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Report

Responses of Emergent Marsh Wetlands in Upstate New York to Variations in Urban Typology

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ABSTRACT. Although it has been repeatedly demonstrated that urbanization has negative environmental consequences, the conversion of land to urban use is increasing worldwide and is not likely to abate. We tested the hypothesis that different urban typologies, i.e., distributions of human population and infrastructure, differentially influence the water quality and ecological functionality of emergent marsh wetlands in New York State's Hudson River Valley. Wetlands were studied in two watersheds, defined as landscapes bounded by ridge lines, containing traditional small-town development and two watersheds containing suburban typologies. Land cover attributes were evaluated by analyzing ground-truthed, orthophotoquad data with a GIS. Water quality, the cover and biomass of emergent vascular plants, phytoplankton biomass, zooplankton biomass, and planktonic trophic transfer efficiency were measured in the wetlands during the fall of 2000, the summer and fall of 2001, and the summer and fall of 2002. Of the 13 variables measured, five exhibited typological differences according to the results of student *t*-tests. The interactions between these variables were quantified by least squares regression. Two key attributes of urban systems, i.e., the amount of vegetated buffer between the urban landscape and receiving waters and the amount of land in urban use, appeared to strongly influence water quality and ecosystem function in the wetlands studied. Nonpoint source loading and the success of exotic emergent macrophytic invasions varied directly with urban land use and inversely with buffer width. Trophic transfer efficiency declined with urban land use and increased with buffer width. The amounts of buffer and urban land use in a watershed appear to vary systematically with urban typology. Thus, watersheds that were developed in accordance with suburban design criteria exhibited more urban land use and less riparian buffering than did watersheds containing comparably scaled traditional small-town typologies.

INTRODUCTION

Decades of research have demonstrated that urbanization stresses and often degrades ecosystems (e.g., Nixon 1980, Scheuler 1994, Fulton et al. 1996, Weinstein 1996, Lerberg et al. 2000, Sala et al. 2000). Nevertheless, policies aimed at regulating development have failed to stay the rate of urbanization anywhere in the world. In fact, the rate of urbanization in the United States doubled during the 1990s (Chen 2000), when more than 615,000 ha of undeveloped and arable land were converted to urban use annually (Fodor 1999, Chen 2000). In 15 states, the loss of prime farmland between 1992 and 1997 more than doubled compared with the amount lost from 1987 to 1992 (American Farmland Trust 2003).

Urbanization is a term that applies to numerous landscape architectures, land use forms, and development strategies. Logically, different kinds of human developments should exhibit different urban attributes or different “intensities” of particular attributes (Kleppel 2002, Kleppel and DeVoe 2000). Whereas different degrees of “urban-ness” might have different impacts on ecosystems within the context of the larger land-use mosaic, few if any studies have addressed the differential impacts of urban development form or “typology” on ecosystems. We are engaged in testing the general hypothesis that different urban scales, i.e., “magnitudes” of urban-ness, and typologies, i.e., architecture and distributions of urban-ness, result in different impacts to ecosystems.

In this paper, we focus on a portion of that larger effort. We describe and compare the impacts of suburban and traditional small-town typologies on wetland ecosystems that drain watersheds in the

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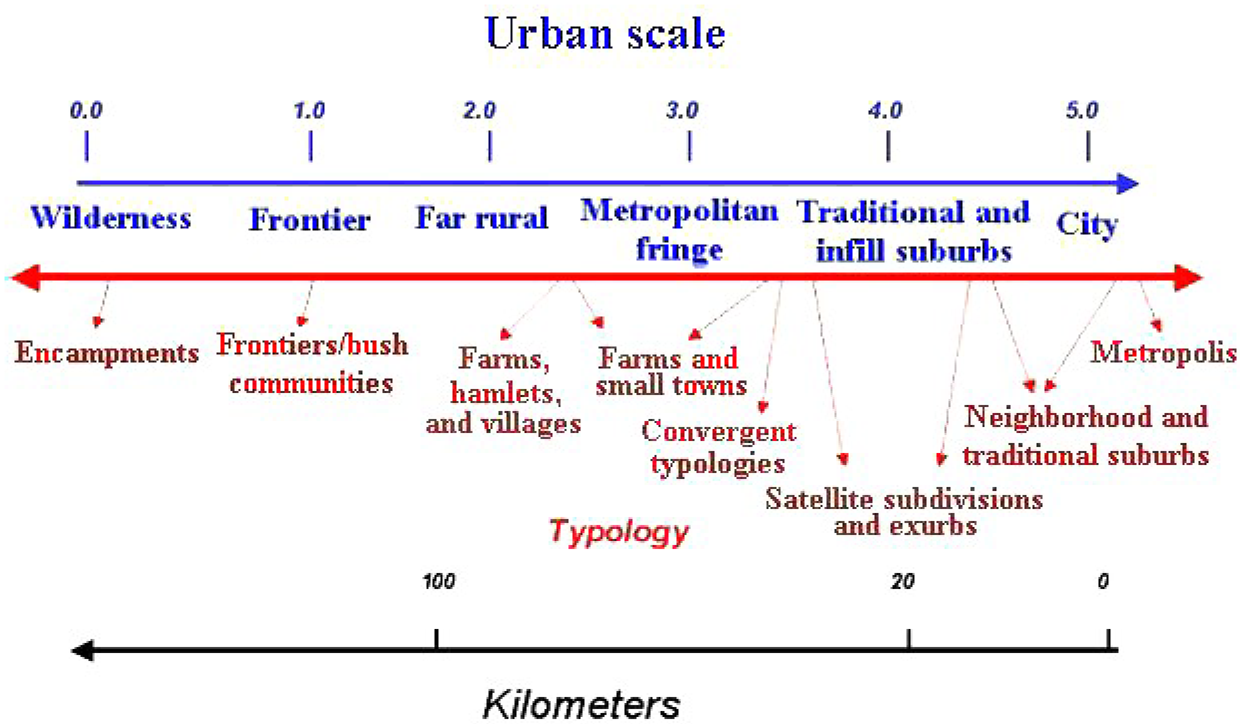
Hudson River Valley in upstate New York. We test the null hypothesis that the impact of urbanization on wetland ecosystems is independent of typology.

URBAN SCALE AND TYPOLOGY

The term "urban" is used here in its broadest sense to refer to the built environment and the extent to which infrastructure and services are provided to the largely sedentary human populations that occupy it. Urban systems are characterized by high human population densities relative to the average densities on the surrounding landscape. In addition, urban systems are enriched with infrastructure, and particular services are provided to their human populations at levels that exceed the average across the landscape. Finally, urban systems are governed by sets of laws and ordinances that are often enforced much more vigorously than on the surrounding landscape.

Fig. 1. A gradient scale describing the magnitudes of urban-ness associated with various kinds of landscapes and the distribution of their urban features or typologies.



Kleppel (2002) suggested that sedentary human systems should be thought to exist along a gradient of urban-ness that can be scaled with respect to a pair of extremes: wilderness and city (Fig. 1, Table 1). By scaling the gradient of urban-ness from wilderness to city, it is possible to compare urban environments and systematically evaluate their attributes and impacts on other systems. Wilderness, with a value of zero along this urban scale, represents the absence of urban-ness. Cities, with values above five, are intensely urban. Thus, although an outpost on the frontier meets the criteria of urban-ness when compared with the surrounding wilderness, it exhibits substantially less urban-ness than does a city or a suburb. Similarly, a rural hamlet or village is an urban environment relative to the agricultural landscape around it. The village is less urban than a large town that, in turn, is less urban than a medium-to-large city. Furthermore, a village 50 km from a city will, in general, exhibit more urban-ness than a village 100 km from a city. Between the small town and the city are suburbs. We recognize three categories of suburbs (see Duany et al. 2001, Kleppel 2002): (1) traditional suburbs connected to central cities, usually built prior to ~ 1950; (2) sprawling satellite subdivisions and commercial districts, i.e., low-density urban development in the metropolitan fringe; and (3) infill suburbs that extend from satellite suburbs toward the urban core.

