute chloride exposure corresponding to mean reach chloride concentrations greater than 860 mg/L (Figure 3.16).

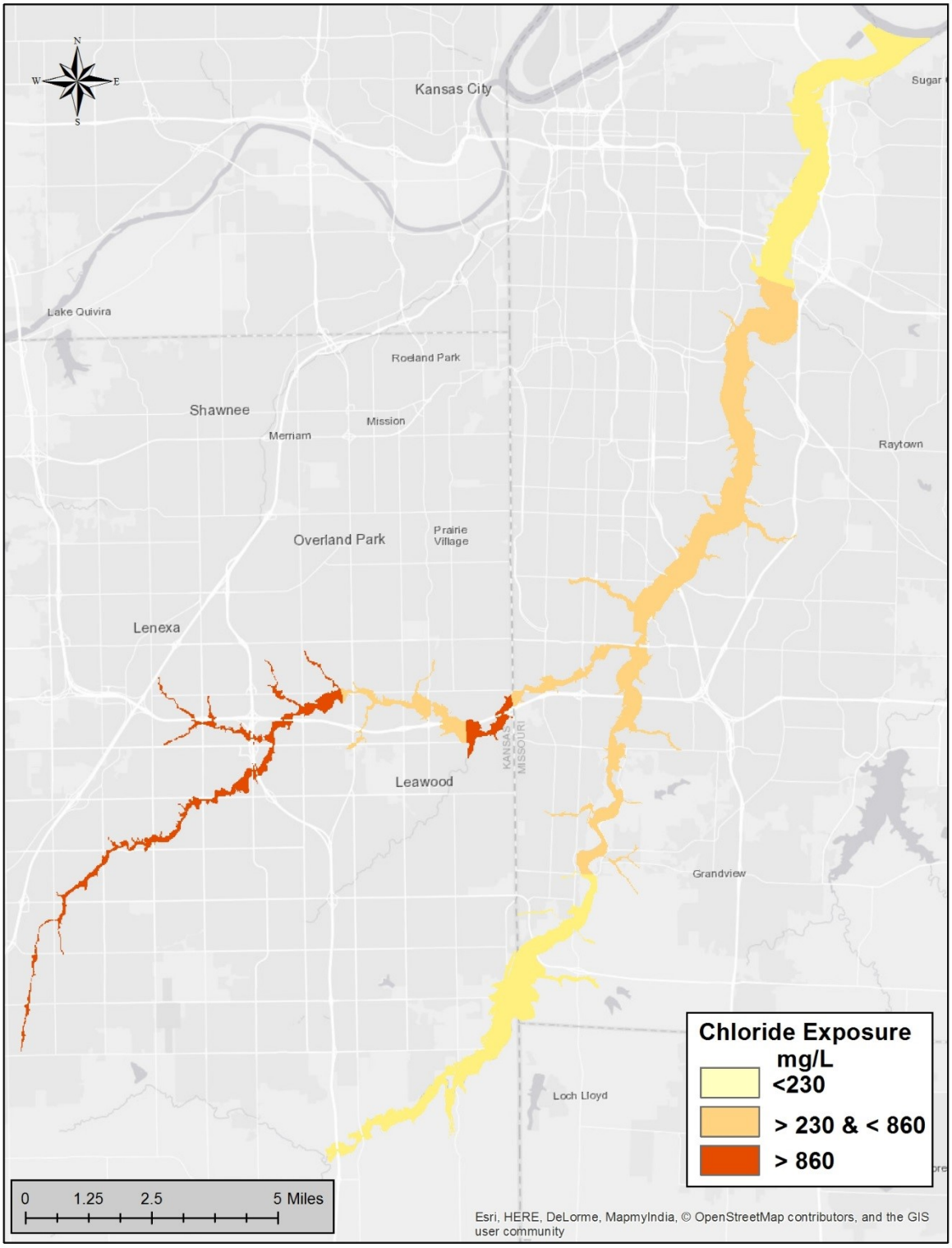


Figure 3.16. Water Quality – Chloride data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

The *E. coli* EIUSH layer was developed based on data collected within the basin (an average of 44 samples per site) between 1999 and 2010 and this data layer was included in the index as the percent of available samples (USGS, 2017b) that were within the state compliance for *E.coli* at selected sample locations (Figure 3.17).

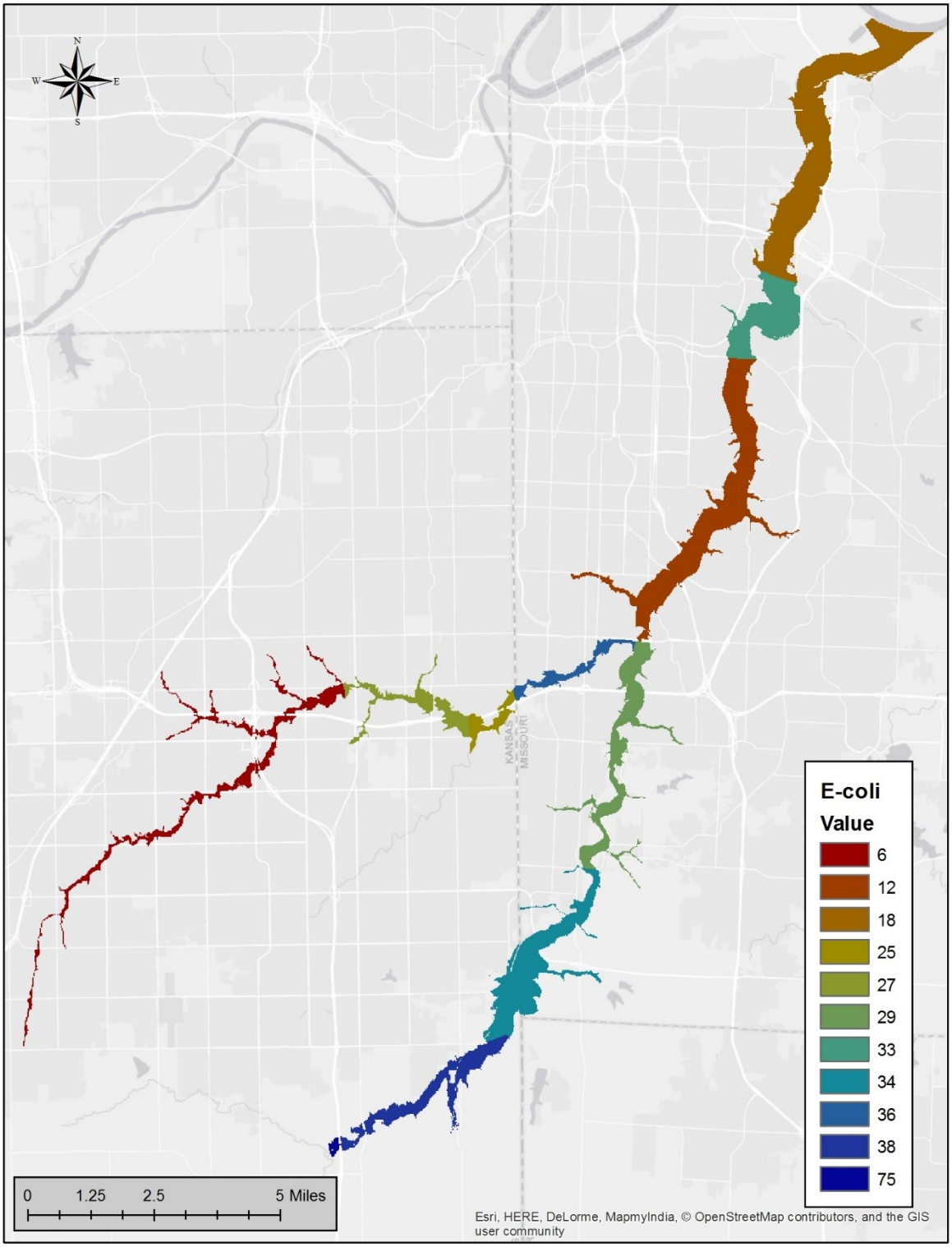


Figure 3.17. Water Quality – E.coli data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

The TWPE layer was developed by summarizing the TWPE values (determined by EPA using pounds of discharge multiplied by the toxicity weighting factor; USEPA, 2017b) within analysis reaches that were defined by streamgage locations. The TWPE data for the period 2007-2009 were used in EIUSH version I, and data for 2014-2016 were used for EIUSH version II. TWPE data were derived from reported and permitted NPDES discharges including nutrients, petroleum products, metals, and other organic compounds. The cumulative TWPE values, by reach, were averaged for the 2007-2009 period and the 2014-2016 periods. The values were included in the index based on the percentile of the mean reach values within the distribution of all individual non-zero TWPE values within the basin for the 2007-2009 (Figure 3.18) and the 2014-2016 periods (Figure 3.19). The index scores were presented as

$$1 - \text{mean TWPE percentile},$$

such that areas with high TWPE index scores corresponded to low levels of TWPE, and low scores were areas with high levels of TWPE.

Index Model Development

The spatially-weighted index models incorporated datasets with varying spatial and temporal extents. The purpose of the EIUSH was to summarize and communicate a broad range of spatial and temporal data and provide a generalized health assessment utilizing available datasets. The index model incorporated each different layer as a contributing weight and scale based on the different feature attributes (Table 3.7), which were numerically combined into a raster layer. Each dataset shared common geography based on the same study extents, while the interior spatial range of each layer varied based on the available distribution of the feature values. The temporal extents also were different, depending on the available monitoring period of the features.

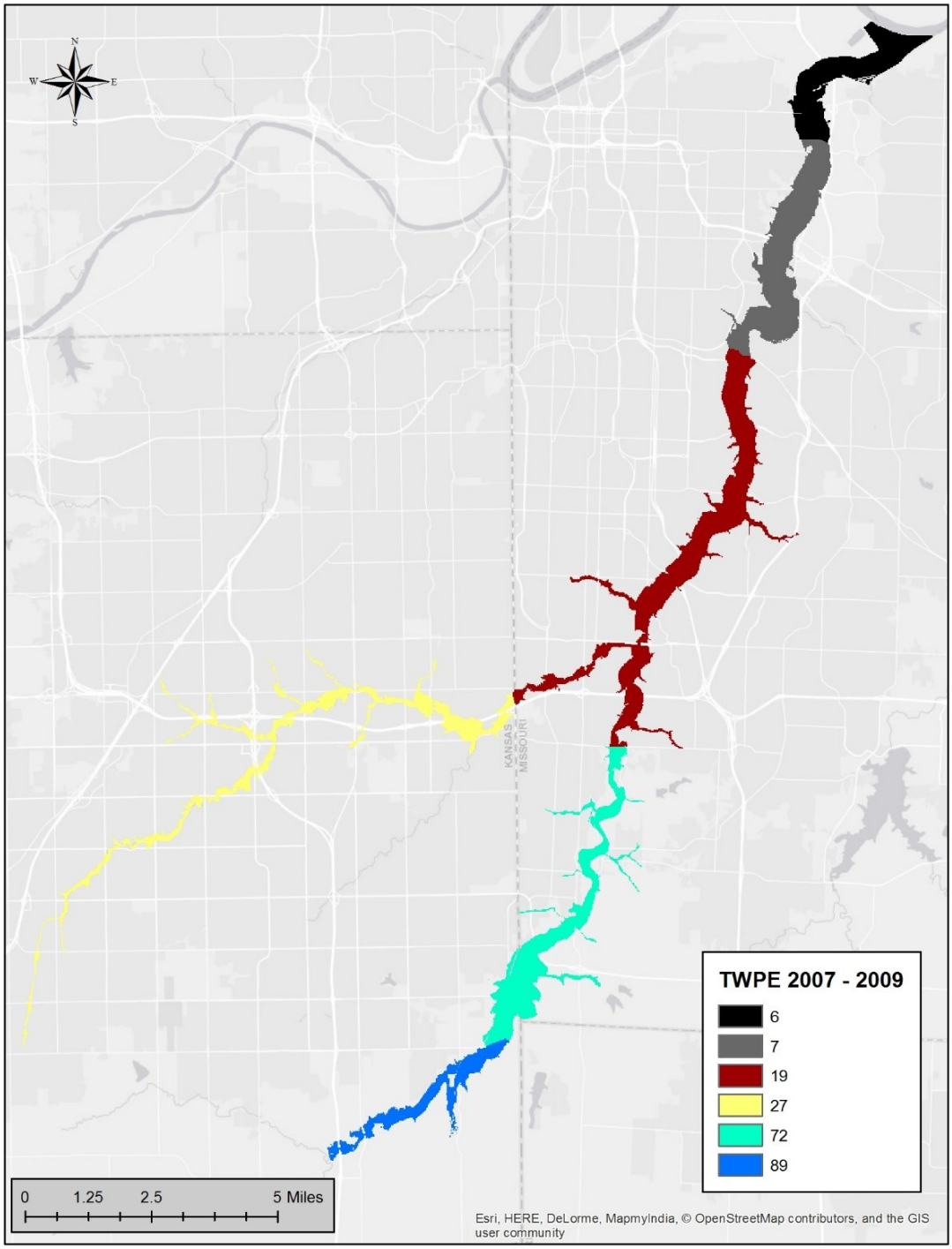


Figure 3.18. Water Quality – TWPE 2007-2009 data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

