

Implications of shared edge length between land cover types for landscape quality: the case of Midwestern US, 1940–1998

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Abstract The north-central region of Indiana in the Midwestern United States was covered by deciduous forest, but was largely cleared for agriculture during the 1800s. The landscape has experienced tremendous change due to forest restoration, urban expansion, and reservoir construction since the early 1900s. At the same time, ecological health and environmental quality have been dramatically degraded in the region. We used simple landscape indices, such as land proportion, TE, and Shared Edge Length (SEL) between any two classes, to examine changes in the spatial patterning of six land cover types, including agriculture, grassland, closed-canopy forest, open-canopy forest, urban, and water, using aerial photographs dating from 1940 to 1998. The landscape's domination by agriculture did not change (65% in 1940 and 57% in 1998), but there were net gains in area for closed-canopy forest (79%), urban (256%), and water (125%). Several landscape indices did not change much but SEL between closed-canopy forest and urban increased over seven fold, and

SEL between water and urban increased over eight fold from 1940 to 1998. More forestlands and water bodies were exposed to human activities. The clumped pattern of forest, water, and urban in a landscape can be ecologically detrimental and should be considered in future land-use decisions.

Keywords Land use · Urban sprawl · Reforestation · Landscape edges · Water quality · GIS · Remote sensing

Introduction

Hardwood forests once covered most the eastern third of the US (Opie 1998). During the east-west migration of European colonists, forests were rapidly cleared, resulting in forest fragmentation, with woodlots scattered across the landscape. Since the 1940s, industrialization and urbanization have increased, agricultural fields have been abandoned and hence some recovery of native forests has occurred (Parker and Merritt 1994; Rudel 1998). Active reforestation and afforestation also have occurred in many places of the country (Simpson et al. 1994; Medley et al. 1995; Brown 2003). The abundance of forest cover in the region is now accounted for almost exclusively by biophysical attributes of the land and landowner socioeconomic characteristics (Frimpong et al. 2006). Although the remaining forests are small in area, they are ecologically significant because they function as

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environmental filters, wildlife habitat, and leisure sites for humans (Ross-Davis et al. 2005). These forests play a critical role in supplying economic, ecological, and quality of life benefits (Gobster and Rickenbach 2004) but are mostly privately owned and are not normally managed by forestry professionals (Butler and Leatherberry 2004).

North-central Indiana is a topographically homogeneous landform covered with contiguous agricultural land, highly fragmented woodland, and sparsely distributed residential areas. Past glaciations followed by water erosion left a landscape with scattered wet depressions, streams, and river valleys throughout the till plain. The Wabash River Basin is the largest river valley that crosses east to west in north-central Indiana (Rizkalla and Swihart 2006). Water pollution is the most serious environmental problem in north-central Indiana. *Escherichia coli*, polychlorinated biphenyls (PCBs), ammonia, lead, and cyanide levels exceed total maximum daily load standards in several stream segments throughout the basin (Indiana Department of Health 1997). Over 20 stream segments are unfit for recreational uses or do not support any aquatic life due to pollution.

Exurban growth has been the dominant rural land-use trend in the United States between 1950 and 2000 (Brown et al. 2005), and urbanization is important at local scales in degrading forest resources (Gobster et al. 2000). Urbanization and fragmentation are important factors in the distribution of exotic species, which threaten forest sustainability by altering ecological processes (McKinney 2006; Radeloff et al. 2005). For example, the invasion of woody juniper (*Juniperus virginiana*) is exacerbating the fragmentation process initiated by previous human activity, and represents a serious threat to the continued integrity and conservation of remaining southern Great Plains grasslands in the United States (Coppedge et al. 2001). Invasive species, such as ailanthus, autumn olive, and bush honeysuckle are dramatically altering a majority of Indiana's forest structure and composition (Woodall et al. 2004). In addition to invasives, urbanization and fragmentation can result in long-term shifts in the composition of wildlife communities. Altered habitat connectivity and capacity are related to reduced occurrences of those organisms sensitive to landscape change, including many native species of fish (Guenther and Spacie 2006), turtles (Rizkalla and Swihart 2006), insects

(Swihart and Verboom 2004), amphibians (Kolozsvary and Swihart 1999), and mammals (Moore and Swihart 2005, Swihart et al. 2003, 2006). Altered hydrology and geomorphology also have resulted in degradation of water quality and associated aquatic environments in urbanizing agricultural landscapes of Indiana (Doyle et al. 2000; Frimpong et al. 2005).

While populations in some of the most rural areas in Indiana have declined, the population of the entire state has nearly doubled during the past half century (<http://www.stats.indiana.edu/>). This suggests an urban migration and a consolidation of farmland in Indiana. Abandonment of low-yield farmland and pastures would seem likely in these areas, leading to an increase in forest. Such conversions of agriculture and grassland to forest in agriculture-dominated landscapes are predicted by the industrialization hypothesis (Otto 1983; Christensen 1989; Rudel 1998), which states that industrialization leads to a reduced rate of population growth and the pressure on farmers to cultivate all of their land, and consequently, some agricultural fields are allowed to return to forest.

Although the forest transition is a common phenomenon in the eastern United States, it is important to enhance our understanding of how the current landscapes developed, how natural vegetation responds to urbanization, and how to protect or manage unique ecosystems (Foster 1992; Donnelly and Marzluff 2004). Simpson et al. (1994) examined two human-dominated landscapes, including a till plain landscape and moraine landscape, in central Ohio, USA, and found different land conversion patterns. Jenkins and Parker (2000) found that fragmentation had greatly decreased and core forest area had greatly increased at the Charles C. Deam Wilderness in Hoosier National Forest, Indiana. Dolan (2004) statistically analyzed the relationships of tree species diversity and various landscape indices in north-central Indiana but found no correlations between them, indicating that the commonly used landscape indices cannot be used to explain the effects of landscape pattern on ecological process in the region.

