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Beyond the urban gradient: barriers and opportunities for timely studies of urbanization effects on aquatic ecosystems

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Abstract. Many studies have shown that streams degrade in response to urbanization in the watershed. These studies often are based on use of biotic and abiotic variables to measure stream health across a gradient of land cover/land use. The results of these studies can be applied to other urban systems, but often fail to provide a mechanistic understanding of the urban impact, in part, because of the nature of the experimental design. We analyzed the advantages and disadvantages of using environmental gradient studies to further understanding of urban stream systems. We also evaluated alternative experimental design approaches, including best management practice monitoring, long-term watershed studies, paired-watershed studies, and before–after control–impact studies, which could be used to complement the gradient approach. We illustrate these theoretical discussions with an urban paired-watershed case study in the Etowah watershed in northern Georgia. Our goal is to move experimental designs in a direction that will further our mechanistic understanding of the effects of existing urbanization on aquatic ecosystems and will provide opportunities to evaluate stream responses to environmentally sensitive urban land cover.

Key words: urbanization, experimental design, paired watershed, environmental gradient, urban stream, development, impacts.

Over 50% of the global population now lives in cities, and this proportion is projected to increase to nearly 70% by 2050 (UN 2007). A detailed understanding of how cities expand, develop, and function as a unit is just beginning to emerge (Bettencourt et al. 2007, Alberti 2008), but the study of urban effects on surrounding landscapes has a longer history. Urban development and urban land conversion have consistently negative effects relative to unaltered ecosystems (Vitousek et al. 1997). Existing urban areas contain ecosystems that have been dramatically and often irreversibly altered (Batty 2008, Grimm et al. 2008). Damage is particularly common for aquatic ecosystems, where the seemingly ubiquitous *urban stream syndrome* is characterized by flashier hydrographs, higher pollutant levels, highly modified channel geomorphology, and decreased biotic richness (Walsh et al. 2005b).

Relationships between aquatic ecosystems and urbanization are complicated by the temporal and spatial heterogeneity in urban areas and the sur-

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rounding landscape. For example, Lohse et al. (2008) assessed the impacts of landuse conversion on the quality of spawning substrate and found that existing urban development had larger marginal impacts than residential or agricultural land use. However, development on large (>0.4 ha) residential or agricultural parcels was a larger overall threat to spawning substrate than was further development in already urban environments, primarily because land in urban sites was unavailable for conversion. This type of result has been used as the rationale for management strategies in which planning recommendations direct new development into areas that already have reached a threshold of buildout density (e.g., 50% of the area's parcels have already undergone development) or impervious surface cover (Allan 2004, CWP 2003, Moglen and Kim 2007).

Our understanding of the urban effect on aquatic ecosystems is largely a result of the way in which experiments or evaluations of the effect have been done. One approach is to use a *space-for-time* substitution (SFT). Study sites are selected based on the degree to which they are urbanized, typically ranging from very high levels of urbanization or impervious surface (e.g., downtown urban core) to very low levels (e.g., old-growth forest) with a range, or gradient of urbanization, between the extremes. The landuse/land cover gradient is then related to a variable or index of interest, such as turbidity, Index of Biotic Integrity, or fish species richness. The resulting signal is used as a measure of the urban effect on the variable or indicator of stream health. Thus, rather than making direct observation of responses to changes in land use/land cover over time, researchers substitute spatial diversity for the temporal change. The urban signal can take a number of forms as development progresses, depending on the level of environmental protection afforded by the development process and the level of historical impacts.

Figure 1 illustrates 3 theoretical ways that urban effects might occur. One form of degradation could be a simple linear response in which increasing levels of urban land use/land cover result in increasing degradation (a, Fig. 1). The SFT approach documents this type of change well (see **SFT Approaches** below). However, when historical land use is taken into account, the relative urban effect across the gradient might be less harmful than was some previous land use or landuse change might already have altered the aquatic ecosystem (e.g., Brown et al. 2009; b, Fig. 1). Use of environmentally sensitive development practices that minimize damaging land cover characteristics, such as the amount of hydrologically connected impervious area, is a 3rd way in which development activity might affect aquatic ecosystem health (c, Fig. 1). In this case, urban land use might increase significantly, but the alteration might be smaller than would have occurred had traditional development forms been used.