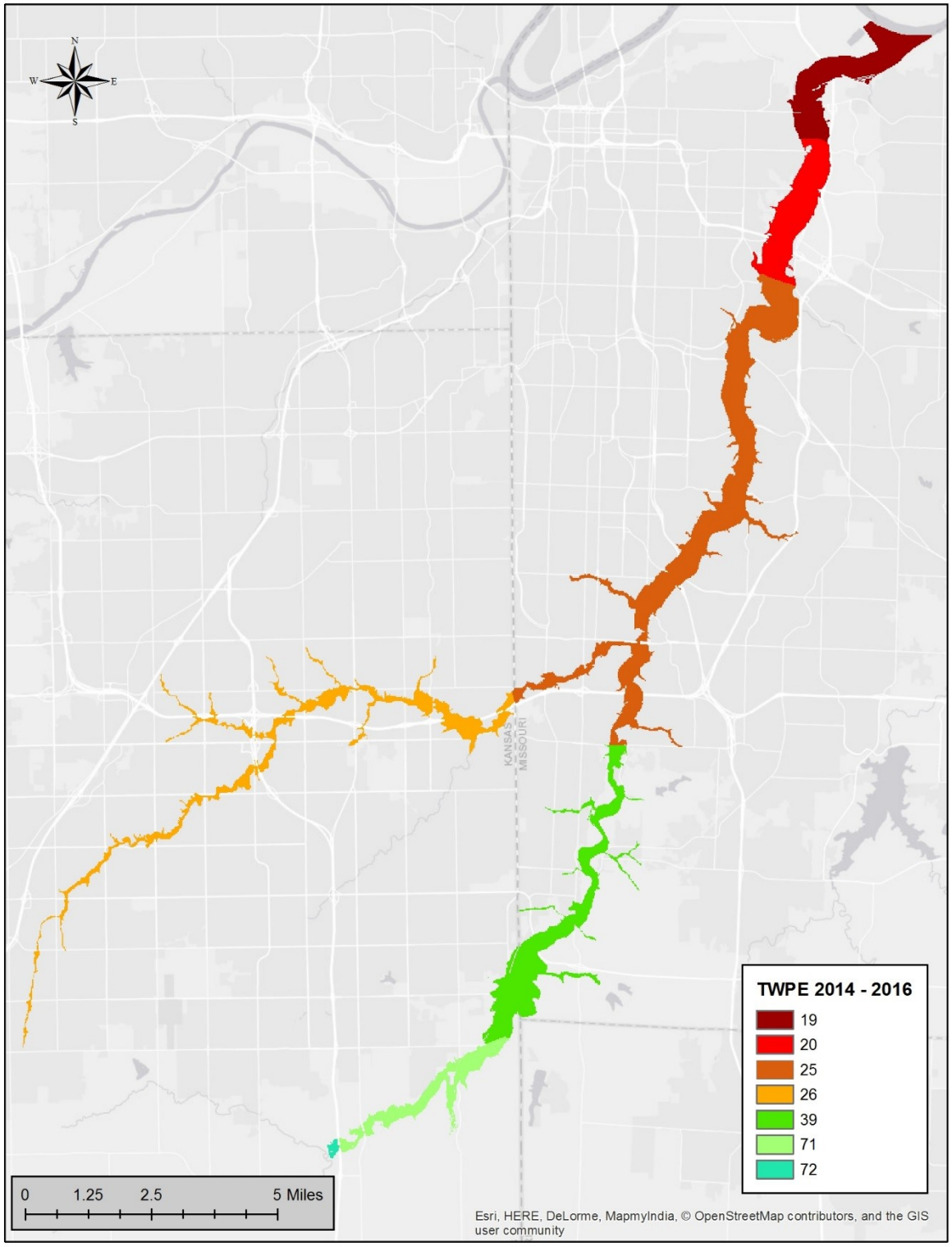


Figure 3.19. Water Quality – TWPE 2014 – 2016 data layer used in the development of the Ecological Index of Urban Stream Health, Kansas City, Missouri, and vicinity.

Table 3.7

The weighted model parameter components and scales used in the development of the Ecological Index of Urban Stream Health (EIUSH) version I and II.

Model Parameter	Feature layer	Units	Categories	Scale Value	Weight
Biological Characteristics	Macroinvertebrates	percent	Nonsupporting (KS, 1/3; MO 4-8/20)	10	0.2
			Partially Supporting (KS 2/3; MO, 10-14/20)	50	
			Supporting (KS, 3/3; MO 16-20/20)	100	
Physical Characteristics	Physical Habitat (2009 - SIR - USGS) 2006 data	percent	0-100 depending on ratio of calculated score to total possible score	#	0.1
	Channelization	percent	substantial substrate changes and total channelization present	0	0.1
			substantial substrate changes and channelization present	20	
			substantial substrate with no or minimal channelization	40	
			moderate substrate changes or channelization	60	
			minimal substrate or channelization	80	
			no substrate or channelization	100	
Water Hydrology	Indicators of Hydrologic Alteration	Hydrology (Indicators of Hydrologic Alteration - Nature Conservancy) 67 ecological factors analyzed (Site data	0-100 depending on ratio of calculated score to total possible score - Inverse (1-# metrics/67)	#	0.2
Land Use Land Cover	National Land Cover Dataset (2006)	na	Open water(11)	80	0.2
			Developed, open space (21)	20	
			Developed, low intensity(22)	20	
			Developed, medium intensity(23)	10	
			Developed, high intensity(24)	0	
			Barren land(31)	40	
			Deciduous forest(41)	70	
			Evergreen forest(42)	70	
			Mixed forest(43)	70	
			Shrub/scrub(52)	60	
			Herbaceous woodland(71)	80	
			Hay/pasture(81)	40	
			Cultivated crops(82)	50	
			Woody wetlands(90)	100	
			Emergent herbaceous wetlands(95)	100	
Water Quality (Non point source)	Chloride	Mg/L	< chronic standard of 230	100	0.05
			> chronic , < acute 860	50	
			> acute 860	0	
	E Coli	%	0-100 based on % of samples below state standards	#	0.05
Water Quality (Point source)	Toxic Weighted Pounds Equivalent (TWPE)	Mass of a pollutant	Inverse of the TWPE (1-#) per stream gage reach	#	0.1

All spatial datasets started as GIS vector files and were processed using ArcMap version 10.3 (ESRI, 2014) GIS modelbuilder. The ArcGIS modelbuilder also was used to preprocess the geospatial data. The vector data layers were all projected to a standard Albers-Conical Equal Area projection and then converted to raster files for inclusion in the index model (Figure 3.20). All input raster (grid) layers used in the development of the EIUSH had a 30-m cell resolution. The numerical values for each feature layer were categorized and assigned a discrete value on a scale of 0 (least suitable) to 100 (most suitable) and each feature was assigned a weighting factor (Table 3.7) based on professional judgment similar to methods by Van Lonkhuyzen, LaGory, and Kuiper, (2004) and White and Fennessy (2005). The final index was calculated using the categorized layer data and weights in the "Weighted Overlay Table" tool in ArcMap (ESRI, 2014) to produce an overall index per raster cell with a score of 0 to 100 that is incorporated into the weighted overlay table (Figure 3.21) and processed through a model (Figure 3.22). The final index scores were split into a range of 4 qualitative classes to define corresponding qualitative levels of impairment. The 0 to 40 score is indicative of a severe level of impairment, a 40 to 60 score is a high impairment level, 60 to 80 score indicates a moderate level of impairment, and a score of 80-100 is a low level of impairment. The index scores were summarized at a study extent level and then were subdivided at a reach level to depict the natural spatial gradients/variation across the study area (Figure 3.23).

The index was "calibrated" by comparing the macroinvertebrate layer scores to the composite EIUSH reach scores without macroinvertebrates. This comparison was used to

simply check the magnitude and distribution of the composite and macroinvertebrate index scores to determine if the correlation appeared to be “reasonable”.

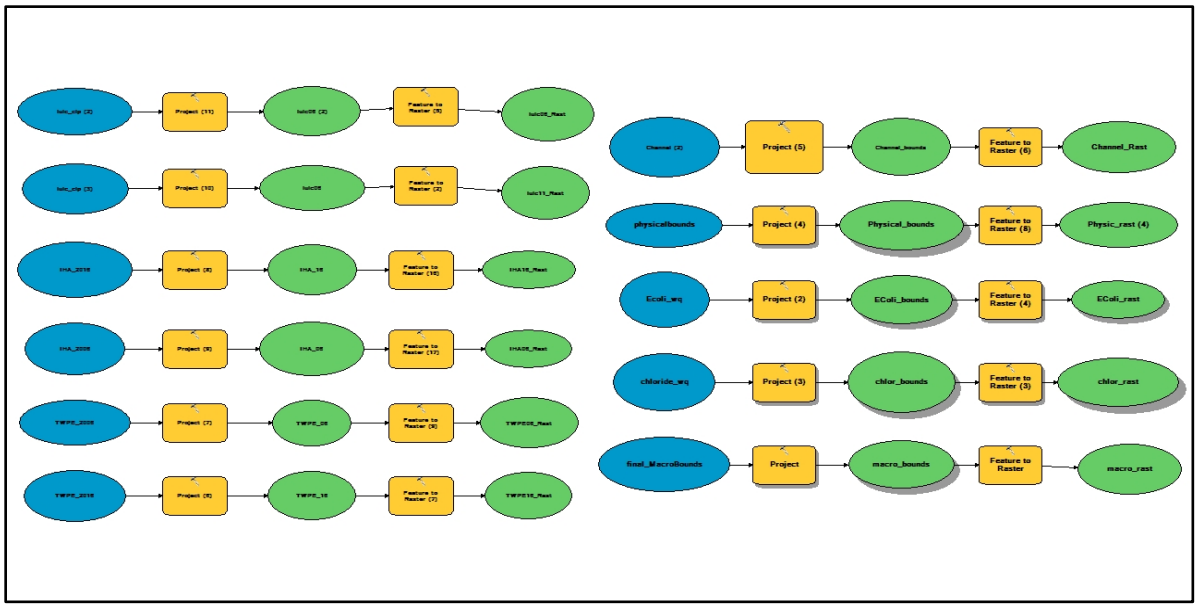


Figure 3.20. ModelBuilder Pre-Processing for EIUSH Models.

