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Recent Land Cover History and Nutrient Retention in Riparian Wetlands

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Abstract Wetland ecosystems are profoundly affected by altered nutrient and sediment loads received from anthropogenic activity in their surrounding watersheds. Our objective was to compare a gradient of agricultural and urban land cover history during the period from 1949 to 1997, with plant and soil nutrient concentrations in, and sediment deposition to, riparian wetlands in a rapidly urbanizing landscape. We observed that recent agricultural land cover was associated with increases in Nitrogen (N) and Phosphorus (P) concentrations in a native wetland plant species. Conversely, recent urban land cover appeared to alter receiving wetland environmental conditions by increasing the relative availability of P versus N, as reflected in an invasive, but not a native, plant species. In addition, increases in surface soil Fe content suggests recent inputs of terrestrial sediments associated specifically with increasing urban land cover. The observed correlation between urban land cover and riparian wetland plant tissue and surface soil nutrient concentrations and sediment deposition, suggest that urbanization specifically enhances the suitability of riparian wetland habitats for the invasive species Japanese stiltgrass [*Microstegium vimenium* (Trinius) A. Camus].

Keywords Nitrogen · Phosphorus · Sediment · Iron · Land cover · Land use · Riparian wetland · *Microstegium vimenium* · Japanese stiltgrass · Invasive plant species

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Introduction

Riparian wetlands occupy transition zones along streams and rivers between upland and aquatic environments, and have the potential to provide ecosystem services that include water quality improvement (nutrient and sediment removal and retention), biodiversity and habitat support, flood mitigation, and groundwater discharge and recharge (Johnston 1991; Walbridge 1993; Vought and others 1994; Zedler 2003). Variations in current and historic land use (LU) and land cover (LC) in the surrounding watershed can affect wetland hydrology and nutrient and sediment loads (Soranno and others 1996; Wickham and others 2000; Winter and Duthie 2000; Horner and others 2001; Groffman and others 2003; Houlahan and Findlay 2004), potentially compromising a wetland's ability to provide important ecosystem services.

Both agricultural and urban LUs are potential sources and can facilitate the transport of excess nutrients [i.e., Nitrogen (N) and Phosphorus (P)] and sediments to aquatic systems via erosion, leaching, and runoff, significantly impacting receiving wetland, stream, or downstream ecosystem health and sustainability (Johnston 1991; Carpenter and others 1998; Reinelt and others 1999; Foley and others 2005). A legacy of past agricultural LU can result in excess nutrient concentrations in soil surface horizons that may be released as sediment from soil disturbance due to agricultural activities such as tilling or grazing or conversion to urban LU (del Campillo and others 1999). In urban watersheds, the primary sources of excess nutrients are lawn and garden fertilizers, leaky sewer lines, home septic systems, and effluents from waste treatment plants (Schueler 1987; Paul and Meyer 2001), and sedimentation is promoted by construction activities and LU that reduce vegetative cover and increase erosion. Further, nutrient and

sediment transport are facilitated in urban watersheds due to increased impervious surface cover (e.g., building footprints, paved surfaces, etc.) (Schueler 1994; Paul and Meyer 2001), and stormwater management infrastructure (including road culverts and curb and gutter systems). Stormwater management often inadvertently directs urban runoff away from historic flow patterns and directly into streams or stormwater management facilities (Schueler 1987; Paul and Meyer 2001), and may therefore reduce hydrologic and nutrient inflows to natural wetlands. As the landscape and flow patterns are altered, ecosystem services performed by riparian wetlands are reduced, and water quality in receiving aquatic ecosystems is likely to decline (Reinelt and others 1999; Groffman and others 2003; Hogan and Walbridge 2007).

The alterations in hydrology and water chemistry resulting from watershed agricultural or urban LU can degrade wetlands, making them susceptible to the colonization and spread of invasive plant species (Ehrenfeld and Schneider 1993; Baldwin 2004). Invasive plants threaten biological diversity and can alter natural ecosystem functions through their impacts on community structure and the vigor of native species (Ehrenfeld and Schneider 1993; Gallaher 2000; Sakai and others 2001; Gibson and others 2002; Simberloff and others 2005). The distribution of the invasive grass, *Microstegium vimenium* (Japanese stiltgrass), has been correlated with disturbed landscapes, wetlands, and elevated soil nutrient concentrations (Sakai and others 2001; Gibson and others 2002). Japanese stiltgrass has the ability to alter resource allocation to maximize growth and reproduction, sometimes forming a monoculture to the exclusion of other species (Gallaher 2000; Claridge and Franklin 2002).

Here we seek to improve the understanding of the consequences of LU change on riparian wetlands and the ecosystem services they provide, by separating the effects of previous agricultural LC history from those associated with the more recent effects of urbanization. We used Fairfax County, VA, a rapidly urbanizing area just west of Washington, DC, as a model, and compared recent (1949–1997) LC with nutrient (N and P) concentrations in tissues of a native and an invasive plant species and in surface soils, and with surface soil Fe chemistry, as indices of eutrophication and hydrologic modification respectively. The two plant species were *Acer rubrum* L. (red maple), a common native deciduous tree that thrives in many different landforms, soil conditions, moisture, and light regimes (Abrams 1998), and *Microstegium vimenium* (Trinius) A. Camus (Japanese stiltgrass), an annual shade-tolerant C_4 grass that has invaded disturbed understory habitats along streambanks and in forests in the eastern United States since its introduction in 1919 (Kartesz 1994; Horton and Neufeld 1998; Gibson and others 2002).

Methods

Riparian Wetland Selection and Analysis

Twenty-one potential forested riparian wetland study sites in Fairfax Co., VA, were identified from National Wetlands Inventory (NWI) maps and verified in the field, and then ranked based on percent impervious surface cover (ISC) in the surrounding watershed, as an index of degree of urbanization (Arnold and Gibbons 1996). Impervious surface cover data were obtained by delineation of 1997 aerial photography, and provided by the Fairfax County Department of Public Works and Environmental Services (DPWES). Sites were categorized as either: (1) slightly urbanized (1.0–6.1% ISC); (2) moderately urbanized (8.6–24.4% ISC); or (3) highly urbanized (24.8–37.6% ISC). These categories are representative of the suburban gradient that existed in 1997 in Fairfax County, VA, a suburb of Washington, DC that has experienced rapid population growth over the past 65 years. Four riparian wetlands were randomly selected from each category for a total of 12 study sites (Fig. 1). Wetlands were delineated in

