Riparian habitat changes across the continental United States (1972–2003) and potential implications for sustaining ecosystem services

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Abstract Riparian ecosystems are important elements in landscapes that often provide a disproportionately wide range of ecosystem services and conservation benefits. Their protection and restoration have been one of the top environmental management priorities across the US over the last several years. Despite the level of concern, visibility and management effort, little is known about trends in riparian habitats. Moreover, little is known about whether or not cumulative efforts to restore and protect riparian zones and floodplains are affecting the rates of riparian habitat change nationwide. To address these issues, we analyzed riparian land cover change between the early 1970s and the late 1990s/early 2000s using existing spatial data on hydrography and land cover. This included an analysis of land cover changes within 180 m riparian buffer zones, and at catchment scales, for 42,363 catchments across 63 ecoregions of the continental US. The total amount of forest and natural land cover (forests, shrublands, wetlands) in riparian buffers declined by 0.7 and 0.9%, respectively across the entire study period. Gains in grassland/shrubland accounted for the 0.2% lower percentage of total natural land cover loss relative to forests. Conversely, urban and developed land cover (urban, agriculture, and mechanically disturbed lands) increased by more than 1.3% within riparian buffers across the entire study period. Despite these changes, we documented an opposite trend of increasing proportions of natural and forest land cover in riparian buffers versus the catchment scale. We surmise that this trend might reflect a combination of natural recovery and cumulative efforts to protect riparian ecosystems across the US. However, existing models limit our ability to assess the impacts of these changes on specific ecosystem services. We discuss the implications of changes observed in this study on the sustainability of ecosystem services. We also recommend opportunties for future riparian change assessments.

Keywords Multi-scale change analysis · Land cover change · Remote sensing · Ecoregions · Riparian habitat change · Ecosystem services

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Introduction

Over the last decade, we have seen an increased awareness of the need to incorporate landscape perspectives and context in environmental sustainability planning (Musacchio 2009a, b; Termorshuizen and Opdam 2009; Wiens 2009; Wu 2010; Pearson and McAlpine this volume, in review). For example, Landscape Conservation Cooperatives, consisting of a number of agencies and stakeholders, are being established to promote and apply landscape perspectives and concepts into conservation in the US (USFWS 2010). Spatially explicit landscape designs that take into account the broader landscape context are seen as critical in mitigating and adapting to broad scale drivers associated with global change (Wiens 2009; Pearson and McAlpine this volume, in review; Verboom et al. this volume, in review). Moreover, maintaining and restoring key landscape elements at multiple spatial scales may be critical in sustaining a wide range of ecosystem services, even within developed landscapes (Grashof-Bokdam et al. 2009; Pearson and Gorman this volume, in review; Ryan et al. this volume, in review; Wu 2010).

Riparian buffers consisting of natural land cover have an ecological importance that often far exceeds their spatial extent in the landscape (Baker et al. 2006). Riparian buffers often serve as corridors for species migration and possess a disproportionately larger number of plant and animal species than adjacent areas, but especially along stream and river corridors that transcend arid, urban, and agricultural landscapes (Jones et al. 1985; Naiman et al. 1993; Knopf and Samson 1994; Spackman and Hughes 1995; Naiman and DeCamps 1997; Storey and Cowley 1997; Skagen et al. 1998; Woinarski et al. 2000; Lee et al. 2001; Groom and Grubb 2002; Boutin and Belanger 2003; Lees and Peres 2008). Intact riparian buffers also provide important ecological functions in catchments, including dissipation of energy associated with flooding events, storage of nutrients and sediments, and filtering of other nonpoint source pollution that would otherwise end up in streams (Swanson et al. 1982; Lowrance et al. 1984; Peterjohn and Correll 1984; Rhodes et al. 1985; Jones et al. 2001; Kiffney et al. 2003; Sweeney et al. 2004; Vidon and Hill 2004; Dwire and Lowrance 2006). Native vegetation cover along the edge of the riparian zones also helps reduce stream bank erosion (Likens and Bormann 1974; Swanson et al. 1982; Lowrance et al. 1984; Rhodes et al. 1985) and water temperature (Chen et al. 1998). Because of these benefits, riparian buffer protection and restoration may be key to off-setting some of the impacts of global climate change, especially in rapidly developing landscapes (Killeen and Solorzano 2008).

There is considerable concern over changes in the extent and condition of riparian ecosystems and the potential loss of ecosystem services (Swift 1984; Stromberg and Patten 1990; Armour et al. 1991; Gregory et al. 1991; Zube and Sheehan 1994; Stromberg et al. 1996; Todd and Elmore 1997; Nilsson and Berggren 2000; Sweeney et al. 2004; Charron et al. 2008). In the western US, there may be as little as two percent of the original forested riparian habitat left (Todd and Elmore 1997), resulting from construction of dams for flood control and water storage, pumping of surface and ground water from floodplains for agriculture and human consumption, and livestock grazing (Lytle and Merritt 2004). Intensive agricultural development in the central and southeastern US over the last 150 years has brought about extensive losses of forested riparian buffers (Jones et al. 1999), and urbanization threatens riparian ecosystems in many other regions of the US (Morgan et al. 2007). Replacement of native vegetation and natural land cover in floodplains not only reduces the ecological benefits of riparian buffers but also can contribute to pollutant loads to streams and rivers (Sweeney et al. 2004) and fragmentation (Spackman and Hughes 1995). Widespread concern of riparian ecosystem impairment and loss has led to a large number of riparian restoration projects across the US (Bernhardt et al. 2005), many of which have shown promise in restoring and protecting biological communities and ecosystem services (Kiffney et al. 2003; Sweeney et al. 2004; Gardali et al. 2006; Lennox et al. 2009). Less well documented is the degree to which restoration projects and changes in environmental policies have influenced changes in the extent and quality of riparian ecosystems at regional and national scales (Bernhardt et al. 2005). Moreover, we lack a systematic assessment of riparian ecosystem trends over broad geographic areas of the US (Bernhardt et al. 2005; Herring et al. 2006).

Using readily available spatial data on land cover trends, and stream, river, and catchment distributions, we analyzed broad-scale changes in riparian buffers between 1972 and 2003. We also compared the amount of land cover change within riparian buffers and at the catchment scale to determine if riparian habitats were changing at a different rate than catchments. A declining rate of natural land cover loss in the riparian zone, as compared to the watershed scale, might reflect a more recent emphasis on riparian protection and restoration. We discuss the implications of observed changes in riparian buffers on ecosystem services and environmental sustainability.