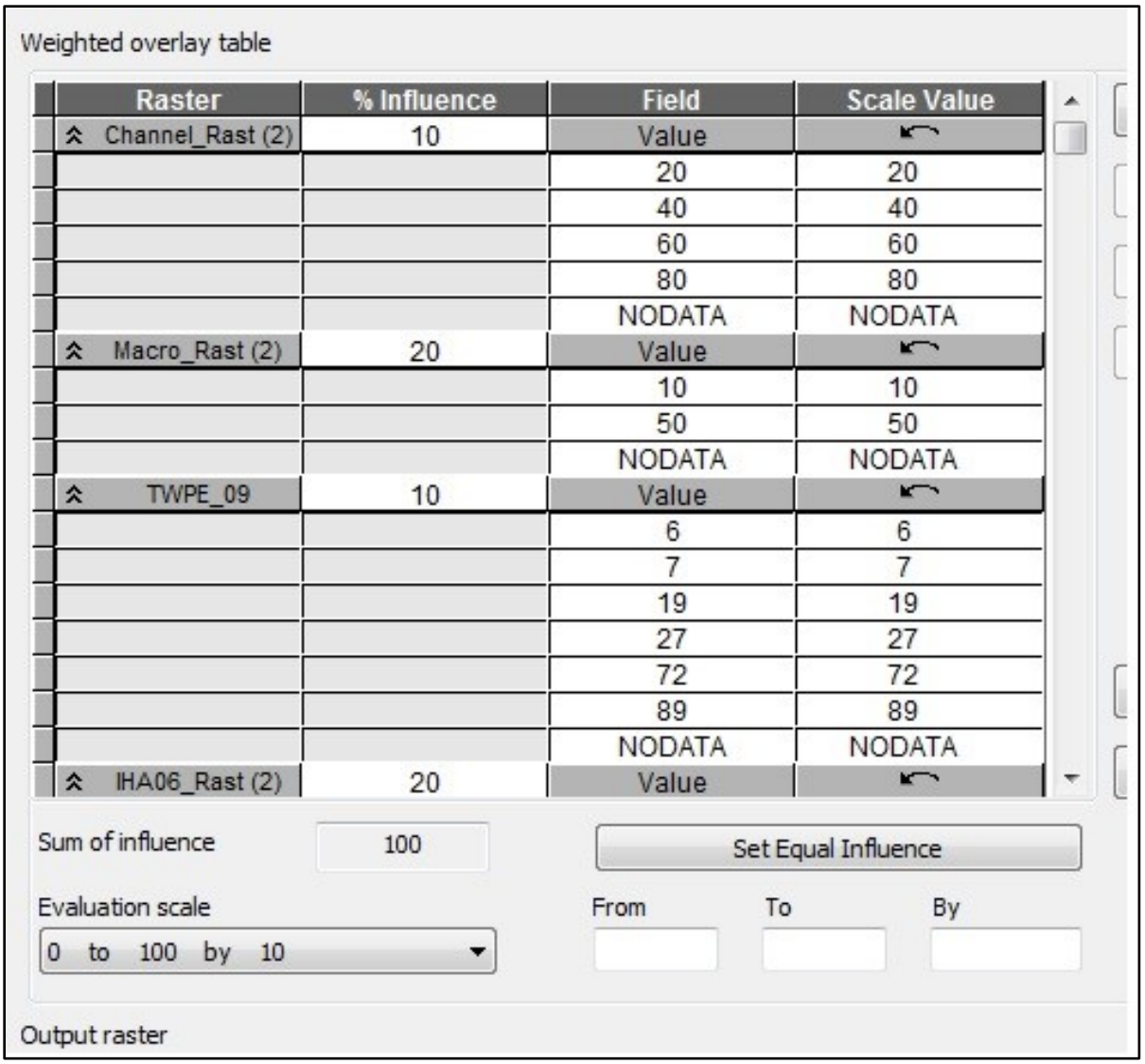


Figure 3.21. Weighted overlay table

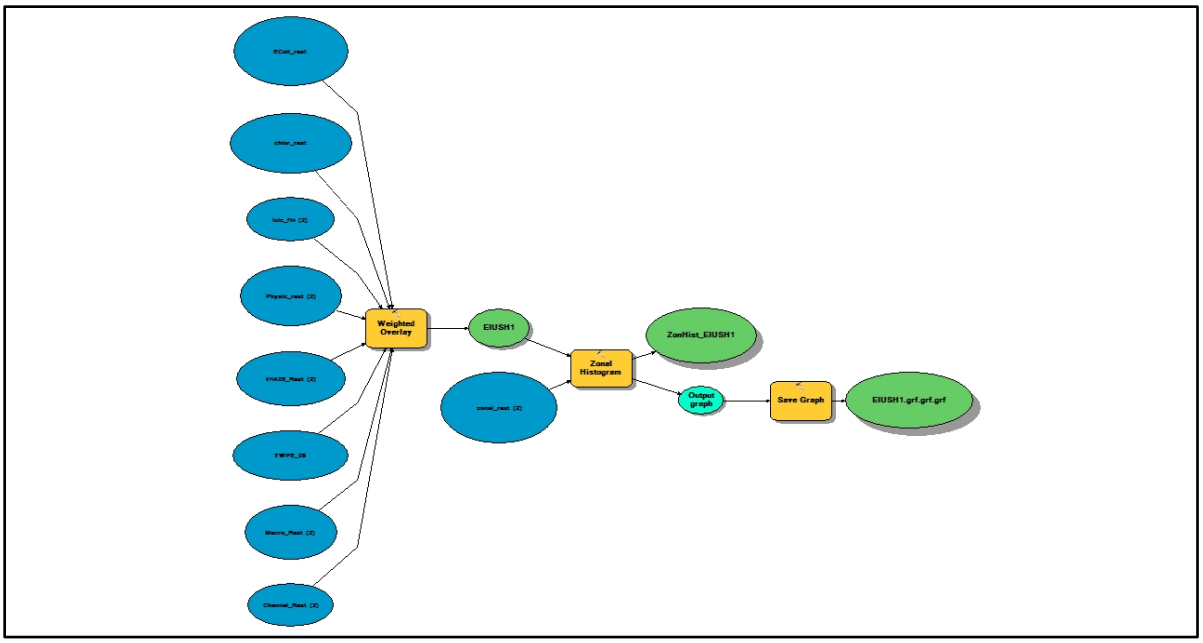


Figure 3.22. ModelBuilder Processing for EIUSH Model I.

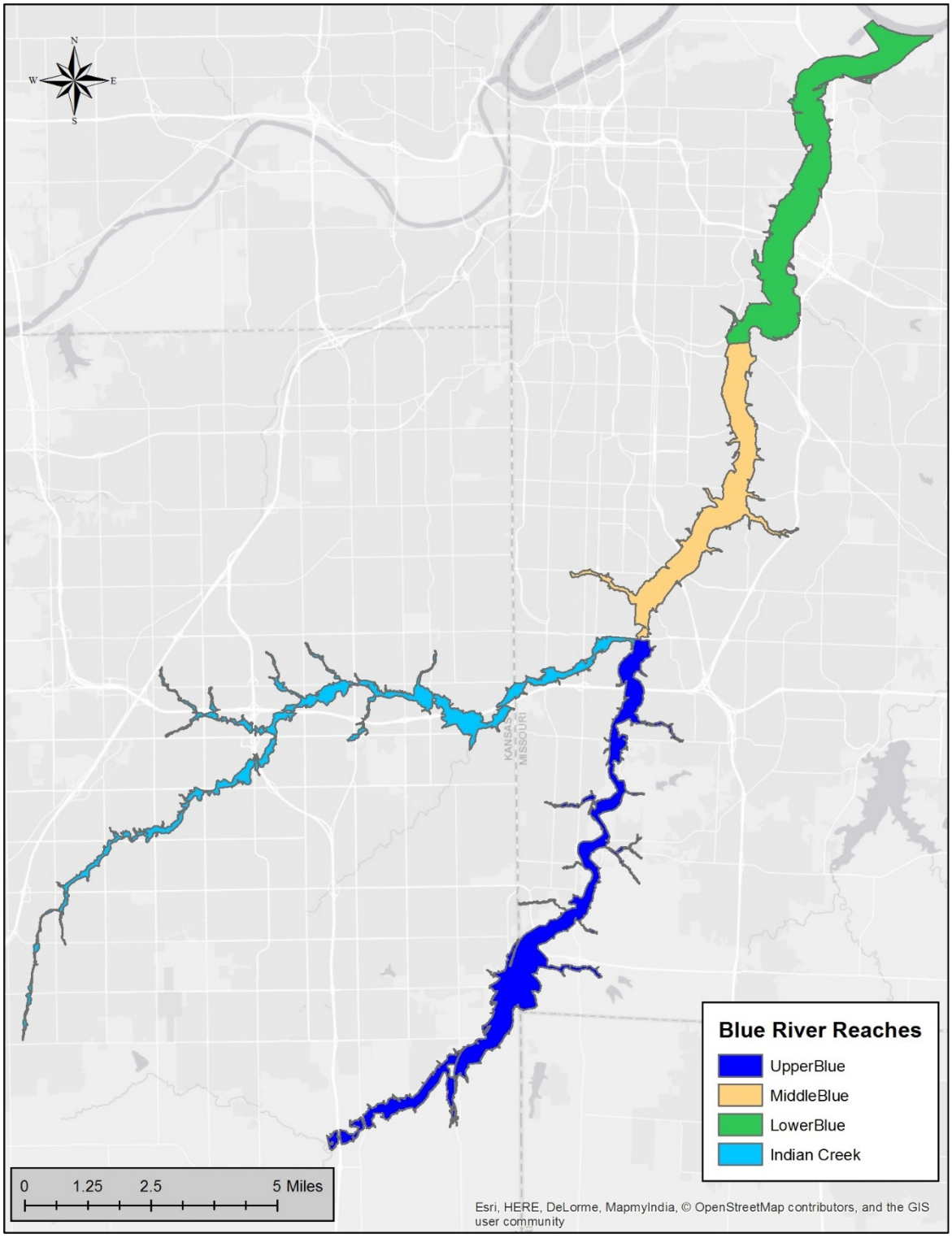


Figure 3.23. Map Reach Extents – EIUSH I and II.

CHAPTER 4

RESULTS

Total Riparian Vegetation Change within the Study Area

There was a net loss in trees and grasses in the study area from 1992 to 2012, and a corresponding increase in developed land-cover during the same period (Table 4.1). The trees land-cover class had a net decrease of approximately 12.5% or 4.15 km^2 over the 20-year period and indicated a consistent incremental loss of trees for each image analysis period. Similarly, the grasses land-cover class lost 9.7% or 3.2 km^2 between 1992 and 2012 within the study area. The greatest change in the land cover classes occurred in developed land, which increased by 22.9% and 7.57 km^2 between 1992 and 2012. The net increase in developed land also was consistent in each discrete analysis increment over the 20-year study period. There was little (<2%) net change in the barren land and water land cover categories over the 20-year study period.

Table 4.1

Land-cover within the total selected study area for pre - and post-streamside ordinance categories and selected years that correspond to SPOT image availability.