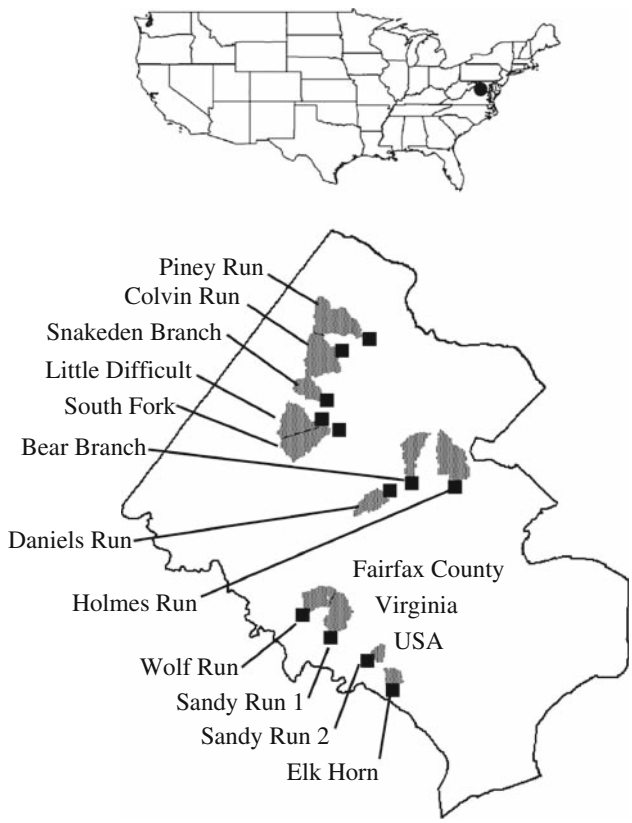

Fig. 1 Map of the United States with Fairfax County marked by the *black dot*. The twelve study sites within Fairfax County are shown; *black squares* represent wetland location and *shaded areas* represent the wetland watersheds. Watershed area and land cover are given in Table 2

the field using US Army Corps of Engineers guidelines (USACE 1987); all riparian wetland study sites are characterized by comparable hydrophytic vegetation, hydric soils, and wetland hydrology. Study sites occurred on either somewhat poorly drained Chewacla (fine-loamy, mixed, active, thermic Fluvaquentic Dystrudepts), or poorly drained Hatboro/Codorus (fine-loamy, mixed, active, mesic Fluvaquentic Endoaquepts/Fluvaquentic Dystrudepts) soils (Porter and others 1963; EFRD 1990; Anonymous 2004).

Samples of red maple (*Acer rubrum* L.) leaf tissue, and Japanese stiltgrass aboveground biomass (*Microstegium vimineum* (Trinius) A. Camus) (Kartesz 1994) were randomly collected from each riparian wetland study site between July 16 and 24, 2002. A 5 × 5 m grid was superimposed over the delineated wetland area, and three plots were randomly selected for the collection of each species ($n = 3$ per site for each species). If plots did not contain the species, neighboring plots were examined systematically until a plot containing the species was found. Red maple leaf samples were collected from the southwest side of the tree in the upper 1/3 of the canopy (full sun) using a 9.1 m tree pole pruner (Erdmann and others 1988). No significant differences were found in average diameter at breast height (dbh) (26.6 ± 2.3 cm, range of 7.5–67.5 cm), or average sampling height (9.0 ± 0.3 m, range of 3.3–11.2 m), of red maple trees across the study sites. Leaves for analysis were selected starting at the bottom of the cut branch, by removing the first leaf to the left of the first terminal bud. In like manner, leaves were removed from the next six terminal buds for a total of 7 leaves per sample ($n = 3$ per site). For Japanese stiltgrass, the entire aboveground parts of 7 plants from each plot were collected, using scissors to cut plants off at ground level ($n = 3$ per site). Total P was determined by sulfuric acid, potassium sulfate, and mercuric oxide digestion, using a Technicon BD40 block digester (Williams and others 1970), and analyzed colorimetrically using a Technicon II Autoanalyzer (Bran and Luebbe Inc. 1989, Method Number 696-82W, 1A and 1C). Total N was estimated using a Carlo Erba Instruments EA1108 CHNS/O Analyzer and standard methods (Nelson and Sommers 1996).

Soil cores (0–15 cm) were collected during summer 2002 by driving a 4 cm diameter polyvinyl chloride (PVC) pipe with a sharpened edge into the ground after brushing away surface litter (USEPA 1989). At each site, a 5 × 5 m grid was superimposed over the delineated wetland area, and four plots were randomly selected. Each plot was divided into four even subplots, and one soil core was collected from each subplot then composited to form one sample per plot ($n = 4$ per wetland study site). Soils were homogenized by hand, removing coarse roots and rocks when present, and moisture content was determined by oven drying ~40 g soil aliquots at 90°C to a constant mass. Sodium hydroxide-extractable P, an estimate of P associated with soil Al and Fe minerals, was estimated by the methods of Bache and Williams (1971) and Hedley and others (1982). Phosphorus extracts were analyzed colorimetrically as orthophosphate by the method of Murphy and Riley (1962), using a Technicon II Autoanalyzer (Bran and Luebbe Inc. 1989, Method Number 696-82W, 1B). Total N and P were determined using the same procedure as for plant tissues. Soil non-crystalline (amorphous), crystalline, and total Fe content were estimated by the method of Darke and Walbridge (1994). With the exception of non-crystalline (oxalate-extractable) Fe, which was analyzed by ICP spectroscopy (Novozamsky and others 1986), soil Fe concentrations were determined using a Perkin Elmer 5100PC atomic absorption spectrometer and standard methods (Perkin Elmer 1982).

Agriculture and Urban Land Cover Indices

Historic LC patterns including type and the amount of time since abandonment of agricultural activities and/or the onset of watershed urbanization can affect current riparian wetland soil chemistries. With this in mind, indices of agricultural and urban LC were calculated by interpretation of historic aerial photography (Table 1) into three LU/LC categories: (1) agriculture—cropland, pasture, and animal farming (includes fields, farm buildings, and field access roads); (2) urban (built-up)—residential, commercial, industrial, institutional, transportation, recreation, and utility LUs; and (3) forest—relatively undisturbed tree-covered areas [includes houses on lots ≥1.6 ha (4 acres), forested parks, and roads bisecting otherwise densely forested areas] (Anderson and others 1976). Although both impervious and pervious surface cover can affect the amount and quality of surface water runoff (including nutrient and sediment loads) (Schueler 1994; Schueler 1995; Arnold and Gibbons 1996), these three categories establish the general effect of changing LC on receiving riparian wetlands.