Table 1. A scale of urbaness based on landscape attributes ranging from the wilderness (0) to the city (5).

Land use category	Urban scale	Natural attributes	Most local government authority	Distance from major city (km)	Relative municipal infrastructure cost†	Population density	People/urban ha
Wilderness (encampment)	0	high	Nation or state	> 150	0	Very low	N/A
Frontier (settlement)	1	High	Nation or state	80 to > 160	< 0.2	Very low	< 10
Rural(farm, small town)	2	Moderate	County or town	> 80	0.2–1.0	Low	30
Metropolitan fringe (farms, suburbs)	3	Low to moderate	County, village, town or city	15 to 80	< 0.2–1.6	Low	10–30
Infill suburbs (high-density suburbs)	4	Low	Town or city	1–30	1.1–1.6	High	6–30
City (small cities, megalopolis)	5	Low	City	0	0.5–1.5	High	> 30

†Values represent the estimated cost of the infrastructure per tax dollar, e.g., a value of 1.6 means that, for every dollar collected in taxes, $1.60 is needed to provide the services people require. Values greater than 1.0 are very common in modern suburbs.

There is presently no convention for assigning values along the urban scale to particular systems. In this study, ground-truthed, remotely sensed land-cover and land-use data (Jensen 1996, Cowen et al. 1999), municipal records, census data, local government and infrastructure attributes, and architectural diagnostics (Katz 1994, Duany et al. 2001) were used to assign values along the urban scale to the urban systems that we considered. Between any two major units on the urban scale, say 3.0 and 4.0, is a series of subunits, e.g., 3.1 to 3.9, that represent typologies or styles of development. Typologies describe the distribution of urban attributes on a landscape at a particular level of urban-ness (Table 2). Because architectural styles, constraints on human mobility, technology, and public ordinances determine the rules that govern landscape design and the distribution of urban attributes, typologies tend to be relatively standardized over a range of a few urban scale units. Since the late 1940s, for instance, the model for urban development has been the satellite suburb (U.S. Census Bureau 2001, Fabozzi 2002, Kleppel 2002), which is organized by single-purpose zoning and subdivision ordinances that define virtually all infrastructure and design specifications for public and private development. In traditional small towns and cities from the pre-World War II era, road networks are laid out in grids, whereas, in suburbs, the highway system tends to be a sparse hierarchy of local roads, feeders, arteries, and major highways.

Various kinds of small towns occur throughout rural America (scale value of 2.0) and the metropolitan fringe (scale value of 3.0). In the metropolitan fringe, hamlets, villages, small towns, and other traditional urban typologies are located between 3.1 and 3.4 on the urban scale. Suburban typologies characterized by “satellite” subdivisions, as in (2) above, are assigned values between 3.6 and 3.9. The value 3.5 is used to designate convergent systems that result from the emergence of suburban subdivisions and strip and "box-store" malls at the edges of small towns. Similar typological values may be appropriate when assigned to rural landscapes, which have an urban scale value of 2.0, and to infill suburbs with a scale value of 4.0. A value of 2.1 might, therefore, represent a hamlet > 100 km from a central city, whereas a value of 3.1 would be a hamlet in the metropolitan fringe. A neighborhood in the region between the metropolitan fringe and the city would receive a score of 4.1. The designation of urban scale and typology thus communicates useful information about the amount and distribution of urban-ness on any landscape. This system was used to designate sites in the present study.

Table 2. Urban typologies. Attributes associated with the distributions of population, infrastructure and urban services, and the dimensions of the urban environment. Special consideration is given to the typologies appropriate to this study.

	Traditional	Neo-traditional	Satellite suburb	Infill suburb	City
Relevant urban scales	2–4	3–4	3	4	5
Typological designation	0.1–0.4	0.1–0.4	0.6–0.9	0.1–0.9	0.1–0.9
Urban population/ha	25–65	25–50	< 12	12–40	> 75
Urban population (order of magnitude)	$< 10^4$	10^4	10^3–10^5	10^3–10^5	10^4–10^7
Percent impervious surface	< 30	< 50	50–70	60–70	> 70
Type of zoning and/or urban land use	3	2–3	1	1–2	3
Jurisdictions	Hamlet, town	Subdivision, town	Subdivision, city	Subdivision, city	Neighborhood, city
Principal road network	Grid	Grid	Sparse hierarchy	Sparse hierarchy to grid	Grid
Transportation scale	Ambulation	Ambulation	Automobile	Automobile, public transit	Automobile, public transit

METHODS

Wetland and watershed attributes

Four watersheds, i.e., small drainage systems defined by ridge-line boundaries, in the Hudson River Valley of New York State were studied. Each watershed is drained by an emergent marsh wetland (Cowardin et al. 1979) and contains either a traditional "small town" (the quotation marks indicate that "small town" is not a political designation) or a suburban development within its boundaries (Figs. 2 and 3, Tables 1 and 2). Wetlands were chosen as the focus of our measurements because, as relatively lotic receiving waters, they should reflect an integrated ecosystem response to the range of land uses in the watershed. Study sites will be referred to by their urban scale and typology designations: 3.1, 3.4, 3.7, and 4.2. Three of these sites, i.e., those with urban scale designations of at least 3.0, are in the metropolitan fringe; the site with a value of 4.2 is in an infill suburb.

Fig. 2. Orthophotoquad images of the watersheds containing small-town typologies in sites 3.1 (A) and 3.4 (B). In the watershed of site 3.1 the urban area is a hamlet; in that of site 3.4, a village. Land cover types are as follows: wetland (W), aquatic (Aq), forest/old field (F/O), agricultural (Ag), and rural residential (RR).

