WATERSHED SCIENCE BULLETIN

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AWSPs
Association of Watershed & Stormwater Professionals

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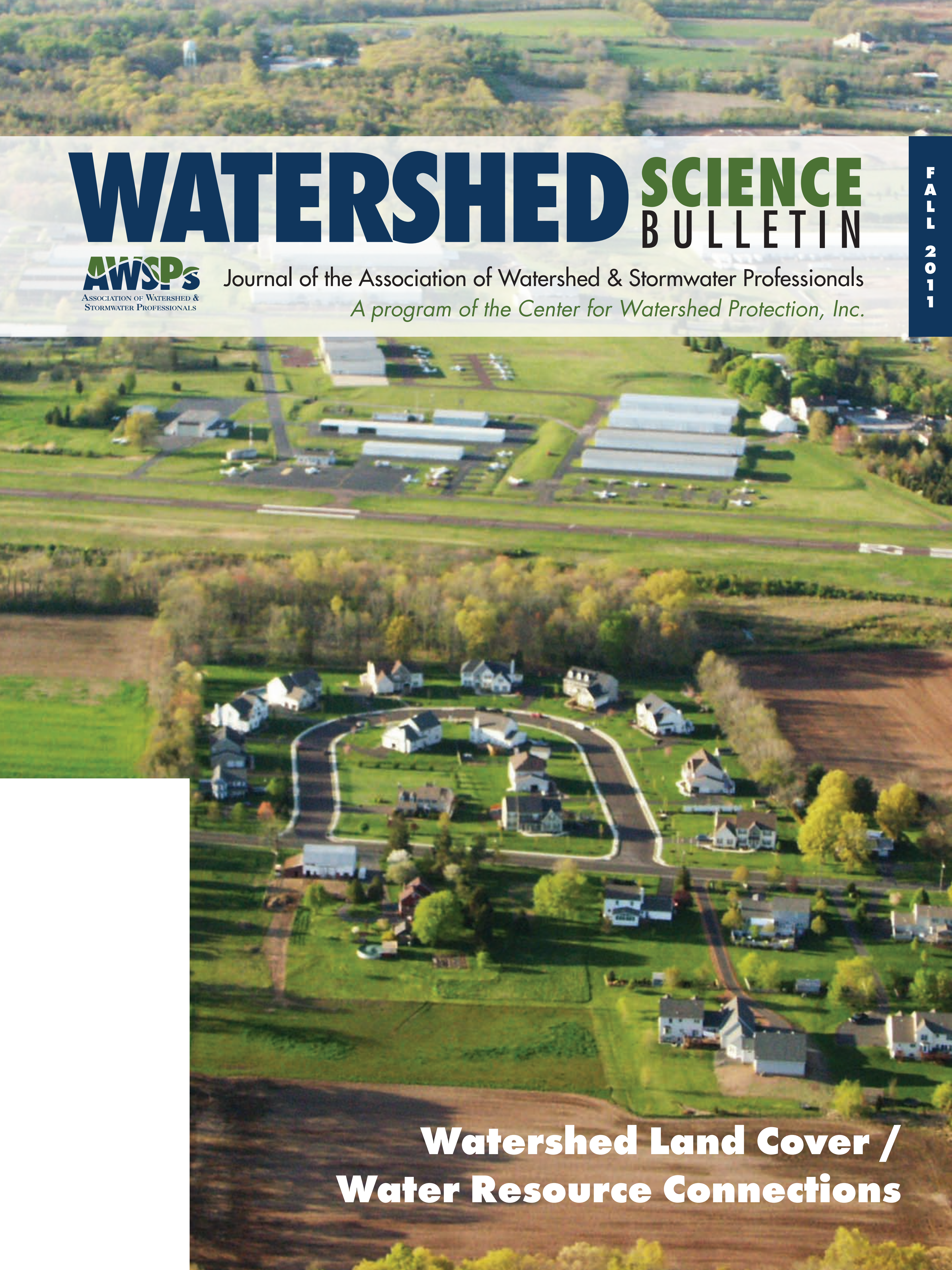
A program of the Center for Watershed Protection, Inc.



Watershed Land Cover / Water Resource Connections

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Cover photo courtesy of Dot Cappiella
This bird's-eye view of Bucks County, Pennsylvania, taken from a hot air balloon, shows the variety of land cover types on this rural and suburban landscape. Trees, turf, pavement, cropland, and even bare soil are present in this fast-developing suburb of Philadelphia.

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From the Editor's Desk

This issue of the Bulletin features research related to the influence of watershed land cover on the condition of downstream water resources and provides examples of how communities across the country are using this information to make better watershed management decisions.

Watershed and stormwater managers use land cover data to guide the restoration and protection of downstream resources. For example, such data are useful for estimating runoff and pollutant loads via watershed models; predicting current and future land use patterns and stream conditions; designing stormwater best management practices (BMPs); designing land use policies and programs (e.g., impervious cover caps, urban tree canopy goals, and stormwater management criteria); and identifying restoration potential. Given the prevalence and importance of these data in managing water resources, this issue of the *Bulletin* focuses on better understanding the various land cover types that are present in our watersheds.

At the Center, the topic of land cover is one that is near and dear to our hearts, given that much of our work is based on the Impervious Cover Model (ICM), first introduced in the Center's journal *Watershed Protection Techniques* in 1994. The basic premise of the ICM is that, as impervious cover increases in a watershed, various hydrologic, physical, chemical, and biological indicators of stream health decline. Many of the studies supporting the ICM are documented in the Center's 2003 publication *The Impacts of Impervious Cover on Aquatic Systems*. The ICM has undergone revisions over the years and has even been a subject of debate, but the appeal of impervious cover as an indicator of stream health persists because it is relatively easy to measure and manage.