For each aerial photograph, riparian wetland watersheds and LC classes were manually delineated using mylar overlays, and a dot grid was used to calculate the percent LC of each of the three categories (Jensen 2000) (Fig. 2). Dots grids at 4, 16, and 64 dots per inch were tested for each photo date (Table 1). No significant differences were found for percent LC classes when using the alternative dots grids; therefore, calculations were done at 4 dots per inch. The minimum mapping unit was 2 ha (5 acres), the smallest unit that could be reliably delineated based on the spatial resolution of the available aerial photography.

To normalize for variations in area across the 12 watersheds, indices were developed based on percent LC in each watershed (Coles and others 2004), by summing the

Table 1 Aerial photography used to delineate watershed land cover history for each of the 12 riparian wetland study sites

Photo date	Source	Film type	Nominal spatial scale	Study sites included
1949	US Geological Survey, project FX000	Black and white positives	24000	10 Sites: Wolf Run, Sandy Run 1, Little Difficult, South Fork, Piney Run, Bear Branch, Daniels Run, Colvin Run, Snakeden Branch, Holmes Run
1954	Fairfax County	Photographic paper print	6000	2 Sites: Sandy Run 2, Elk Horn Run
1964	US Geological Survey, project VBBMO	Black and white positives	24000	10 Sites: Wolf Run, Sandy Run 1, Little Difficult, South Fork, Piney Run, Bear Branch, Daniels Run, Colvin Run, Snakeden Branch, Holmes Run
1968	Fairfax County	Mylar film, half tone positives	6000	2 sites: Sandy Run 2, Elk Horn Run
1977	US Geological Survey, project VEHOO	Black and white positives	80000	11 sites: Sandy Run 2, Wolf Run, Sandy Run 1, Little Difficult, South Fork, Piney Run, Bear Branch, Daniels Run, Colvin Run, Snakeden Branch, Holmes Run
1978	Fairfax County	Mylar film, half tone positives	6000	1 site: Elk Horn Run
1988	US Geological Survey	Virginia DOQQ[a]	12000	All 12 sites
1997	US Geological Survey	Virginia DOQQ[a]	12000	All 12 sites

[a] DOQQ = Digital Orthophoto Quadrangles from the APSRS (Aerial Photo Summary Reference System)

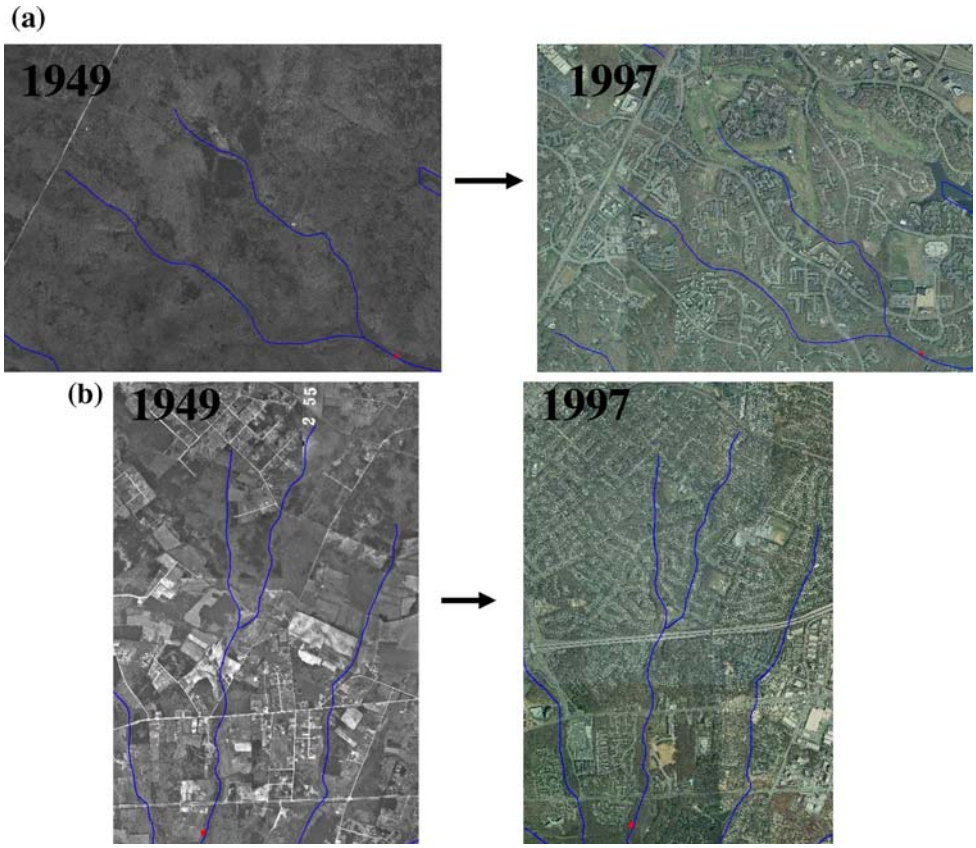

Fig. 2 Two examples of the land cover changes observed in the watersheds of the riparian wetland study sites; Snakeden Branch at 97% forested land cover in 1949 to 70% urban land cover in 1997 (**a**), and Bear Branch at 36% agricultural land cover in 1949 to 85% urban land cover in 1997 (**b**). *Blue lines* represent stream location and red dots indicate the location of riparian wetland study sites

annual percentages for all years (1949 or 1954 through 1997) separately for agricultural or urban LC (interpolating for years between aerial photo dates), then dividing the sum by the number of years of record for each site (to correct for the two sites where data from 1949 was not available). This provides a composite index describing the % LC and the number of years of agriculture or urban LC in each watershed, and assumes that the rate of LC change between photo dates is constant and linear.

Statistical Analyses

Both linear and polynomial regressions were used to compare soil and plant tissue chemistries as a function of

historic agricultural or urban LC in the surrounding watershed. Analyses were performed using Microsoft® Excel 2000 spreadsheets (Microsoft Corporation 2000) and JMP® (SAS 2002), considering significant differences at $P \leq 0.05$.

Results and Discussion

Historic Land Use Patterns

In Fairfax County, VA, conversion from a forested to an agricultural landscape began in the eighteenth century; prior to the 1940s, forest and agriculture were the two predominant LCs in the County (Netherton and others 1978). All of the riparian wetlands in this study were located in watersheds with an agricultural history, and all have experienced decreases in agriculture and increases in urban development since the early 1950s (Table 2). Agricultural abandonment and urban development have been observed in numerous areas of the US, including south Florida, the eastern shore of Maryland, Phoenix, and Baltimore (Walker and others 1997; Benitez and Fisher 2004; Berling-Wolff and Wu 2004; Grimm and Redman 2004). In some studies, the direct conversion of agricultural lands to urban LUs has been the dominant trend (e.g., Walker and others 1997); others report initial conversion of agricultural to urban LUs, followed by the newest housing established primarily on previously non-utilized lands (Grimm and Redman 2004). In Fairfax County, VA, roughly equal percentages of agricultural and forest lands have been converted to urban use over the last 50–55 years (25.75 and 26.83% respectively, averaged across the 12 study sites) (Table 2). While forests converted to urban use may have been agricultural lands at some point prior to 1949, sufficient time elapsed following agricultural abandonment (Netherton and others 1978) for the development of closed canopy (i.e., >20 years old) second-growth forests on these sites prior to urban development.