Generalized land-cover class [1]	1992 (0% of area within streamside ordinance protection)		2003 (1.64% of area within streamside ordinance protection)		2009 (43% of area within streamside ordinance protection)		2012 (43% of area within streamside ordinance protection)	
	Area (km^2)	**Percent cover**	**Area (km^2)**	**Percent cover**	**Area (km^2)**	**Percent cover**	**Area (km^2)**	**Percent cover**
Barren Land	0.90	2.72%	0.72	2.18%	0.05	0.16%	0.53	1.59%
Developed Land	5.71	17.22%	8.83	26.65%	11.11	33.52%	13.28	40.08%
Grasses	12.42	37.48%	9.94	30.00%	9.47	28.58%	9.22	27.82%
Trees	12.81	38.65%	12.01	36.23%	10.64	32.10%	8.66	26.14%
Water	1.30	3.94%	1.64	4.94%	1.87	5.64%	1.45	4.38%
Total	33.14	100.00%	33.14	100.00%	33.14	100.00%	33.14	100.00%

[1] User-defined land-use/land-cover classes developed from SPOT imagery

Riparian Vegetation Change by Municipality and Ordinance Protection

City of Kansas City

There were declines in the trees and the grasses cover classes and increases in developed land in the Kansas City portion of the study area throughout the analysis period regardless of ordinance protection category. The rate of loss in the trees class following ordinance protection was similar to that of the pre-ordinance periods at about -0.56%/year (yr), but was substantially less than the loss (-2.67%/yr) in the non-ordinance area in the 2009-12 post-ordinance period (Figure 4.1). Although declines in the grasses cover class continued throughout the analysis period, there was a modest decline in the rate of loss in the post-ordinance period. The rate of loss of the grasses cover in the post-ordinance period

(-0.58%/yr) also was less than that of the non-ordinance area (-0.86%/yr). The rate of change in the developed cover class increased between pre- and post-ordinance periods from about 0.9%/yr in the 1992-2002 period to 1.85%/yr in the 2009-12 post-ordinance period. The greatest annualized rate of change in trees (-2.67%, 2009-2012), grasses (-1.22%, 1992-2003), and developed land (3.3%, 2009-2012), all were in the non-ordinance protected area (Figure 4.1). The area in the northern portion of the Blue River Basin that is outside of the Kansas City ordinance protection buffer zones provides an example of this pattern for the 2009-12 period, as land-cover changed from trees and grasses to developed land (Figure 4.2).

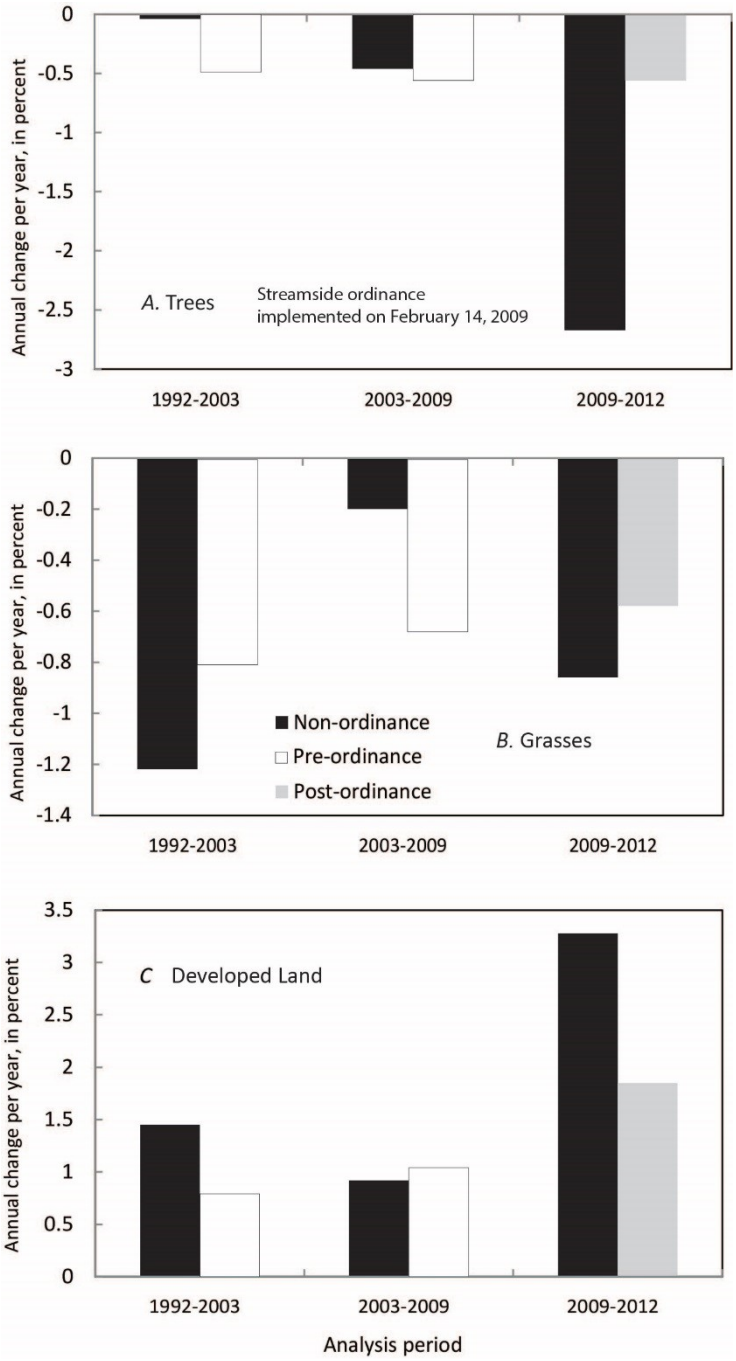


Figure 4.1. Change in *A*) Trees; *B*) Grasses; and *C*) Developed land within the city of Kansas City, Missouri inside and outside the streamside ordinance area during pre-ordinance (1992-2009) and post-ordinance (2009-12) periods.

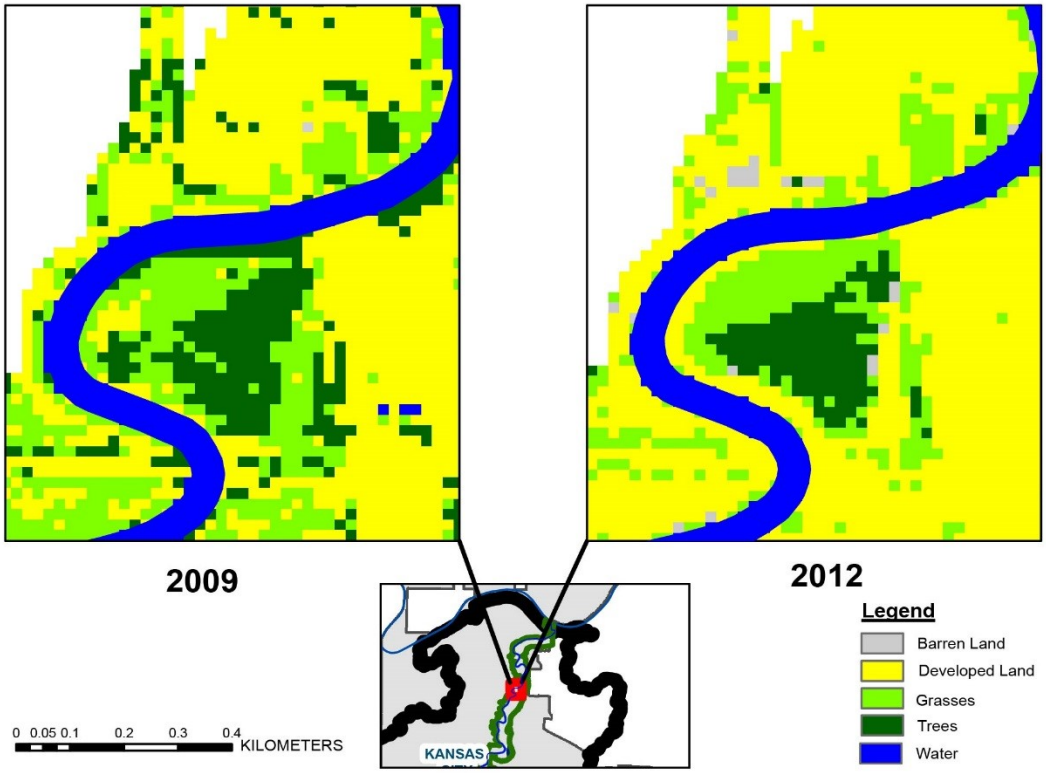


Figure 4.2. Land-cover change classification 2009 and 2012 Kansas City, Missouri.

There generally was consistency in the direction (increase, decrease) of temporal change in land cover among buffer protection zones within the city of Kansas City, but results indicated variability in the magnitude of land cover change by zone. Generally, the least change in the land cover classes occurred in zone 1—the zone nearest the stream channel. The greatest loss (-2.2%/yr) in the trees landcover occurred in zone 3 following ordinance protection (2009-12), while the greatest loss (-2.8%/yr) in the grasses cover class occurred in zone 2, also following ordinance protection. The greatest increase in developed land within the three Kansas City protection zones occurred in zone 2 from 2009-12 (3.9%/yr, Figure. 4.3) and corresponded with similar losses in the grasses and tree cover classes.

City of Overland Park

There were substantial temporal changes in the land cover classes within the city of Overland Park with changes similar within post-ordinance and non-ordinance areas. Tree loss was greater in the post ordinance period (2003-12) compared to pre-ordinance period for both ordinance protected and non-ordinance areas (Figure 4.4). There generally was an increase in the grasses cover class and developed land in the post-ordinance period for both ordinance protected and non-ordinance areas. The rate of increase in developed land area was greater during the post-ordinance period (2003-12) in the ordinance-protected area (0.8-2.1%/yr) compared to that of the non-ordinance area (0.7 – 1.4%/yr) (Figure 4.4). The Overland Park ordinance area experienced the greatest loss in trees cover class from 2003-09, at 2.06% per year, while the non-ordinance protected area experienced the greatest rate of loss of 3.2% per year from 2009-12.

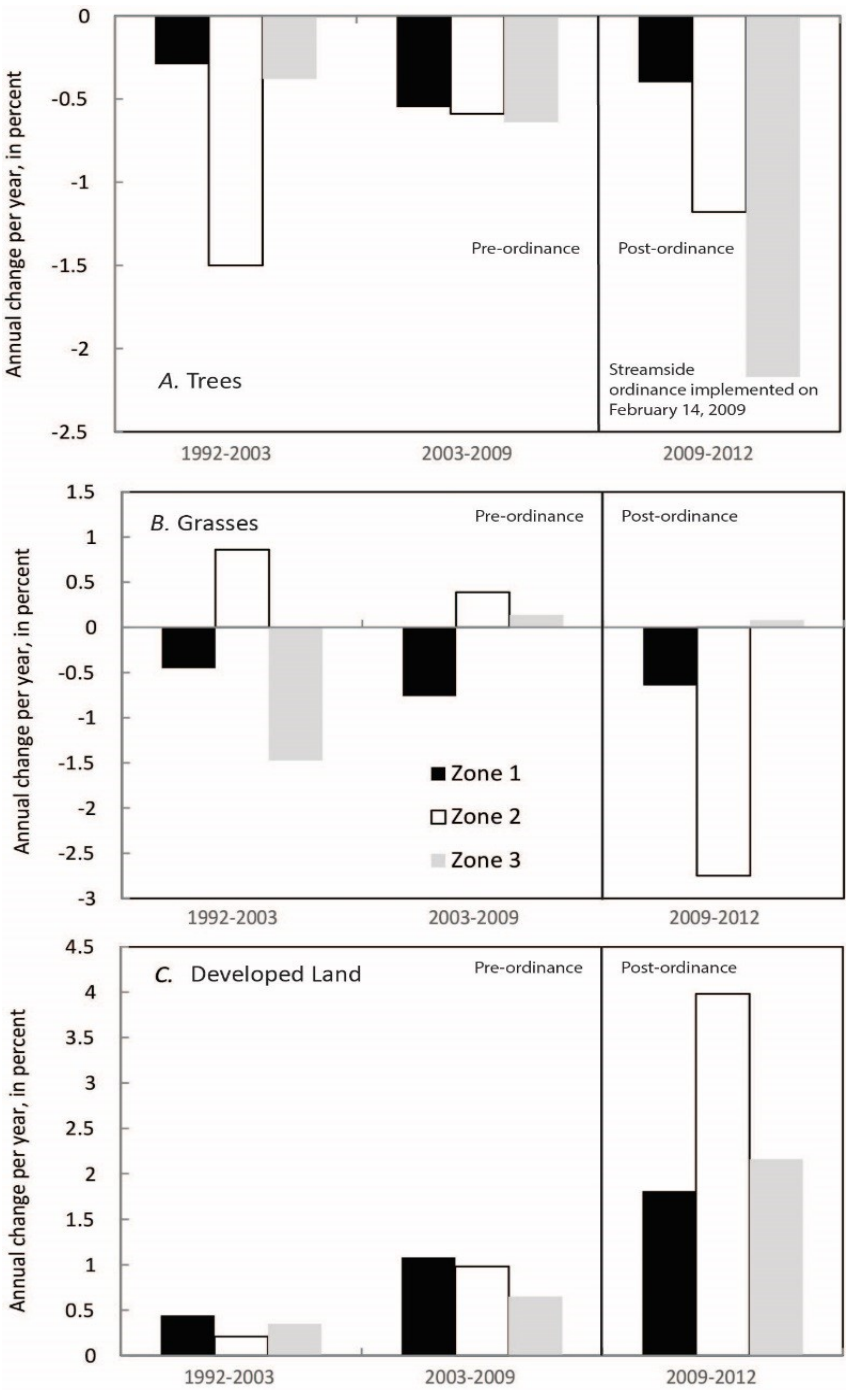


Figure 4.3. Change in *A*) Trees; *B*) Grasses; and *C*) Developed land within the city of Kansas City, Missouri inside the three streamside ordinance zones and outside the ordinance area during pre-ordinance (1992-2009) and post-ordinance (2009-12) periods.