Methods

We analyzed landscape change at catchment and riparian scales (180 m buffers) for 42,363 watersheds distributed across the conterminous US in 63 different Omernik Level III Ecoregions (Omernik 1987, Fig. 1) by combining and comparing data from the USGS Land Cover Trends (LCT) project and the USGS National Hydrography Dataset (NHD) program. Ecoregions are defined as areas with spatial correlation in characteristics of geographical and biological phenomena associated with differences in the quality, health, and integrity of ecosystems (Omernik 2004). Ecoregions capture the essential factors of the physical landscape including the social, environmental and economic factors (Turner and Meyer 1991). Because of this variability, ecoregions may capture differences in dynamics and drivers of landscape change at relatively course scales (Loveland et al. 2002). The 63 ecoregions sampled in this study represent a wide range of biophysical and socio-ecological settings, ranging from mountainous areas of the Pacific Northwest to agricultural landscapes in the western and southern Great Plains to mixed landscapes of the Piedmont of the eastern United States.

The land cover data used in this project comes from the USGS Land Cover Trends (LCT) program which is a research project focused on understanding the rates, trends, causes, and consequences of contemporary land use and land cover change throughout the US (Loveland et al. 2002). Land use and land cover change is a constant environmental and societal phenomenon that modifies land cover characteristics which in turn affects a broad range of socio-economic, biological, geological, and hydrological systems and ecological processes. Understanding the causes and impacts of land use and land cover change on environmental systems requires a detailed understanding of the rates, patterns, and drivers of past, present, and future landscape change (Loveland et al. 1999; Drummond and Loveland 2010). The LCT land cover data used were based on randomly selected sample blocks within each of the 63 ecoregions (1840 10 km and 92 20 km sample blocks). Differences in

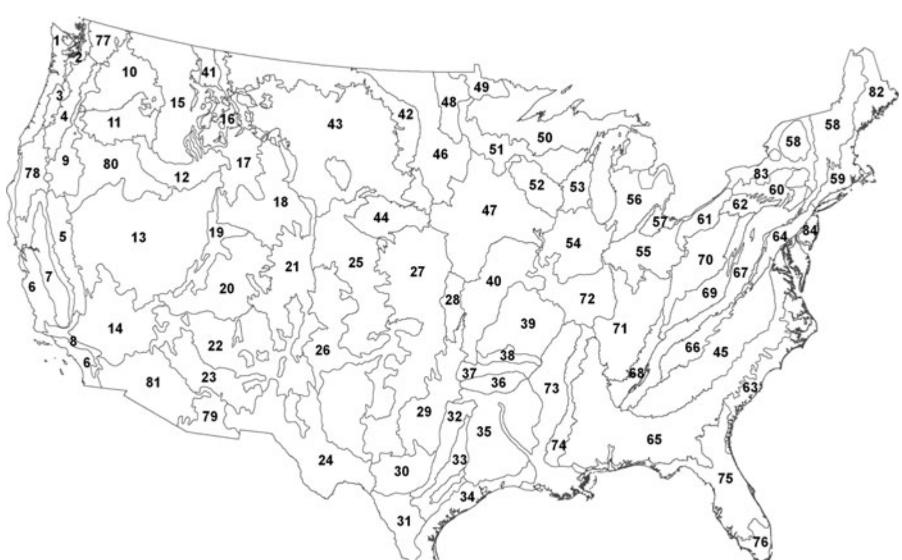


Fig. 1 Map of Level III Omernik Ecoregions for the lower 48 United States. Full descriptions of ecoregions can be downloaded from the US EPA (http://www.epa.gov/wed/pages/ecoregions/level_iii.htm)

sample block size resulted from an early adjustment to the sample design strategy (Griffith et al. 2003). Landsat Multispectral Scanner (MSS), Landsat Thematic Mapper (TM), Landsat Enhanced Thematic Mapper (ETM), and historical aerial photography were used to create 11-class digital land cover maps (similar to Anderson et al. 1976 Level I classifications, Table 1) for each sampling block for five temporal windows; 1972–1976, 1979–1983, 1985–1987, 1991–1993, and 1999–2002 with a data aggregation to a minimum mapping unit of 60 m. Manual interpretation/mapping methods have been shown to provide the highest quality and the most reliable method for deriving land-cover and land-cover change information (Loveland et al. 2002).

Data were prepared by geo-referencing the Landsat MSS, TM and ETM to a root-mean-square-error of one pixel or better and translating all data to an Albers Equal Area projection. Aerial photographs were also acquired for each data source and typically included photos from both the National Aerial Photography Program (NAPP) and the National High Altitude Program (NHAP). Because land cover change is often localized and spectrally ambiguous, land clover classes were manually interpreted and screen digitized. The 1992 National Land Cover Database (NLCD) (Vogelmann et al. 2001) was used as an initial baseline. After verifying and adjusting the 30 m NLCD data to a 60 m product, the land cover for early 1970s, late 1970s, mid 1980s, early 1990s, and late 1990s/early 2000s were compiled by back or forward analysis of the baseline land cover and land cover change for each sample block. Historical aerial photographs from roughly equivalent times were used as ancillary information sources to assist in the land cover analysis and mapping.

Table 1 Land cover classification used in the study (based on Loveland et al. 2002)

Land cover type/index	Type	Abbreviation
Natural index	Natural	Nat
Forests and woodlands	Natural	For
Grassland/shrubs	Natural	NG
Wetlands	Natural	Wet
Snow and ice	Natural	
Natural barren	Natural	
Non-mechanically disturbed	Natural	
Human use index	Anthropogenic	HUI
Developed (urban and built-up)	Anthropogenic	Urb
Agriculture (cropland and pasture)	Anthropogenic	AG
Mines and quarries	Anthropogenic	
Mechanically disturbed	Anthropogenic	Mech
Water bodies		

The human use index is the sum total of all anthropogenic land cover types and the natural index the sum total of all natural land cover types. Land cover types without abbreviations were not analyzed separately but were included in the human use and natural indices. Water bodies were excluded from any of the analyses

Validation was conducted by multiple peer-reviews of land cover analysts. An accuracy assessment of the Land Cover Trends data was conducted by creating a 1 km sub-grid within each sampling block and then stratifying the sub-blocks into classes of high, moderate and low classes of change. High-resolution aerial photographs (~1:40,000–1:80,000 scale) for the appropriate time periods were acquired and land cover data were created for twenty random samples of the sub-grid using manual methods. These results were compared against the classifications derived from the satellite data. Details of the classification and change detection approach can be found in Loveland et al. (2002) and Sohl et al. (2004).

Catchment boundaries and stream and river network data (1:24,000 scale) were downloaded from the National Hydrography Database websites (http://www.horizon-systems.com/nhdplus/ for catchments and ftp://nhdftp.usgs.gov/ for stream and river digital line data). NHD catchments represent catchment areas for individual stream segments across the US (total of 2.3 million across the conterminous US). We used only those catchments that intersected the LCT program sample blocks and then these catchments were used to clip stream and river network data (Fig. 2). Catchments used in this study ranged in size from 100 to 9994 ha.