Without studying the changes in landscape pattern over time, it is difficult to understand why ecosystem quality has been declining (Wu and Hobbs 2002). Our objective was to examine geospatial change in forest, urban, water and other land cover types in north-central Indiana between 1940 and 1998. We especially intended to study the spatial relationships

of the land cover types and the trend of their changes with proportion and edge landscape indices. We then wanted to interpret the computed landscape indices for better understanding the ecological impacts of landscape change and for relating ecological considerations with land use issues in the region.

Methods

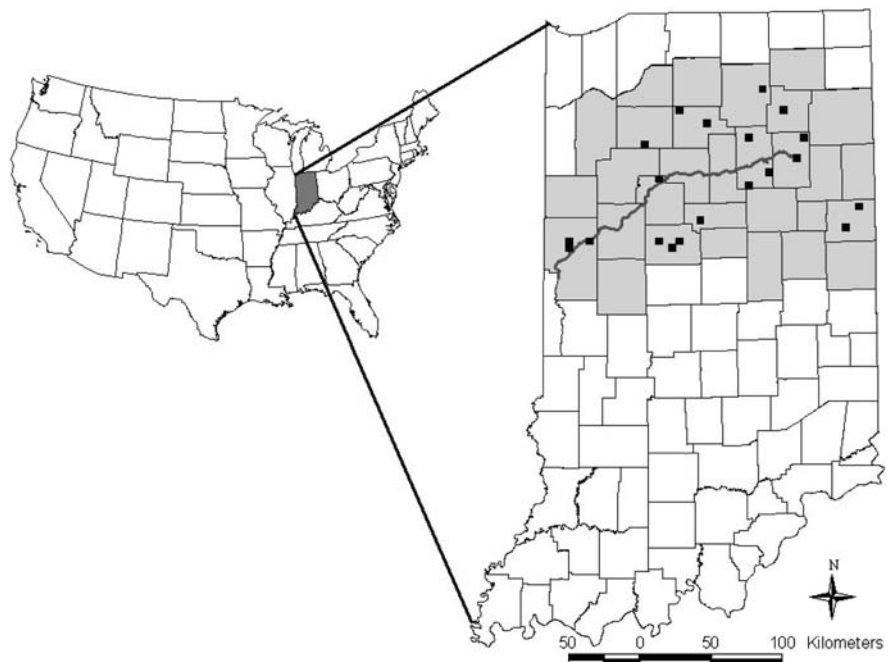
Data

The Wabash River Basin in north-central Indiana was divided into 783 4.83×4.83 km (3×3 mile) cells, and 23 cells were randomly selected based on a prior analysis of broader-scale landscape pattern (Moore and Swihart 2005; Rizkalla and Swihart 2006). Such a sample size could represent the overall landscape features in the region. The earliest time for which aerial photographs existed for the study sites was around 1940. The time span for this “1940” data set actually ranged from August 13, 1938 to July 30, 1941. Aerial photos were obtained from the Purdue University Earth and Atmospheric Sciences Library, as well as from the Soil and Water Conservation Districts in the region. These photographs were mostly at a 1:20,000 scale. A total of 300 aerial

photographs were obtained for 20 sites (Fig. 1), with three sites having no historic aerial coverage. These were then scanned and rectified by referring to the USGS 1-meter-resolution Digital Orthophoto Quadrangle (<http://www.edc.usgs.gov/products/aerial/doq.html>) acquired in 1998 (Lillesand and Kiefer 2000). The rectification process was performed with ERDAS Imagine 8.5/8.6 (<http://www.gis.leica-geosystems.com/>). The rectified images for each test site were then mosaiced together.

While the 1940 aerial photographs were used for generating the 1940 land use maps, the USGS Digital Orthophotos in 1998 were used for creating the 1998 land use maps. The classification system contains six land cover types: agriculture, grassland, closed-canopy forest (more than 80% canopy closure), open-canopy forest (between 20 and 80% canopy closure), urban, and water. Agriculture included all the cropland recognizable from aerial photographs. Soybean and corn are the major crop types. Grassland included permanent and temporary grasslands. These two land cover types were sometimes difficult to separate. The natural forest vegetation in the basin is beech-maple and oak-hickory forests (Omernik 1987). Open-canopy forest also included shrubby land. Urban included developed/built-up area such as commercial, industrial, and residential areas. Areas of sparse

Fig. 1 The location of 20 study sites in north-central Indiana, USA. The gray area indicates counties within the Wabash River watershed, dark-gray line within the area is the Wabash River