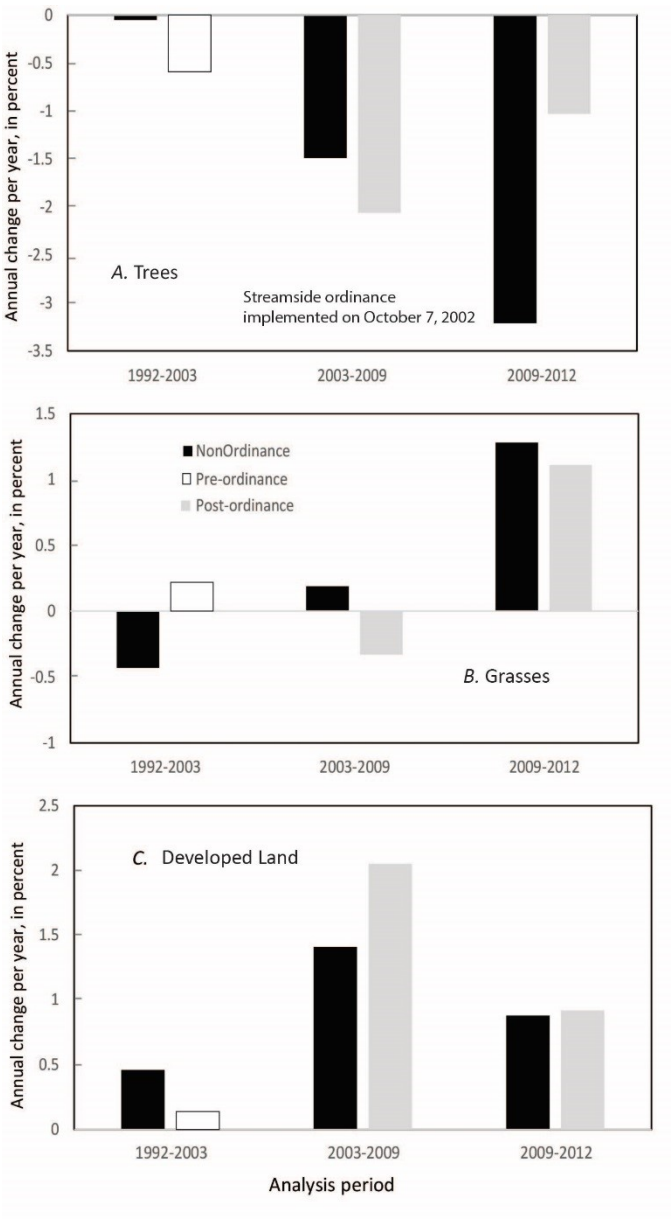


Figure 4.4. Change in *A*) Trees; *B)* Grasses; and *C*) Developed land within the city of Overland Park, Kansas inside and outside the streamside ordinance area during pre-ordinance (1992-2009) and post-ordinance (2009-12) periods.

Unincorporated Johnson County, Kansas

The unincorporated Johnson County portion of the study area, an area without ordinance protection, showed the greatest changes in land-cover during the analysis period. There was a consistent loss of trees during each discrete analysis period with the greatest rate of tree loss (-4%/yr) occurring in the 2009-12 period. The grasses land cover increased from 2003-12 by a maximum of 3%/yr in 2009-12. The developed land-cover increased during each analysis period, with the greatest increase (1.71%/yr) occurring during 2003-09 and the least (0.2%/yr) during the 2009-12 period (Figure 4.5).

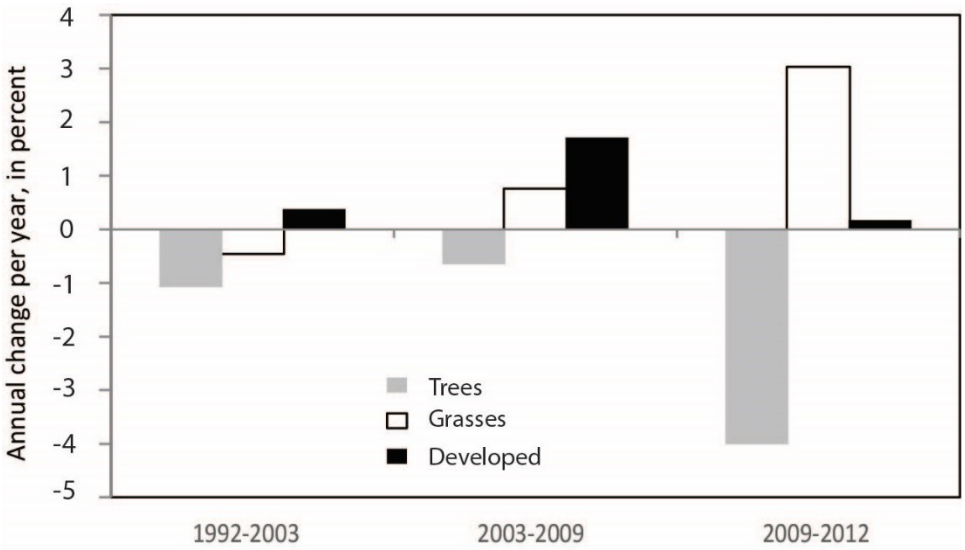


Figure 4.5. Change in selected land-cover categories within unincorporated Johnson County, Kansas, during 1992-2012.

Ecological Index of Urban Stream Health

The ecological health of the Blue River and major tributary, Indian Creek, were assessed using a GIS-based index incorporating data layers representing five major environmental factors. The index can be used as a quantitative indication of the spatial and temporal changes in the ecological integrity of the study area.

The resulting EIUSH maps (Figure 4.6, Figure 4.7) show the spatial distribution of calculated values with a possible range of 0 (lowest ecological health) to 100 (greatest ecological health). No portion of the study area attained the maximum potential index value of 100. The overall range of index scores in EIUSH I was 25 (high impairment) to 82 (low impairment) (Figure 4.6) and the mean index value was 45 (Table 4. 2, Figure 4.8). Areas mapped with an EIUSH value of 31-40 increment (severe impairment) represented the largest areal extent covering about 31 percent of the study area (Figure 4.8). Overall, nearly 48 percent of the flood plain in the analysis area has an EIUSH value of greater than 40 (moderate impairment) and about 0.01 percent had an index score of greater than 80 that was within the low impairment category (index >80) (Table 4.2).

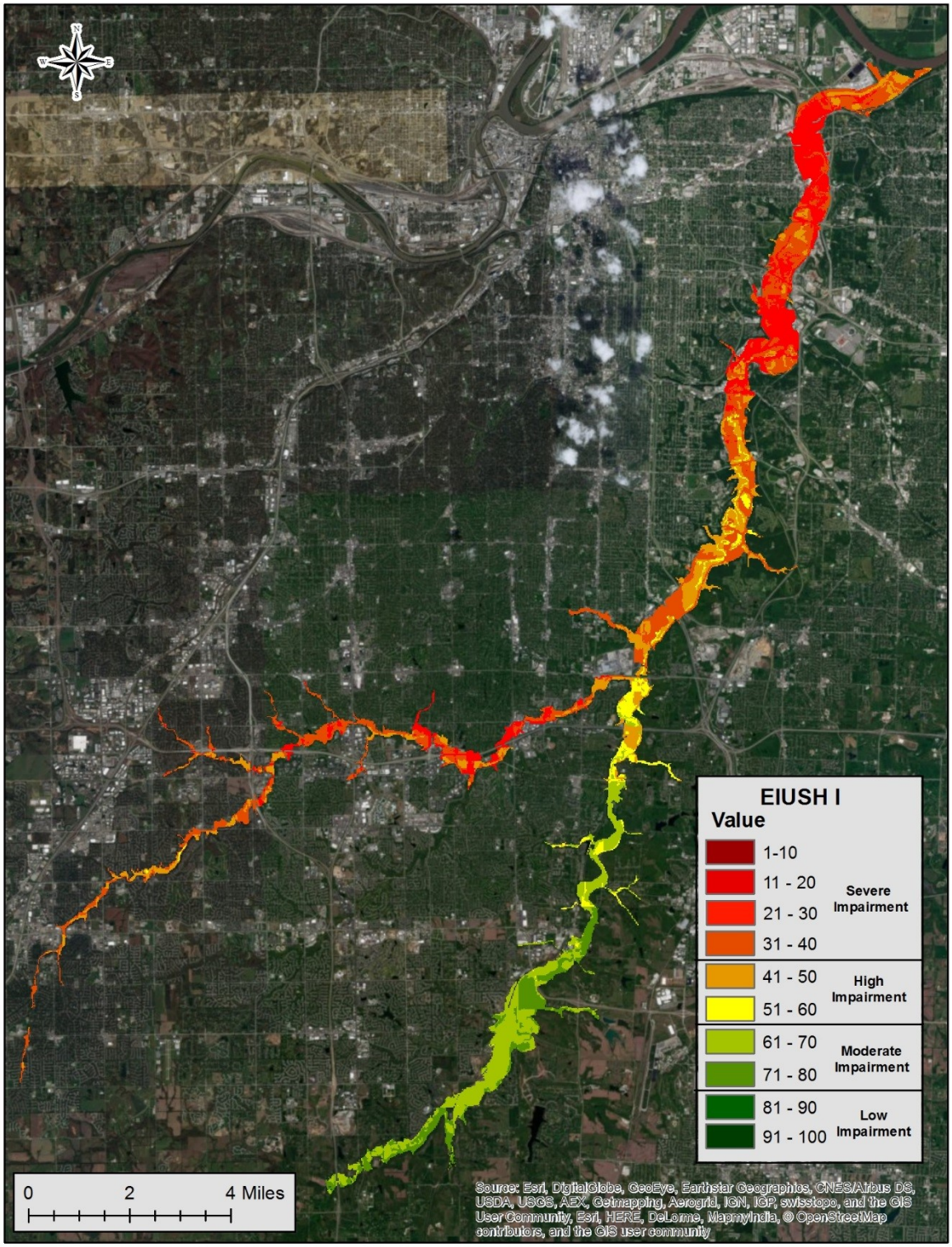


Figure 4.6. Map of EIUSH I, 1999 to 2013, Blue River, Kansas City metropolitan area.