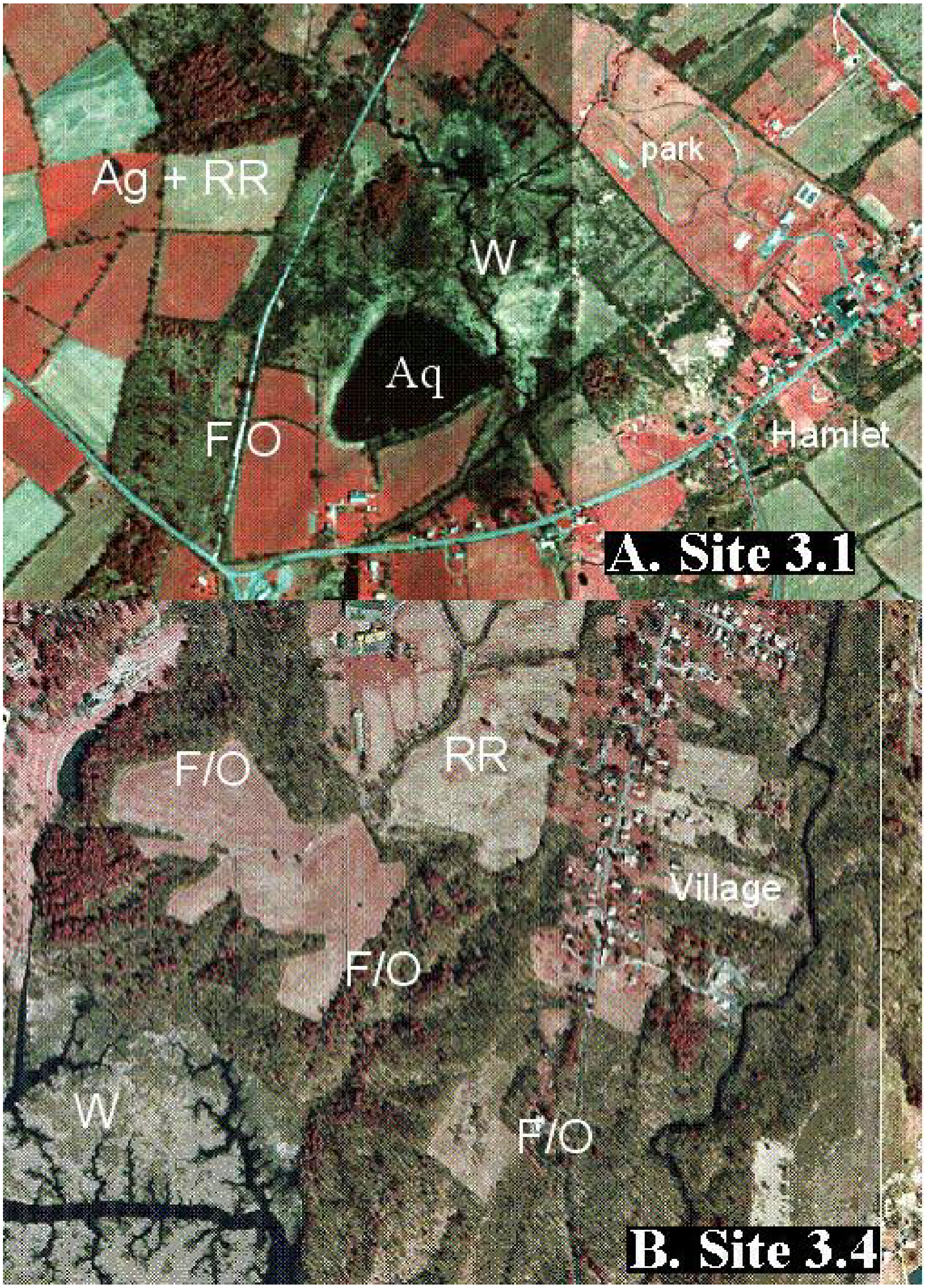
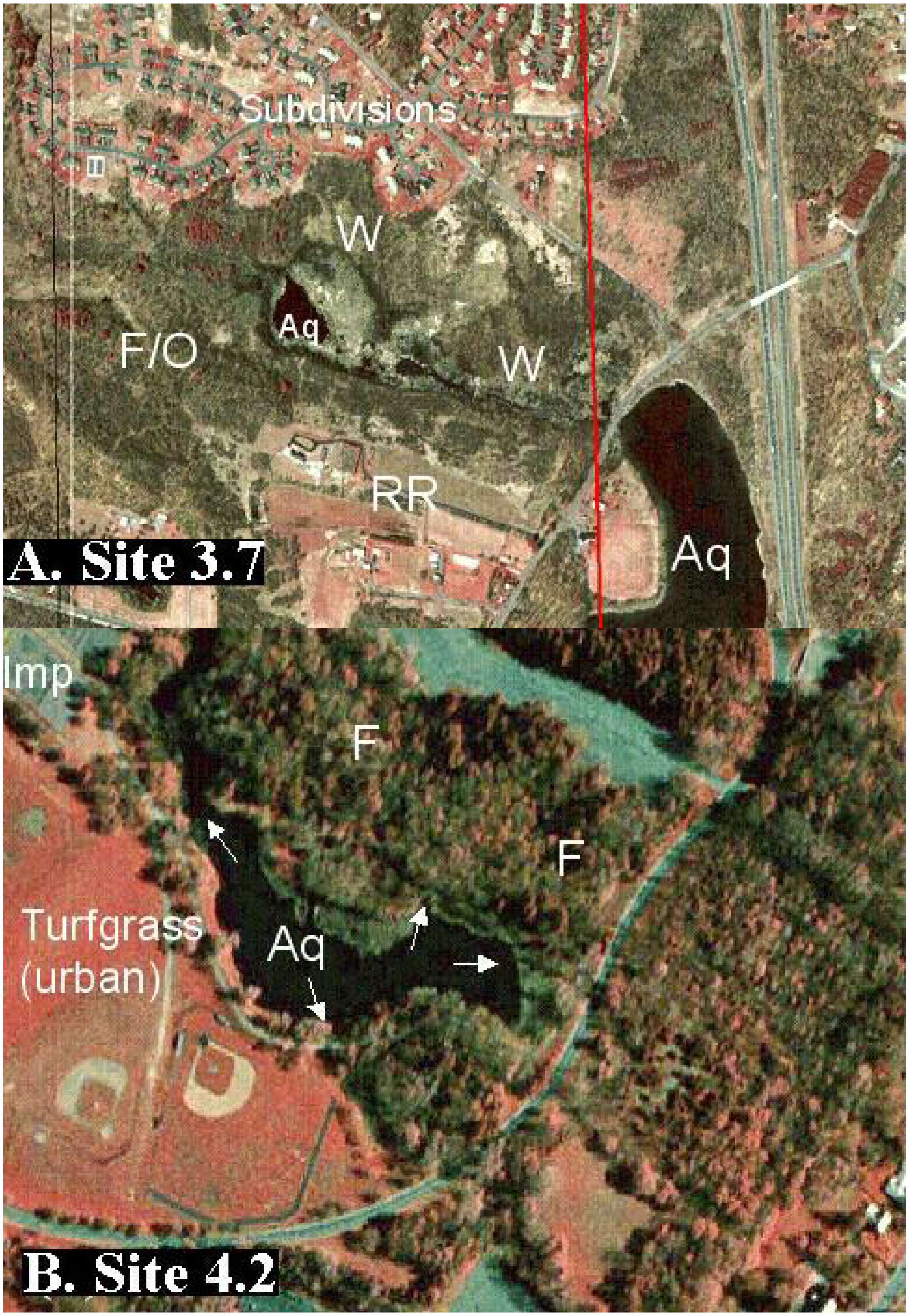


Fig. 3. Orthophotoquad images of the watersheds containing small-town typologies in sites 3.7 (A) and 4.2 (B). In the watershed of site 3.7 the urban area is a satellite suburb; in that of site 4.2, an infill suburb. Land cover types are as follows: wetland (W), aquatic (Aq), forest (F), forest/old field (F/O), agricultural (Ag), rural residential (RR), and impervious surface (Imp).



All of the emergent marshes in the study are located on silt-clay or loamy clay soils. Wetland 3.1 is stream-fed. In addition to marsh, it contains shrub-scrub, forested wetland, and wet meadow areas that were not studied. This wetland is influenced by the limestone-karst geology of the Helderberg escarpment (Driscoll and Childs 2002). Wetlands 3.4 and 3.7 are adjacent to the Hudson and Mohawk Rivers, respectively; wetland 3.4 receives input from two creeks in the watershed. Wetland 4.2 is part of an artificially created impoundment that emerges from a spring-fed stream that drains, via a spillway, to a creek that joins a tributary of the Hudson River.

To characterize the land cover and land use attributes within each watershed, orthophotoquads from the National Aerial Photography Program that were archived by the New York State GIS Clearinghouse were downloaded to ESRI™ software packages for processing geographic information. Initially, ArcView® version 3.2 was used; more recently, Arc GIS® version 8.1 was implemented. Watersheds were delineated by topographic ridge lines and by flow projections from digital elevation models obtained from the GIS Data Depot Website and processed with the Spatial Analyst accessory. Land cover attributes in the imagery were classified according to a scheme modified from Anderson et al. (1976). Four land-cover and land-use categories were evaluated: (1) urban, (2) rural-residential and agricultural, (3) forest and old field, and (4) wetland and aquatic. Classifications were confirmed during site visits (Table 3).

Table 3. Land cover and population distributions for the four watershed study sites 3.1, 3.4, 3.7, and 4.0.