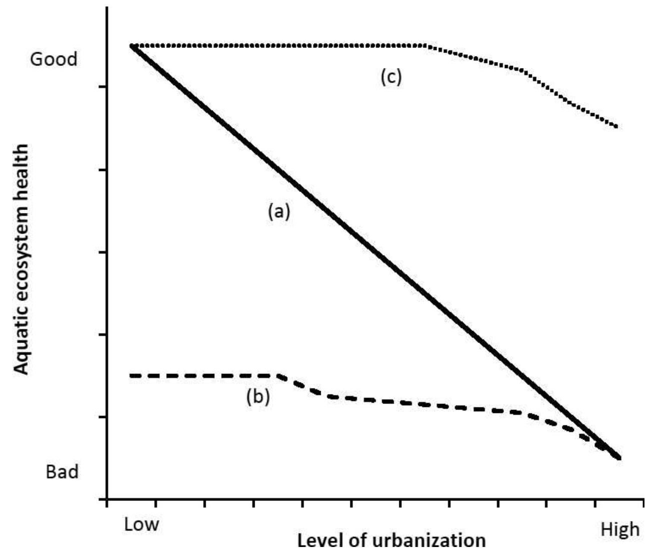

FIG. 1. Projected changes in aquatic ecosystem health with increasing urbanization: a) conversion of native land to conventional urban development, b) conversion of land with historic legacy of other human land uses to conventional urban development, and c) hypothesized trajectory following conversion of native land to innovative forms of urban development.

Our 1st goal was to demonstrate the utility of SFT approaches for providing novel and useful insights into the forms and functions of urban aquatic ecosystems. We synthesized recent literature based on the gradient approach and identified the advantages and limitations of these studies, particularly within the context of novel forms of development. Our 2nd goal was to identify and evaluate other research approaches that can complement and augment SFT studies by testing variables that are difficult to tease apart with a single method. Our 3rd goal was to illustrate how one of these approaches, a paired-watershed design, is currently being implemented in the Etowah watershed north of Atlanta, Georgia. The limitations identified in this case study demonstrate how a mixed-method approach might overcome inherent flaws in single study designs.

SFT Approaches

Many studies have evaluated the urban signal with the SFT approach. Paul and Meyer (2001), Allan

(2004), and Walsh et al. (2005b) reviewed and synthesized much of the existing work on urban streams and landuse/land cover change before 2004. Rather than repeat their analyses, we reviewed studies published in 2006 to 2008.

Abiotic effects

The major physical effects reported in recent studies of urban impacts are consistent with those identified in past studies. However, surprisingly few recent studies targeted physical effects as their primary study focus, possibly because relating urban effects to physical alteration of the environment is mechanistically straightforward. One of the more detailed evaluations of urban hydrologic effects was done in 18 watersheds in western Georgia, 6 of which had urban land use that ranged from 1.8 to 41.9% impervious surface (Schoonover et al. 2006). Urban watersheds had higher flow pulses and peak discharges than did watersheds with other land uses/land covers, and baseflow was less seasonally variable in urban than in other watersheds. Poff et al. (2006) reported that hydrologic and geomorphic effects were likely to be similar between watersheds with agricultural and urban land use when impervious surface was >15% and agricultural land use was >25% (Poff et al. 2006). In another study, habitat quality declined when total impervious area (TIA) reached 5% and was consistently degraded when TIA was >10% (Schiff and Benoit 2007). Negative effects on total P and, to a lesser extent, on total N in streams occur at low levels of urbanization relative to streams in forested watersheds (e.g., Wickham et al. 2008, Shields et al. 2008, Kaushal et al. 2008, Bernhardt et al. 2008).

Biotic/ecosystem effects

Much of the recent work on the effects of urban land use on biotic assemblages was focused on macroinvertebrate or fish assemblages or the associated biotic indices (e.g., Goddard et al. 2008, Lussier et al. 2008, Stepenuck et al. 2008). A fish index of biotic integrity (FIBI) and a benthic index of biotic integrity (BIBI) were linearly and negatively related to degree of urbanization in western Washington, USA (Matzen and Berge 2008). However, the relationships were significantly variable when TIA was between 30 and 40%, and Matzen and Berge (2008) concluded that TIA is inadequate for predicting human influence at the subwatershed scale at that level of urbanization. In a study of 53 streams in southern Illinois, the largest percentage of tolerant macroinvertebrate taxa, highest nutrient concentrations, and poorest habitat quality were found in streams draining the densest urban areas (Heatherly et al. 2007). Macroinvertebrate assemblage structure was strongly modified and seasonality was reduced at sites in heavily urban watersheds compared to sites in all other landuse/land cover categories in streams in 18 watersheds in western Georgia (Helms et al. 2009). Catford et al. (2007) used a mechanistic approach to separate the effects of light and nutrient availability on benthic algal biomass across an urbanization gradient. They found that light was a secondary factor, and nutrient availability probably was a more important driver of biomass.