More recently, the Center has been on a quest to identify other important relationships between watershed land cover and stream health to help guide policies and decisions on where and how to develop and where to mitigate urban impacts or protect undeveloped watersheds. Numerous studies support the strong link between watershed forest cover and water resource condition, but other land cover types, including turf, grasslands, and wetlands, have received less research attention at the watershed scale. In addition, while most studies focus on quantifying specific relationships between watershed land cover and indicators of receiving water condition, parsing out specific variables that may influence this relationship (e.g., distance from the stream or the presence of response "thresholds") is more of a challenge.

Although land cover is one of the most studied aspects of watersheds, our understanding of how different land covers across the nation impact water resources continues to evolve. The articles in this issue of the *Bulletin* represent two of the most common lines of research on land cover and watersheds:

- studies that improve our understanding of the rates and patterns of land cover conversion for use in predicting future changes and impacts on water resources and
- research to better characterize the influence of land covers on runoff quantity and quality and how this may change under varying management scenarios.

Fraley-McNeal and others address the question of how forest cover varies by land use for one primarily rural Maryland county. The authors present *a method for deriving forest cover coefficients* to use in estimating a watershed's future forest cover under "build out" conditions on the basis that watershed forest cover is strongly related to water quality. **Goetz and others** summarize their use of the SLEUTH model to predict growth in the Upper Delaware River basin using a stakeholder-driven process to develop projections of future urban land cover under different scenarios. The authors use the results as inputs to a hydrologic model with which they evaluate the impacts of each scenario on runoff, baseflow, and sediment loads. The findings provide further support for *land use policies that limit the footprint of urban land cover* by reducing impervious surfaces, encouraging clustered developments, and protecting natural lands. **Wilson and others** describe a statewide study of land cover change in Connecticut's riparian corridors based on the importance of forest cover in these zones as determinants of stream condition. This study is unique in that the *watershed-scale estimates of forest loss are used to target outreach programs* for the protection and restoration of riparian forest buffers.

Hubbart and others evaluate the potential for a very specific type of land cover—bottomland hardwood forests—to be used as a watershed management practice. The authors collected data describing the floodplain vegetation and soil characteristics associated with a bottomland hardwood forest in an urban impaired watershed in Missouri. The results demonstrate the *great potential for the restoration of bottomland hardwood forests in urban floodplain* areas to improve water quality and stream condition. **Stier and Soldat** present a horticulturalist's and soil scientist's view of lawns, a land cover that is receiving increasing attention by watershed managers because of its prevalence in the urban landscape. In a review of the literature on lawns and their contribution to urban runoff, the authors conclude that, perhaps surprisingly, *properly developed and managed lawns are generally effective for retaining nutrients* and can help mitigate urban runoff problems in a watershed. The vignette, **Regional Effects of Land Use Change on Water Supply in the Potomac River Basin**, summarizes the effects of land use and land cover change on river flows and its implications for water supply management under future growth conditions. Another vignette, **The Curve Number Method in Watershed Management and Watershed Health**, discusses a specific method for predicting the impact of land cover on watershed hydrology. Findings, cautions, and suggestions for users of the Curve Number Method, which predicts runoff volume based on rainfall depth, land cover, land use, and soils, are described based on its use over the past 50 years.

The *Bulletin's* vignettes provide tangible examples of science-based approaches for the management of watershed land use and the associated impacts on water resources. And what better region to highlight as a case study than Portland, Oregon, which has long been known for its Smart Growth attitude? In this issue, **Metropolitan Portland, Oregon, Urban Growth Boundary: A Land Use Planning Tool Protecting Farms, Forests, and Natural Landscapes** describes the urban growth boundary adopted in Portland, Oregon, as a highly successful land use policy designed to concentrate growth and protect the outlying rural lands. The resulting dense development within the city of Portland—and, specifically, the impervious surfaces—creates ten billion gallons of runoff that must be managed each year. Therefore, **Grey to Green: A Watershed Approach to Managing Stormwater Sustainably** describes the City's aggressive watershed-based approach to dealing with this runoff using green infrastructure.

And let's not forget about agriculture. The huge variety of agricultural land cover types makes it difficult, if not ill-advised, to treat them as one common land cover category when assessing potential impacts. In Ask the Experts, we summarize interviews with three agricultural research and extension specialists who discuss challenges and successes in managing agricultural impacts. Common themes include *the need for watershed-scale research; efforts that target BMPs to the sites that pollute the most; and collaborative, local solutions to water quality problems*.

I hope you will find that, by informing policy, models, engineering solutions, and even outreach strategies, the research presented in this issue can be used to help manage watershed impacts. Please feel free to send along any tidbits on your endeavors in applying this research or post additional resources on our AWSPs Facebook page at http://www.facebook.com/AWSPs. Thanks for reading!

–Karen Cappiella, *Editor-in-Chief*

Estimating Forest Loss with Urbanization:

An Important Step toward Using Trees and Forests To Protect and Restore Watersheds

Lisa M. Fraley-McNeal,[a*] Julie A. Schneider,[b] Neely L. Law,[c] and Adam W. Lindquist [d]

Abstract

Watershed forestry is a watershed-based approach for the management of trees and forests that acknowledges their importance in protecting water resources. In urban and urbanizing watersheds, this approach involves developing watershed-based goals and strategies for managing the urban forest as a whole, rather than on a site-by-site or jurisdictional basis. This paper presents a method to derive forest cover coefficients that represent the proportion of a particular area of land use that is covered by forest, using an example from Frederick County, Maryland. In an application of this method, we use the coefficients from the leaf-out analysis to evaluate changes in forest cover. We used the results in the Watershed Treatment Model to estimate pollutant loading under current conditions and under scenarios of future development for the Linganore Creek watershed, a drinking water source within the county.