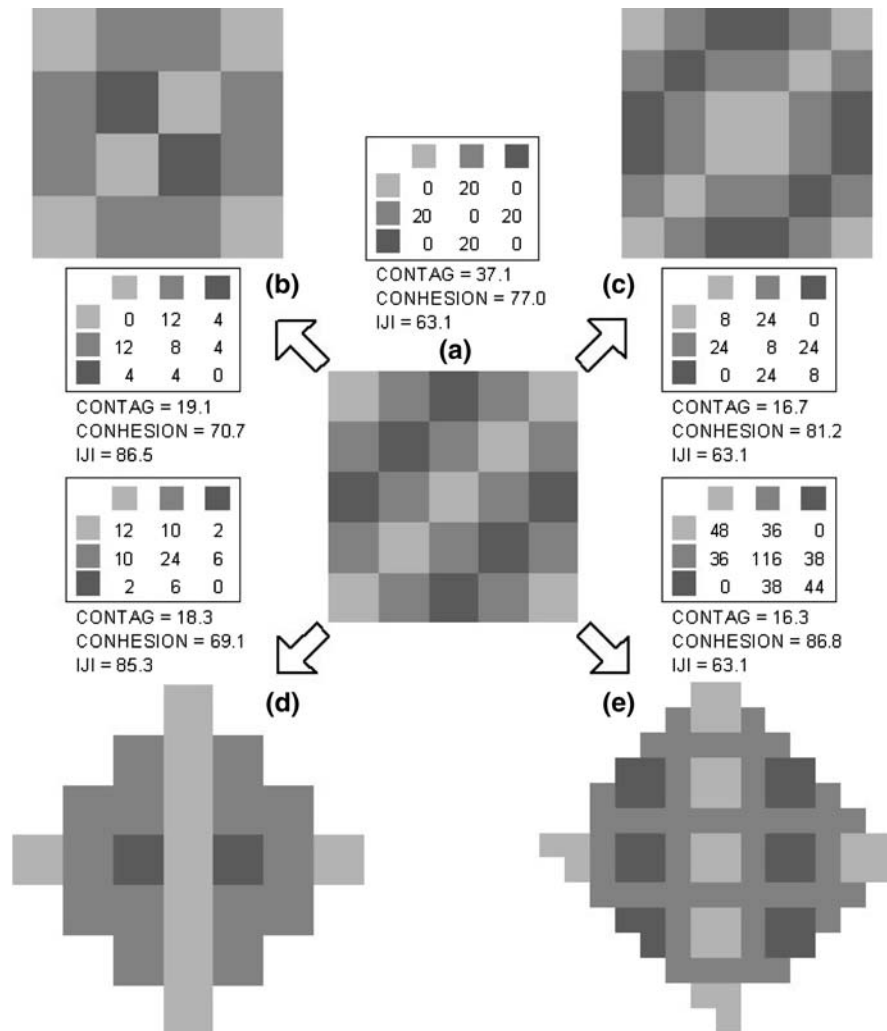
residential land use, such as farmsteads, were also included in Urban. Water included lakes, reservoirs, rivers, and creeks that were wide enough for digitizing as polygons.

Change analysis

Landscape quantifications can be made with a variety of landscape indices but the interpretation of the indices is not as easy as the proliferation of the indices might suggest (Gustafson 1998). Li and Wu (2004) suggest that simple indices measured directly from fields or maps are more likely to generate meaningful inferences. This study was intended to quantify changes in landscape pattern with vector land cover

data. However, most of spatial-relationship landscape indices, such as Contagion Index (CONTAG), Patch Cohesion Index (COHESION), and Connectance Index (CONNECT), are based on the adjacency matrix that can be generated only with raster data and our direct land cover maps after digitizing were in vector format. As illustrated by Fig. 2, changes in raster data structure can lead to different results of adjacency matrix and relevant landscape index. This study employed the proportions of area by land cover types, total edge length of each land cover type, and Shared Edge Length (SEL) between any two land cover types. The major advantage of SEL over other spatial landscape indices is that it can be computed with both vector and raster data and it contains land-cover-pair-explicit connectance information. The computations

Fig. 2 An illustration of different raster data structures and their effects on the adjacency matrix and selected spatial landscape indices: (a) is original data, (b) and (c) are re-sampled data from (a); (d) is a 45° rotation of (a); and (e) is a rotated and re-sampled data after (a)



of SEL are not affected by the exterior background. To calculate SEL with ArcInfo vector coverage, the polygon attribute table (PAT) and arc attribute table (AAT) need to be linked. When raster data is used, SEL can be derived from the adjacency matrix. To obtain supporting information from raster-data-based landscape indices, the original land cover data were converted into 2 and 5 m resolution raster data. Four selected landscape indices, including Total Edge (TE), CONTAG, Interspersion & Juxtaposition Index (IJI), and COHESION, which do not require user-specified distances, were computed with FRAGSTATS 3.3 (McGarigal et al. 2002).

A total of 36 from-to land cover transition classes were created by overlaying the 1940 and 1998 land cover maps. The total area of each transition class was summarized. For each land cover type, the net growth in area was computed by subtracting its total area in 1940 from that in 1998. The net growth can be positive (i.e., a gain in the area of the land cover type over the time period) or negative (i.e., a loss over the time period). The net growth in edge length was computed the same way as area. A comparison between net growth in area and net growth in edge length is helpful for depicting fragmentation status for each land cover type.

SEL was computed between every possible pair of land cover types (agriculture vs. grassland, agriculture vs. closed-canopy forest, etc.) for both the vector and raster datasets. We then compared SEL for each pair of land cover types between the two time periods to determine the change in edge, particularly SEL between forest and urban, between forest and water, and between urban and water. Data on land cover conversions and changes in SEL between any two

land cover types were combined to provide an overall picture of landscape change in space between 1940 and 1998. For land use planning purposes, the proportions of SEL for every combination of land cover types in 1998 were summarized.

Results

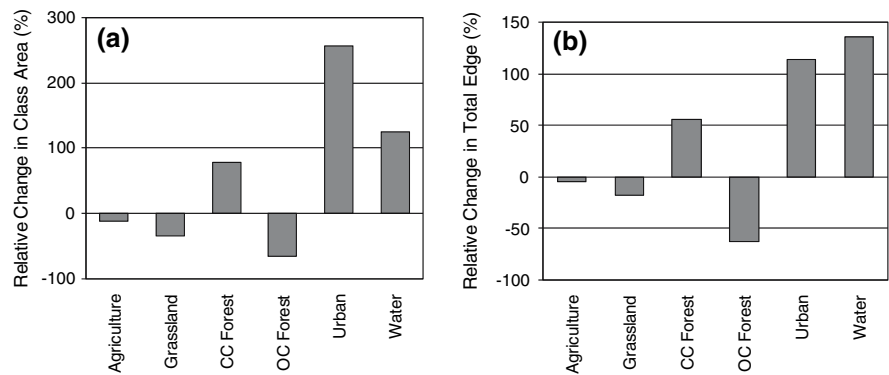
In both 1940 and 1998, agriculture was the dominant land cover type (65% in 1940 and 57% in 1998) in the study area (Table 1). The total area of agriculture was 4.2 times higher than the 2nd largest land cover type (grassland) in 1940 and 3.2 times higher than the 2nd largest land cover type (closed-canopy forest) in 1998. The least dominant land use types were water in 1940 and open-canopy forest in 1998.