Catchment- and riparian-scale (180 m buffers along streams and rivers) land cover composition was calculated for each of the five temporal windows (early 1970s, late 1970s, mid 1980s, early 1990s, late 1990s/early 2000s) in a Geographic Information System (GIS) using the Automated Tools Interface for Landscape Assessments (ATtILA, Ebert and Wade 2004). Riparian zones were delineated in ATtILA as 180 m wide buffer zones centered on the stream and river lines. This provided a buffer that

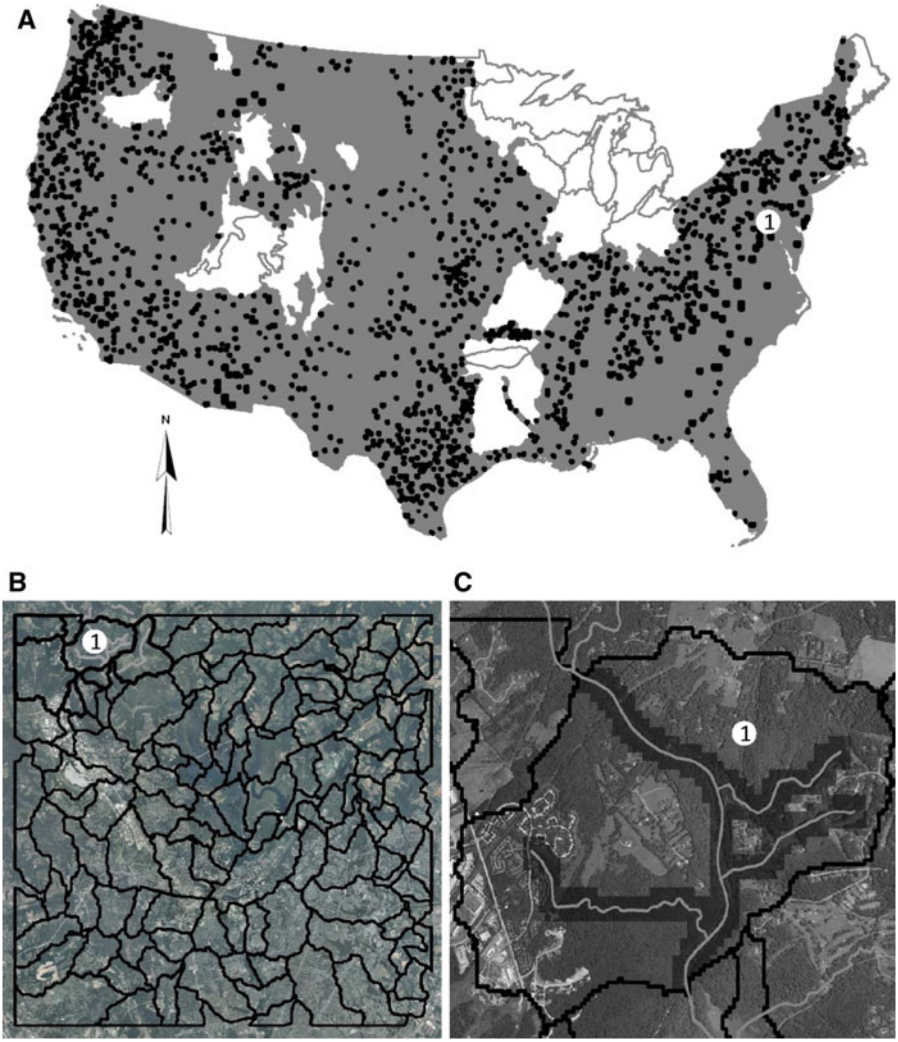


Fig. 2 Map illustrating extent and scales of catchments and riparian buffer analyses. **A** national samples, **B** example catchments at the 10 × 10 km land cover trends block scale, and **C** an individual catchment illustrating the 180 m wide riparian buffer zone along a stream reach. Small black dots in A denote locations of catchment samples, gray areas denote ecoregions that were sampled, and white areas denote ecoregions that were not sampled. The number 1 identifies the location of an example catchment at the three different scales

was three times the width of the minimum land cover resolution (60 m). This buffer size was selected as a compromise between reducing the impact of potential errors in the spatial registration of the stream and land cover data on the land cover metrics and providing a spatial representation of the riparian zone. ATtILA was used to calculate the proportion of each land cover type at the catchment and riparian buffer zone scales (Fig. 2). Land cover types analyzed in this study included forests, wetlands, grasslands/shrublands, urban, mechanically disturbed, and agriculture (Table 1). Mechanically disturbed lands are those in altered and often unvegetated states that, due to disturbances by mechanical means, are in transition from one cover type to another. These disturbances include forest clear-cutting, earthmoving, scraping, chaining, reservoir drawdown, and other similar human-induced changes. We also used ATtILA to calculate a natural index (all natural land cover types found in Table 1) and a human use or U Index (all anthropogenic land cover found in Table 1).

We analyzed land cover change at catchment and riparian buffer zone scales between four change periods: the early 1970s to the late 1970s, the late 1970s to the mid 1980s, the mid 1980s to the early 1990s, and the early 1990s to the late 1990s/early 2000s. We calculated differences between the percentage of land cover in the riparian zone and at the catchment scale for each catchment and evaluated changes among the different periods. This was calculated by subtracting the percentage of a particular land cover type at the catchment scale from the percentage in the riparian buffers. A positive number meant that there was a greater percentage of a particular land cover type in riparian buffers than at the catchment scale in which those riparian buffers were imbedded. A negative number meant that there was a higher percentage of a particular land cover type at the catchment scale. The aim of this analysis was to determine if there was greater or less loss of natural land cover in riparian buffers versus the amount at the catchment scale, and whether those differences changed over time. We analyzed changes across the entire dataset and by ecoregion and time. We used analysis of variance (Proc GLM SAS v9.2) to test the hypothesis that variance within ecoregions for natural land cover change was less than variance among ecoregions. This helped us evaluate ecoregions as strata to sample riparian land cover change, and as strata to evaluate changes in differences between riparian and catchment scale land cover percentages. A multiple comparison of means (Proc GLM with LSMEAN option; SAS v9.2) was used to evaluate significant differences in land cover changes between time periods. We used a stepwise regression (Proc Reg; SAS v9.2) to evaluate relationships among time periods for riparian land cover and differences between land cover proportions at the catchment versus riparian buffer zone scales.