Introduction

Nearly 0.4 million ha (1 million acres) of forest were converted to developed uses each year in the 1990s, with increased conversion rates through 2001 (Stein et al. 2005; Natural Resources Conservation Service 2001). Stein et al. (2005) estimate that an additional 9.3 million ha (23 million acres) of forest may be lost as a result of development by 2050. Areas experiencing the most forest loss are often suburban and urbanizing communities where municipal staff may not have the tools (or priorities) necessary to fully evaluate forest loss at the watershed scale. The projected increase in development and subsequent forest loss over the next four decades reinforces the need for better forest planning and management.

The important link between forests and the condition of streams in a watershed has been well documented. Booth (2000) found that at least 65% watershed forest cover is needed for the presence of a healthy aquatic insect community. Other researchers have determined that riparian forest cover is an important factor in maintaining stream geomorphology and various indices of biotic integrity (Moore and Palmer 2005; Goetz et al. 2003; Wang et al. 2003). And riparian forest cover can mitigate, to a certain extent, the impacts of impervious surfaces that are constructed as a watershed develops (Walsh et al. 2007; McBride and Booth 2005). Watershed forestry is a watershed-based approach for the management of trees and forests that acknowledges their importance in protecting water resources. In urban and urbanizing watersheds, this approach involves the development of watershed-based goals and strategies for managing the urban forest as a whole, rather than on a site-by-site or jurisdictional basis.

This paper presents a method to derive forest cover coefficients (FCCs) and to use them to estimate, on a watershed basis, existing forest cover and the potential forest loss likely with future development. Through a case study of the Linganore Creek watershed in Frederick County, Maryland, we illustrate an application of FCCs using the *leaf-out analysis* (Cappiella et al. 2005) to evaluate changes in forest cover under current conditions and under scenarios of future development. These methods provide planning-level estimates commensurate with commonly available data sources. Used in such a way, FCCs can play a key role in the identification of proactive measures needed to protect existing forest cover and watershed health.

Study Area

With a drainage area of 217 km^2 (83.8 square miles) and 336 km (209 miles) of streams, the Frederick County portion of the drinking water source area of the Linganore Creek watershed was the focus of this study (Figure 1). Lake Linganore, an impoundment of Linganore Creek that is classified by the State of Maryland as a recreational trout water body, provides recreational opportunities within the watershed. In addition, the County has designated the land area draining to the lake a source water protection area because the lake is a major drinking water supply serving residents in Frederick County and the City of Frederick. As the largest impoundment in the Monocacy River basin, the lake currently stores about 2.8 billion liters (729 million gallons) of water (Perot et al. 2002). Lake Linganore is also listed by the State as impaired for sediment and phosphorus, and the

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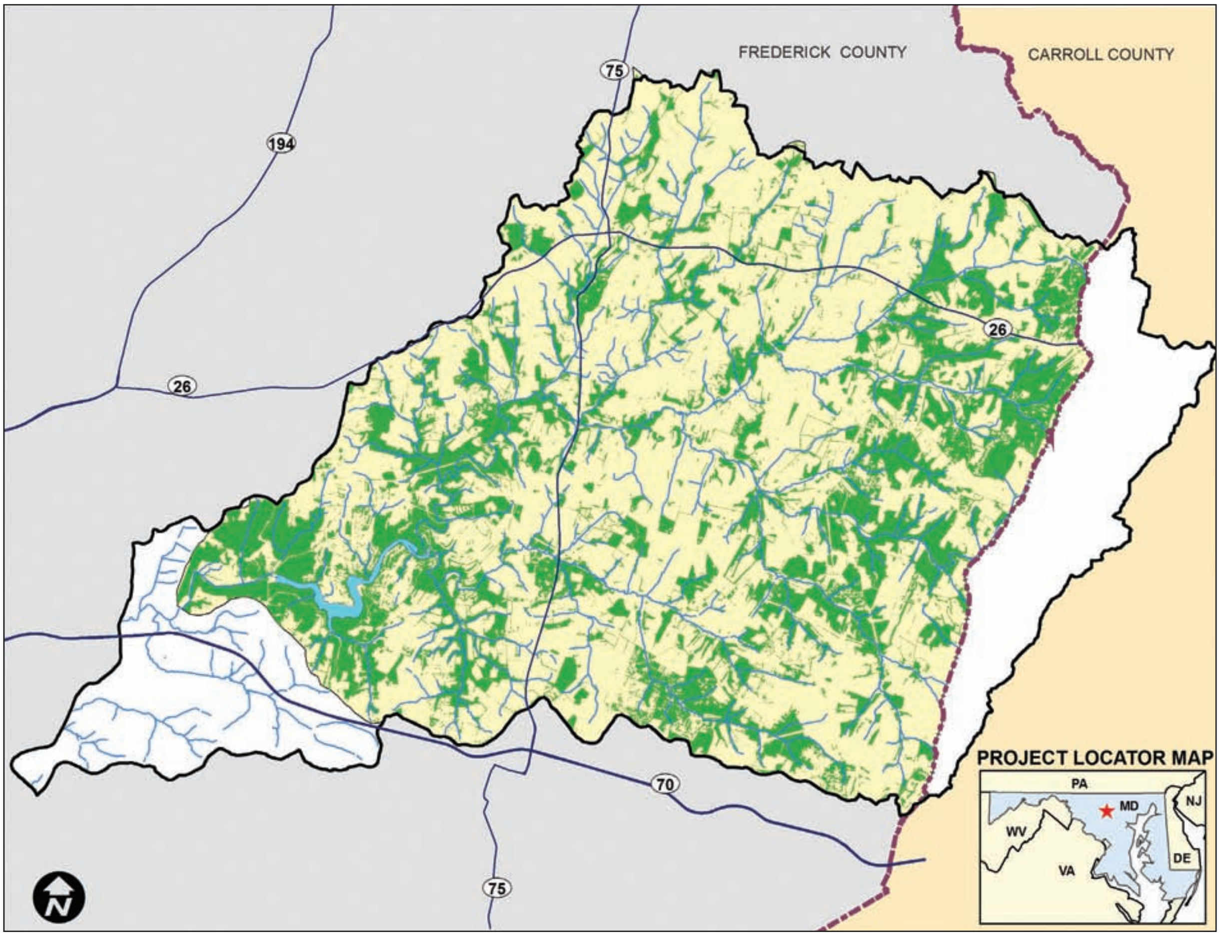