	Traditional typology				Suburban typology			
Attributes	Hamlet	Village	Mean	SD	Satellite	Infill	Mean	SD
Designation	3.1	3.4			3.7	4.2		
Watershed (ha)	100.4	200.0	150.2	49.8	182.9	70.2	126.6	56.4
Land cover use (%)								
Urban	1.7	12.6	7.2	5.5	41.0	85.8	58.9	26.9
Agricultural	50.1	49.8	49.5	0.2	1.5	0.8	0.8	0.0
Forest/old field	19.6	26.0	22.8	3.2	29.2	12.7	26.2	13.5
Wetland/aquatic	28.6	11.6	20.1	8.5	28.3	1.5	14.9	13.4
Population density in watershed (X 1000)	0.3	1.5	1.1	0.5	1.5	2.0	1.8	0.3
People/urban ha	175.0	59.5	117.3	57.8	20.0	33.3	26.7	6.7

Urban landscapes in land cover category (1) include both small-town and suburban typologies. They were classified on the basis of the relative density and distribution of residential, commercial, and institutional development; percentages of impervious surfaces in the watersheds (see Table 3); and evidence of urban services such as street and traffic lights, fire hydrants, police stations, fire and EMS stations, and libraries. Rural residential parcels, (2), are typically outside hamlet or village limits. Lot sizes are 0.8–4.0

ha or larger but are not cultivated other than for ornamental and small vegetable gardens, and urban services are typically sparse or distant. Agricultural land consists of the working landscape, including pastures, crop rows, barns, pens and other fenced areas, and silos and residences, but excluding woodlots and fields left fallow. The locations of forests and old fields, (3), were resolved in the orthophotoquads and verified during site visits. Wetlands and aquatic features (4) were also identified in the orthophotoquads and were verified by vegetation analysis (Reschke 1990) and determination of soil hue and chroma with Munsell charts (Tiner1999).

The proportion of each land cover type was quantified in the imagery with ESRI™ software (see above). Vegetated riparian buffer widths were estimated at three to eight randomly selected locations in each image. At each location, the linear distance between the edge of a wetland and the edge of the area designated as urban land cover that contained forest and/or old field was measured, and the mean distance was computed.

Urban development within study watersheds

Watersheds ranged from 70.2 to 200.0 ha, with means of 150.4 ha for those containing a traditional small-town development (sites 3.1 and 3.4) and 126.6 ha for those developed with a suburban typology (sites 3.7 and 4.2, Tables 2 and 3). Mean population densities were 1100 and 1800 for small-town and suburban watersheds, respectively.

Two of the wetlands in the metropolitan fringe drain watersheds that contain urban development in the traditional “small-town” typology (Fig. 2, Table 3). One of the traditional communities, in watershed 3.1, is a hamlet of approximately 300 people; the other is a village of about 1500 people. Typologically, both communities exhibit traditional building and landscape architectures (Table 2; also see Katz 1994, Duany et al. 2001). The hamlet is simply a concentration of small businesses, churches, municipal buildings such as a town hall and a fire department, and other urban features, e.g., two community parks and two cemeteries, along a two-lane state highway. It is the seat of a town government, but it is not self-governing.

The other community, in watershed 3.4, is a village. It is larger than the hamlet, with an off-highway Main Street, shops and residences, public buildings, and several predominantly residential side streets. It is a self-governing jurisdiction. Beyond the urban boundaries of both communities, land uses include some agriculture, several large estates (rural residential), and smaller rural residential parcels. Watershed 3.4 contains a New York State conservation area (forest) and an old field.

The third wetland, in watershed 3.7 (Fig. 3A), also in a metropolitan fringe, drains a watershed that contains several subdivision developments comprising approximately 500 single and multifamily residential units. The typology, based on building and landscape architecture and local ordinance structure, is suburban (see Table 3; also Duany et al. 2001). Commercial development is prohibited within the single-purpose residential zone in the watershed.

The fourth wetland, in watershed 4.2 (Fig. 3B), is on a university campus in an older infill suburb built during the 1960s and 1970s within 10 km of a mid-sized city of approximately 96,000 residents (U.S. Census Bureau 2001). The watershed drains an institutional landscape, principally by surface runoff and stormwater sewerage. Approximately 2000 residents live within the watershed.

The actual locations of the watersheds in this study have not been disclosed to protect the privacy of local property owners. Specific information can be requested from the senior author.

Sampling

Data collection began in September 2000 at sites 3.1 and 4.2. Sites 3.4 and 3.7 were added in July 2001. All four sites were visited in September and October of 2001 and in July, September, and October of 2002. At each wetland, temperature, conductivity, dissolved oxygen (DO), pH, turbidity, and chlorophyll fluorescence were measured at two to four randomly selected sites with standing water or ponds approximately 0.5 m from the land-water interface. On each visit, three or four 1-m^2 plots of emergent vegetation were selected at random along the land-water interface by tossing a 1 x 1 m^2 frame constructed of three PVC tubes. The frame was open on one side and, when tossed open end first, enclosed an area of 1 m^2 upon landing. Vascular plant cover was assessed as the percentages of the following categories: cattails and other large native herbaceous and small woody species; grasses, sedges, rushes, and small herbaceous species; invasive exotics; and litter, bare ground, or standing water. To estimate emergent macrophyte biomass, the plants within a quadrat randomly selected from among those assessed for cover attributes were cut at ground level and returned to the laboratory, where they were dried and weighed.

Temperature, conductivity, and DO were measured with a YSI model 85 multimeter, and pH was measured with a Hanna Instruments (HI 9023) portable pH meter. Turbidity, in nephthal turbidity units, and relative chlorophyll fluorescence were measured with a Turner Designs Aquafluour 8000 series handheld fluorometer. Relative chlorophyll fluorescence data were converted to estimates of chlorophyll *a* (chl) concentration by regression, based on the fluorescence of serially diluted (90% acetone), authentic standards (Sigma Corporation). These concentrations were then confirmed by spectrophotometry using a Pharmacia Biotech Ultraspec 1000 (Strickland and Parsons 1972, Kleppel et al. 1985) and converted to estimates of phytoplankton carbon (C) biomass by multiplying by measured C:chl ratios (Kleppel 1992).

Zooplankton biomass at each wetland was estimated with collections from casts in standing water of a net with a mouth diameter of 25 mm and a mesh of 63 μm. These were returned to the laboratory for determination of dry weight during 2000 and 2001 or displacement volume during 2002. Dry weight or displacement volume was converted to zooplankton carbon biomass using Eqs. 1A or 1B as follows (Wiebe 1988):

$$\log(C) = 1.009 \log(DW) - 0.504 \quad (1A)$$

and

$$\log(C) = 1.220 \log(DV) + 1.749, \quad (1B)$$

where C = mg carbon/m^3, DW = mg dry weight/m^3, and DV = ml displacement volume/m^3.

The modification in protocol from dry weight to displacement volume was made for expediency, but is valid because Eqs. 1A and 1B are intercalibrated.

In the evaluation of ecosystem functionality, the efficiency of energy flow between trophic levels is arguably as important as the production of biomass within a trophic level (Lindeman 1942; also Kleppel, *unpublished manuscript*). Therefore, a planktonic trophic transfer function or efficiency, K_1, was estimated as:

$$K_1 = C_{II}/C_I, \quad (2)$$

where C_{II} and C_I represent the biomasses of zooplankton and phytoplankton, respectively (see Odum 1973).