Agriculture and Urban Land Cover Indices

Six sites were characterized by a low level of continuing (or recent abandonment of) agriculture with a range of years since the onset of urban development (Bear Branch, Elk Horn, Piney Run, Sandy Run 2, Wolf Run, and Sandy Run 1, represented by squares, Table 2). Five sites were characterized by the early onset of urban development, with a range of times since agricultural abandonment (Daniels Run, Holmes Run, Colvin Run, Bear Branch, and Elk Horn, represented by closed symbols). In addition, four sites were characterized by an intermediate amount of time since agricultural abandonment, with a range of times since the onset of urban development (Colvin Run, Little Difficult, South Fork, and Snakeden Branch, represented by triangles).

Recent Watershed Land Cover Trends and Riparian Wetland Elements

Increasing agricultural LC indices for the surrounding watershed was correlated with significant increases in N and P concentrations in native red maple foliage (Fig. 3a, b), but not in the invasive Japanese stiltgrass (Hogan 2005). Agricultural LC was not associated with significant increases in N or P concentrations in surface soils, the saturation of soil P sorption capacity, or concentrations of crystalline or total Fe that would suggest increased sediment deposition (Hogan 2005). Since Fairfax County has a long agricultural history, considerable sediment deposition likely occurred much earlier in the County's history, with legacy effects that may be so widespread by now that they are not discernible when based on the history of recent agricultural activity (Langland and Cronin 2003). However, as a whole, agricultural activity during the past 50–55 years seems to have had relatively little effect on riparian wetland ecosystem structure and function in these watersheds, at least in terms of the variables examined in this study.

Increasing urban LC indices for the surrounding watershed was significantly correlated with increases in P concentrations, and decreases in N:P ratios, in aboveground biomass of Japanese stiltgrass, an invasive species (Fig. 4), but not in foliage of the native red maple (Hogan 2005). Highest P concentrations were associated with the early onset of urban development (closed symbols) (Fig. 4a); as the index of urban LC increased, N:P ratios in Japanese stiltgrass leaf tissues decreased (Fig. 4b). These data suggest that urbanization increases P availability, altering the relative availability of P versus N. Unlike the increases in both N and P availability associated with agriculture, it is the invasive, rather than the native, species, that responds to this specific increase in P availability.

Non-native plants in disturbed sites may respond very differently to altered nutrient and hydrologic conditions as compared to native flora, both enhancing the ability of invasive species to establish and thrive and affecting plant community structure (Ehrenfeld and Schneider 1993; Gibson and others 2002; Baldwin 2004). In both New Jersey and in riparian forests in southern Manitoba, urbanization was shown to alter both water chemistry and hydrology, with associated decreases in the number and diversity of native species and an increased proportion of exotic species (Ehrenfeld and Schneider 1993; Moffatt and others 2004). The spread of the invasive Japanese stiltgrass has been linked to increased nutrient availability under

Table 2 Recent agriculture, urban, and forest land cover (LC) in the watersheds of the riparian wetland study sites

Site	Watershed area (ha)	Photo year	LC (%)			Agriculture LC index	Urban LC index	Figure symbol
			Agriculture	Urban	Forest			
Sandy Run 2	165.6	1954	15	0	85			
		1968	15	0	85			
		1977	4	1	95	9.5	1.0	□
		1988	7	2	91			
		1997	5	3	92			
Wolf Run	406.5	1949	24	0	76			
		1964	30	0	70			
		1977	27	4	69	24.5	2.1	□
		1988	17	4	79			
		1997	19	3	78			
Elk Horn Run	204.3	1954	26	5	69			
		1968	15	9	76			
		1978	12	10	78	14.5	15.5	■
		1988	10	30	60			
		1997	10	30	60			
Sandy Run 1	608.6	1949	13	0	87			
		1964	17	0	83			
		1977	11	12	77	11.7	9.0	□
		1988	8	17	75			
		1997	4	26	70			
Little Difficult	602.1	1949	39	0	61			
		1964	33	2	65			
		1977	25	18	57	21.9	22.1	▲
		1988	0	55	45			
		1997	0	58	42			
South Fork	706.5	1949	30	0	70			
		1964	29	0	71			
		1977	13	29	58	16.4	22.0	Δ
		1988	0	42	58			
		1997	0	62	38			
Piney Run	930.5	1949	63	0	37			
		1964	62	0	38			
		1977	50	6	44	43.9	16.9	□
		1988	18	36	46			
		1997	0	79	21			
Bear Branch	552.8	1949	36	8	56			
		1964	19	45	36			
		1977	6	62	32	13.2	53.7	■
		1988	2	77	21			
		1997	0	85	15			
Daniels Run	397.3	1949	48	2	50			
		1964	0	72	28			
		1977	0	75	25	7.8	63.1	●
		1988	0	77	23			
		1997	0	80	20			

Table 2 continued

Site	Watershed area (ha)	Photo year	LC (%)			Agriculture LC index	Urban LC index	Figure symbol
			Agriculture	Urban	Forest			
Colvin Run	892.3	1949	30	3	67			
		1964	43	6	51			
		1977	26	42	32	23.6	36.5	▲
		1988	0	80	20			
		1997	0	75	25			
Snakeden Branch	348.6	1949	3	0	97			
		1964	15	0	85			
		1977	8	69	23	6.7	38.3	Δ
		1988	0	70	30			
		1997	0	70	30			
Holmes Run	837.5	1949	33	27	40			
		1964	9	57	34			
		1977	0	77	23	8.0	66.0	●
		1988	0	84	16			
		1997	0	91	9			

Figure symbol shape represents the time (years) since complete agricultural abandonment in the watershed: early (20–33 years ago) by *circles*, intermediate (9 years ago) by *triangles*, or late (0–1 years ago) by *squares*. Figure symbol *shading* represents the onset of urbanization in the watershed: early (43–48 years ago) by *closed symbols*, intermediate (33 years ago) by *shaded symbols*, or late (20 years ago) by *open symbols*