Other studies linking abiotic and biotic elements focused on spatial relationships between urbanization of watersheds and riparian areas. Roy et al. (2007) found that fish assemblage metrics were more strongly related to watershed-scale land use/land cover than to riparian-scale land use/land cover and that watershed-scale effects overwhelmed riparian forest benefits at high levels (>15%) of watershed urbanization. In contrast, Horwitz et al. (2008) found that watershed-scale effects did not overwhelm riparian-scale effects on fish assemblage metrics and hypothesized that riparian interactions were most important at intermediate levels of urbanization. Riparian-zone effects on macroinvertebrate assemblages in a large Australian river were weaker than the effects of an urbanization gradient (Walsh et al. 2007). Shandas and Alberti (2009) focused on riparian- and watershed-scale vegetation patterns and found strong correlations between a BIBI and both % riparian forest and watershed-scale vegetation fragmentation.

Metrics of ecosystem function and health are inversely related to urbanization and increases in impervious surface cover (Imberger et al. 2008, Scott 2006). In some cases, the highest levels of macroinvertebrate biomass and taxon richness correspond to intermediate levels of TIA, but generally the trend is an inverse relationship between these variables (Chadwick et al. 2006). Brown et al. (2009) evaluated hydrology, habitat, water chemistry, and biological data from sites across the country and found a negative effect of urbanization. They also found that regional variation, scale, and landuse legacies were important predictors of physical and biological conditions in aquatic ecosystems.

Benefits and Limitations to the SFT Approach

The general story is the same across studies: urban land use is bad for aquatic ecosystems. In all cases, regardless of the metric used, study areas with the highest levels of urbanization had the poorest environmental quality. This result shows the benefits and the limitations of using the SFT approach to study urban effects. The SFT approach provides the ability to capture a great deal of variability along the sampling gradient. By selecting numerous sites that span the gradient, the researcher can generate a robust data set across a range of conditions in a cost-effective timeframe and need not spend years waiting for effects of ongoing development to manifest at one or a few sites. At a coarse scale, the SFT approach can enable investigators to describe trends over a large spatial extent, and thus, the results can be generalized relatively easily. Moreover, once the gradient has been established and sampled, resampling can provide insights not captured by a single survey (Foster and Tilman 2000).

The SFT approach also has serious limitations, some of which are inherent to the method, whereas others are caused by the way the approach is applied. The SFT approach has long been used by forest ecologists to study vegetation succession (e.g., Cowles 1899, Bakker et al. 1996). The following criticisms of the SFT approach are similar to those in the forest ecology literature.

Urbanization alters stream ecosystems by simultaneously increasing peak flows, destabilizing stream channels, adding toxins, increasing nutrient concentrations, and changing other inputs of mass and energy (see Wenger et al. 2009 for a more complete accounting of mechanisms by which urbanization alters streams). However, how the urban environment is built and managed (e.g., application of structural stormwater mitigations, amount of soil compaction, maintenance of riparian buffers, direct manipulation of channels, application of chemicals to landscaping) and the amount and type of pre-urbanization disturbance are highly variable. SFT studies often are built on the assumption that urbanization effects can be evaluated along a simple gradient of proportion of impervious surface or fraction of land converted from native vegetation (e.g., Wang et al. 2001). Because of the large number of stressors activated by urbanization, the collinearity of some of these stressors, the difficulty of quantifying certain stressors, and the small number of streams available for analysis in any urban area, SFT studies typically have very few degrees of freedom with which to separate the effects of different stressors (Pickett 1989).

In addition, when SFT studies are based on a single measure of urbanization, the urban effect is treated as a black box, and accounting for the heterogeneity that can be found in urban land use/land cover becomes difficult. Thus, results of SFT studies are difficult to use for predicting effects of nontraditional or innovative types of urban development and of restoration activities. To overcome this limitation, researchers have begun to separate land cover and land use when the SFT approach is used. For example, Cadenasso et al. (2007) developed a new land-cover classification to account for heterogeneity in urban systems that better predicted NO_3^- yield in a metropolitan area than did traditional classification schemes. Others have demonstrated that particular features or characteristics of the urban landscape, such as density of urban land cover or stormwater drainage networks, are better predictors of indicators of aquatic ecosystem decline than the more general variable, urban land cover (see Hatt et al. 2004, Walsh et al. 2004). Changes in the way we think about land use/land cover from the urban vs nonurban dichotomy to a more nuanced land-cover classification or targeted landscape characteristics in urban areas might enable users of SFT studies to identify better particular mechanisms that affect urban streams (e.g., Walsh and Kunapo 2009).