Results

Riparian land cover change

The majority of catchments exhibited only small amounts of natural land cover change within riparian buffers during any of the time periods. There was relatively continuous loss of riparian natural and forest land cover over the entire study period (approximately 0.7 and 0.9%, respectively), although natural land cover loss slowed between the mid 1980s and the early 1990s (Fig. 3). Riparian forest loss slowed somewhat between the early and late 1990s (Fig. 3). Riparian grassland/shrubland and wetlands changed the least over the study period (Fig. 3). The riparian human use index showed an identical opposite change pattern to that of the natural index (Fig. 4). Riparian urban land cover exhibited the greatest amount of percent change over the study period (approximately 1.1%), with the largest amount of change occurring between the early and late 1990s (Fig. 4). Mechanically disturbed land increased significantly between the mid 1980s and early 1990s and then dropped slightly by the late 1990s (Fig. 4). Riparian agriculture decreased by approximately 0.6% (Fig. 4).

Changes in riparian land cover among years were relatively unrelated ($r^2 < 0.15$); as such, changes

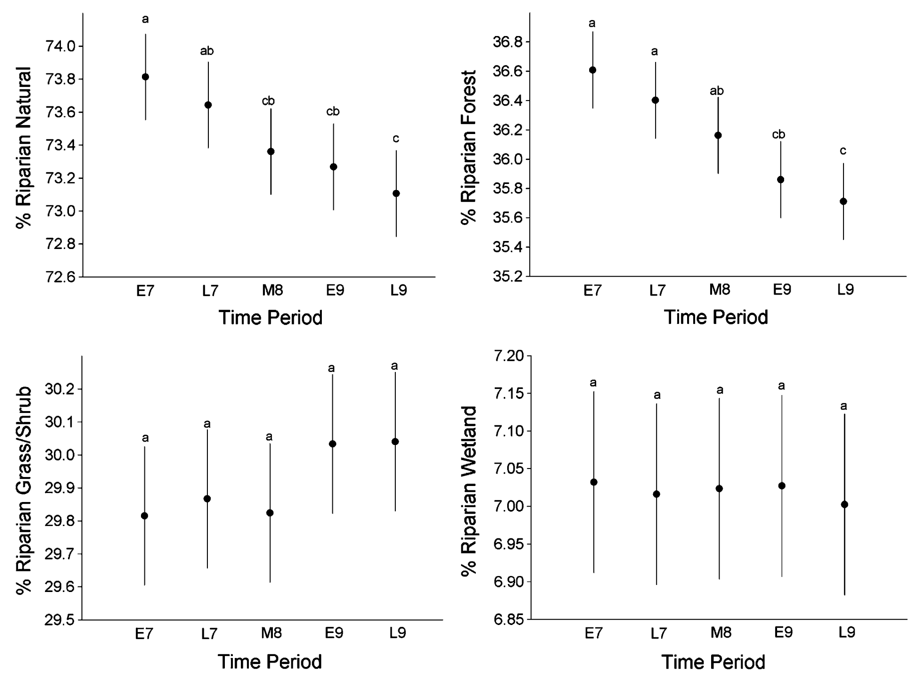


Fig. 3 Mean natural land cover composition in riparian buffers for each time period. Closed circles denote means, vertical lines denote ±1 standard error. E7 = early 1970s, L7 = late 1970s, M8 = mid 1980s, E9 = early 1990s, L9 = late 1990s/early 2000s. Vertical lines with the same letter indicate no significant difference ($P < 0.05$) based on analysis of variance-multiple means comparisons

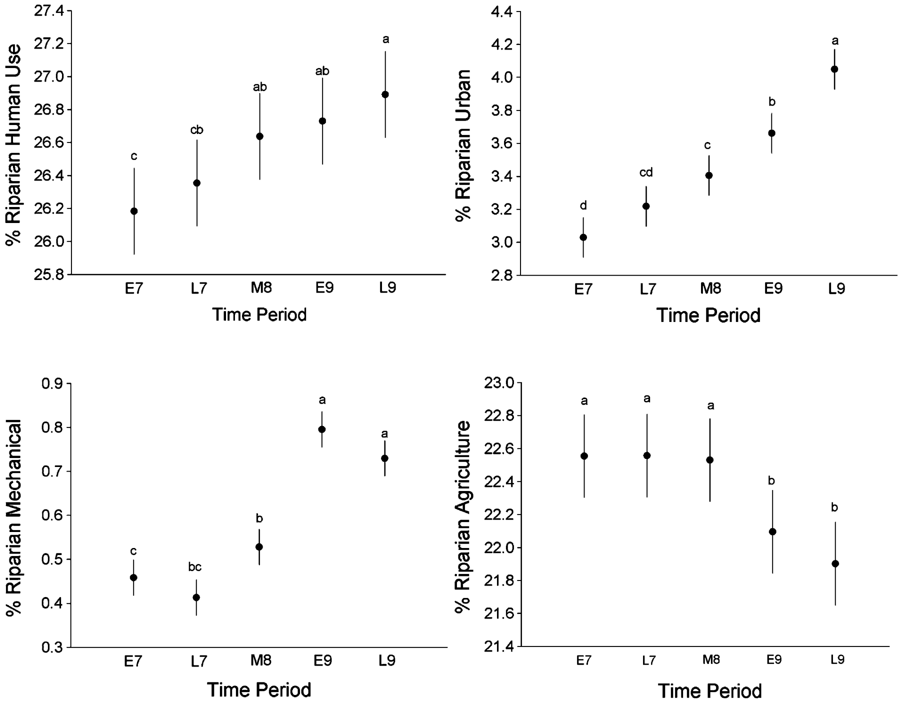


Fig. 4 Mean anthropogenic land cover composition in riparian buffers for each time period. Closed circles denote means, vertical lines denote ±1 standard error. E7 = early 1970s, L7 = late 1970s, M8 = mid 1980s, E9 = early 1990s, L9 = late 1990s/early 2000s. Vertical lines with the same letter indicate no significant difference ($P < 0.05$) based on analysis of variance-multiple means comparisons

from one time period were generally not reflected in subsequent time periods. Additionally, stepwise regression indicated that riparian land cover changes were unrelated to land cover changes at the catchment scale. As such, there were no good predictors of riparian land cover change.