Data analysis

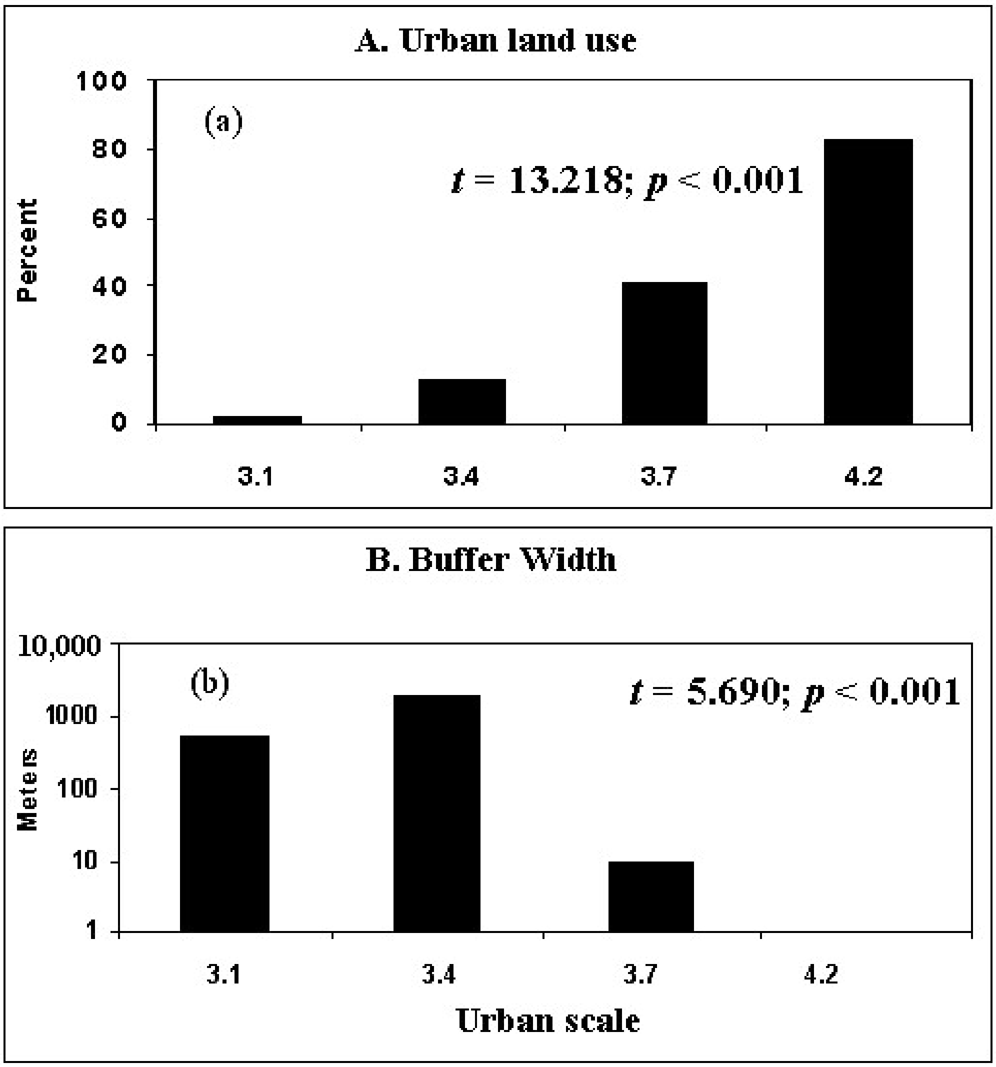
Data from the two wetlands that drain the traditional small towns, 3.1 and 3.4, were pooled to produce a data set that could be compared with the pooled data from the two wetlands that drain suburban watersheds, 3.7 and 4.2, to test the null hypothesis of no difference between attribute means distinguished by urban typology. Student *t*-tests were performed with SPSS software to detect differences between typologies with regard to land use, water quality, and ecological variables. Quantification of the relationships among the variables that the *t*-tests revealed to be typologically distinct was accomplished by least squares regression analysis. Data expressed as percentages, e.g., urban land use, invasiveness, were log-transformed prior to analysis.

RESULTS

Landscape attributes

In the watersheds that support typologies 3.1 and 3.4, an average (mean ± standard error) of 7.2 ± 5.5% of the land cover was urban (Table 3), compared with 58.9 ± 26.9% urban land cover in the suburban, i.e., 3.7 and 4.2, watersheds. The difference between the amount of land in urban use in small-town and suburban watersheds was significant (Fig. 4A; t = 13.22, $p < 0.001$). Suburban watersheds contained no agricultural and little rural-residential land cover. On average, about half as much forest and old field were present in suburban watersheds as in watersheds containing traditional typologies. Depending upon their locations, forests and old fields may buffer receiving waters from runoff and other impacts introduced by contact with the urban landscape. Mean vegetated buffer widths at sites 3.1 and 3.4 were approximately 500 and 2000 m, respectively. At site 3.7, forest and old-field buffer widths were drastically reduced, with turf grass strips < 10 m wide separating residential units from the wetland boundary. The mean buffer width was 3.1 ± 1.2 m. At site 4.2, much of the buffering capacity of the forested and turf grass landscapes was obviated by steep slopes and six storm sewers that drain impervious surfaces on the university campus directly into the wetland and adjacent pond. Typological differences between buffer widths were significant (Fig. 4B; $t = 5.69$, $p < 0.001$).

Fig. 4. Percentage urban land cover (A) and mean vegetated buffer width (B) in each of the four watersheds in this study. Student *t*-tests were performed on pooled data from each typology, i.e., sites 3.1 and 3.4 together vs. sites 3.7 and 4.2 together, to test H_0 of no difference between typological means. The dark bars indicate that typological differences between watersheds with small-town typologies and those with suburban typologies were significant.