Furthermore, seasonal fluctuations and cycles reduce the power of the SFT approach to test our mechanistic understanding of the urban effect on functional variables, such as nutrients, trophic interactions, and flow. The SFT approach, by definition, substitutes space for time. Thus, the gradient typically is not studied over long time scales, and the period of study might include anomalous weather patterns that could skew results. Pickett (1989, p. 116) wrote, "…given this uncertainty, SFT may be more appropriate in generating hypotheses about functional parameters than it is in testing them". However, this limitation is not inherent to SFT because the problem lies in how SFT studies usually are done, i.e., over short periods of time. Other approaches can be used for long-term study, but they also can have this limitation if funding constraints require a shorter-than-optimal experimental design. Methods are available for detecting trends in water-chemistry and biological data sets (e.g., Webb and King 2009), but predevelopment data are rarely available, and biological data sets tend to cover short time spans.

Another major flaw of the SFT approach is its inability to uncouple the historical legacy of past conditions at a site from its current condition, one of the major challenges to understanding urban effects on stream systems (Allan 2004). Many urban areas are on land formerly used for agriculture. Thus, many urban waterways have landuse legacies that might have had dramatic impacts before the urban area existed (Trimble 1974, Walter and Merritts 2008; Fig. 1).

Last, the SFT approach is difficult to use for determining whether degradation occurred during the construction phase of development or after the development was completed. Environmental stressors caused by site clearing differ from stressors that might arise once the site has stabilized (Wolman 1967). For example, sediment transport into the stream system occurs primarily when soil is exposed when the site is initially cleared for development. Once the site has stabilized, stressors caused by sediment are less likely to originate at the site, but the sediments that were deposited during clearing might be mobilized and resuspended from the stream bed and banks, thereby creating a legacy effect from the earlier disturbance. Sediment movement might be exacerbated by the altered hydrology in the post-developed condition, but the proximate source of the sediment will be different from the original source. Distinguishing present-day from legacy effects with an SFT approach would be difficult, but possible, if the design included gradients across urban to rural land use/land cover and across stages of development. This design could be implemented in large, phased subdivisions where the predevelopment land-use/land-cover history is known and full development has not yet occurred.

Complementary Approaches for Studying Urban Effects on Aquatic Ecosystems

Several other methods provide effective ways to evaluate how urbanization might affect stream systems. Some of these methods include SFT techniques and demonstrate how a mixed method can be a useful approach.

Monitoring the effectiveness of best management practices

Billions of dollars are spent each year on building structural stormwater controls to mitigate the effects of residential, commercial, industrial, and transportation development on water quality. Stormwater engineers and planners need some assurance that such measures have tangible benefits. Thus, engineers and hydrologists have conducted numerous comparisons of water quality upstream vs downstream of specific best management practice (BMP) structures, including infiltration ponds, detention ponds, wet ponds, biofilters, media filters, hydrodynamic devices, porous pavements, and others. Results from most such studies have been compiled in the International Stormwater BMP Database (www.bmpdatabase.org).

Performance of BMP structures typically is quantified by comparing influent and effluent event (storm) mean concentrations of pollutants and calculating % reductions. These studies show that most BMP structures substantially reduce pollutant concentrations during most storms. However, % reductions in pollutant concentrations vary with storm characteristics, influent concentrations, influent sediment particle-size distributions, position of the facility in the watershed, facility size, hydrodynamics, age, and maintenance regime (e.g., McCuen 1979, Anderson et al. 2002, Hossain et al. 2005, Chen et al. 2007). In addition, most of these performance tests are done on individual BMP structures in isolation from the receiving watershed. Researchers have attempted to overcome this flaw by modeling linked BMP structures and broadening the analysis to the watershed scale (Emerson et al. 2005).