Although there were differences among ecoregions in riparian land cover change, ecoregions did little to reduce the overall variance of land cover change across the U.S (based on the analysis of variance tests). Some ecoregions had greater amounts of change than others (Figs. 5, 6). The Northern Cascade Mountains (eco 77) and Klamath Mountains of Northern California (eco 78), both forested ecoregions, had the largest positive gain in natural land cover between the early and late 1970s, whereas the Central California Valley (eco 7), a large agricultural ecoregion, had the greatest loss of natural land cover (Figs. 5, 6). Change between the late 1970s and mid 1980s was mostly negative (Fig. 6). The Western Corn belt ecoregion (eco 47) had the greatest positive natural riparian land cover change (only 0.34%) whereas the Central California Valley and East Cascade Mountain and Foothill ecoregions (eco 7 and 9) lost the most (3.09 and 2.11, respectively, Figs. 5 and 6). The Western High Plains of Colorado (eco 25) and Texas and the Snake River Plain of Idaho (eco 12) had the largest gains in natural riparian land cover between the mid 1980s and the early 1990s (3.98 and 1.96%, respectively), and the Puget Sound Lowlands near the Seattle, Washington area (eco 2) and the Southern Texas Plains (eco 31) had the greatest losses (3.91 and 2.19%, respectively, Figs. 5 and 6). The Idaho Batholith ecoregion of west central Montana (eco 16) and the Northwestern Great Plains of Wyoming, eastern Montana, and western South and North Dakota (eco 43) had the largest positive changes between the early and late 1990s (1.38 and 1.39%, respectively, Figs. 5 and 6). The largest losses during this time period were in the Central California Valley (3.95%) and Western High Plains (2.95%, Figs. 5, 6). The Central California Valley was the only ecoregion to experience consistent declines (large changes in 3 of the 4 change periods) in riparian natural land cover (Figs. 5, 6).

Riparian versus catchment land cover change

The percentage of natural land cover in riparian buffers versus the catchment scale increased over the study period, although there were small changes between the late 1970s and early 1990s (Fig. 7).

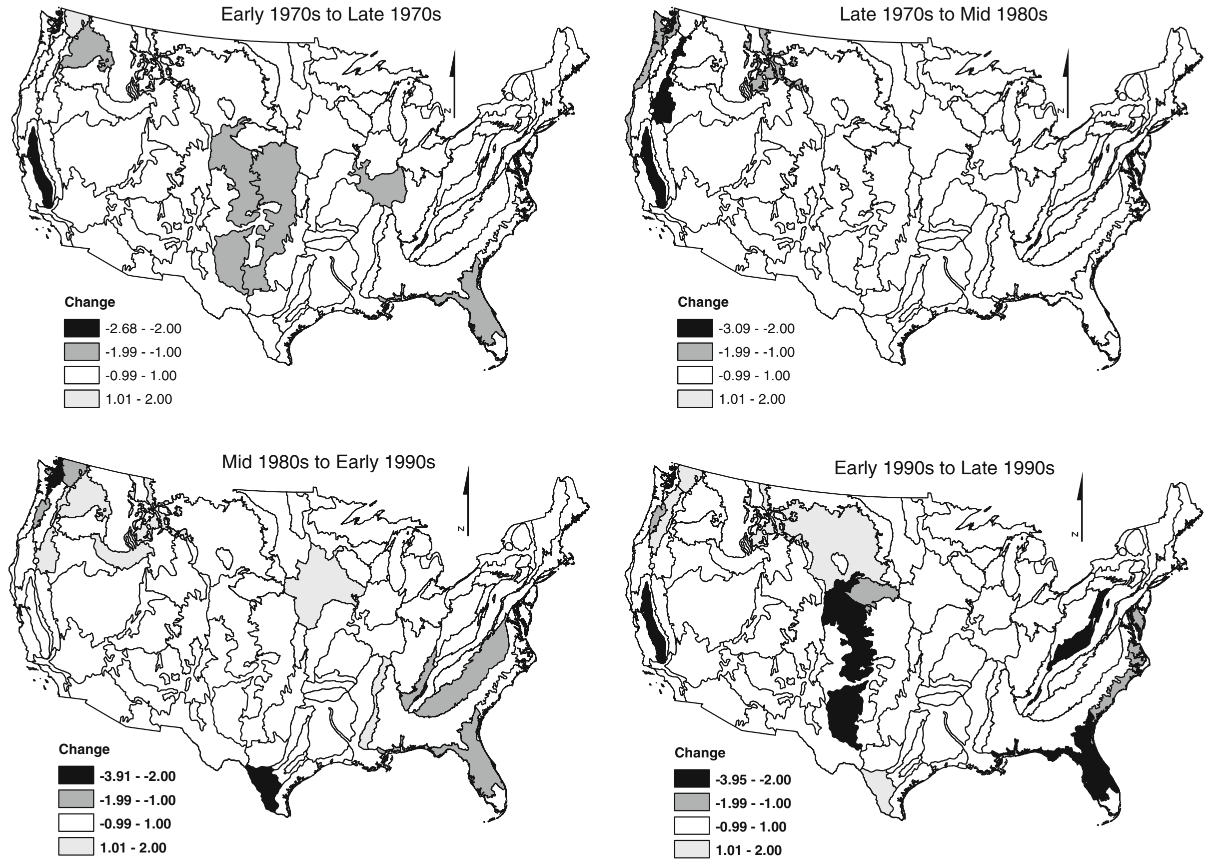
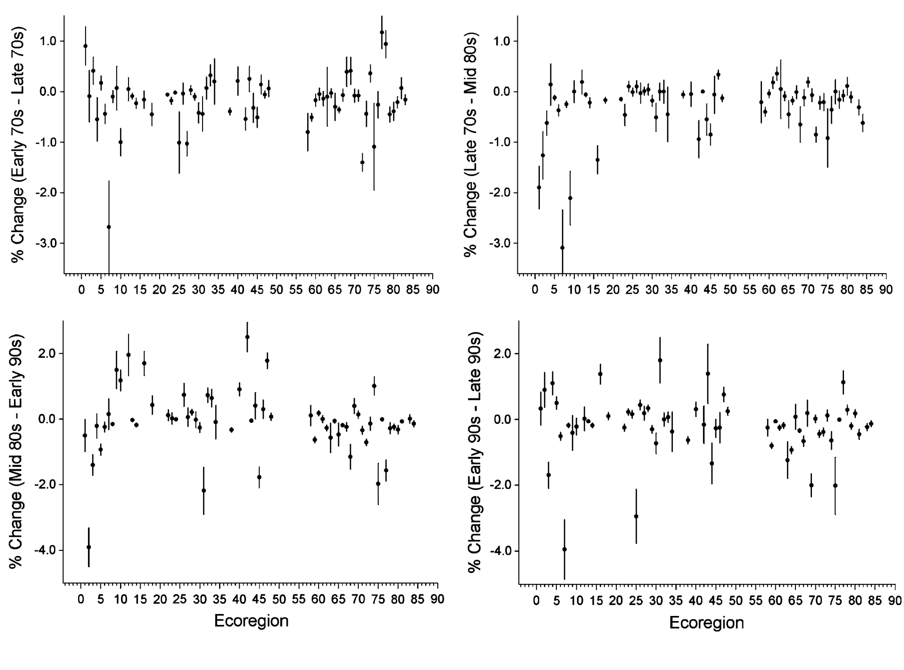


Fig. 5 Map illustrating ranges of mean riparian natural land cover change by ecoregion for each of the change periods

Fig. 6 Mean changes in riparian natural land cover by ecoregion for each of the change periods ±1 standard error



Similarly, forests increased across the study period (from 1.52 to 1.91%, Fig. 7). Grasslands/shrublands became more common at the catchment scale across the study period whereas wetlands remained largely unchanged (Fig. 7). Except for agricultural, the difference between anthropogenic land cover at the riparian versus catchment scale decreased over the study period (Fig. 8). The decrease in differences between agriculture in riparian buffers versus the catchment scale resulted from larger decreases at the catchment scale rather than increases in agriculture in riparian buffers. The percentage of agriculture in the riparian zone decreased during the study period (Fig. 4).