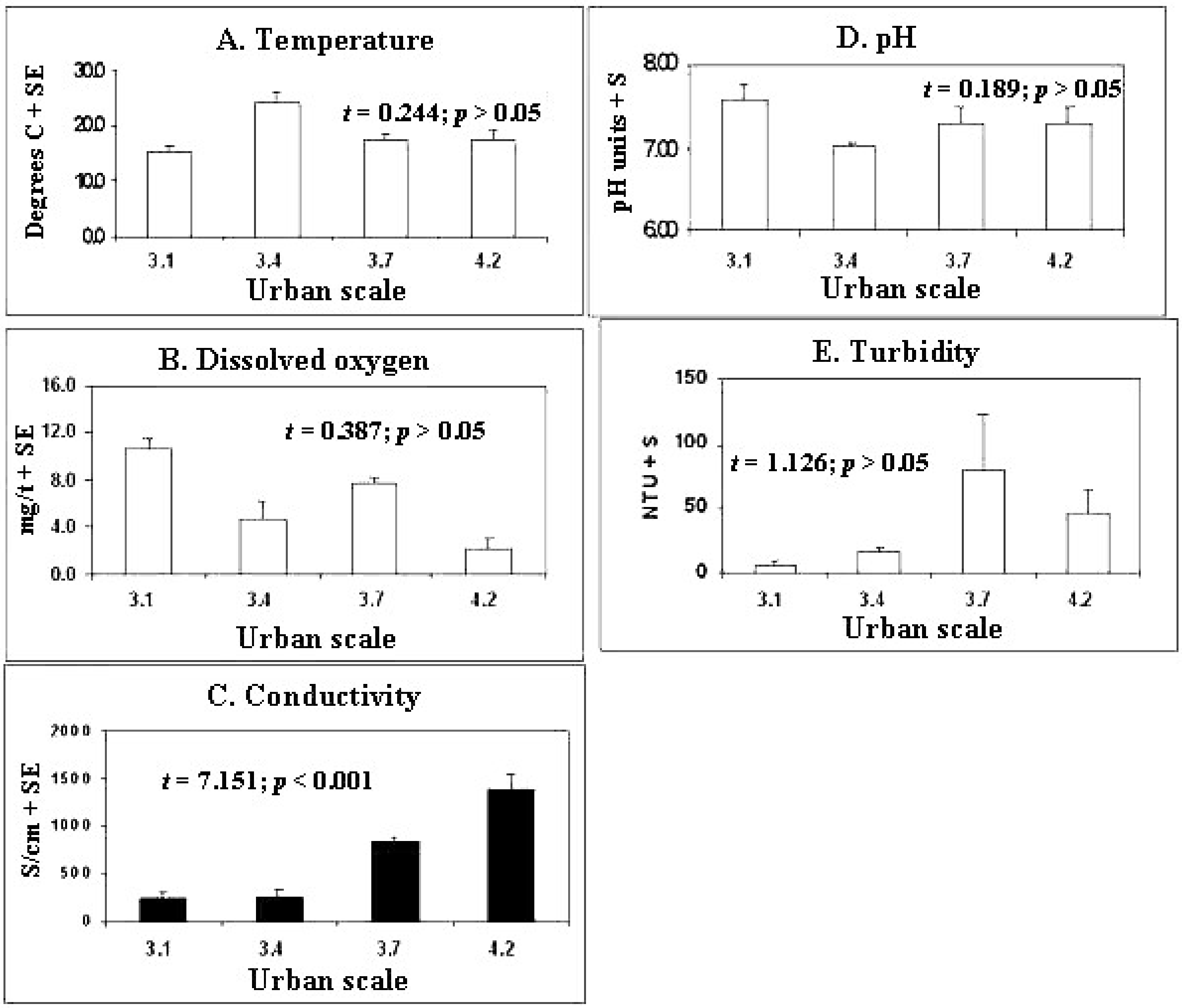
Water quality

Conductivity, which we assume here to be reflective of the magnitude of nonpoint source loading (Herlihy et al. 1998, Nuñez-Delgado et al. 2001, Yuan and Norton 2003), was distinguished by urban typology (Fig. 5C; $t = 7.15$, $p < 0.001$). Mean conductivity was higher in suburban (1152.8 ± 429.9 μS/cm) than in traditionally developed (239.1 ± 189.5 μS/cm) watersheds. Temperature, dissolved oxygen, pH, and turbidity were not distinguishable between typologies (Figs. 5A,B,D,E; $p > 0.05$).

Ecosystem attributes

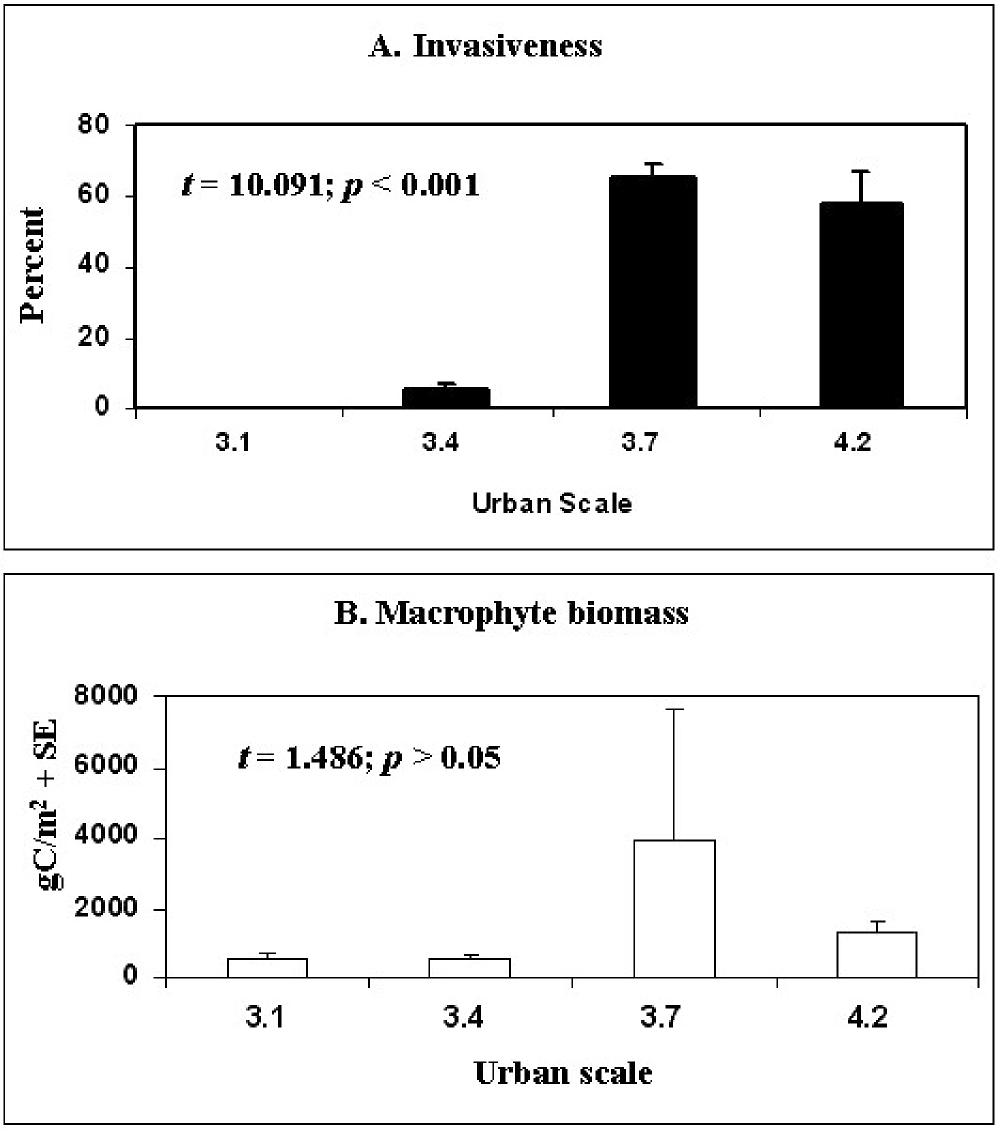
The emergent plant communities of the wetlands in the suburban watersheds appeared more susceptible to successful invasion by exotic species than did those in the watersheds in which traditional typologies characterized the urban landscape (Fig. 6A). At site 3.7, purple loosestrife (*Lythrum salicaria*), an aggressive native of eastern Europe, composed, on average, 65.0 ± 22.2% of the emergent vascular plant cover. At site 4.2, the invasive common reed (*Phragmites australis*) composed, on average, 60.9 ± 25.1% of the emergent plant cover. Conversely, wideleaf cattail (*Typha latifolia*) and various native grasses were dominant at site 3.1, and *T. latifolia* and arrow arum (*Peltandra virginica*) were dominant at site 3.4.

Fig. 5. Water quality data (mean ± 1 SE) in the wetlands for each study site. Students *t*-tests were performed on pooled data from sites with either small-town or suburban typologies, i.e., sites 3.1 and 3.4 together vs. sites 3.7 and 4.2 together, to test H_0 of no difference between typological means. The dark bars indicate that typological differences between watersheds with small-town typologies and those with suburban typologies were significant. Light bars indicate that the difference between typological means was not significant.



The mean biomasses of emergent macrophytes, phytoplankton, and zooplankton in the wetlands of traditional and suburban watersheds were not statistically distinguishable (Figs. 6B, 7A,B). However, the large variability in macrophyte biomass within typologies may have been influenced by our sampling scheme, which did not account for seasonal growth patterns or the loss of biomass between summer and fall. Typological differences in mean trophic transfer efficiencies, K_1, represented by the ratio of zooplankton to phytoplankton biomass, were significant (Fig.7C; $t = 4.844$, $p < 0.05$).

Fig. 6. (A) Emergent vascular plant (macrophyte) biomass (mean ± 1 SE) at wetlands in each of the four study sites, and (B) percentage of invasive macrophyte species (mean ± 1 SE) at wetlands in each of the four study sites. Student *t*-tests were performed on pooled data from sites with either small-town or suburban typologies, i.e., sites 3.1 and 3.4 together vs. sites 3.7 and 4.2 together, to test H_0 of no difference between typological means. The dark bars indicate that typological differences between watersheds with small-town typologies and those with suburban typologies were significant. Light bars indicate that the difference between typological means was not significant.



Interactions between urban typology, water quality, and ecosystem function

We used regression to quantify the relationships between the five variables that were distinguished typologically by the *t*-tests. The equation of the line or curve of best fit was determined for each interaction. The analysis demonstrated that, as urban land use increased (Figs. 8A,B,C) and buffering decreased (Figs. 8D,E,F), important changes occurred in water quality and ecosystem structure and function. Conductivity rose, emergent plant communities became more susceptible to invasion, and the efficiency of energy flow between trophic levels in the plankton declined exponentially. These alterations led to secondary associations between trophic transfer efficiency, conductivity, and invasiveness (Figs. 8G,H,I).