Between 1940 and 1998, net losses occurred for agricultural land (−12%), grassland (−35%), and open-canopy forest (−66%). Net gains occurred for closed-canopy forest (79%), urban (256%), and water (125%) (Fig. 3a). The lost agricultural land became mainly grassland, urban, and closed-canopy forest; the lost grassland became agricultural land and closed-canopy forest; and the lost open-canopy forest became closed-canopy forest. Gains in closed-canopy forest came mainly from agricultural land, grassland, and open-canopy forest; gains in urban came from agriculture and grassland; and gains in water came from agriculture land, closed-canopy forest, and grassland (Table 1). As expected, urban and water underwent the lowest conversions to other land cover types, whereas grassland and open-canopy forest experienced the highest conversions to other land cover types (Table 1). In 1940, 67% of the forest was

Table 1 A matrix table of land use change (in km²) between 1940 and 1998 for the study area

		1940						
		Agriculture	Grassland	CC Forest	OC Forest	Urban	Water	Total
	Agriculture	225.4	31.1	4.4	4.2	1.4	0.2	266.7
1	Grassland	30.7	11.6	1.5	2.3	0.7	0.1	46.8
9	CC Forest	18.9	16.3	33.9	13.1	0.7	0.5	83.5
9	OC Forest	3.1	2.7	1.3	1.1	0.1	0.0	8.3
8	Urban	22.2	7.5	2.7	2.7	9.2	0.1	44.3
	Water	4.3	2.4	2.9	0.9	0.4	7.1	17.9
	Total	304.5	71.5	46.8	24.3	12.5	8.0	467.6

Fig. 3 A comparison of changes in area (a) and edge length (b) during 1940–1998 between land cover types (based on the vector land cover data)



closed canopy; in 1998, closed-canopy forest comprised 91% of the total forest (Table 1).

The three land cover types with net losses in area lost edge while the three land cover types with net gains in area gained edge (Fig. 3). The relative gains in edge length of the closed-canopy forest and urban were 56% and 114%, much smaller than those (79% and 256%) in area. The TE of the landscape was 2,666 km in 1940 and 2,947 km in 1998, increased by 11% between the two times (Table 2, Fig. 4). However, the increase in SEL between urban and water was 812% (Figs. 4, 5a) and that between closed-canopy forest and urban was 775% (Figs. 4, 5b and c), which were much greater than the maximum increase in total edge length for any single land cover type (Fig. 3). SEL between water and grassland, and between urban and grassland were more than doubled from 1940 to 1998. The negative growth in SEL took place mainly between

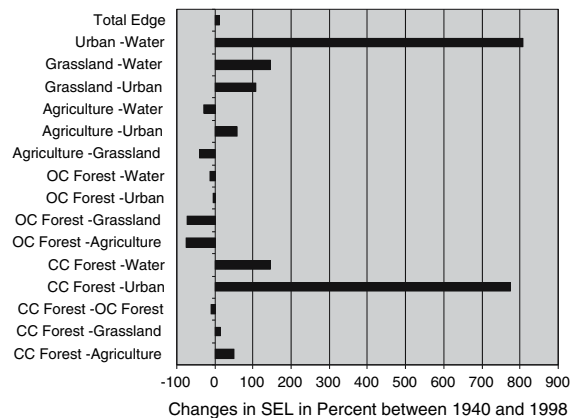


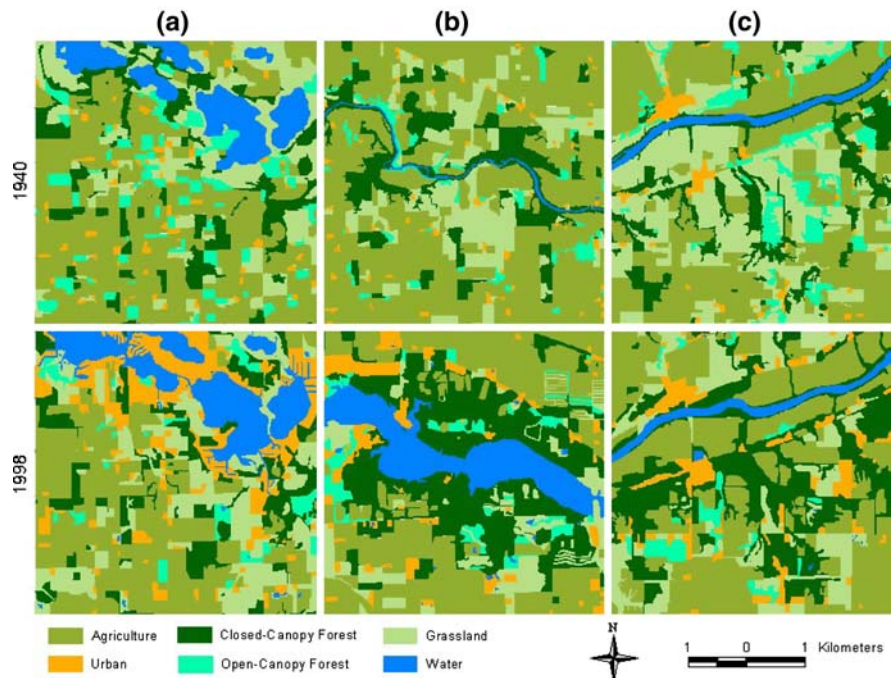
Fig. 4 Changes in SEL between land cover types for the study area from 1949 to 1998

open-canopy forest and agriculture, between open-canopy forest and grassland, between agriculture and grassland, and between agriculture and water.