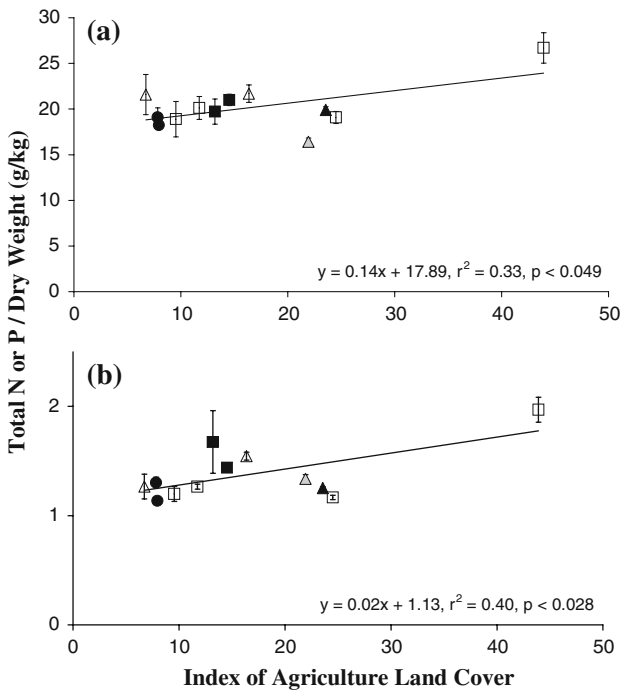

Fig. 3 Total nitrogen (**a**) and total phosphorus (**b**) concentrations (g/kg) in red maple leaf tissue as a function of the index of agricultural land cover. *Closed* (▲ and ■), *shaded* (▲), and *open* (○, Δ, and □), symbols represent early, intermediate, or late onset of urban development in the surrounding watershed, respectively. *Circles* (○), *triangles* (Δ, ▲, and ▲), and *squares* (□ and ■) represent early, intermediate, or late agricultural abandonment in the surrounding watershed, respectively (Table 2). Error bars represent ± 1 SE

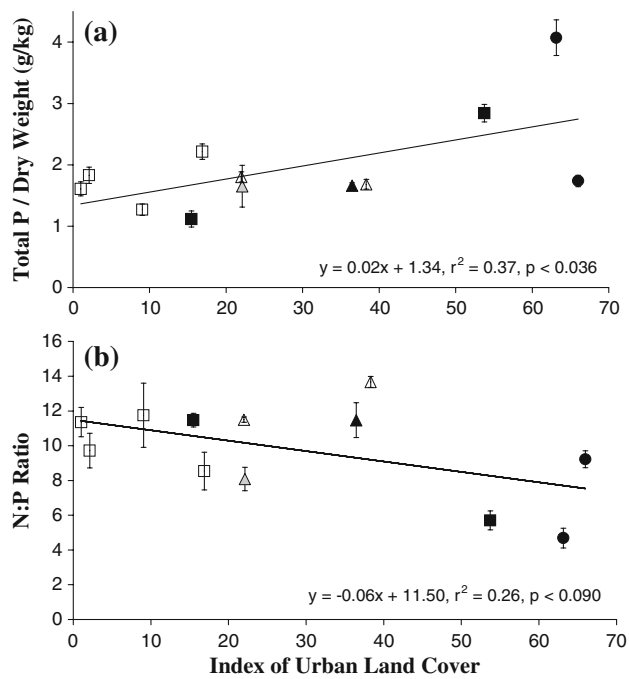

Fig. 4 Total phosphorus concentration (g/kg) (**a**) and N:P ratios (**b**) in Japanese stiltgrass as a function of the index of urban land cover. *Closed* (▲ and ■), *shaded* (▲), and *open* (○, Δ, and □), symbols represent early, intermediate, or late onset of urban development in the surrounding watershed, respectively. *Circles* (○), *triangles* (Δ, ▲, and ▲), and *squares* (□ and ■) represent early, intermediate, or late agricultural abandonment in the surrounding watershed, respectively (Table 2). Error bars represent ± 1 SE