Figure 1. Forest cover (green) within the Frederick County portion of the Linganore Creek drinking water source area boundary (beige) in 2005. Approximately 90% of the Linganore source area is located within Frederick County, and the remaining 10% is within Carroll County. White areas are the portions of the Linganore Creek watershed that are either outside of the drinking water source area boundary or are in the Carroll County portion of the watershed; these areas were not analyzed in this study.

Maryland Department of the Environment (2002) has developed a total maximum daily load that will require measures to reduce sediment and phosphorus loads to the lake.

The Linganore Creek watershed had 28.2% forest cover in 2007, according to Maryland Department of Planning (MDP) land use/land cover data (Table 1). Most of the watershed's forest was cleared for agriculture by 1910. Agriculture continues to be the dominant land use within the watershed, especially in the northern and eastern portions; however, much of the land in the southern part of the watershed, along the I-70 corridor, is classified as low-, medium-, or high-density residential. Fifteen percent of the land use—primarily in areas adjacent to the lake—is urban. Although population data are not available for the watershed, the County population is expected to increase 38% by 2030 (Frederick County Government 2005, 2011), indicating considerable development pressure. This watershed has significant areas of highly erodible soils and steep slopes, exacerbating the sediment inputs to the lake any time land is disturbed. County watershed managers and environmental groups are concerned about the impact on these erodible soils of additional development that may further reduce the forest cover in the watershed and exacerbate erosion and phosphorus loadings to the creek and lake. Table 1 summarizes the land use distribution in the watershed.

Table 1. Land use/land cover distribution in the Linganore Creek watershed in 2007.

Type of Land Use/Land Cover	Percentage of Watershed
Urban/Suburban/Open Urban	15.2
Agricultural	55.7
Forest	28.2
Wetlands	0.04
Open Water	0.43
Barren/Transitional	0.52

Source: Derived from Maryland Department of Planning (forthcoming) 2007 land use/land cover data.

The management of forests in Frederick County is guided by the 1991 Maryland Forest Conservation Act (FCA). Md. Code Ann. [Nat. Res.] §5 1601–1613 (1991). The FCA requires local governments to develop forest conservation programs that must include an ordinance establishing standards for fulfilling forest conservation, reforestation, and afforestation requirements for certain land use categories and regulated activities. *Id.* § 5 1603–1612. In Frederick County, the local ordinance established under the FCA is the Forest Resource Ordinance (FRO), which was adopted in 1992. In 2007, significant and unique changes were made to the FRO that resulted in conservation requirements that are more stringent than what is mandated by the state law. Developers may choose to meet FRO requirements by purchasing forest banking credits or by paying a per-square-foot fee of required forest mitigation into a fee-in-lieu fund. In 2010, the Board of County Commissioners authorized the use of a portion of fee-in-lieu funds to purchase forest conservation easements along certain stream segments in the Linganore Creek watershed.

Methodology

To derive Frederick County FCCs that represent the proportion of a land use parcel covered by forest, we used ESRI ArcGIS© software and the basic protocol described below. Additional details are available from the Center for Watershed Protection (CWP 2011). Data used in this analysis include 1973 and 2007 (forthcoming) MDP land use/land cover data, 2005 planimetric data, 2008 parcels and tax points, 2008 subdivisions, and aerial photographs from 1988 to 2007. The County's 2005 planimetric data were the most recent and the most accurate available representation of forest cover for the study area. The data were delineated by Frederick County Department of Public Works staff from true color orthophotography with a 6-inch ground pixel resolution.

The first step in the FCC analysis was to select the targeted land use categories and the number of sampling units. The sampling units used in this study were polygons of homogeneous land use. The study used eight land use categories that corresponded to those defined by Cappiella and Brown (2001) for impervious cover coefficients. The purpose of aligning land use categories with this prior study was twofold: first, the categories are general enough to be readily transferable to other jurisdictions, and second, this approach should facilitate future land cover estimates that focus on impervious cover and forest cover using consistent land use categories and methods.

We delineated all sample polygons for those areas developed between 1973 and 2005. This time frame was based on the availability of 1973 land use/land cover data from MDP and the 2005 forest cover data derived from the County's planimetric data. It also corresponds to a period when Frederick County experienced significant urban development. From the 1970s to 1980s, the County population increased by 35.2%, from 84,927 to 114,792 (Frederick County Division of Planning 2004). The majority of the urban land created during this time period was for residential use.

Delineation of the sample polygons followed a set of criteria outlined in CWP (2011) with a brief description provided here. The polygons generally followed parcel boundaries; aerial photographs and parcel data, such as business or owner name, helped verify land use. The sample polygons included local and arterial roads where the parcels bordering each side of the road had the same land use. Local and arterial roads were included in the sample polygons if the parcels bordering each side of the road had the same land use. If a local or arterial road bordering a parcel had a different land use bordering the other side of the road, only half the road was included in the polygon. The polygons did not include interstate or state highways. For residential land uses, polygons followed the lot lines of contiguous parcels that correspond to that specific type of residential land use category (e.g., one-quarter-acre lots) and generally follow subdivision boundaries rather than individual parcels. Figure 2 shows an example of a residential polygon delineation.