Table 2 Shared Edge Length (km) between any two land cover types and TE (km) in brackets for each land cover type in 1940 and 1998 for the study area

	Agriculture	Grassland	CC Forest	OC Forest	Urban	Water
1940						
Agriculture	(1815)	611.6	564.6	342.9	270.2	25.3
Grassland		(1227)	312.6	191.7	86.2	24.8
CC Forest			(1065)	86.4	21.2	79.9
OC Forest				(662)	28.5	12.9
Urban					(414)	7.4
Water						(150)
1998						
Agriculture	(1733)	362.5	844.8	80.8	426.8	18.0
Grassland		(1011)	355.3	54.0	178.5	61.1
CC Forest			(1659)	77.1	185.5	196.4
OC Forest				(250)	27.4	11.1
Urban					(886)	67.5
Water						(354)

Fig. 5 Landscape changes between 1940 and 1998 for three of 20 study sites at 4.83×4.83 km: (a) urban development around lakes, (b) reforestation and urbanization around a reservoir, and (c) forest expansion and urban development along creeks but forest along Wabash River was largely unchanged



In 1940, the top two prevalent SEL occurred between agriculture and grassland (611.6 km) and between agriculture and closed-canopy forest (564.6 km); in 1998, the top two prevalent SEL took place between agriculture and closed-canopy forest (844.4 km) and between agriculture and urban (426.8 km) (Table 2). In other words, agriculture edges were shared mainly with grassland and closed-canopy forest in 1940 but shared mainly with closed-canopy forest and urban in 1998. All the land cover types except water were framed mainly by agriculture in both 1940 and 1998. Water was neighbored mainly with closed-canopy forest.

In north-central Indiana, the net gains of closed-canopy forest, urban, and water came mainly from agriculture lands while agriculture and grassland were mutual providers or receivers (Fig. 6). The range expansions for forest, urban, and water resulted in increases in SEL between them. The large magnitude of the increase in edge suggests that the range expansions of the three land cover types happened in clumps and were becoming more geospatially adjointed.

When the raster data were used, TE was increased by 8–10% from 1940 to 1998 depending on spatial resolutions: the coarser the resolution, the greater the change (Table 3). The raster data overestimated TE

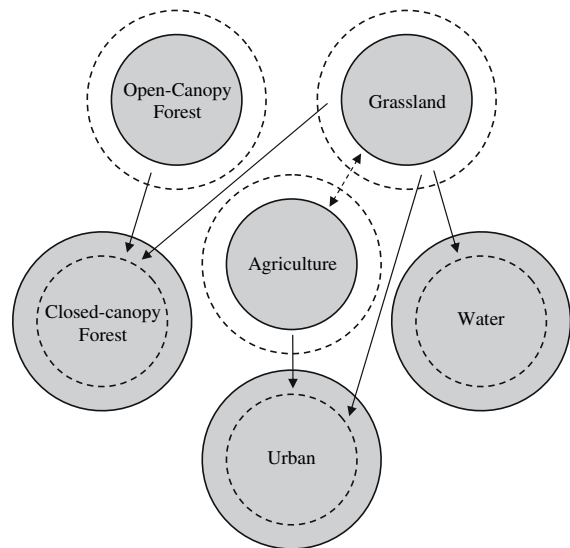


Fig. 6 A summary of major transitions and neighborhood relationships between six land cover types in the study area from 1940 to 1998. Agriculture, grassland, and open-canopy forest experienced net losses and closed-canopy forest, urban, and water experienced net gains within an agriculture-dominated landscape. The three gainers were getting spatially closer to each other

by up to 13% over the vector data in 1940 and 10% in 1998. CONTAG values were ranged from 64–66% in 1940 to 60–63% in 1998, depending on spatial

Table 3 A comparison of selected landscape indices between two times and between two spatial resolutions in the study area

	2 m Pixel size			5 m Pixel size		
	1940	1998	Change	1940	1998	Change
TE (km)	3,009	3,237	7.6%	2,893	3,172	9.6%
CONTAG (%)	66.1	62.7	-5.1%	64.0	60.1	-6.1%
IJI	69.8	71.1	1.9%	70.4	71.1	1.0%
COHESION	94.9	94.8	-0.1%	94.6	94.6	0.0%

Table 4 Changes in selected two class-level landscape indices derived from the 2 m resolution data between 1940 and 1998 in the study area

	IJI (%)			COHESION		
	1940	1998	Change	1940	1998	Change
Agriculture	78.4	68.1	-13.1%	94.9	94.9	0.0%
Grassland	64.7	73.5	13.6%	94.5	94.4	-0.1%
CC Forest	59.9	68.9	15.0%	94.5	94.7	0.2%
OC Forest	63.8	74.3	16.5%	94.1	94.1	0.0%
Urban	53.5	67.7	26.5%	93.6	94.3	0.7%
Water	44.6	53.2	19.3%	83.6	93.9	12.3%

resolutions. IJI and COHESION underwent minor changes between two times or between two spatial resolutions (Table 3). IJI values averaged 71 and COHESION values averaged 95. At the class level, IJI decreased for agriculture but increased for other five land cover types between 1940 and 1998 (Table 4). Only water experienced an increase in COHESION and the values of COHESION of other five land cover types did not change between 1940 and 1998.