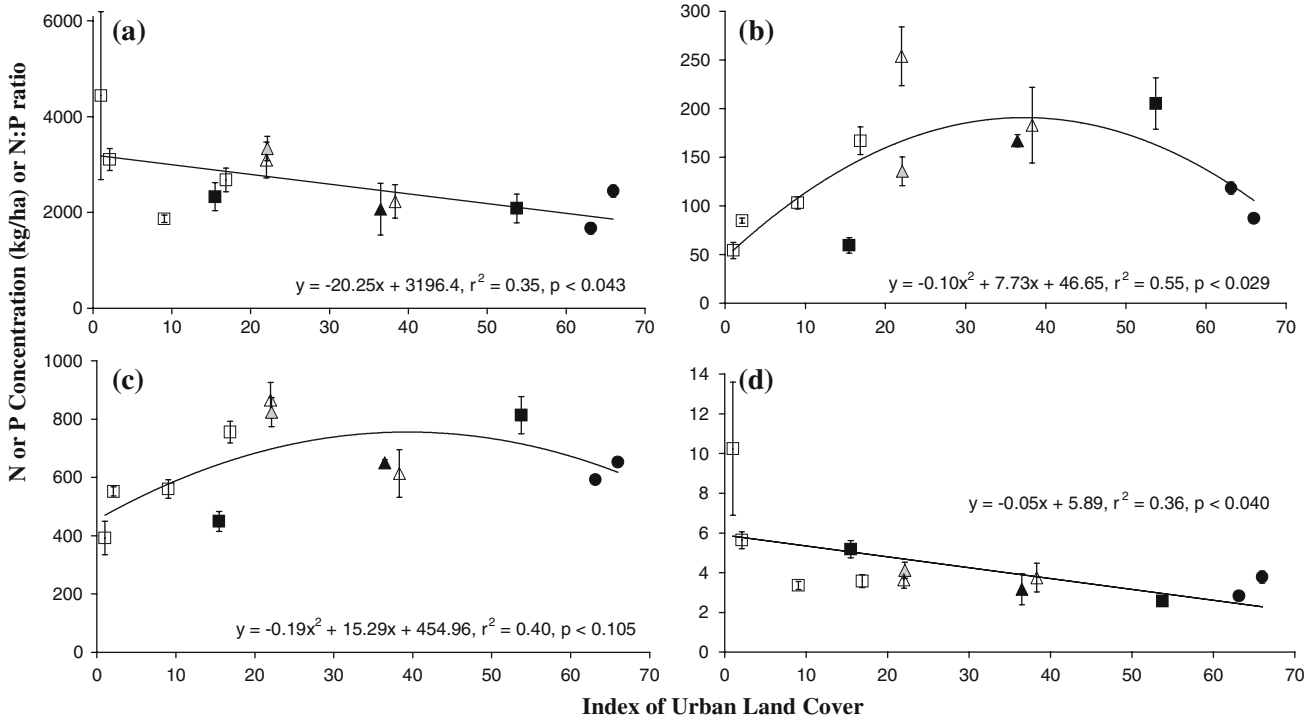

Fig. 5 Total nitrogen (**a**), sodium hydroxide extractable-P (**b**), total phosphorous (**c**), and N:P ratio (**d**) in riparian wetland surface soils as a function of the index of urban land cover in the surrounding watershed. *Closed* (▲ and ■), *shaded* (▲), and *open* (○, Δ, and □) symbols represent early, intermediate, or late onset of urban development in the surrounding watershed, respectively. *Circles* (○), *triangles* (Δ, ▲, and ▲), and *squares* (□ and ■) represent early, intermediate, or late agricultural abandonment in the surrounding watershed, respectively (Table 2). Error bars represent ± 1 SE

forest canopies, and it is well adapted to take advantage of disturbed habitat conditions (Gallaher 2000; Ehrenfeld and others 2001; Claridge and Franklin 2002; Gibson and others 2002).

The changes in the relative concentrations of foliar P versus N as the urban LC index increased, were consistent with decreases in total N and N:P ratios in surface soils, with the lowest values associated with the early onset of urban development (closed symbols) (Fig. 5a, d). Decreasing pools of N in surface soils reflect a change in the ecosystem nutrient balance, and could be due to the increased sediment deposition to riparian wetlands associated with the onset of urbanization in the surrounding watershed, or to decreases in soil saturation. Sediment mobilization associated with urban development is often linked to construction activities, and thus the sediments may originate from subsurface soils, which have generally lower N concentrations than surface soils (Groffman and others 2003). The lower water tables and reduced soil saturation characteristic of many urban wetlands has been shown to result in increases in N mineralization and nitrification, with a higher probability of N export from the watershed (Zhu and Ehrenfeld 1999; Groffman and others 2002; Faulkner 2004).

In contrast to N, NaOH extractable-P, an estimate of P bound to soil Al and Fe minerals, exhibited a non-linear pattern with urbanization, with maximum P concentrations associated with an intermediate number of years since the onset of urban development (triangles) (Fig. 5b). This relationship was best described by a 2nd order polynomial, significantly improving ($P < 0.05$) the linear model for NaOH extractable-P. Total P in surface soils varied similarly, but not significantly (Fig. 5c). The lower soil P concentrations observed at both low and high indices of urban LC were associated with either late agricultural abandonment followed by late onset of urban development (open squares), or early agricultural abandonment followed by early onset of urban development (closed circles) (Fig. 5b, c).

Watershed development requires use of stormwater management to prevent flooding of urban areas. The non-linear trend for riparian wetland NaOH extractable-P concentrations suggests an increase in P loads with increasing urban LU in the surrounding watershed (i.e., Schueler 1994; Soranno and others 1996; Carpenter and others 1998; Reinelt and others 1999; Wickham and others 2000; Winter and Duthie 2000; Horner and others 2001; Houlahan and Findlay 2004), combined with a decrease in

riparian wetland surface soil P concentrations at higher levels of watershed urbanization, as the primary effects of urbanization shift from increased nutrient loads to hydrologic modification (Fig. 5b). Stormwater management infrastructure may inadvertently direct these waters away from riparian wetlands in more urbanized watersheds, causing nutrient inflows to riparian wetlands to also decrease. Despite this trend for NaOH extractable-P, surface soil N:P ratios still declined significantly, likely primarily due to decreasing N concentrations, as urbanization increased (Fig. 5a, d).

Finally, as indices of urban LC increased, crystalline and total Fe concentrations in riparian wetland surface soils increased significantly (Fig. 6a, b), and relative Fe crystallinities (indexed as oxalate extractable-Fe/total Fe) decreased significantly (Fig. 6c). Maximum crystalline and total Fe concentrations, and minimum relative Fe crystallinities, were associated with the early onset of urban development (closed symbols) (Fig. 6). No significant relationships for pool sizes of oxalate extractable-Fe were found as a function of urban LC indices in the surrounding watershed ($r^2 = 0.11$). The increased Fe crystallinity (i.e., decreased relative Fe crystallinity; Fig. 6c) was driven primarily by significant increases in crystalline and total Fe associated with increased urban LC indices. These data suggest significant deposition of crystalline forms of Fe to riparian wetlands from watershed sources, likely associated with the deposition of sediments derived from subsurface soils that are produced during the onset of urbanization. Other studies have also reported increased mobilization of terrestrial sediments to riparian systems in urbanizing watersheds, associated with development and altered watershed hydrology (Wolman and Schick 1967; Paul and Meyer 2001; Groffman and others 2003). In addition to their potential to alter N:P balance in surface soils and facilitate invasions of non-native species, increases in sediment deposition can also decrease the effective area of a wetland, reducing both water and nutrient retention capacities (Thom and others 2001).

Taken as a whole, recent urbanization in Fairfax County, VA, appears to have had a much more dramatic effect on ecosystem structure and function in riparian wetlands than recent agricultural activity. Observed changes in nutrient balance are directional [i.e., favoring the increased availability of one nutrient, P, versus another (N)] and appears to benefit an invasive plant species. Invasive plants threaten native species and biological diversity and can alter natural ecosystem function and community structure (Ehrenfeld and Schneider 1993; Gallaher 2000; Sakai and others 2001; Gibson and others 2002; Simberloff and others 2005). Further, soil Fe content suggests that increasing urban LC is associated with significant deposition of terrestrial-derived sediments to the riparian wetland surface. Because