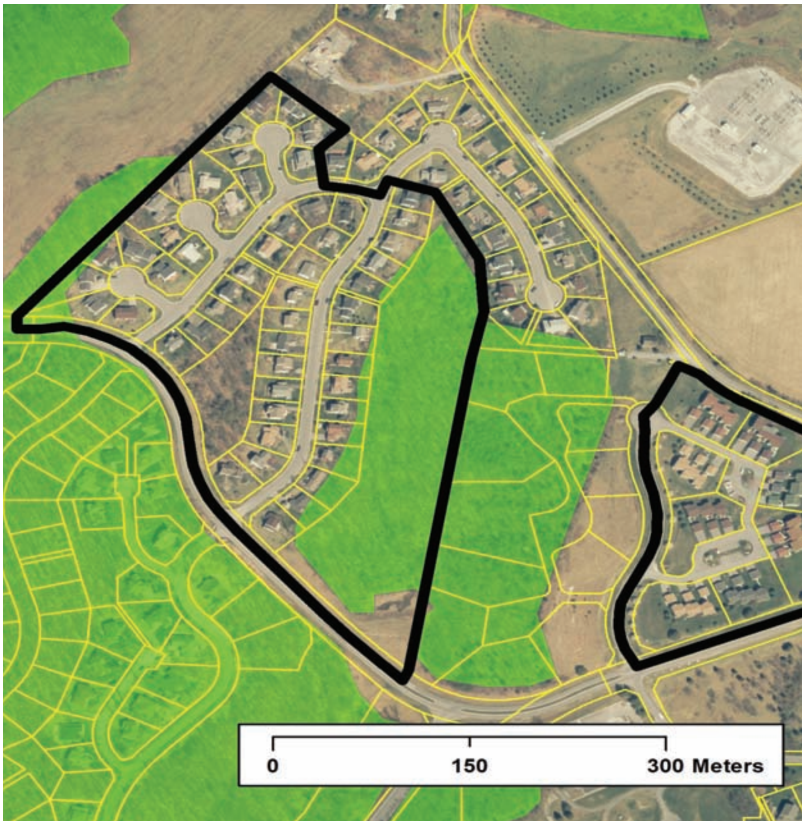

Figure 2. Example of a residential sample polygon delineation for Frederick County. Parcels are shown as yellow lines, forest cover in green, and the sample polygon delineation in black.

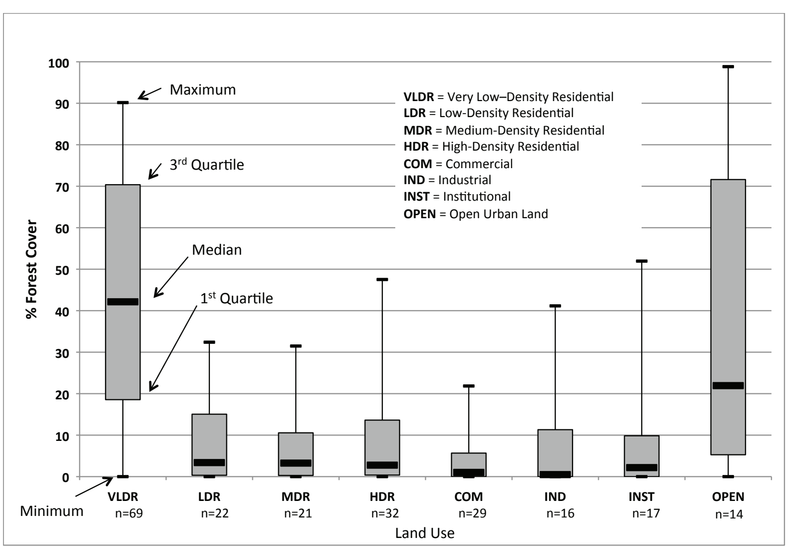

Figure 3. Box-and-whisker plot showing the percentage of 2005 forest cover across land use categories in Frederick County (n indicates the number of sample polygons delineated for each land use category).

In the final step, determining the percentage of forest cover for each delineated sample polygon involved calculating the area of each sample land use polygon and intersecting the layer with both the 2005 forest cover layer and the predevelopment forest cover layer extracted from the 1973 land use/land cover data. This allowed for calculations of the area of pre- and post-development forest cover within each land use polygon. We then divided the area of forest cover within each polygon by the sample polygon area to determine the percentage of forest cover before and after development.

Because of the high degree of variability in the data, the median proved to be a better measure of central tendencies as it discounts the importance of numbers outside the data range, whereas the mean tends to be affected by outliers. Figure 3 shows the median forest cover for all land uses. The data for the very low-density residential (VLDR) and open urban land (OPEN) land use categories have the most variability because they are influenced by the amount of predevelopment forest cover, as described further below.

For each land use category, we plotted the predevelopment and post-development forest cover data to determine whether the amount of forest cover present before development is influential in the amount that remains after development. Table 2 shows the results of a linear regression fitted to each plot, using the forest cover for the two time periods (1973 and 2005). Pre- and post-development forest cover were strongly correlated for only two land use categories, VLDR and OPEN. Low-density residential (LDR), industrial (IND), and institutional (INST) regressions were statistically significant at the 95% confidence level, but showed low correlation according to the R-squared values. Significant relationships for the remaining land use categories were not apparent. However, we found that the sample polygons for the OPEN land use category represented two distinct types of land use (i.e., recreational vs. passive) that should ideally be analyzed further to provide a more accurate estimate.

Table 2. Linear regressions comparing the percentage of forest cover before and after development.