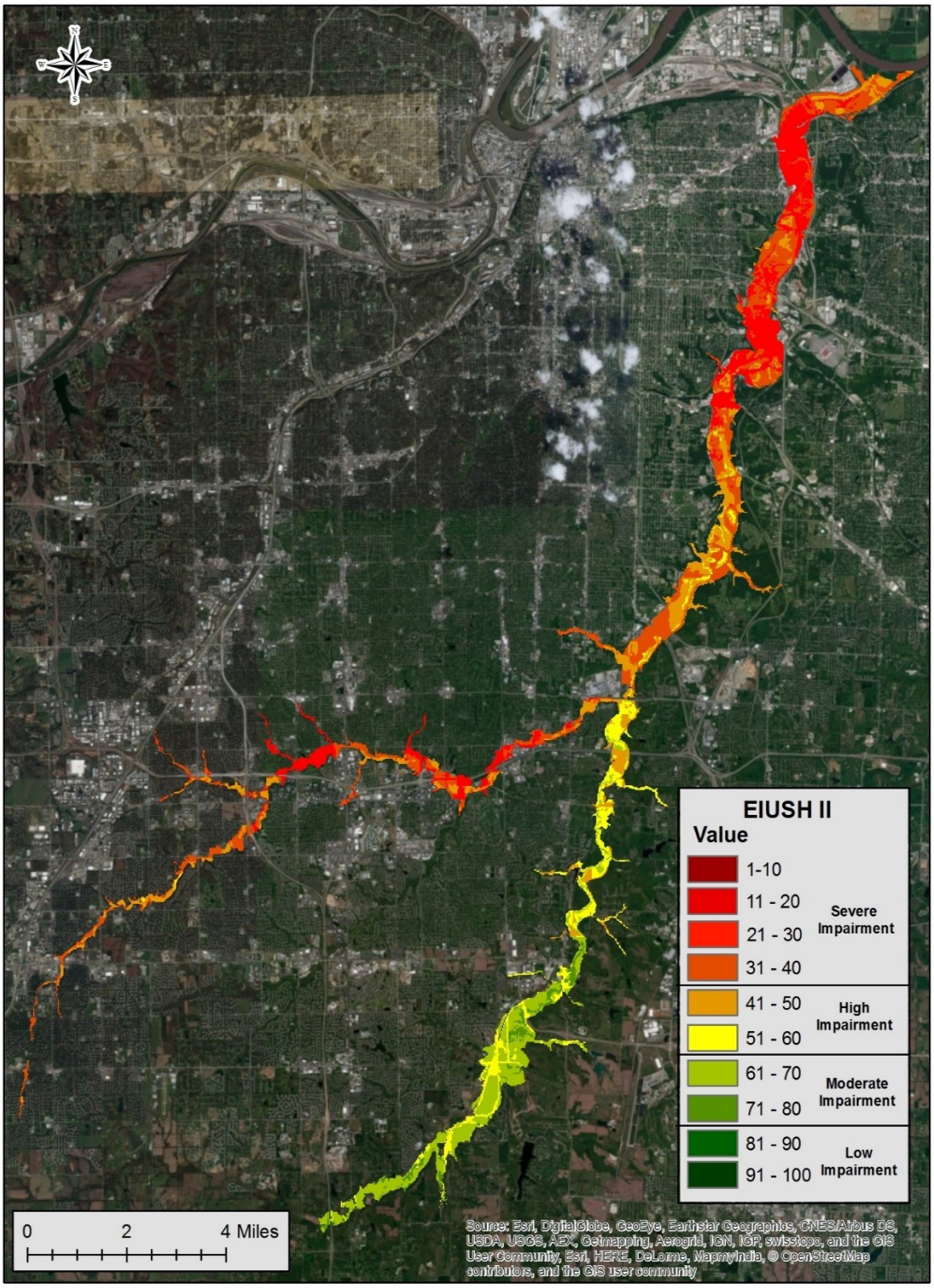


Figure 4.7. Map of EIUSH II, 1999 to 2016, Blue River, Kansas City metropolitan area.

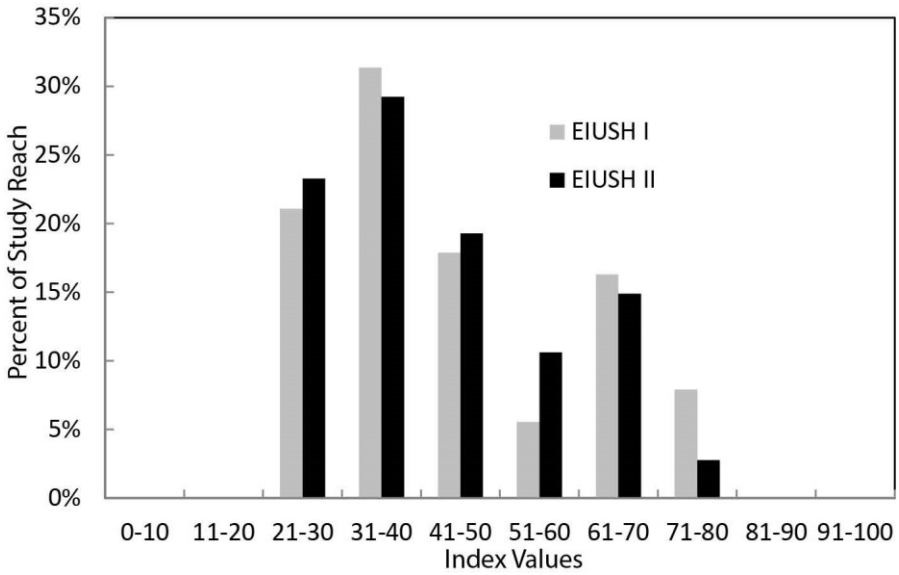


Figure 4.8. Distribution of Ecological Index of Urban Stream Health (EIUSH) values within the study area for EIUSH I and EIUSH II.

Table 4.2

EIUSH I scores by reach and index increment (Indian Creek, Upper Blue, Middle Blue, Lower Blue)

	Percent of area within EIUSH I score incremental category									
	Impairment scale									
	Severe				High		Moderate		Low	
Stream reach	0-10	11-20	21-30	31-40	41-50	51-60	61-70	71-80	81-90	91-100
Overall	0.00%	0.00%	21.21%	31.03%	18.59%	4.73%	16.61%	7.82%	0.01%	0.00%
Indian Creek	0.00%	0.00%	21.77%	52.39%	25.37%	0.46%	0.00%	0.00%	0.00%	0.00%
Upper Blue River	0.00%	0.00%	0.00%	0.00%	5.81%	12.07%	55.53%	26.56%	0.03%	0.00%
Middle Blue River	0.00%	0.00%	9.35%	44.70%	41.26%	4.69%	0.00%	0.00%	0.00%	0.00%
Lower Blue River	0.00%	0.00%	49.04%	39.08%	11.88%	0%	0.00%	0.00%	0.00%	0.00%

The distribution of EIUSH I index scores by reach indicates that the upper Blue River was the only reach with a mean index score greater than 40 as the mean value in this reach was 65.8 (Table 4.3, Figure 4.9). The lower Blue River reach had the lowest mean index score 33.1 and the maximum score did not exceed 50, whereas the mean index scores in Indian Creek and middle Blue River were 36.0 and 40.3, respectively, and maximum index values were 53 and 54, respectively (Table 4.3, Figure 4.9).

Table 4.3

Summary of EIUSH I scores by reach (Indian Creek, Upper Blue, Middle Blue, Lower Blue)

EIUSH I reach summary scores			
General River Segment	**Minimum**	**Maximum**	**Mean Value**
Overview	25	82	44.8
Indian Creek	25	53	36.0
Upper Blue	42	82	65. 8
Middle Blue	26	54	40. 3
Lower Blue	26	48	33.1

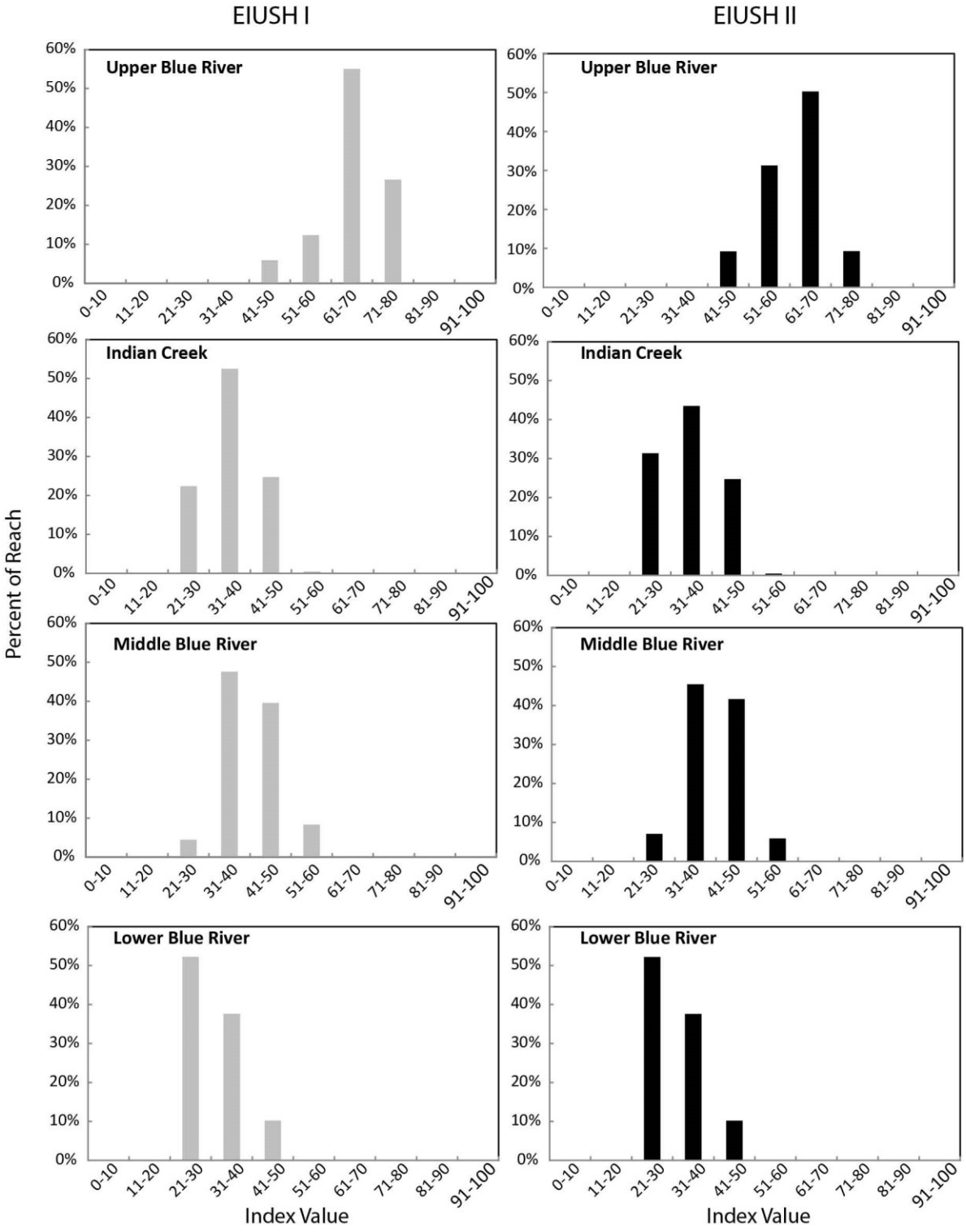


Figure 4.9. Distribution of Ecological Index of Urban Stream Health (EIUSH) by reach, within Blue River study area.

The EIUSH II results, representing temporal revisions to three (hydrology, water quality, land use) of the five primary environmental factors with 2011-2016 datasets. The overall results were similar to those of EIUSH I with a minor increase in impairment (Figure 4.8). The overall range of index scores was 24 – 78 with a mean index score of 43.2. As with EIUSH I, the majority of the study area in EIUSH II was in the 31-40 index increment, but none of the area in EIUSH II obtained an index score of greater than 80 (low impairment category) (Figure 4.8, Table 4.4).