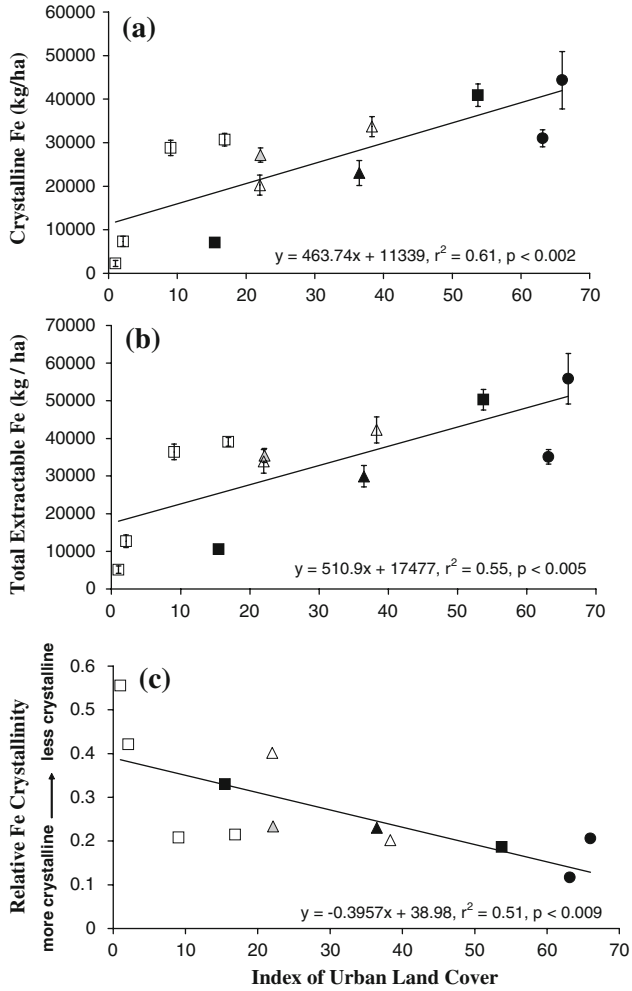

Fig. 6 Crystalline (**a**) and total (**b**) Fe, and relative Fe crystallinity (indexed as oxalate extractable-Fe/total Fe) (**c**), in wetland surface soils as a function of the index of urban land cover. *Closed* (▲ and ■), *shaded* (▲), and *open* (○, Δ, and □) symbols represent early, intermediate, or late onset of urban development in the surrounding watershed, respectively. *Circles* (○), *triangles* (Δ, ▲, and ▲), and *squares* (□ and ■) represent early, intermediate, or late agricultural abandonment in the surrounding watershed, respectively (Table 2). Error bars represent ± 1 SE for total and crystalline Fe

of these rather dramatic changes in ecosystem structure and function in comparison with the effects observed in response to agriculture, it seems logical that, were the effects of urbanization in the surrounding watershed to be reduced or ameliorated, the recovery or restoration of associated riparian wetland ecosystems would, at best, be slow (Baldwin 2004; Faulkner 2004). Collins and others (2001) suggested that urban ecosystems may function differently from other ecosystems. Observations made during this study support this view for riparian wetlands in urbanizing watersheds.

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References

Abrams MD (1998) The red maple paradox. BioScience 48:355–364

Anderson JR, Hardy EE, Roach JT, Witmer RE (1976) A land use and land cover classification system for use with remote sensor data. US Department of the Interior, US Geological Survey Professional Paper 964

Anonymous (2004) Official soil series descriptions. Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. http://soils.usda.gov/soils/technical/classification/osd/index.html

Arnold CL Jr, Gibbons CJ (1996) Impervious surface coverage: the emergence of a key environmental indicator. Journal of the American Planning Association 62:243–258

Bache BW, Williams FJ (1971) A phosphate sorption index for soils. Journal of Soil Science 22:289–300

Baldwin AH (2004) Restoring complex vegetation in urban settings: the case of tidal freshwater marshes. Urban Ecosystems 7: 125–137

Benitez JA, Fisher TR (2004) Historical land-cover conversion (1665–1820) in the Choptank watershed, eastern United States. Ecosystems 7:219–232

Berling-Wolff S, Wu J (2004) Modeling urban landscape dynamics: a case study in Phoenix, USA. Urban Ecosystems 7:215–240

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559–568

Claridge K, Franklin SB (2002) Compensation and plasticity in an invasive plant species. Biological Invasions 4:339–347

Coles JF, Cuffney TF, McMahon G, Beaulieu KM (2004) The effects of urbanization on the biological, physical, and chemical characteristics of coastal New England streams. US Department of the Interior, US Geological Survey, Reston, VA (Professional Paper 1695)

Collins JP, Kinzig A, Grimm NB, Fagan WF, Hope D, Wu J, Borer ET (2000) A new urban ecology. American Scientist 88:416–425

Darke AK, Walbridge MR (1994) Estimating non-crystalline and crystalline aluminum and iron by selective dissolution in a riparian forest soil. Communications in Soil Science and Plant Analysis 25:2089–2101

del Campillo MC, Van der Zee SEATM, Torrent J (1999) Modelling long-term phosphorus leaching and changes in phosphorus fertility in excessively fertilized acid sandy soils. European Journal of Soil Science 50:391–399

Ehrenfeld JG, Schneider JP (1993) Responses of forested wetland vegetation to perturbations of water chemistry and hydrology. Wetlands 13:122–129

Ehrenfeld JG, Kourtev P, Huang W (2001) Changes in soil functions following invasions of exotic understory plants in deciduous forests. Ecological Applications 11:1287–1300

Erdmann GG, Crow TR, Rauscher HM (1988) Foliar nutrient variation and sampling intensity for *Acer rubrum* trees. Canadian Journal of Forest Research 18:134–139

Environmental and Facilities Review Division (EFRD) (1990) Fairfax County soil survey update. Department of Public Works and Environmental Services. Fairfax County Government Center, Fairfax, VA

Faulkner S (2004) Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems 7:89–106

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, Helkowski JH, Holloway T, Howard EA, Kucharik CJ, Monfreda C, Patz JA, Prentice IC, Ramankutty N, Snyder PK (2005) Global consequences of land use. Science 309:570–574

Gallaher K (2000) The effects of shading and nutrients on resource allocation in an exotic invasive, *Microstegium vimineum*. Journal of the Tennessee Academy of Science p 18

Gibson DJ, Spyreas G, Benedict J (2002) Life history of *Microstegium vimineum* (Poaceae), and invasive grass in southern Illinois. Journal of the Torrey Botanical Society 129:207–219

Grimm NB, Redman CL (2004) Approaches to the study of urban ecosystems: the case of central Arizona–Phoenix. Urban Ecosystems 7:199–213

Groffman PM, Boulware NJ, Zipperer WC, Pouyat RV, Band LE, Colosimo MF (2002) Soil nitrogen cycle processes in urban riparian zones. Environmental Science and Technology 36: 4547–4552

Groffman PM, Bain DJ, Band LE, Belt KT, Brush GS, Grove JM, Pouyat RV, Yesilonis IC, Zipperer WC (2003) Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment 1:315–321

Hedley MJ, Stewart JW, Chauhan BS (1982) Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubation. Soil Science Society of America Journal 46:970–976

Hogan DM (2005) The effects of urbanization on wetland ecosystem structure and function—a case study of freshwater riparian wetlands in Fairfax County, Virginia. PhD Dissertation. George Mason University, Fairfax, VA