Similar to total natural riparian land cover changes, ecoregions provided little differentiation of changes between riparian buffers versus catchment scale riparian land cover percentages. Regression analyses yielded very small r^2 values (all less than 0.01) when considering ecoregions. However, a few ecoregions showed greater gains or losses in differences between riparian and catchment natural land cover, although these changes were not consistent across the four different change periods (Figs. 9, 10). Between the early and late 1970s, the Southern Coastal Plain of Florida (eco 75), which consists of several expanding urban areas, had an increase of 0.98%, whereas the Interior River Valleys of Southern Illinois (eco 72) had a 0.33% decrease (Figs. 9, 10). The Southern Coastal Plain of Florida (eco 75) was the only ecoregion to have an increase of greater than 1% between the late 1970s and mid 1980s (Figs. 9, 10). Only the East Cascade Mountain and Foothill ecoregion had a loss of more than 0.5%. The Mid-Atlantic Coastal Plain (eco 63) had the largest increase (0.85%) between the mid 1980s and early 1990s, and the Western High Plains of Colorado (eco 25) had the largest decrease (2.51%, Figs. 9, 10). Finally, during the last change period, the Western High Plains of Colorado (eco 25) had the greatest increases (2.25%) and the Northwestern Great Plains (eco 43) decreased the most (0.71%, Figs. 9, 10).

Changes in land cover percentages in the riparian buffers versus the catchment scale among years were unrelated (r^2 values were less than 0.10). Similarly, changes in riparian/catchment land cover percentages were unrelated to land cover composition or changes at the catchment scale.

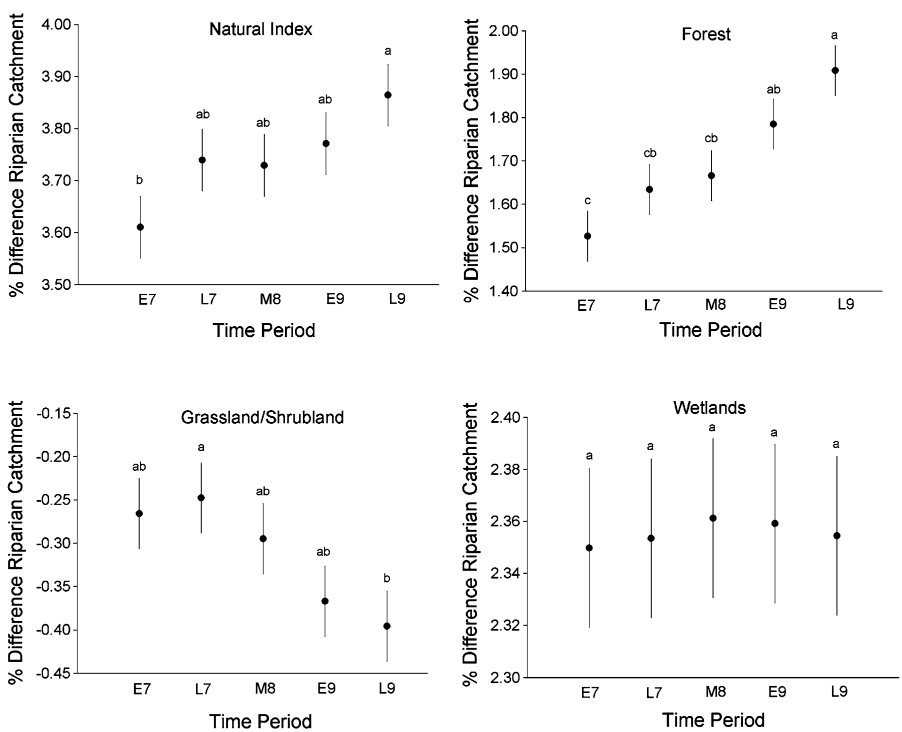


Fig. 7 Mean differences between percent composition of natural land cover types at the riparian buffer versus catchment scale. A positive number indicates a greater percentage of a specific land cover in the riparian buffer zone than at the catchment scale and a negative number indicates the reverse. Closed circles denote means; vertical lines denote ±1 standard error. Vertical lines with the same letter indicate no significant difference ($P < 0.05$) based on analysis of variance-multiple means comparisons. E7 = early 1970s, L7 = late 1970s, M8 = mid 1980s, E9 = early 1990s, L9 = late 1990s/early 2000s