Land Use Category	Linear Regression	R^2	Significance F
VLDR	Y = 0.0071X + 0.0397	0.86	0.00
LDR	Y = 0.0014X + 0.0043	0.24	0.02
MDR	Y = 0.0008X + 0.0048	0.09	0.20
HDR	Y = 0.0008X + 0.0066	0.06	0.17
COM	Y = 0.0003X + 0.0024	0.08	0.14
IND	Y = 0.0003X + 0.0024	0.62	0.00
INST	Y = 0.0062X + 0.0033	0.46	0.00
OPEN	Y = 0.0087X + 0.0142	0.98	0.00

Notes: Y, coefficient for post-development forest cover; X, % predevelopment forest cover. Significance F is the probability that the equation does not explain the variation in Y. If the significance F is less than 0.1, the correlation is significant.

Results

Results of the analysis show that the median percentage of 2005 forest cover best represents the post-development forest cover for the following land use categories: LDR, medium-density residential (MDR), high-density residential (HDR), commercial (COM), IND, and INST. For the VLDR and OPEN land use categories, the linear regression equation resulted in the most reliable estimate of post-development forest cover. Table 3 presents FCC recommendations for Frederick County.

Table 3. Recommended forest cover coefficients for Frederick County.

Land Use Category	Land Use Category Description	Forest Cover Coefficient	Measure of Variance
VLDR	Single-family residential development with a density of less than 1 dwelling unit per acre	Y = 0.0071X + 0.0397	0.110
LDR	Single-family residential development with a density of 1–4 dwelling units per acre	0.034	0.147
MDR	Single-family and attached residential development with a density of 5–10 dwelling units per acre	0.033	0.103
HDR	Residential development with a density of > 10 dwelling units per acre, generally multifamily development	0.028	0.132
COM	Retail, small office, and business uses	0.010	0.057
IND	Manufacturing and industrial facilities, including associated warehouses, storage yards, and research laboratories; business, professional, and corporate office parks	0.005	0.113
INST	Schools, churches, government offices, and facilities	0.022	0.098
OPEN	Golf courses, parks, recreation areas, and game preserves (except areas associated with schools or other institutions)	Y = 0.0087X + 0.0142	0.065

Notes: Y = coefficient for post-development forest cover; X = % predevelopment forest cover. Acres were used as opposed to hectares in the land use category descriptions because that is the unit of measure used by the County. Interquartile range, a measure of statistical dispersion defined as the difference between the third and first quartiles, is used as a measure of variance for the LDR, MDR, HDR, COM, IND, and INST land use categories, for which FCCs represent the median of the sample data. Variance for the VLDR and OPEN land use categories is the standard error of the linear regressions that are used to calculate these FCCs.

Table 4. Comparison of urban forest cover for various Maryland communities and the forest cover coefficients derived for Frederick County, expressed as percentages.

Community	Agriculture	Right of Way	Commercial	Industrial	Institutional/ Government Services	Apartments /Condos	Townhomes	Single-Family Residential
Frederick County[a]	—	—	10%	0%	2%	3%[c]	3%[c]	3%[c]
Frederick County – Brunswick[b]	2%	23%	6%–8%	10%	18%	—	—	30%–34%
Frederick County – Frederick[b]	—	7%	10%–17%	7%	9%	16%	16%	14%–20%
City of Baltimore[b]	5%	37%	24%	27%	32%	29%–33%	13%	53%
Anne Arundel County – Annapolis[b]	—	25%	20%	27%	34%	37%–40%	—	54%
Allegany County – Cumberland[b]	69%	28%	38%	26%	47%	18%–33%	—	57%
Howard County[b]	39%	37%	34%	28%	44%	36%–48%	33%	56%
Montgomery County – Rockville[b]	5%	37%	24%	27%	32%	29%–33%	13%	53%
Prince George's County – Bowie[b]	—	25%	20%–31%	49%	38%	47%	47%	47%
Prince George's County – Greenbelt[b]	—	43%	17%–28%	17%–24%	24%	64%	64%	64%
Prince George's County – Hyattsville[b]	—	28%	5%–17%	12%	24%	53%	53%	53%

Notes: — = no data. Ranges exist where data from two zoning categories were included under one land use classification for purposes of comparison in the table (e.g., Bowie does not have a general commercial zoning category, but instead uses retail trade and office buildings).

[a] FCCs derived as part of this study.

[b] Forest cover data developed by the US Department of Agriculture Forest Service and the University of Vermont SAL.

[c] Apartments/condos and townhouses are assumed to correspond to County medium-density residential (MDR) and high-density residential (HDR) data. Single-family residential is assumed to correspond to low-density residential (LDR). Very low–density residential (VLDR) is not included in this table.

Table 4 compares the derived FCCs, expressed as a percentage, with forest cover data for other Maryland communities that the US Department of Agriculture (USDA) Forest Service and the University of Vermont Spatial Analysis Lab (SAL) developed using high-resolution satellite imagery following the methodology presented in Grove et al. (2006) for New York City.

Frederick County communities generally have less forest (regardless of the data source) than other communities across land use categories. As expected, the USDA Forest Service/SAL estimates are higher than the FCCs because of the difference in the resolution of forest cover data. The resolution of the data used by the USDA Forest Service/SAL is such that it captures individual trees, whereas the data used to derive FCCs was a generalization of forest cover based on a minimum mapping unit. However, even considering the difference in data resolution, Frederick County forest cover is comparatively low. Another explanatory factor for the difference between the derived FCCs and the USDA Forest Service/SAL data is that the latter analysis includes forest cover on all lands within a given zoning category, whether the land is developed or not, whereas the FCC derivation was limited to parcels of developed land within a zoning category.