Table 4.4

EIUSH II scores by category and reach increment (Indian Creek, Upper Blue, Middle Blue, Lower Blue)

	Percent of area within EIUSH II score incremental category									
	Impairment scale									
	Severe				High		Moderate		Low	
Stream reach	0-10	11-20	21-30	31-40	41-50	51-60	61-70	71-80	81-90	91-100
Overall	0.0%	0.0%	23.8%	29.2%	18.9%	10.6%	14.8%	2.7%	0.0%	0.0%
Indian Creek	0.0%	0.0%	31.4%	43.5%	24.7%	46.0%	0.0%	0.0%	0.0%	0.0%
Upper Blue River	0.0%	0.0%	0.0%	0.0%	9.2%	31.3%	50.2%	9.3%	0.0%	0.0%
Middle Blue River	0.0%	0.0%	12.0%	44.9%	38.5%	4.0%	0.0%	0.0%	0.0%	0.0%
Lower Blue River	0.0%	0.0%	50.0%	38.3%	11.7%	0.0%	0.0%	0.0%	0.0%	0.0%

Similar to EIUSH I, the distribution of EIUSH II index scores by reach indicated that the upper Blue River reach had the greatest mean index score (61.7, moderately impaired), and the lower Blue River had the lowest mean score (33.1)(table 4.5). The middle Blue River reach mean index score declined to within the severely impaired qualitative category with a mean score below 40 (38.9). The greatest change by reach between EIUSH I and II was in the

Table 4.5

Summary of EIUSH II scores by reach (Indian Creek, Upper Blue, Middle Blue, LowerBlue)

EIUSH II reach summary scores			
General River Segment	**Minimum**	**Maximum**	**Mean Value**
Overview	24	78	43.2
Indian Creek	24	53	35.3
Upper Blue	41	78	61.7
Middle Blue	26	55	38.9
Lower Blue	26	48	33.1

upper Blue River reach as there were no areas with an index value greater than 80 in EIUSH II as there were in EIUSH I, and the percent of area with scores >60 declined from 82.1 in EIUSH I to 59.5 in EIUSH II (Table 4.2; Table 4.4). The EIUSH II index-area distribution for the lower Blue River and Indian Creek reaches were unchanged from the EIUSH I values, whereas the middle Blue River reach indicated a small (2.9 percent) increases in the area with index scores less than 40 (Tables 4.2, 4.4, and Figure 4.9). The mean index score of 33.1 and maximum score of 48 in the lower Blue River were similar in EIUSH I and II, as were the mean index scores (35.3, 38.9) and maximum values (53, 55) in Indian Creek and middle Blue River reaches, respectively (Table 4.3, 4.5).

The EIUSH, which is weighted to five environmental factors associated with stream health, provides a means of identifying areas that are most ecologically impaired within the

basin and the primary contributing categories to impairment. Conversely, those areas with EIUSH scores of 50 or greater, for example, could be identified as areas of greater ecological integrity that are susceptible to further impairment and warrant protection.

CHAPTER 5

DISCUSSION AND CONCLUSIONS

Evaluation of Streamside Ordinances

Urbanization results in major changes to the natural vegetation and land cover, which in turn, can directly or indirectly affect each of the five major environmental factors (aquatic biological communities, hydrology, LULC, physical habitat, water quality) determining stream health. One of the means of protecting riparian vegetation and, therefore, aspects of all five factors, is by means of streamside ordinances. There is a paucity of current research on the effectiveness of streamside ordinances and differences in performance of ordinances implemented by multiple jurisdictions on the same stream system. The results of this study provide additional information that can be used to evaluate the streamside ordinances and management of riparian resources along the Blue River in Missouri and Kansas.

Total Riparian Vegetation Change within the Study Area

The results indicate that streamside ordinances did not prevent the loss of riparian vegetation, or the buildup of developed land in the study area. In certain extents, tree loss increased following the implementation of ordinances. This finding is similar to the results of Yeakley et al (2006) and Ozawa and Yeakley (2007), which indicated continued stream buffer loss in selected urban areas in the Portland, Oregon, metropolitan area resulting from development, regardless of the level of regulatory protection.

The temporal trends in the trees and developed land cover classes in this study are similar to those determined in riparian areas on a national scale by Jones et al (2010), which indicated a 0.2% loss in riparian forest cover and a 0.3% increase in the human-use land cover between the early 1990s and the early 2000s. This study estimated a 9.4% loss in riparian

forest cover and a 2.4% increase in developed lands between 1992 and 2003 for the study extents. Annualized rates of tree loss in this study were greater than the loss rates determined in the Yeakley et al (2006) and Ozawa and Yeakley (2007) studies, which presented a rate of vegetative cover loss of 1.5% in the 7.5-m buffer study area and a 9% rate of loss in the vegetative cover at the citywide level from 1990 to 2002. The Blue River study area, however, was much more limited in size compared to the study area in Yeakley et al (2006) and Ozawa and Yeakley (2007). The analysis of the rate of change before and after streamside ordinance implementation in this study, as well as looking at areas with and without streamside ordinances, expands on the previous knowledge by incorporating the before and after consideration of change.

The streamside ordinances implemented in the study area included exemptions to allow for alterations to the land cover, in the case of flood abatement projects through city council approval (**88-415-08-B)**, or mitigation offsets **(88-415-07-C)** by landowners. This is one possible explanation for the continued loss in riparian vegetation following the implementation of streamside ordinances. Urbanization and the loss of riparian vegetation is associated with an increase in impervious cover, which is one of the primary factors in urban areas for increases in the magnitude and frequency of flooding (Leopold, 1968; Wibben, 1976; Shuster, Bonta, Thurston, Wamemuende, & Smith, 2005; Jacobson, 2011). Stage records from the USGS streamgage at the Blue River near Stanley, Kansas, (USGS, 2017b), in the upstream part of the Blue River Basin, indicated that the National Weather Service flood stage (National Weather Service, 2017) was exceeded in 1993, 1995, 2004, 2005, 2008, and 2010 within the study period. Frequent flooding is an impetus for flood abatement activities that may include the direct removal of high-roughness riparian tree cover or indirect activities that can result in

vegetation change including channel alterations (construction of levees and stream channelization).

Riparian Vegetation Change by Municipality and Ordinance Protection

In addition to differences in ordinance protection by municipality, there are several additional upstream-downstream gradients within the study area that affect the extent of implemented ordinances and the distribution and rate of change in riparian vegetation. These gradients include drainage area, history of city establishment, and local economic factors. The ecologically defined extent of the riparian zone, as determined by the aquatic-terrestrial transition, will increase proportionally to the drainage area and may exceed the width of ordinance protection. The ordinance-protected streamside width can be absolute or be proportional to drainage area up to an arbitrarily defined maximum. The city of Kansas City was established many decades prior to Overland Park, and ordinance protection within the city of Kansas City portion of the study area was established well after development and growth along the Blue River floodplain. In Overland Park, development is continuing within the basin during the post-ordinance period. The unincorporated Johnson County, Kansas, reach along the Blue River largely is transitioning from rural to suburban. This disparity in the timing and extent of urban development results in substantial differences in land values along the Blue River corridor, which in turn, can affect the cost of vegetation protection.

The city of Kansas City provides the greatest lateral width of streamside vegetation protection in the study area, but it still provided a non-continuous and patchwork application of the ordinance. The results indicated that a lower rate of land cover change, including vegetation loss, occurred in the ordinance-protected area compared to the non-ordinance area, however, the greatest rate of tree loss and increase in developed land occurred following the

implementation of the streamside ordinance in 2009-12. Another consideration in explaining the rate and timing of vegetation changes within the Kansas City ordinance zones is the Blue River Channel Modification Project (USACE, n.d.). The project was authorized by Congress in 1970 at a cost of over $300 million in Federal funds, with the goal of reducing the risk of flooding in the lower Blue River in Kansas City. Implementation extended from 1983 through 2016 and included three stages of construction covering 12.5 miles of the lower Blue River channel. Modifications to the channel included channelization, streambank armoring, and alterations to vegetation within the near-channel riparian areas. Flooding in 2010 along the lower Blue River led to increased efforts in the flood abatement project including bank stabilization projects that could account for a part of the increased rate of vegetation loss and increased rate of development from 2009 to 2012.

Unlike the city of Kansas City, which has an absolute buffer width, the city of Overland Park implemented an ordinance zone that was proportional to the drainage area of the stream. The maximum width, however, remained constant in streams with drainages over 5,000 acres leaving a substantial part of the study area extent outside of the ordinance area within the Overland Park reach. Real estate values also may be a factor in determining the extent of riparian protection along the Blue River riparian area. For instance, the median house value in Overland Park was over $100,000 greater than the median house value in Kansas City (USCB, 2010). This disparity in the value of protected lands also could account for differences in the extent of ordinance protection.

The unincorporated Johnson County, Kansas, does not have a streamside ordinance to prevent the alteration or removal of vegetation within the riparian corridor. Such a lack of protection facilitates the temporal and downstream-to-upstream spatial progression of riparian

vegetation alterations that have occurred in the basin. The greatest rate of tree loss occurred in this rural segment of the study area but the losses did not correspond to increases in developed land but rather to an increase in the grasses cover. This area was the only area within the study area that had a net increase in grasses cover. Possible explanations for the increase in grass cover may be from clearing of treed lands for recreational or flood reduction purposes. Ordinance protected areas within Kansas City (Zone 3) had greater increases in developed land cover than did the unincorporated Johnson County and this is likely the result of the pre-urbanized state of this reach.

The results demonstrated that a water body and associated riparian area such as the Blue River that spans many jurisdictional boundaries can have varying levels of protection resulting from inconsistencies in the content and implementation of streamside ordinances. In such instances, established ordinances can potentially limit further degradation of impaired water bodies but, afford little remediation to existing impairments or offer insubstantial effects on upstream sources of impairment (Deksissa, Ashton, & Vanrolleghem, 2003). The ordinance provisions should address the primary focal issues of the jurisdictional area whether it be erosion control, vegetation loss, or flood purposes. The implementation of streamside ordinances alone will not provide protection of riparian vegetation without consistent and unified effective oversight (USEPA, 2002). To be most effective, the ordinances need to be implemented with adequate spatial coverage, to contain provisions that address protection needs, and to be enforced to ensure that the policies are actually being used for intended governance.

The results from the streamside ordinance analysis showed that the ordinances were ineffective at stopping the rate of change to the LULC within the protected areas. The highest

rate of vegetation change within ordinance-protected areas in this study generally occurred following the implementation of streamside ordinances. The study period included an economic recession and corresponding decline in commercial and housing development, which could have influenced the results due to economic incentives and the timing of the market correction. The positive outcome was that the rate of change within the protected study areas showed a lower rate of change than the rate of change in areas that did not have ordinances implemented. Future studies could incorporate economic data at a suitable scale to determine possible drivers of the timing and extent of riparian vegetation loss.

EIUSH I and II

The EIUSH provides an indication of the ecological health of a river system and a higher overall score provided by the aggregation of the various layers, indicates a lower level of impairment and better health of the river system. The results of EIUSH I and II indicate that the ecological integrity in most of the study area is highly impaired. Overall, the Blue River within the study area is a highly impaired system as the mean basin index score was 45. The temporal change in impairment can be quantified through updating one, or preferably more, of the eight composite data layers. The updated EIUSH II, with three layers updated to show a 5-10 year passing of time, indicated that the overall change in ecological impairment using the revised land use, water quality, and hydrology layers was small but changes varied substantially by reach. The mean score decreased from 45 in EIUSH I to 43 units in EIUSH II (both in the high level of impairment category) and the overall range in EIUSH scores shifted toward greater impairment as the range of scores decreased from 25 to 82 in EIUSH I to 24 to 78 in EIUSH II.