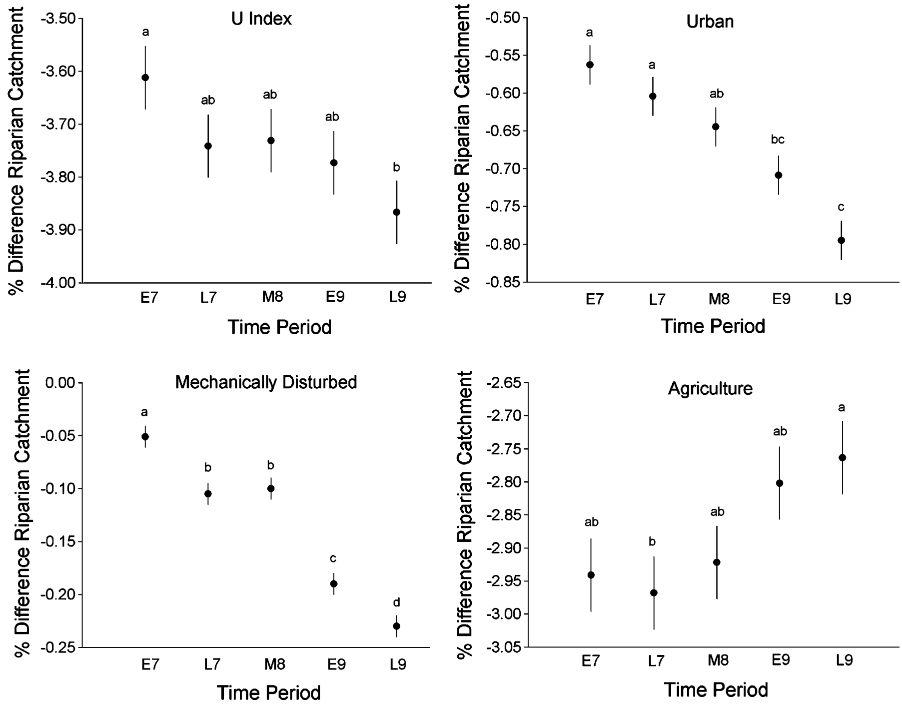


Fig. 8 Mean differences between percent composition of anthropogenic land cover types at the riparian buffer versus catchment scale. A positive number indicates a greater percentage of a specific land cover in the riparian buffer zone than at the catchment scale and a negative number indicates the reverse. Closed circles denote means; vertical lines denote ±1 standard error. Vertical lines with the same letter indicate no significant difference ($P < 0.05$) based on analysis of variance-multiple means comparisons. E7 = early 1970s, L7 = late 1970s, M8 = mid 1980s, E9 = early 1990s, L9 = late 1990s/early 2000s

Discussion

Our results suggest that riparian forests have declined slowly since the early 1970s, from an average of about 36.6 to 35.8%, while anthropogenic land cover in riparian buffers has increased (26.2 to 26.8%), primarily as a function of increasing urban and mechanically disturbed land cover. This mirrors concerns about national-scale urbanization (Theobald 2005; Theobald and Romme 2007) and forest loss and fragmentation (Wickham et al. 2008; Heinz 2008). Moreover, loss of agriculture from riparian zones between the mid 1980s and late 1990s (22.6 to 21.9%) reflects broader scale concerns of agricultural losses during this period of rapid urban and exurban expansion (Heinz 2008). However, the patterns of change were spatially and temporally variable. Some ecoregions exhibited higher percentages of losses or gains than other ecoregions during particular time periods, but only a few ecoregions exhibited repeated patterns of gains or losses (e.g., losses in the Central California Valley ecoregion). Urbanization is one of the primary causes of natural land cover loss, but urban land cover tends to occur in unevenly distributed clusters both within and among ecoregions (Theobald 2005; Theobald and Romme 2007). Therefore, as urban and exurban areas expand, the pattern of change tends to be clustered. Moreover, differences in local scale drivers of land development (e.g., zoning, parcelization, floodplain protection, etc.) may account for increased spatial and temporal variation in change and the lack of any strong predictors of change at national and regional scales (Riitters et al. 2006). Natural disturbances such as periodic flooding help maintain healthy riparian ecosystems and these disturbances can lead to episodic changes in land cover (Charron et al. 2008; Jones et al. 2008). For example Jones et al. (2008) showed that horizontal migration of the San Pedro River in response to flooding resulted in gains and losses of riparian vegetation. This process would result in periodic gains in forests in different parts of the river basin. Therefore, natural disturbances may also increase spatial and temporal variability of change. Finally, certain types of land use (e.g., livestock grazing, groundwater withdrawal, impoundments) can also contribute to spatial and temporal variation in riparian land cover change (Armour et al. 1991; Stromberg et al. 1996). In many of these cases (but especially in the western US), forested riparian zones are converted to grasslands and shrublands. In our study, we observed an increase in the grassland/