The standard method for reporting structure performance in terms of % reductions in concentrations poses problems for interpreting and using results. As pointed out by Barrett (2005), % reduction of pollutants is not independent of the influent concentration; % reduction is higher for higher influent concentrations. Another way to evaluate performance would be to estimate time-integrated (stormflow and baseflow) influent and effluent loads and to determine the annual load trapped or transformed by the structure. Such information would be more useful for evaluating watershed-scale pollutant budgets, but accurate load estimation requires expensive high-density flow and water-quality data (Johnes 2007). A 3rd possible technique for evaluating performance of BMP structures would be to compare effluent concentrations to freshwater standards, but most US states do not have such standards for most stormwater pollutants. An ecological performance standard for BMPs was developed recently by Walsh et al. (2009), and this method shows promise as a way to link performance of BMP structures to ecological conditions in the stream.

Assessments of BMP structures suffer other problems. Most monitoring campaigns are of limited duration because of time and cost constraints and, thus, do not capture a wide variety of storm sizes, do not allow load estimation, and do not account for possible seasonal effects. These studies generally have not included biological assessments, so the ability of structural BMPs to protect stream biota is unknown (e.g., Carter and Rasmussen 2006, Dreelin et al. 2006). Exceptions to this rule do exist. Maxted and Shaver (1997) examined biotic conditions downstream of BMP structures and found no benefit of the BMP structures compared to similar sites without them. Horner et al. (2001) found that BMP structures mitigated only a small portion of stormwater inputs to streams and, thus, benefits to stream biota were difficult to discern. Horner et al. (2001) also noted that the design and frequency of BMP structures across the watershed precluded evaluation of their effectiveness at a watershed scale. Therefore, much of potential of BMP structures to mitigate effects of stormwater on stream biota remains unidentified at the watershed scale.

Long-term watershed studies

Long-term monitoring of urban watersheds, analysis of temporal trends in chemical, physical, and biological indicators of stream condition, or comparisons between water-quality conditions and standards might provide useful information to managers and scientists (e.g., Hottenroth et al. 1999). For example, planners for the City of Portland, Oregon (USA) found it difficult to determine the effectiveness of stormwater BMPs on a city-wide basis (Hottenroth et al. 1999). City planners selected the 58-ha Parkrose watershed for intensive long-term monitoring to determine if BMP implementation led to measureable improvements in water quality. After 4 y of monitoring, watershed-scale effects of BMP implementation were not detectable, and the city was relying on models to estimate BMP effectiveness. Other attempts to monitor BMP effectiveness have been based on statistical and modeling approaches, such as Bayesian hierarchical trend analysis, which was used to identify trends in stream biological health from records of only 10-y duration (Webb and King 2009), and comparison of trends in data from watershed monitoring to water quality simulated by a model calibrated to base conditions (Hartley and Funke 2001). Unfortunately, most such studies are commissioned by municipalities and are published only in reports from those municipalities or their consultants. Obvious problems with such studies are the need for long-term monitoring data to resolve water-quality trends or changes and lack of control of the variables that affect water quality, which results in little ability to discriminate among possible causes of those trends and changes.

Paired-watershed studies

Forest hydrologists have used the paired-watershed approach to elucidate how changes in vegetation and creation of road networks altered watershed hydrology (e.g., Bates and Henry 1928, Hornbeck et al. 1997, Lewis et al. 2001), how timber management and stand composition affected nutrient concentrations (e.g., Swank et al. 2001), how riparian management and conditions affected stream temperatures (e.g., Brown and Krygier 1970, Hewlett and Fortson 1982) and channel habitat (e.g., Froehlich 1973), how cumulative water-quality effects of forestry affected fish productivity (Gregory et al. 2008), and how various BMP implementations altered inputs of sediments from forestry operations to streams (e.g., Lieberman and Hoover 1948, Arthur et al. 1998, Wynn et al. 2000). Together, and in concert with process-based studies, paired-watershed studies have led to demonstrably improved forestry BMPs.

Paired-watershed experiments are conducted by selecting nearby watersheds that are similar with respect to variables that control hydrology and water quality (watershed area, soils, topography, vegetation, land use, and aspect), monitoring flow and selected water-quality variables for a calibration period of 1 to 3 y before treating 1 of the watersheds, and monitoring both for another 1 to 3 y. Treatment effects can be quantified by comparing changes in flow and water-quality relationships between the 2 watersheds for the pre- and posttreatment periods. The strength of this method is that the design controls for variability in climate and, to a lesser extent, watershed factors not related to the treatment.