Application of Forest Cover Coefficients in the Linganore Creek Watershed

We estimated current forest cover (as of 2010) in the Linganore Creek watershed by subtracting the area of forest cleared between 2005 and 2009, documented by the County's FRO, from the area of forest in the 2005 forest cover layer. This application used the FCCs to estimate future forest cover in the Linganore Creek watershed via the GIS-based leaf-out analysis (Cappiella et al. 2005). Data used to complete the leaf-out analysis included: developed/undeveloped land, land use designations from the County's comprehensive plan, and protected land (e.g., conservation easements). The leaf-out analysis assumes that (1) no changes will occur in current zoning, (2) land cover on developed land will remain the same, and (3) buildable land will be developed according to the County's comprehensive plan. In addition, we assumed that all future growth within the Linganore Creek watershed would occur within the community growth areas identified in the comprehensive plan.

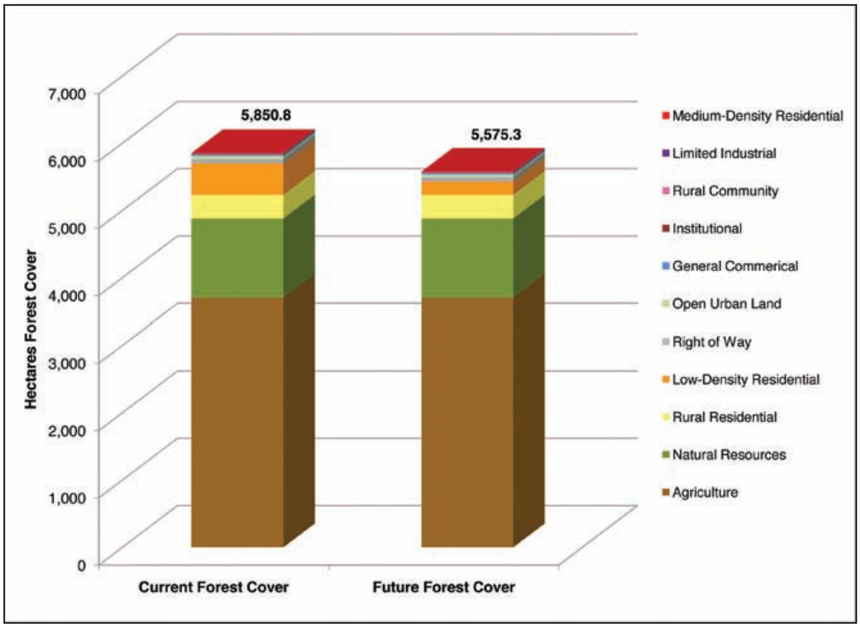

Figure 4. Current (as of 2005) and future forest cover (with watershed build-out) by comprehensive plan land use designations in the Linganore Creek watershed. Current forest cover is representative of 2005 instead of 2010 because the FRO estimates of forest loss are tallied on a watershed scale, and therefore could not be subtracted from the individual land use designations to obtain an estimate of forest cover for 2010.

The leaf-out analysis for the Linganore Creek watershed included the following steps:

- identify buildable land
- calculate the area for each comprehensive plan category for buildable land
- multiply the buildable land in each comprehensive plan category by the corresponding FCCs
- calculate total forest cover on developed land
- sum future forest cover on buildable and developed land

The results of the leaf-out analysis (Table 5) show that, for the entire Linganore Creek watershed, the estimated 2010 forest cover is 30.0%, and forest cover is likely to decrease to 28.6% under the future build out scenario, resulting in a loss of 256.9 ha (634.8 acres) or 4.4% of 2010 forest cover. At the watershed scale, forest loss is minimal; however, this loss is more substantial within the community growth areas. Figure 4 shows the distribution of forest cover by the different comprehensive plan land use designations under current and future build out conditions for the Linganore Creek watershed. The greatest loss in forest cover will occur with the development of the LDR land use, with a loss of 270.2 ha (667.7 acres). This is followed by OPEN (2.7 ha [6.6 acres]), right-of-way (1.9 ha [4.8 acres]), and COM (0.5 ha [1.3 acres]) land uses.

Table 5. Summary of current and future forest cover for the Linganore Creek watershed.

Forest Cover	Hectares	%
2010 Forest Cover	5,832.2	30.0
Future Forest Cover	5,575.3	28.6 (potential error of −0.1% to +0.3%)[a]
Loss in Forest Cover	4.4%	

[a]The potential error was calculated using the first quartile of the sample data for the low-end estimate of error and the third quartile for the high-end estimate of error. The exception was for OPEN, for which the forest cover coefficient was calculated by a linear regression that used standard error as opposed to the quartiles to estimate error.

In addition to the leaf-out analysis, we assessed runoff volume and pollutant loadings using the Watershed Treatment Model (WTM; Caraco 2010) for three scenarios—Predevelopment (99.7% forest), Existing Development (30.0% forest), and Future Build Out (28.6% forest) of the watershed based on comprehensive land use plan designations. For each scenario, we calculated annual loading rates for total nitrogen (TN), total phosphorus (TP), and total suspended sediment (TSS). This analysis used GIS and 2007 land use/land cover data for Frederick County (MDP forthcoming). While the runoff and pollutant estimates for the existing and future scenarios do not reflect absolute values—because they do not account for secondary sources of pollution (i.e., non–land use factors such as road sand, septic systems, and channel erosion) or the presence of management practices to treat runoff—the results of this analysis show the relative change in runoff and pollutants associated with land use changes in the watershed.