Fig. 7. (A) Mean phytoplankton biomass (mean ± 1 SE), (B) mean zooplankton biomass (mean ± 1 SE), and (C) mean trophic transfer function (mean ± 1 SE) at wetlands in each of the four study sites. Student *t*-tests were performed on pooled data from each typology, i.e., sites 3.1 and 3.4 together vs. sites 3.7 and 4.2 together, to test H_0 of no difference between typological means. The dark bars indicate that typological differences between watersheds with small town typologies and those with suburban typologies were significant. Light bars indicate that the difference between typological means was not significant.

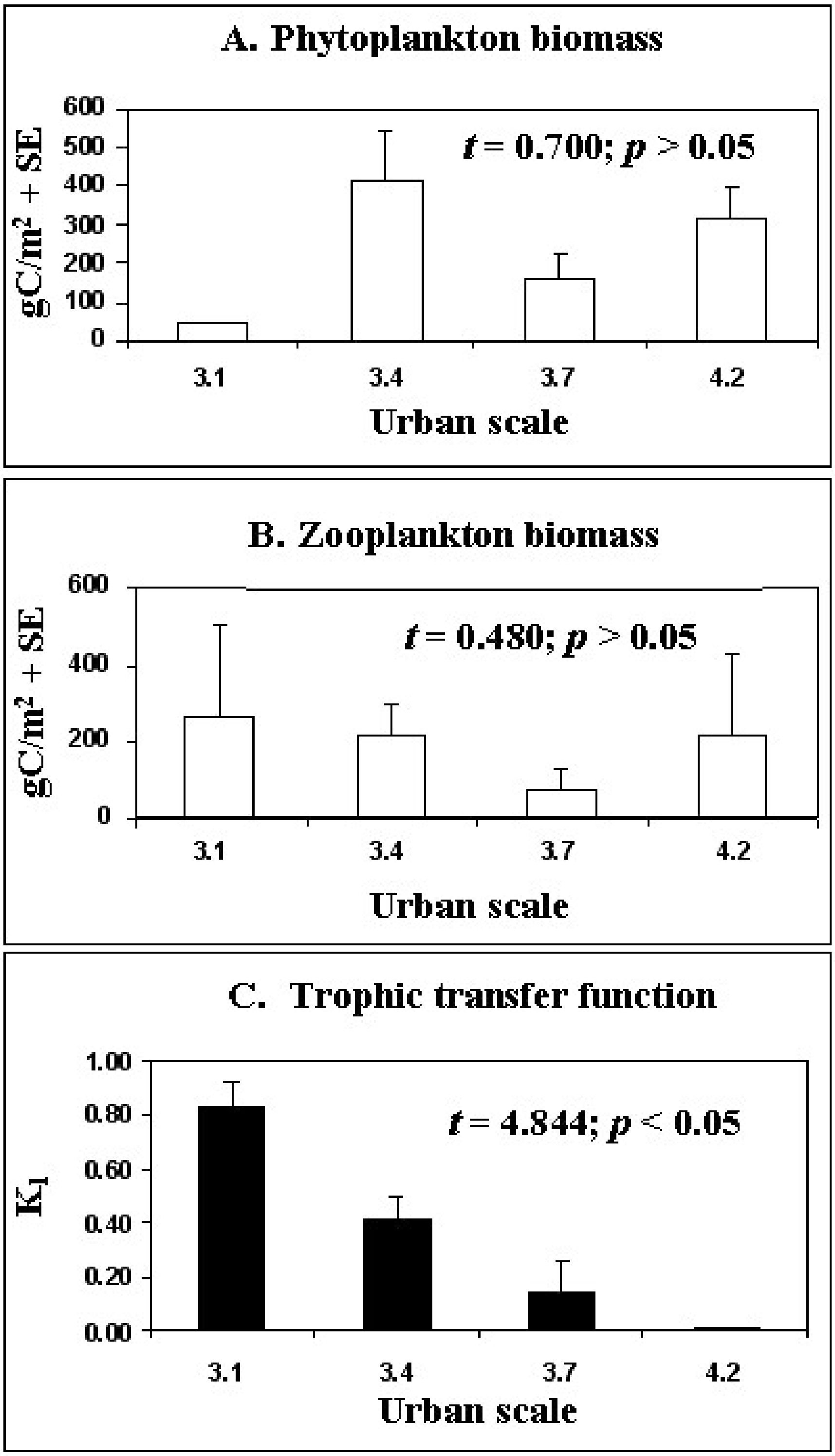
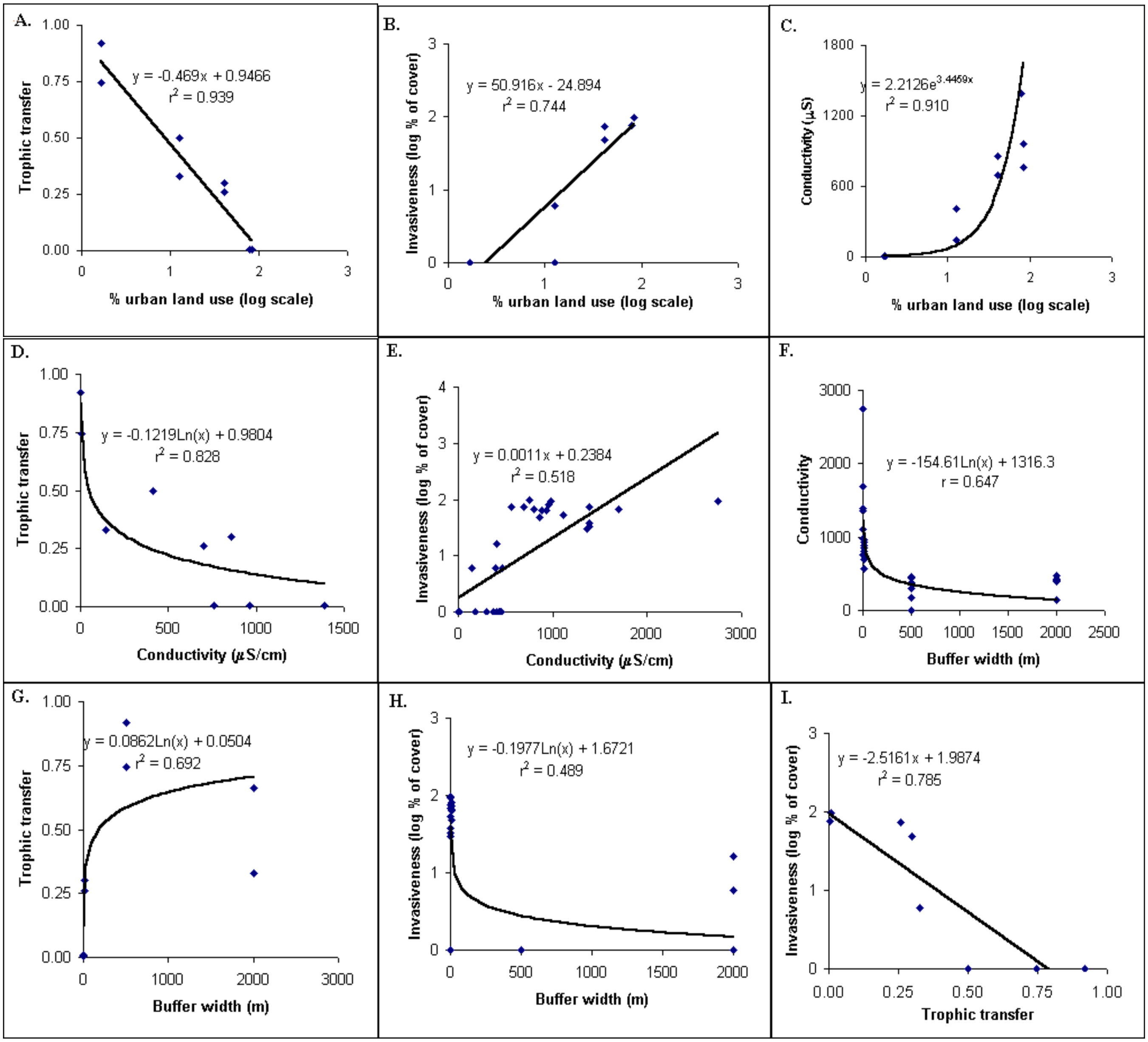


Fig. 8. Interactions between variables identified as having typologically distinct means. Analysis was by least squares regression. Equations describe the curve of best fit. All slopes were differed significantly from zero (p < 0.05).