The benefits of the paired-watershed approach are many, but the approach has limitations. Paired-watershed experiments are costly in terms of time and money. They can be difficult to replicate and statistical analysis of the results can be challenging. Moreover, paired-watershed studies do not incorporate the natural variability of hydrologic and water-quality variables inherent in forested watersheds, the variability in treatment application, or the variability in legacy effects of prior land uses. These drawbacks have been acknowledged and debated for decades (Hewlett et al. 1969), but the method continues to be used because it continues to create practical knowledge.

Many limitations, including the scale of analysis, need for cooperating landowners, the long pretreatment monitoring period, expensive and extended monitoring during and after development, and lack of scientific control over treatments prevent their widespread application to urban studies. We were unable to find any record of use of the paired-watershed design to study effects of urbanization on water-quality relative to undeveloped reference watersheds. In 3 studies, comparisons were made between watersheds that were developed using conventional or low-impact development (LID) strategies. In each case, investigators reported significant hydrologic and water-quality benefits from LID (Lloyd et al. 2002, Dietz and Clausen 2008, Selbig and Bannerman 2008). However, predevelopment baseline monitoring was not done in the watersheds undergoing development

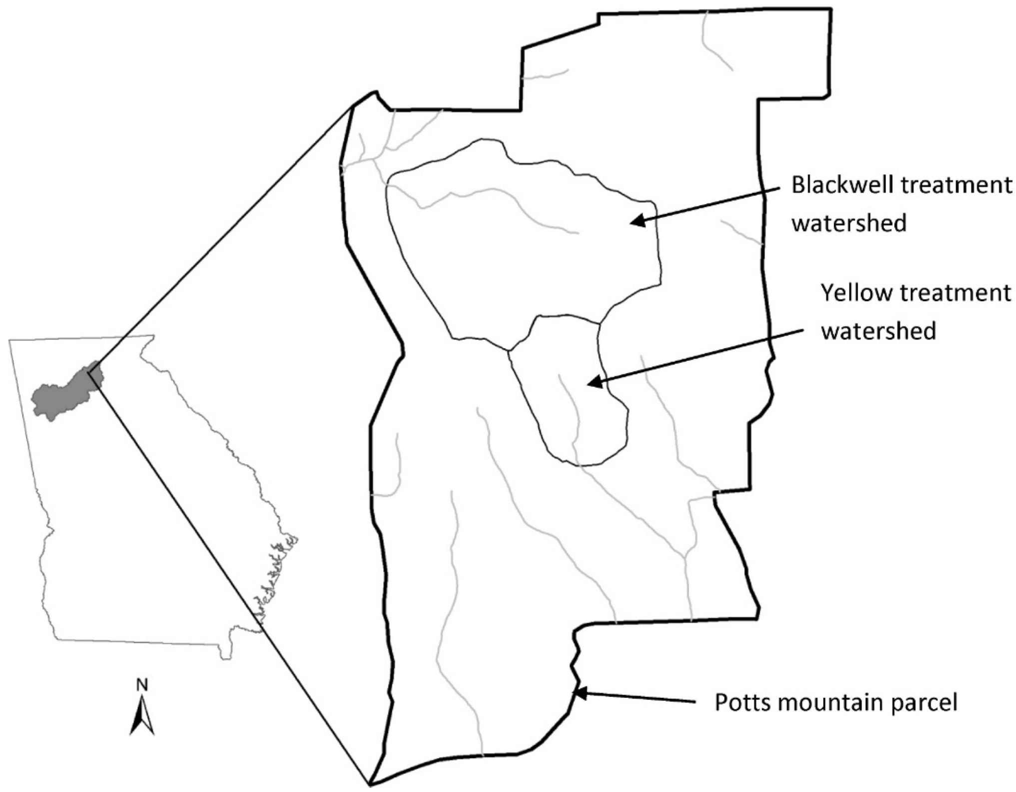

FIG. 2. Study sites for the paired-watershed study in the proposed Potts Mountain development in the Etowah watershed near Atlanta, Georgia. Blackwell watershed is being developed for low-density residential land use, whereas Yellow watershed is being developed for higher-density, mixed land use.

and no reference or undeveloped watersheds were used as controls in these studies. Thus, the results were of limited application for developing a mechanistic understanding of urban treatment effects.