The upstream part of the Blue River Basin showed the lowest levels of impairment as indicated in the Upper Blue River EIUSH scores, and the level of impairment increased in severity downstream to the Middle Blue River reach and the Lower Blue River reach, as both reaches maintained a severe level of impairment in the EIUSH I and EIUSH II models. The Indian Creek reach showed widespread impairment as well.

The overall comparison between EIUSH I and II indicated little change, however, the reach by reach scores indicate the change was not spatially consistent. The reach by reach break out of the scores of both EIUSH models indicated that the lower Blue River and Indian Creek reaches had the lowest index scores, while the Upper Blue River had the highest index score within the study area. The upper Blue River reach experienced the greatest overall change (increased impairment) in the mean score between EIUSH I and EIUSH II. Impairments from all five factors in the lower Blue reach contributed to the severely impaired rating based on the EIUSH I index score (Table 5.1). The water quality and hydrology factors largely contributed to the severe impairment rating for the Indian Creek reach, whereas the physical habitat factors were a positive influence on the index value for that reach. The high impairment rating of the middle Blue reach resulted from the physical and biological characteristic factors. The upper Blue reach EIUSH I index score is indicative of a moderate level of impairment, and was most negatively impacted from the *E. coli* layer and the low macroinvertebrate layer scores, whereas the scores from the other six layers were not substantial contributors to the impairment. The upper Blue reach level of impairment increased from EIUSH I to EIUSH II as a result of a 22-unit decline in the TWPE water quality layer score between EIUSH I and II indicating greater point sources of contaminants in this reach (Table 5.1, 5.2). This is significant because the results show that the change in impairment is

greatest in the areas previously identified in EIUSH I as having the least impairment in the study area. It also is significant because it indicates that the source of the additional impairment largely is from a point-source location rather than a systemic issue. This can also be used to improve riparian function by directing intervention towards the most significant parameters.

Table 5.1

EIUSH I summarized layer scores by category and reach

EIUSH I TABLE

Factors	Layers	Upper Blue River	Indian Creek	Middle Blue River	Lower Blue River
Biological Characteristics	*Macroinvertebrates*	*37.2*	*20.3*	*10.0*	*10.0*
Physical Characteristics	*Physical Habitat*	*72.6*	*61.2*	*57.5*	*49.0*
	Channelization	*79.3*	*65.7*	*64.1*	*20.0*
Water Hydrology	*Hydro-IHA*	*96.0*	*39.0*	*61.0*	*61.0*
Land Use Land Cover	*NLCD*	*56.3*	*35.3*	*43.0*	*37.0*
Water Quality	*Chloride*	*100*	*00.0*	*50.0*	*50.0*
	E.coli	*33.8*	*17.5*	*13.0*	*21.6*
	TWPE	*67.6*	*25.8*	*18.8*	*6.60*

Table 5.2

EIUSH II summarized layer scores by category and reach

EIUSH II TABLE

Factors	Layers	Upper Blue River	Indian Creek	Middle Blue River	Lower Blue River
Biological Characteristics	*Macroinvertebrates*	*37.2*	*20.3*	*10.0*	*10.0*
Physical Characteristics	*Physical Habitat*	*72.6*	*61.2*	*57.5*	*49.0*
	Channelization	*79.3*	*65.7*	*64.1*	*20.0*
Water Hydrology	*Hydro-IHA*	*87.0*	*36.0*	*54.0*	*54.0*
Land Use Land Cover	*NLCD*	*56.2*	*34.6*	*41.3*	*27.8*
Water Quality	*Chloride*	*100*	*0.00*	*50.0*	*50.0*
	E.coli	*33.8*	*17.5*	*13.0*	*21.6*
	TWPE	*44.8*	*25.9*	*25.0*	*20.8*

The benefits of EIUSH

The EIUSH model is a useful tool for conveying ecological impairment data to the public and to decision makers. The data that were included can be collected or obtained from existing monitoring by local agencies and/or community groups. It is adaptable in that the included layers can be updated to reflect changes in the system or alternative methods of assessing existing components. The index can be updated on a partial basis without having to update all layers, and it allows for the comparison of change over time. The multiple ecological categories represented in EIUSH provide an advantage over a single category index in that not

only the magnitude of impairment can be estimated, but the possible causes of the impairments also can be assessed. The quantitative values from multiple layers also limits subjectivity that can be associated with any single category assessment.

The EIUSH can be used to spatially identify low impairment areas that can be targeted for protection. Alternatively, EIUSH can be used to target areas of high impairment and identify the probable source of impairment. Targeting these areas of low or high impairment can provide an informed means of addressing issues of protection and remediation.

The limitations of EIUSH

The limitations of the EIUSH index are, in part, related to the complexity of the data requirements. It could prove difficult in the future to update all eight layers for all reaches within the study area, or to apply the index to other rivers which may not have the full datasets available to populate all eight layers. Additionally, some of the layers such as the aquatic biology and physical habitat layers, may require specialized training and expertise in order for users to obtain and work with the datasets.

Additional possible limitations of the index are that it may not include the correct layers for all study locations in order to fully assess impairment, and this could either overestimate or underestimate the true level of impairment. The initial weighting of the variables may not accurately quantify interactions amongst the variables and serve as a starting point for possible future modifications. Despite the possible limitations, the EIUSH was created using the rich data sets available for the Blue River and these layers were specifically selected to be indicators of impairment in urban settings. The intention was to provide a general assessment, and EIUSH is not intended to be an end point, but rather a starting point and benchmark for documenting and communicating ecological impairment in the Blue River Basin.

It is also important to understand for future studies that the EIUSH index is constrained by the design and inclusion of layers and their assigned weighting factors. Depending on the objectives of the index study, the temporal time period may not meet those objectives. The index needs to be carefully implemented with consideration as to what data layers are available and what the desired outcome is, whether to determine average conditions or "worst case" conditions for the study area.

Future studies can utilize the ecological index model to analyze other basins and to incorporate ecological parameters both spatially and temporally. The data layers used in the index can be modified or refined to further simplify data collected and index updates. For example, perhaps more readily attainable measures of specific conductance could be substituted for chloride concentrations in the index layer. If some layers have stabilized with time, then others may be added or substituted to be able to determine future temporal and spatial changes in index layers.

Use of Results in Basin Management

The change in riparian land use/land cover in the study area over the 20-year analysis period showed a steady decline in riparian vegetation and an increase in impervious surfaces through development. A basin-scale adaptive management process is one possible approach to reverse or slow this trend of vegetation loss and development of impervious surfaces by using the EIUSH results and the results of the effectiveness of streamside ordinances (Figure 5.1). One potential scenario used in the process of determining future outcomes is to extend the findings into the future without altering the current rate of change. A second potential scenario could be to incorporate the EIUSH and creating ordinances that are designed for adaptive management. At regularly defined intervals, the ordinances could be evaluated for their

effectiveness and then widespread and informed adjustments could be made in the management of EIUSH components including adjustments to streamside ordinances by the interagency decision team. True change in the levels of impairment could be proven attainable if such scenarios are monitored over time and adaptations in management implemented. The products presented in this study could be used to monitor and communicate spatial and temporal changes to indicate environmental improvements or declines. These are just some of the possible ways that adaptive management could prove to be useful for changing urban riparian land-use policies, and more specifically incorporating changes to streamside ordinances and management of the Blue River corridor.

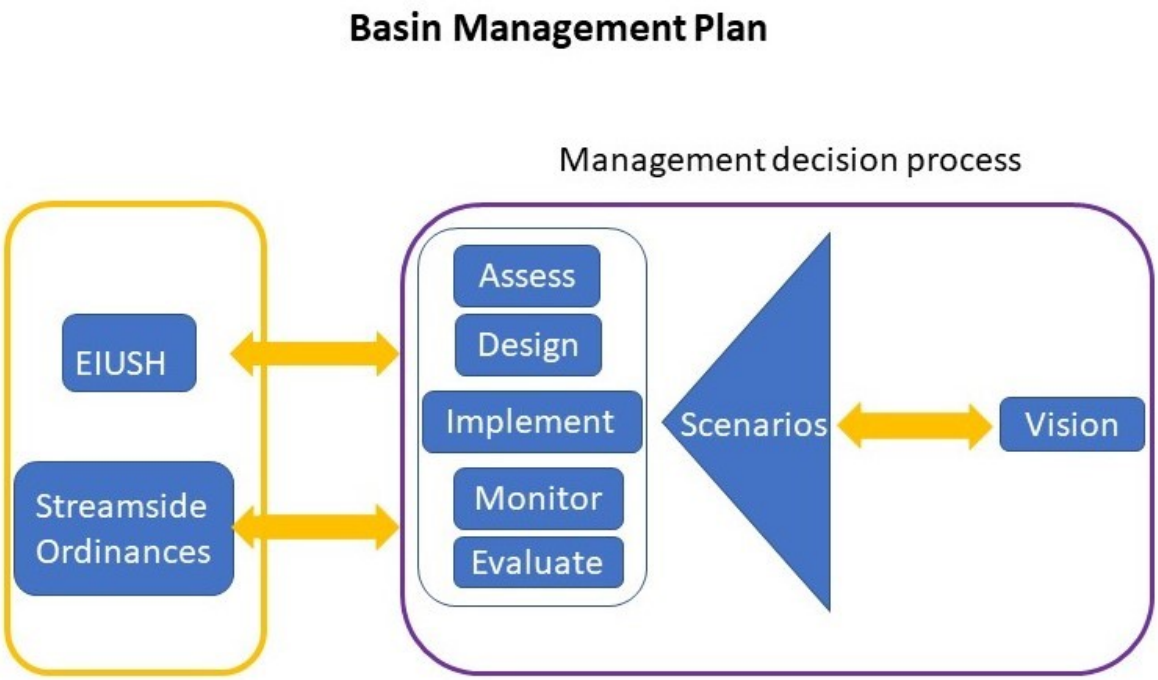


Figure 5.1. Use of study results in management decision process.

Conclusions

The rates of the land cover changes in the streamside ordinance study area were not consistent either temporally or spatially. The streamside ordinances did not result in a complete reduction in the rate of vegetation change and losses continued within the tree cover class and developed lands continued to increase following the implementation of ordinances. In fact, the rate of tree loss and gains in developed land classes increased following the implementation of ordinances, albeit at a lower rate compared to areas without ordinance protection.

The lack of a holistic basin management approach in the Blue River hinders the ability to effectively manage the ecological integrity of the river including the maintenance and

function of riparian vegetation in the basin despite that 42% of the area currently is under ordinance protection. As each jurisdiction allocates for its own impacts, careful consideration should factor into what is happening upstream and downstream to ensure that their management plans are appropriate and adaptable. The values in several EIUSH layers (hydrology, TWPE, chloride concentrations, E. coli densities) are cumulative from the upper Blue River to lower Blue River reaches so impairments in the middle and lower Blue River are caused, in part, by upstream contributions. The EIUSH results document upstream-downstream gradients in water quality (TWPE, chloride, *E. coli*), and hydrology layers. The increase in impairment of these factors from upstream to downstream sampling locations were related to an increase in developed areas, a loss in riparian vegetation, and increases in point and non-point sources of contaminants. The implementation and enforcement of consistent streamside ordinances and management goals along the entire longitudinal extent of a stream could afford a more effective means of protection or rehabilitation of such gradients. In contrast, a patchwork and inconsistent implementation of basin-wide goals, including streamside ordinances, likely will reduce the effectiveness of protection efforts.

An assessment of ecological index factors including the effectiveness of streamside ordinances should be a part of the river protection management. To ensure the success of such assessment, the approaches need to be designed and developed with proper spatial and temporal considerations. Geospatial techniques including remote sensing and GIS analysis prove to be useful in addressing the needs of spatial data and temporal analysis in such assessments.

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