DISCUSSION

Urban attributes, typology, and impacts to wetlands

Five of the variables measured in this study—urban land cover, buffer width, conductivity, i.e., nonpoint source (NPS) loading, macrophyte invasion, and trophic transfer efficiency—were differentiated by typology. The observed interactions among variables in the regression analyses were not unexpected. For example, as buffer widths decrease, one would expect NPS loading to increase (Correll 1997, Wenger 1999). Similarly, the direct correlation between NPS loading and urban land use in a watershed is well established (Scheuler 1994, Lerberg et al. 2000), and the susceptibility to biological invasion that accompanies the loss of buffering capacity is predictable (Elton

1958, Correll 1991, Scheuler 1995, Mack et al. 2000). It was unclear why temperature, pH, dissolved oxygen (DO), and turbidity were not distinguished typologically (Figs. 5A,B,D,E), because these variables are frequently used to detect human disturbance. In part, seasonal variability, particularly in temperature, and complex interactions among forcing functions such as atmospheric or geologic factors may have added unexplained variability to pH. Nor was the influence of the algal bloom cycle on DO and turbidity, and to a lesser extent pH, extracted from the data. However, as data collection at the study wetlands continues, we expect to more clearly resolve true typological differences in environmental quality if they exist.

What is novel about our findings is that the observed relationships between land use attributes and environmental quality were consistently associated with typology. That is, the way that people and infrastructure are distributed on the landscape seems to influence the kinds and magnitudes of the impacts that occur. Although in this study traditional small-town and suburban typologies supported populations of a similar size, the traditional typology required considerably less urban land cover to do so. Thus, mean urban population densities in traditional small-town typologies (117.3 ± 57.8 people/urban ha) were more than four times higher than in suburban typologies (26.7 ± 6.7 people/urban ha), but, as seen in Table 3, the number of people per hectare of watershed was, on average, twice as high in suburban watersheds (14.2 people/ha) as in traditionally developed watersheds (7.3 people/ha). This is because almost eight times as much land was characterized as urban in the suburban watersheds as in watersheds containing small towns. Further, the engineering and landscape architecture of suburban systems seems more likely to promote environmental and ecological impacts than do traditional typologies.

For example, in the suburban typologies (Figs. 2A,B) in this study, buffers were compromised by building close to wetlands, as at site 3.7, and by delivering runoff directly to receiving waters via storm sewers at site 4.2. This is not to suggest that development decisions made in small towns are, or historically have been, necessarily environmentally enlightened. The design of America's small towns has rarely been purposefully conscientious or conscious of environmental quality. Small towns and cities were often built close to the water, vegetation was removed, and marshes were drained or dredged to expedite commerce.

When possible, however, development was set back from the water. Urban development in flood plains was thought to be unwise. Wetlands were generally considered undesirable for development and were avoided or drained. However, during the 20th century, and particularly since World War II, the enormous increase in the use of pesticides to control annoying and disease-carrying insects (Eisenberg 1998), along with the passage of the National Flood Insurance Program of 1968, have permitted encroachment into riparian and wetland areas that were once inaccessible. Similarly, 19th century planners did not seek to restrict the urban landscape or the spread of impervious surfaces. However, constraints on mobility in urban environments demanded an efficiency of scale that led to what today is referred to as compact or cluster and mixed-use development (Arendt 1996). The rescaling of the American landscape to the automobile (Downs 1992, Kunstler 1994) and the advent of single-purpose zoning (Duany et al. 2001) have permitted the spread of urban land-use attributes to an extent that would have limited the accomplishment of normal business a century ago, given the available forms of transportation.

The availability of the automobile has meant that urban development in the United States is now associated with the extensive conversion of land to impervious surfaces. Prior to World War II, the ratio of urban land per person was 0.1; today it is between 2 and 8 depending upon location (Berkeley-Charleston-Dorchester Council of Government 1997, Fabozzi 2002, Kleppel 2002; see also J. Allen, *personal communication*). The impervious surfaces that are being created are incapable of absorbing and processing runoff and other products of the urban environment, which are being released into the environment at rates that are at least an order of magnitude higher than they were a century ago (Berkeley-Charleston-Dorchester Council of Government 1997, Eisenberg 1998, Fabozzi 2002). The impacts on natural habitats, biological diversity, and environmental quality of this post-World War II rescaling of the American urban landscape are increasingly documented and uniformly negative (Wahl et al. 1996, Weinstien 1996, Kleppel 2002). For example, in 1999, land uses in watersheds along the Okatee River estuary in South Carolina consisted largely of agriculture, small towns, and “fishing villages.” Water quality in the river was good, even though little was done to minimize the impacts of local land uses (Van Dolah et al. 2000). However, between 1999 and 2002, the amount of urban development, largely in the form of the suburban typology described in (2) above, increased by more than 500% (J. Holloway, *personal communication*). During the same period, water quality declined significantly in the Okatee (G. Scott, *personal communication*), although it is unclear whether this change is caused simply by increased population or by typology. Probably both are involved. Also, although local climatological factors, particularly drought, may have contributed to changes in water quality, there is evidence from coastal South Carolina that the same human population might have a smaller ecological impact if suburbs were replaced by traditional urban typologies (Kleppel et al. 2004).

Implications for land use policy

Although federal and state policies strongly influence the way land is used in the United States (Salkin 2002), the vast majority of land use decisions are made at the local level (Dale et al. 2000). Even though the property owner generally has jurisdiction, land use decisions at this level are strongly influenced by local ordinances and information arriving from the market (Eppli and Tu 1999, Hulse and Ribe 2000). Despite considerable evidence that urbanization has negative ecological impacts, there is a paucity of data on whether alternative development styles might provide a more sustainable relationship with adjacent ecosystems. Thus, even neo-traditional typologies, which emphasize classical landscape design (Katz 1994, Table 2), have been limited in what they can accomplish with compact development approaches (Arendt 1992, Duany et al. 2001) by the lack of data linking typology with impact. Evidence of typologically distinguishable environmental impacts must be available before municipal authorities will be inclined to grant variances or create ordinances legitimizing novel stormwater management schemes, the reduction of impervious surfaces, or the protection of riparian buffers.

CONCLUSION

The growth of urban systems is a global phenomenon. It is unrealistic to believe that urban development will be constrained to any great extent merely by demonstrating environmental impacts. Instead, we suggest that different styles of development may have different impacts on ecosystems. Typologies characterized by reduced urban land use, i.e., more compact distribution of urban features, and extensive buffering appear less stressful to ecosystems than those that expand the urban landscape and/or compromise the buffering capacity of natural vegetation. Typology is increasingly recognized at local, state, regional, and federal levels as a factor to be considered when addressing many of the social and economic consequences of urbanization and sprawl (American Farmland Trust 2001, Salkin 2002). Until now, evaluations of how different urban typologies relate to ecosystems have been lacking. The present study suggests that typological differences in urban development may help to reduce negative impacts on wetlands and possibly on other aquatic systems. However, it should be emphasized that the present study represents a very limited first step in the evaluation of the hypothesis of no difference in the impact of urban typology on ecosystem integrity. Many more samples and a larger range of ecosystem types must be evaluated. Studies should be conducted in other geographic regions and over a wider range of seasons. Although a great deal remains to be learned about the relationship between how we build and the impact we have on ecosystems, the results of this study suggest that the investigation of the ecological impacts of different urban typologies is worth pursuing.

Responses to this article can be read online at:
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