Before–after control–impact studies

Before–after control–impact (BACI) study designs have been used to detect anthropogenic changes (Underwood 1994, Benedetti-Cecchi 2001), but have been not been used extensively in urban stream studies. A notable exception is a study by Roy et al. (2006), who used a BACI design to test effects of retrofitting a watershed with stormwater BMPs on stream hydrology, ecology, and water chemistry. Roy and colleagues monitored preinstallation conditions in the watershed, then distributed the BMPs via a public reverse auction (property owners submitted bids for compensation to have the BMPs installed), and currently are monitoring postimplementation conditions (Thurston et al. 2008). Walsh et al. (2005a) also alluded to a BACI study design after modeling outcomes of opportunities for stormwater retrofitting, and they recently began a modified BACI study with multiple control and treatment streams (C. J. Walsh, University of Melbourne, personal communication).

An Urban Paired-Watershed Study: Setting the Stage in the Etowah Watershed

Site context

We developed a paired-watershed study to evaluate the effects of LID on stream systems in the Etowah watershed north of Atlanta, Georgia, USA. We chose this watershed because its high aquatic biodiversity and high development pressure caused local governments to draft a Habitat Conservation Plan (HCP) (www.etowahhcp.org). The HCP included stormwater management standards that encouraged use of LID in sensitive areas of the watershed. We partnered with a developer who owns an ~445-ha parcel in the Etowah watershed (Potts Mountain) that contains more than twenty 1st- and 2nd-order stream systems.

In May 2007, we established 2 watershed pairs (Blackwell and Yellow). Each pair consisted of 1 watershed that would be developed and a reference undeveloped watershed with similar physical characteristics (Fig. 2). The treatment watersheds were projected to undergo development starting in June 2008 with development to continue for $\geq$7 y of the study period and all projected development to be completed in 2015. However, because of current economic conditions, initial construction on the treatment sites is now projected to begin in autumn 2009.

Both treatment watersheds will meet the stormwater performance standards of the HCP through a combination of engineered stormwater controls and nonstructural stormwater controls. In the low-density treatment watershed, this standard is the equivalent of the stormwater volume that would leave the site for a storm with a 2-y recurrence interval if the site had 5% impervious cover and 95% forest cover. In the higher-density treatment watershed, the standard is to match the stormwater volume that would leave the site for a storm with a 2-y recurrence interval if the site had 50% of the actual postdevelopment impervious surface cover. These standards are intended to encourage sustainable construction practices and establishment of smart growth strategies that will protect the imperiled fish species in the Etowah watershed.

Monitoring

We outfitted streams with pressure transducers, rain gauges, flumes, and water-quality samplers at the watershed outlets and are monitoring hydrologic and water-chemistry conditions. A tipping-bucket rain gauge linked to the sampler will enable collection of detailed rainfall information. We are making geomorphic measurements for detailed physical stream mapping and biological assessments, including epilithon, macroinvertebrate, and salamander sampling for aquatic ecosystem foodweb and energetic analysis.

Lessons to date

The Etowah project already has generated interesting results. We have only initial baseline data, and development has not yet occurred, but we see indications that the baseline data collection required by the paired-watershed approach will be valuable. For example, we found unusually large larval salamander sizes in the Yellow Creek treatment watershed that would not be expected without some historical effect. In an SFT study, this result would have been attributed to an urban effect (J. Maerz, University of Georgia, personal communication). Anecdotal accounts suggest that extremely high amounts of sediments (in some cases, high enough to raise concern about weir clogging; CRJ, personal communication) move through the low-density treatment watershed. Again, this result might have been interpreted as an urban effect had baseline monitoring not occurred. Last, one of the streams in a reference watershed dried because of the drought conditions in Georgia. This event will have legacy effects (e.g., at least 1 year class of salamanders was eliminated) that will alter biological assemblages and could have been misinterpreted as an effect of urbanization.

The scope of this study required involvement of a multidisciplinary team of researchers early in the experimental design process to outline the diverse data necessary to monitor the physical, chemical, and biological components of the aquatic ecosystem. Different experimental designs and sampling protocols were necessary to meet different disciplinary protocols. For example, ecologists in the group were more concerned about the statistical limitations of the paired-watershed approach than were hydrologists. With adequate preplanning, the team was able to establish monitoring protocols that maximized resources and produced defensible results. These protocols included use of >1 treatment and reference watershed and ensuring adequate time before development commenced to collect pretreatment data across seasons.

The process also has demonstrated a number of the limitations of the paired-watershed approach. First, developers are driven by market conditions and consumer demand, and the research team cannot control when development will occur. In the present case, development has been delayed by nearly 2 y. The delay was beneficial because it allowed collection of more baseline data, but is a detriment for practical reasons, including attraction of funding. Second, communication with developers is critical if researchers are to know when or where a site might be developed so that baseline monitoring can be established. Moreover, access to sites must be obtained from the developer before sites are disturbed. Third, many developments occur on relatively small sites where treatment effects from areas outside the site cannot be controlled. We avoided this problem by limiting our study watersheds to those drained by 0-, 1st-, or 2nd-order streams even though the total development area was relatively large. We did not work on larger streams and rivers because the potential for landuse changes other than urbanization and that >1 developer will have control increases as the size of the watershed increases.