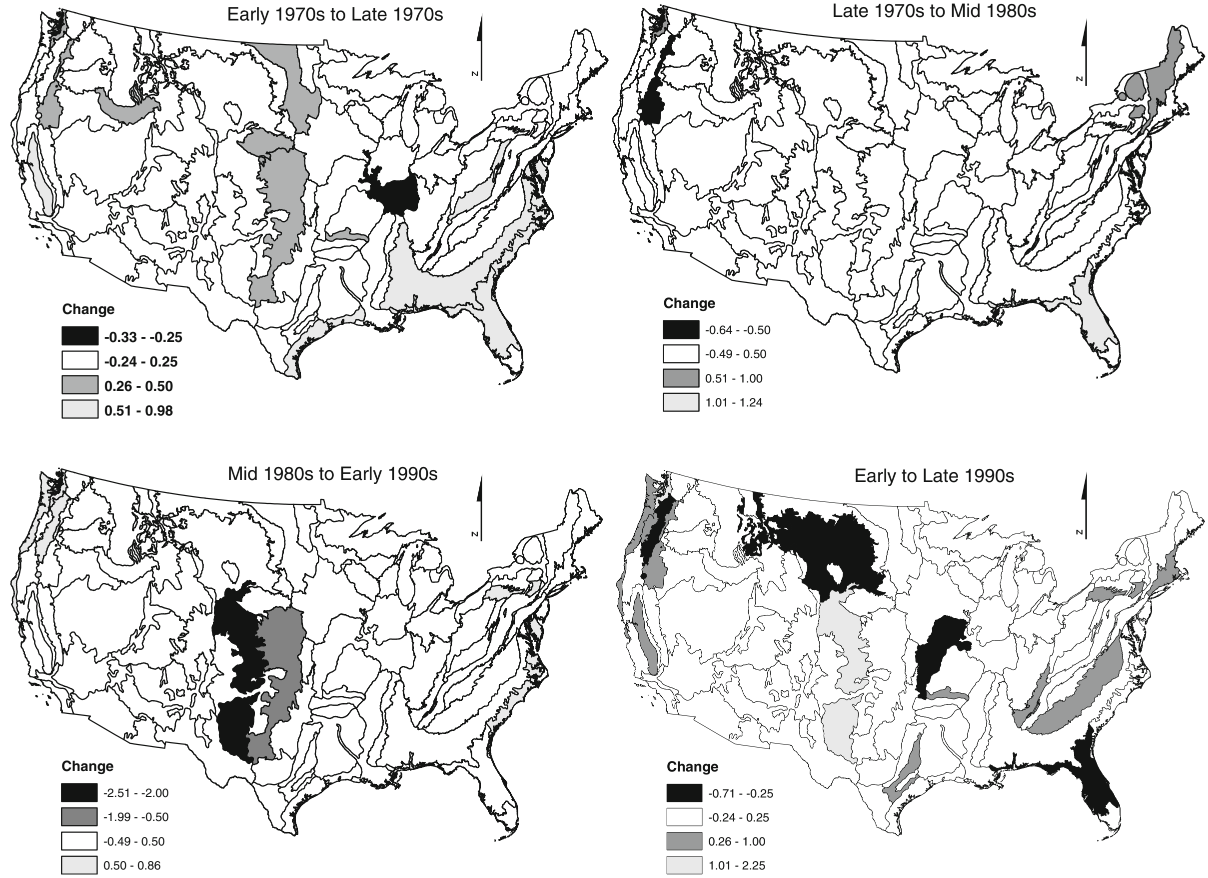


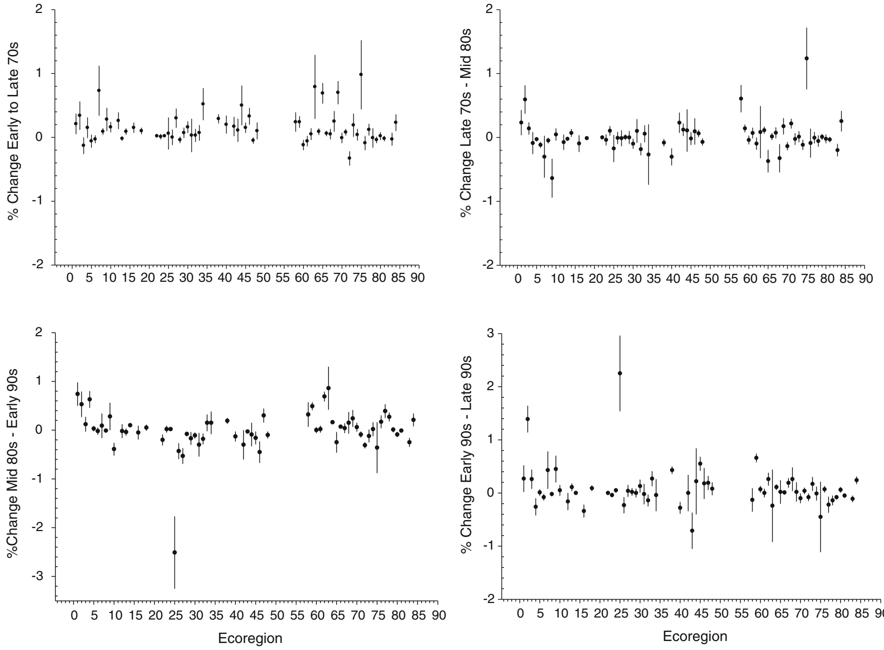
Fig. 9 Map of ranges of mean changes in percent differences between riparian and catchment scale natural land cover for each of the four change periods by ecoregion

shrubland land cover types between the mid 1980s and early 1990s, especially in the western U.S.

Our results showed that the proportion of total natural land cover and forests at the riparian versus catchment scale increased over the 31 years of this study, whereas the proportion of urban and mechanically disturbed land cover decreased. This increase may reflect greater protection of total natural land cover and forests within riparian zones at the national scale. Several potential factors may have contributed to these trends including floodplain protection (zoning), conservation and water quality improvement projects, education programs at state, county, and city levels, conservation reserve programs, and greater general public awareness of the importance of riparian ecosystems. Some of the changes may have resulted from land abandonment, although there is no evidence to suggest that land abandonment is greater in riparian areas than at the catchment scale. Although the changes were relatively small, they do suggest an increasing awareness and action toward protection of riparian ecosystems. Relatively stable numbers for wetlands over the study period may reflect wetland protection programs that apply equally to all areas with wetlands. Additional studies are needed to determine the drivers of observed changes in riparian ecosystems.

Protection and restoration of riparian buffers may be the single most important conservation action to sustain a range of ecosystem services at catchment and basin scales (Iverson et al. 2001; Bernhardt et al. 2005). Riparian land cover changes measured through moderate resolution satellite imagery (e.g., 30–60 m) may provide a way to measure changes in bundles or stacks of ecosystem services. Numerous studies have documented riparian buffer size requirements in relation to specific ecosystem services. The following are examples of riparian buffer widths and attributes

Fig. 10 Mean changes in the percent differences between riparian and catchment scale natural land cover for each of the four change periods by ecoregion. Closed circles denote means; vertical lines denote ±1 standard error



reported for specific ecosystem services and benefits: (a) 25–176 m to achieve 90% nitrate removal in conductive soils (Vidon and Hill 2004); (b) 30 m to maintain fish populations (Jones et al. 2006); (c) 35 m or more to sustain aquatic ecosystems (Barker et al. 2006); (d) 200 m buffers extending 2–3 km to enhance fish populations; (e) 50 m or more to maintain benthic communities (Scarsbrook and Halliday 1999); (f) 75–175 m to include 90% of the bird diversity, 15–30 m to maintain 90% of plant species (Spackman and Hughes 1995); (g) 200 m or more to maintain intact riparian communities that have complete species assemblages (Lees and Peres 2008). The results of these studies suggest that relatively large buffer widths (>200 m) and extents (a few to several kilometers) may be needed to sustain multiple ecosystems services, especially in landscapes with increasing amounts of anthropogenic land cover. Although the scale of riparian buffer changes observed in this study may be insufficient to assess changes in specific ecosystem services, it may be sufficient to address changes in bundles of ecosystem services. However, many existing process models used in assessing ecosystem services lack riparian parameters. These models are needed to interpret the potential responses of specific ecosystem services to riparian land cover change. Lack of in situ data on specific ecosystem services makes development of multi-scaled riparian models difficult. More emphasis is needed on monitoring programs that measure indicators of ecosystem services along riparian landscape gradients (e.g., different sized riparian characteristics in different landscape settings).

Databases, including the National Land Cover Database (NLCD, Vogelmann et al. (2001), now make it possible to conduct wall-to-wall assessments of riparian habitat change at 30 m. The NLCD also includes additional classes of forest and wetlands, and includes separate classes for desert shrub and grassland. Moreover, high resolution imagery (e.g., 1 m aerial photography) is now available for large areas of the US. These data permit analyses of finer resolution riparian landscape features (Goetz 2006).

Future riparian habitat change analyses could be improved by including measurements of riparian land cover extent, since extent has been tied to aquatic ecosystem condition (Storey and Cowley 1997; Scarsbrook and Halliday 1999; Lee et al. 2001). Moreover, new methods are available that evaluate riparian zone ecological services without using fixed buffer widths or dimensions (Baker et al. 2006).

Finally, there is a need to more effectively communicate results of regional and national assessments to local stakeholders in terms of the ecological services and benefits provided by certain landscape configurations and patterns, and what is lost when certain types of changes occur (Termorshuizen and Opdam 2009). The ways in which people interact with and perceive their landscape can significantly influence overall environmental quality (Atwell et al. 2009). Therefore, results from multi-scaled analyses of landscape change, if effectively communicated to stakeholders, could potentially lead to more sustainable environmental practices (Termorshuizen and Opdam 2009).

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