Discussion

Conclusions about change in landscape spatial pattern differed among the indexes used. The slight increase in TE, slight decrease in CONTAG, and stable IJI and COHESION values at the landscape level indicated that fragmentation status of the landscape stayed nearly unchanged. The stable landscape structure conflicts with evidence of continued degradation of forest ecosystems in the region, many of which have been correlated with edge effects and proximity to urban areas or other human activity (Dolan 2004;

Woodall et al. 2004). Degradation has been documented in the form of reduced habitat quality and population viability for a variety of forest vertebrates (Duchamp 2006; Swihart et al. 2003), invasions into disturbed sites that could impede forest regeneration and reduce species diversity (Goheen and Swihart 2003; Lamprecht 2005; Swihart et al. 2007), increases in pathogenic parasites (Page et al. 2001), and altered interspecific interactions within and between trophic levels (Moore and Swihart 2005; Nupp and Swihart 2001; Govindan unpubl. data). In each of these examples, edge effects have been implicated as factors leading to degradation, consistent with other studies of edge effects on species viability (Gates and Gysel 1978), composition (Harper et al. 2005), interactions (Cronin 2003; Fagan et al. 1999; Sisk and Battin 2002), and invasions (Fraver 1994). Not surprisingly, these effects can be dependent on landscape context (Ewers and Didham 2006; Swihart et al. 2006).

At the class level, the consistent increases in IJI for all the land cover types except agriculture suggest that the five landscape cover types became more interspersed with other land cover types. Changes in SEL provided explicit information about the dynamics of class neighborhood relationships. The seven-fold increase in SEL between closed-canopy forest and urban likely has contributed to the degradation in forest ecosystems described above. On the other hand, closed-canopy forest was increased by 79% in area, supporting observations during the past decade by Parker and Merritt (1994) and Rudel (1998). Such mixed changes in the landscape structure and composition have to be explained with combined landscape indices.

Although agriculture was still the dominant land cover type in north-central Indiana, its total area decreased 12% between 1940 and 1998 (Table 1). About 30% of the loss in agriculture became forest. The conversions from agricultural land, grassland, and open-canopy forest to closed-canopy forest together promoted a 79% increase in its area between 1940 and 1998 and elevated closed-canopy forest to be the 2nd dominant land cover type in 1998. The TE of closed-canopy forest was increased by 56% but SEL between closed-canopy forest and water was more than doubled from 1940 to 1998. The increase in SEL between closed-canopy forest and water was resulted from the efforts in reforestation in

north-central Indiana. Such an expansion of forest range and increase of canopy closure are ecologically encouraging in the region.

The expansion of urban came mostly at the expense of agriculture and grassland (Table 1), but the greatest increases in urban edge were those adjoining water and closed-canopy forest (Fig. 4). Additionally, urban area was more than tripled (Fig. 2a) but its total edge length was more than doubled (Fig. 2b). The first phenomenon suggests that some of the urban development was made along forest edges, resulting in additional increases in forest-urban interface; the second phenomenon suggests that major urban sprawl took place in conjunction with the existing urban areas. Forests bordering urban areas are likely to be more stressed by human activities than forests bordering other land cover types (Edwards and Bliss 2003). The ecological value of increased forest area may be tempered by negative attributes associated with adjoining urban areas. Thus, the increase in forest area and canopy closure that we observed likely has not resulted in a corresponding increase in forest health and wildlife habitat because the additional forest is more exposed to human activities. Urban sprawl is also contributing to increasing forestland parcelization, making it difficult to implement ecologically and economically sound forest management (Butler and Leatherberry 2004). The majority of forest edges were shared with agriculture (51%) and grassland (21%) in 1998 (Fig. 4). These forestlands have developed a dense wall of bordering vegetation that blocks deeper invasion of exotic plants (Brothers and Spingarn 1992). However, the adjacent agricultural areas and grasslands have good potential for residential development. If this development occurs, the length of forest-urban edges will continue to increase.

Water management in north-central Indiana is another important issue. During the past decades, reservoirs have been built in north-central Indiana by the US Army Corps of Engineers, leading to substantial increases in water surface area. The creation of the Salamonie and Mississinewa reservoirs was the major cause for a 125% increase in water area in this study. The doubled area led to a 12% increase in COHESION, meaning that water bodies became more clumped. The reservoirs mainly flooded agriculture, grassland, and closed-canopy

forest (Table 1). However, water edges shared with agriculture were decreased and those shared with grassland and closed-canopy forest were more than doubled (Table 2, Fig. 5a). This demonstrated the progress of the Best Management Practice in the region, which is important for maintaining and improving water quality for fish (Frimpong et al. 2005). The sharp increase in water-urban edges was a negative indicator for the increased water bodies. The forest shared more than a half of total water edges (Table 2). If any of the forested water banks are developed, which is possible according to the past trend, there will be negative effects on both water bodies and forest ecosystems. This is an important concern for future land use practice in the region.

The overall effects of the changes in the landscape depend on the contributions of individual change elements. Most of the gain in forest has been along water edges (Fig. 5b). These riparian forests are important for predicting a fish index of biotic integrity (Frimpong et al. 2005). Because forestlands are highly fragmented and outer-core area ratio is relatively high for forest patches in Indiana (Jenkins and Parker 2000), the increase of forest edges and area may not be as ecologically beneficial as it seems (Donnelly and Marzluff 2004). Lamprecht (2005) suggested that native herbaceous species richness increased with woodland size and distance to woodland edge within woodlands in north-central Indiana. Urban expansion, on the other hand, has some obvious negative consequences. Kline et al. (2004) suggested that the increase in forest-urban interface threatens forests as productive socio-economic and ecological resources. The spread of invasive species is affected by landscape structure such as habitat edges (With 2002). By developing agriculture and grassland within/near forest and along water banks, the negative effects of urban expansion could be magnified. Land use planning in north-central Indiana needs to incorporate ecological constraints (Theobald et al. 2005). As development continues to increase (Alig et al. 2004), more attention is needed to managing forests with human interfaces for the purposes of habitat protection and wildlife conservation. More efforts are needed to expand forest cover along rivers and creeks (Fig. 5c), and sensitive forests and water banks should be strictly protected from development in north-central Indiana.