Conclusions

SFT-based studies of effects of urbanization on aquatic ecosystems show that urbanization has detrimental ecological effects. However, until recently, effects on aquatic systems were rarely considered in development planning. Now, many municipalities have regulations that require (at least) stormwater management, and in more progressive jurisdictions, might require application of LID standards (e.g., City of Lacey, Washington, USA). Booth et al. (2002) recommended the following minimum requirements for protecting water quality in urbanizing basins: 1) cluster development to maintain 50% natural vegetative cover, 2) limit total impervious surface coverage to 20%, 3) provide detention or infiltration structures to control durations of geomorphically significant flows, 4) protect riparian areas and wetlands and minimize stream crossings, and 5) avoid construction on steep slopes. The Center for Watershed Protection (http://www.cwp.org/) has recommended 8 tools for watershed protection, including landuse planning, land conservation, aquatic buffers, better site design, erosion and sediment control, stormwater BMPs, improved point-source discharges, and watershed stewardship programs. Walsh et al. (2009) developed a theoretical retention capacity index as a design objective that accounts for the effects of disconnecting impervious surfaces that are hydrologically connected to a receiving water body through pipes or channels and allows flexibility in meeting stormwater management goals. Thus, present-day urbanization projects constructed under new regulatory frameworks might have significantly different effects on aquatic systems than the historical effects of urbanization. Whether watersheds developed according to all the above guidelines maintain good aquatic ecosystem conditions remains untested and unknown. Multidisciplinary retrospective comparisons of biotic assemblages, channel morphology, and water chemistry in such streams and in similar streams draining undeveloped areas could be used to quantify and detail the degree of our ability to protect streams with current knowledge. These new methods of investigation are needed if we are to understand the effects of environmentally sensitive development on aquatic ecosystems.

We have outlined a number of approaches to study the effects of urban development. The most promising approaches are based on monitoring responses to development through time or include sites that have sufficient predevelopment data to allow evaluation of postdevelopment changes by comparison with reference watersheds. These approaches permit tests of several hypotheses. The 1st hypothesis is that sustainable construction practices, such as those encouraged under the Etowah Aquatic HCP, will result in urban developments that maintain predevelopment hydrology and cause fewer alterations to aquatic and riparian ecosystems than conventional development. This hypothesis already has been tested partially in the few studies that have compared LID and traditional development (Dietz and Clausen 2008, Selbig and Bannerman 2008). A 2nd hypothesis is that the postdevelopment structure and function of stream and associated riparian ecosystems will be within the range of variation of predevelopment measurements. A 3rd hypothesis is that ecological condition will decline during grading and construction stages of development and will recover after the site is fully developed.

However, the limitations of the paired-watershed approach should not be understated. Finding 2 watersheds with identical landuse histories is extremely unlikely. Thus, careful consideration must be paid to calibration during the baseline monitoring phase. Other logistical constraints must be overcome (see *Lessons to date* above). Paired-watershed studies have statistical limitations, and their inferential power is greatly improved by including additional control watersheds (Underwood 1994).

A combined experimental design or mixed methods might provide the best opportunity to overcome limitations of the SFT and paired-watershed approaches. Long-term paired watershed data could be combined with short-term surveys along urban gradients around the paired watersheds so that both types of data would be available to test the hypothesis of interest. In addition, mechanistic (e.g., BMP structural, hydrologic, or geomorphologic) models of physical processes could be combined with paired-watershed approaches to overcome the statistical and experimental limitations of the paired-watershed approach alone. Long-term monitoring at sites undergoing urbanization or long-term gradient studies that include sites developed with innovative and environmentally sensitive development techniques will help build understanding of the mechanisms that lead to degradation of aquatic systems in urban areas.

As urban areas continue to expand, innovative forms of development must be used and outcomes must be monitored for effectiveness in protecting aquatic ecosystems. Such monitoring will require that researchers move beyond the classic SFT approach if the modern urban effect is to be understood fully. Both funding and accessibility issues associated with researching these forms of construction and with long-term monitoring programs can be overcome by developing innovative research partnerships with the development community.

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