Hogan DM, Walbridge MR (2007) Urbanization and nutrient retention in freshwater riparian wetlands. Ecological Applications 17(4):1142–1155

Horner RR, Cooke SS, Reinelt LE, Ludwa KA, Chin NT, Valentine M (2001) Effects of watershed development on water quality and soils. In: Azous AL, Horner RR (eds) Wetlands and urbanization: implications for the future. Lewis Publishers, Boca Raton, Florida, pp 237–254

Horton JL, Neufeld HS (1998) Photosynthetic responses of *Microstegium vimineum*, (Trin.) A. Camus, a shade-tolerant, C_4 grass, to variable light environments. Oecologia 114:11–19

Houlahan JE, Findlay CS (2004) Estimating the 'critical' distance at which adjacent land-use degrades wetland water and sediment quality. Landscape Ecology 19:677–690

Jensen JR (2000) Remote sensing of the environment: an earth resource perspective. Prentice Hall, Upper Saddle River, New Jersey, p 544

Johnston CA (1991) Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Critical Reviews in Environmental Control 21:491–565

Kartesz JT (1994) A synonymized checklist of the Flora of the United States, Canada, and Greenland. University of North Carolina Press, Chapel Hill, North Carolina, 542 pp

Langland M, Cronin T (eds) (2003) A summary report of sediment processes in Chesapeake Bay and Watershed. US Department of the Interior, US Geological Survey Water-Resources Investigations Report 03-4123

Luebbe Bran Inc. (1989) Technicon autoanalyzer II methods. Technicon Instruments Corporation, Buffalo Grove, Illinois

Microsoft Corporation (2000) Getting results with microsoft office. Microsoft Corporation Document No. X03-21975-0397. Printed in the United States of America

Moffatt SF, McLachlan SM, Kenkel NC (2004) Impacts of land use on riparian forest along an urban-rural gradient in southern Manitoba. Plant Ecology 174:119–135

Murphy J, Riley J (1962) A modified single solution method for the determination of phosphate in natural waters. Analytica Chimica Acta 27:31–36

Nelson DW, Sommers LE (1996) Total carbon, organic carbon, and organic matter. In: Bigham JM (ed) Methods of soil analysis: part 3–chemical methods. Soil Science Society of America, Inc., Madison, Wisconsin, pp 961–1010

Netherton N, Sweig D, Artemel J, Hickin P, Reed P (1978) Fairfax County, Virginia: a history Fairfax County board of supervisors. Fairfax County, Virginia, p 784

Novozamsky I, van Eck R, Houba VJG, van der Lee JJ (1986) Use of inductively coupled plasma atomic emission spectrometry for determination of iron, aluminium and phosphorus in Tamm's soil extracts. Netherlands Journal of Agricultural Science 34: 185–191

Paul MJ, Meyer JL (2001) Streams in the urban landscape. Annual Reviews Ecology and Systematics 32:333–365

Perkin Elmer (1982) Analytical methods for atomic absorption spectrophotometry. The Perkin-Elmer Corporation. Norwalk, Connecticut

Porter HC, Derting JF, Elder JH, Henry EF, Pendleton RF (1963) Soil survey of Fairfax County, Virginia. United States Department of Agriculture Soil Conservation Service, Virginia Agricultural Experiment Station and Fairfax County, VA. Series 1955, No. 11

Reinelt L, Horner R, Azous A (1999) Impacts of urbanization on palustrine (depressional freshwater) wetlands—research and management in the Puget Sound region. Urban Ecosystems 2: 219–236

Sakai AK, Allendorf FW, Holt JS, Lodge DM, Molofsky J, With KA, Baughman S, Cabin RJ, Cohen JE, Ellstrand NC, McCauley DE, O'Neil P, Parker IM, Thompson JN, Weller SG (2001) The population biology of invasive species. Annual Reviews of Ecological Systems 32:305–322

SAS Institute Inc. (2002) JMP® introductory guide, Version 5. The SAS Institute Inc., Cary, NC

Schueler TR (1987) Controlling urban runoff: a practical manual for planning and designing urban BMPs. Department of Environmental Programs, Metropolitan Washington Council of Governments, Washington, DC

Schueler TR (1994) The importance of imperviousness. Watershed Protection Techniques 1:100–111

Schueler TR (1995) The peculiarities of perviousness. Watershed Protection Techniques 2:233–238

Simberloff D, Parker IM, Windle PN (2005) Introduced species policy, management, and future research needs. Frontiers in Ecology and the Environment 3:12–20

Soranno PA, Hubler SL, Carpenter SR, Lathrop RC (1996) Phosphorus loads to surface waters: a simple model to account for spatial pattern of land use. Ecological Applications 6:865–878

Thom RM, Borde AB, Richter KO, Hibler LF (2001) Influence of urbanization on ecological processes in wetlands. In: Wigmosta MS, Burges SJIII (eds) Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas. Water Science and Application Volume 2. American Geophysical Union, Washington DC, pp 5–16

United States Environmental Protection Agency (USEPA) (1989) Soil sampling quality assurance user's guide, 2nd edn. Environmental Monitoring Systems Laboratory, Las Vegas, NV EPA/600/8-89/046

US Army Corps of Engineers (USACE) (1987) USACE wetlands delineation manual. Technical Report Y-87-1, Environmental Laboratory, US Army Corps of Engineers Waterway Experimental Station

Vought LBM, Dahl J, Pedersen CL, Lacoursiere JO (1994) Nutrient retention in riparian ecotones. Ambio 23:342–348

Walbridge MR (1993) Functions and values of forested wetlands in the southern United States. Journal of Forestry 91:15–19

Walker RT, Solecki WD, Harwell C (1997) Land use dynamics and ecological transition: the case of South Florida. Urban Ecosystems 1:37–47

Wickham JD, Riitters KH, O'Neill RV, Reckhow KH, Wade TG, Jones KB (2000) Land cover as a framework for assessing risk of water pollution. Journal of the American Water Resources Association 36(6):1417–1422

Williams JDH, Syers JK, Walker TW, Rex RW (1970) A comparison of methods for the determination of soil organic phosphorus. Soil Science 110:13–18

Winter JG, Duthie HC (2000) Export coefficient modeling to assess phosphorus loading in an urban watershed. Journal of the American Water Resources Association 36:1053–1061

Wolman MG, Schick AP (1967) Effects of construction on fluvial sediment, urban and suburban areas of Maryland. Water Resources Research 3:451–464

Zedler JB (2003) Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Environment 2:65–72

Zhu WX, Ehrenfeld JG (1999) Nitrogen mineralization and nitrification in suburban and undeveloped Atlantic white cedar wetlands. Journal of Environmental Quality 28:523–529