Conclusions

Using simple landscape indices was effective for exploring the insight of landscape change and its ecological implications. The proportion in area explains the composition of a landscape, changes in class and landscape TE reveal the trend of landscape fragmentation, and SEL provides insights on neighborhood relationships between land cover types. All these effective landscape indices can be easily derived from vector land cover data. In contrast, some raster-based, complicated landscape indices, such as CONTAG, JIJ and COHESION do not contain information about spatial relationships between specific land cover types. Their computations are affected by spatial resolution and their results are difficult to interpret.

Changes in each landscape index may be interpreted as positive or negative signals for ecological health and environmental quality in a landscape. Among the six land cover types, closed-canopy forest, urban, and water had net gains in area while agriculture, grassland, and open-canopy forest had net losses in area between 1940 and 1998. Urban had the largest gain in area, which was 256%, followed with water (125%) and closed-canopy forest (79%). These changes in proportion from 1940 to 1998 contained both positive and negative signals. An 11% increase in landscape TE meant a moderate trend toward fragmentation. A seven-fold increase in SEL between closed-canopy forest and an eight-fold increase in SEL between water and urban were a serious concern for landscape structure. The degradation of landscape structure depicted with SEL coincided with continued sharp declines in ecosystem integrity and environmental quality in north-central Indiana, suggesting that SEL is an effective landscape index for quantifying landscape pattern and explaining landscape quality.

Although forestland had increased by 78%, the landscape was still overwhelmingly dominated with agriculture in north-central Indiana. The limited forest ecosystems in the region will continue to function as invaluable environmental filters, wildlife habitat, and leisure sites for humans. The existing landscape structure provides potential for additional lands for urban sprawl near forest and water. If trends in land use remain the same, the clumped pattern of closed-canopy forest, urban, and water will be

expanded, and more ecological damage and environmental degradation are likely for forest and aquatic ecosystems. More importantly, this ecologically unbalanced landscape pattern directly threatens the sustainability of the environment and natural resources at a regional level (Wu 2006). Additional efforts are needed to expand forest cover and sensitive forests and water banks should be strictly protected from development in north-central Indiana.

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References

- Alig RJ, Kline JD, Lichtenstein M (2004) Urbanization on the US landscape: looking ahead in the 21st century. *Landsc Urban Plan* 69:219–234
- Brothers TS, Spingarn A (1992) Forest fragmentation and alien plant invasion of central Indiana old-growth forests. *Conserv Biol* 6:91–100
- Brown DG (2003) Land use and forest cover on private parcels in Upper Midwest, USA, 1970 to 1990. *Landsc Ecol* 18:777–790
- Brown DG, Johnson KM, Loveland TR, Theobald DM (2005) Rural land-use trends in the conterminous United States, 1950–2000. *Ecol Appl* 15:1851–1863
- Butler BJ, Leatherberry EC (2004) American's family forest owners. *J Forest* 102:4–9
- Christensen NL (1989) Landscape history and ecological change. *J Forest History* 33:116–124
- Coppedge BR, Engle DM, Fuhlendorf SD, Masters RE, Gregory MS (2001) Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. *Landsc Ecol* 16:677–690
- Cronin JT (2003) Patch structure, oviposition behavior, and the distribution of parasitism risk. *Ecol Monogr* 73:283–300
- Dolan BJ (2004) Ecosystem classification and succession of northeast Indiana. Purdue University, Thesis 50272 PhD, 172 pp
- Donnelly R, Marzluff JM (2004) Importance of reserve size and landscape context to urban bird conservation. *Conserv Biol* 18:73–745
- Doyle MW, Harbor JM, Rich CF, Spacie A (2000) Examining the effects of urbanization on streams using indicators of geomorphic stability. *Phys Geogr* 21:155–181
- Duchamp JE (2006) Modeling bat community structure and species distribution across fragmented landscapes within the upper Wabash River Basin. Purdue University thesis, PhD, 115 pp
- Edwards KK, Bliss JC (2003) Practicing forestry at the urban fringe. *J Forest* 101:6–11
- Ewers RM, Didham RK (2006) Confounding factors in the detection of species responses to habitat fragmentation. *Biol Rev* 81:117–142

- Fagan WF, Cantrell RS, Costner C (1999) How habitat edges change species interactions. *Am Nat* 153:165–182
- Foster DR (1992) Land-use history (1730–1990) and vegetation dynamics in central New England, USA. *J Ecol* 80:753–771
- Fraver S (1994) Vegetation responses along edge-to-interior gradients in the mixed hardwood forests of the Roanoke River basin, North Carolina. *Conserv Biol* 8:822–832
- Frimpong E, Sutton TM, Engel BA, Simon TP (2005) Spatial-scale effects on relative importance of physical habitat predictors of stream health. *Environ Manage* 36:899–917
- Frimpong EA, Ross-Davis AL, Lee JG, Broussard SR (2006) Biophysical and socioeconomic factors explaining the extent of forest cover on private ownerships in a Midwestern (USA) agrarian landscape. *Landsc Ecol* 21:763–776
- Gates JE, Gysel LW (1978) Avian nest dispersion and fledging success in field-forest ecotones. *Ecology* 59:871–883
- Gobster PH, Haight RG, Shriner D (2000) Landscape change in the Midwest: an integrated research and development program. *J Forest* 98:9–14
- Gobster PH, Rickenbach MG (2004) Private forestland parcelization and development in Wisconsin's Northwoods: perceptions of resource-oriented stakeholders. *Landsc Urban Plan* 69:165–182
- Goheen JR, Swihart RK (2003) Food hoarding behavior of gray squirrels and North American red squirrels in the central hardwoods: implications for forest regeneration. *Can J Zool* 81:1636–1639
- Guenther CB, Spacie A (2006) Changes in fish community structure upstream of impoundments within the upper Wabash River basin, Indiana. *Trans Am Fisher Soc* 135:570–583
- Gustafson EJ (1998) Quantifying landscape spatial pattern: What is the state of the art? *Ecosystems* 1:143–156
- Harper KA, MacDonald SE, Burton PJ, Chen J, Brososke KD, Saunders SC, Euskirchen ES, Roberts D, Jaiteh MS, Esseen P-A (2005) Edge influence on forest structure and composition in fragmented landscapes. *Conserv Biol* 19:768–782
- Jenkins MA, Parker GR (2000) Changes in the forest landscape of the Charles C. Deam wilderness, Southern Indiana, 1939–1990. *Natural Areas J* 20:46–55
- Kline JD, Azuma DL, Alig RJ (2004) Population growth, urban expansion, and private forestry in western Oregon. *Forest Sci* 50:33–43
- Kolozsvary MB, Swihart RK (1999) Habitat fragmentation and the distribution of amphibians: patch and landscape correlates in farmland. *Can J Zool* 77:1288–1299
- Lamprecht CJ (2005) Influence of site and landscape factors on the distribution of woodland herbaceous species within the upper Wabash basin of Indiana. Purdue University, Thesis 50499 MS, 137 pp
- Li HB, Wu JG (2004) Use and misuse of landscape indices. *Landsc Ecol* 19:389–399
- Lillesand TM, Kiefer RW (2000) Remote sensing and image interpretation. John Wiley & Sons, Inc., New York, 724 pp
- McGarigal K, Cushman SA, Neel MC, Ene E (2002) FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: www.umass.edu/landeco/research/fragstats/fragstats.html
- McKinney ML (2006) Urbanization as a major cause of biotic homogenization. *Biol Conserv* 127:247–260
- Medley KE, Okey BW, Barrett GW, Lucas MF, Renwick WH (1995) Landscape change with agricultural intensification in a rural watershed, southwestern Ohio, USA. *Landsc Ecol* 10:161–176
- Moore JE, Swihart RK (2005) Modeling site occupancy by forest rodents: incorporating detectability and spatial autocorrelation with hierarchically structured data. *J Wildlife Manage* 69:933–949
- Nupp TE, Swihart RK (2001) Assessing competition between forest rodents in a fragmented landscape of midwestern USA. *Mammalian Biol* 66:345–356
- Omerik JM (1987) Ecoregions of the conterminous United States. *Ann Assoc Am Geogr* 77:118–125
- Opie J (1998) *Nature's Nation: an environmental history of the United States*. Harcourt Brace and Company, Fort Worth, pp 15–17
- Otto JS (1983) The decline of forest farming in Southern Appalachia. *J Forest History* 27:18–27
- Page LK, Swihart RK, Kazacos KR (2001) Changes in transmission of *Baylisascaris procyonis* to intermediate hosts as a function of spatial scale. *Oikos* 93:213–220
- Parker GR, Merritt C (1994) The central region. In: Barrett JW (ed) *Regional silviculture of the United States*. John Wiley & Sons, New York, pp 129–172
- Radeloff VC, Hammer RB, Stewart SI, Fried JS, Holcomb SS, McKeefry JF (2005) The wildland-urban interface in the United States. *Ecol Appl* 15:799–805
- Rizkalla CE, Swihart RK (2006) Community structure and differential responses of aquatic turtles to agriculturally induced habitat fragmentation. *Landsc Ecol* 21:1361–1375
- Ross-Davis AL, Broussard SR, Jacobs DF, Davis AS (2005) Afforestation motivations of private landowners: an examination of hardwood tree plantings in Indiana. *Northern J Appl Forestry* 22:149–153
- Rudel TK (1998) Is there a forest transition? Deforestation, reforestation, and development. *Rural Sociol* 63:533–552
- Simpson JW, Boerner REJ, Demers MN, Berns LA, Artigas FJ, Silva A (1994) 48 Years of landscape change on 2 contiguous Ohio landscapes. *Landsc Ecol* 9:261–270
- Sisk TD, Battin J (2002) Habitat edges and avian ecology: geographic patterns and insights for western landscapes. *Studies Avian Biol* 25:30–48
- Swihart RK, Verboom J (2004) Assessing biodiversity consequences of land-use decisions: a role for ecologically scaled landscape indices. In: Swihart RK, Moore JE (eds) *Conserving biodiversity in agricultural landscapes: model-based planning tools*. Purdue University Press, West Lafayette, Indiana, pp. 81–101
- Swihart RK, Gehring TM, Kolozsvary MB, Nupp TE (2003) Responses of “resistant” vertebrates to habitat loss and fragmentation: the importance of niche breadth and range boundaries. *Diversity Distributions* 9:1–18
- Swihart RK, Lusk JJ, Duchamp JE, Rizkalla CM, Moore JE (2006) The roles of landscape context, niche breadth, and range boundaries in predicting species responses to habitat alteration. *Diversity Distributions* 12:277–287

- Swihart RK, Goheen JR, Schnelker SA, Rizkalla CE (2007) Testing the generality of patch and landscape-level predictors of tree squirrel occurrence at a regional scale. *J Mammal* 88:564–572
- Theobald DM, Spies T, Kline J, Maxwell B, Hobbs NT, Dale VH (2005) Ecological support for rural land-use planning. *Ecol Appl* 15:1906–1914
- With KA (2002) The landscape ecology of invasive spread. *Conserv Biol* 16:1192–1203
- Woodall C, Brand G, Vissage J, Gallion J (2004) Indiana's Forest Resources in 2003. North Central Research Station, United States Department of Agriculture Forest Service, Resource Bulletin NC-238, 37 pp
- Wu JG (2006) Landscape ecology, cross-disciplinarity, and sustainability science. *Landsc Ecol* 21:1–4
- Wu JG, Hobbs R (2002) Key issues and research priorities in landscape ecology: an idiosyncratic synthesis. *Landsc Ecol* 17:355–365