…for the entire Linganore Creek watershed, the estimated 2010 forest cover is 30.0%, and forest cover is likely to decrease to 28.6% under the future build out scenario…

The WTM calculates annual runoff and pollutant loading rates based on annual rainfall, pollutant concentrations, and land cover coefficients for forest, impervious cover, and turf using the modified simple method equation described by the Virginia Department of Conservation and Recreation (2011). This analysis used event mean concentrations, derived from a watershed assessment of Lower Linganore Creek (Perot et al. 2002), for TN, TP, and TSS instead of the default values provided with the WTM. For the Existing Development scenario, we determined land cover distribution across urban land uses by multiplying the total acreage of each land use by the FCCs and a Frederick County–specific impervious cover coefficient for each of the eight land use categories, with the exception of VLDR and OPEN. For VLDR and OPEN, we derived land cover distributions directly from the GIS. The analysis assumed that land cover not classified as forested or impervious was turf. For the Future Build Out scenario, we also used land cover coefficients to determine the distribution of forest, impervious cover, and turf across urban land use types. To solve the linear regression formula for VLDR and OPEN in the Future Build Out scenario, we used the 2005 forest cover layer to calculate predevelopment forest cover as an input to the equation. One could derive a more accurate estimate of urban land cover for the Existing Development scenario by directly deriving impervious and forest cover from the GIS. However, use of the land cover coefficients allowed for consistency with the Future Build Out scenario since the primary goal of this exercise was to evaluate relative changes in pollutant loads under different land use scenarios; in addition, the use of land cover coefficients provides a reasonable approximation of land cover distribution.

The results, presented in Table 6, show that pollution loading increases as forest cover is replaced with agriculture and urban uses. Comparing Predevelopment to Existing Development reveals that TN increased 82%, TP increased 289%, and TSS increased 30%. Comparing Existing Development to Future Build Out reveals that TN may increase an additional 3%, TP may increase by 5%, and TSS may increase another 3%.

Discussion

A number of data limitations were apparent in the Frederick County study. For example, the 1973 forest cover derived from MDP data are mapped at a lower spatial resolution

Table 6. Estimated annual land use–based pollutant loadings for the Linganore Creek watershed.

Land Cover Scenario	Annual Runoff (m^3/year)	TN (kg/year)	TP (kg/year)	TSS (kg/year)
Predevelopment (99.5% forest cover)	4,019,918 (3,259 acre-feet/year)	55,560 (122,487 lbs/year)	4,392 (9,683 lbs/year)	2,186,447 (4,820,210 lbs/year)
Existing Development (30.0% forest cover)	15,166,894 (12,296 acre-feet/year)	101,001 (222,665 lbs/year)	17,106 (37,712 lbs/year)	2,846,646 (6,275,675 lbs/year)
Future Build Out (28.6% forest cover)	16,589,099 (13,449 acre-feet/year)	103,706 (228,628 lbs/year)	17,939 (39,549 lbs/year)	2,934,435 (6,469,212 lbs/year)

(i.e., a 4-ha [10-acre] minimum mapping unit) than the 2005 forest cover derived from the planimetric data; therefore, the predevelopment forest cover is typically over- or underestimated (e.g., forest tracts less than 4 ha [10 acres] are not mapped). Further, many areas reforested as part of the FCA are not reflected in the 2005 forest cover data because they had not yet matured enough for mapping methods to classify these areas as forest. New plantings typically take an estimated 5–10 years to become established and 15 years until they are identifiable using moderate-resolution remote-sensing imagery. However, the ability to identify and map individual tree canopy is also dependent on the spatial resolution of the remote-sensing imagery (e.g., 30-m Landsat compared to digital aerial imagery at a resolution of less than 0.3 m).

Many variations on the methods described here to derive FCCs are possible, depending on the available data, the scale at which the coefficients will be applied, and the representative land use categories chosen. The methods chosen to delineate sample polygons may also affect the derivation of FCCs. For example, delineation of sample polygons for determining FCCs can be done by (1) using individual parcels, (2) lumping parcels in the same land use category (as we did for Frederick County), or (3) on a broader scale, analyzing all areas of the same land use together. Delineation based on individual parcels is a good way to evaluate land cover for a large number of parcels within each land use type. However, the ability to account for the land cover changes associated with urbanization taking place outside of individual parcels (e.g., road networks created to sustain urban development) is limited under this approach. Lumping of parcels in the same land use category can be used to capture these changes, but is a more time-intensive process because it requires the development of criteria for delineating land use polygons, which then need to be hand-delineated. A subdivision is an example of a case in which aggregation of individual parcels into one land use polygon is applicable.

Whichever method is chosen to delineate sample polygons, the number of sampling units chosen for each land use category should be based on the frequency and variability of land uses or zoning categories. For example, a larger sample size would be needed with greater variability of land cover within a given land use. For this analysis, we initially targeted 10 sample polygons for each land use. Statistical analysis showed that the data did not follow a normal distribution; therefore, it was not possible to accurately predict the sample size needed to provide a statistically significant result. As an alternative, all possible sample polygons that were developed between 1973 and 2005 were delineated; this provided the maximum possible sample size.

Comparing Predevelopment to Existing Development reveals that TN increased 82%, TP increased 289%, and TSS increased 30%.

One option for improving the FCC methodology in Frederick County is to delineate a larger number of sample polygons built after establishment of the FRO. Originally, we attempted this as part of the FCC derivation for Frederick County, but not enough sample polygons were delineated for areas developed after establishment of the FRO to yield reliable results. This expanded analysis would help determine forest cover impacts that can be attributed to the FRO. In addition, one could incorporate the age of the development into sample polygon delineation to determine how age affects the FCCs. One would expect that older developments should have greater percentages of forest cover because trees in such developments have been growing for a longer duration. Last, one could obtain a more accurate FCC estimate for OPEN by analyzing the different types of land use within this category, such as parks, golf courses, and playgrounds.

Results of the FCC derivation in Frederick County show that post-development forest cover for all land uses except VLDR and OPEN is less than 5%, suggesting that forest loss in response to development is substantial, despite pre-existing land use conditions and the requirements of the County's FRO and the Maryland FCA. One potential explanation is that many areas reforested on-site as part of the FRO are not reflected in the 2005 forest cover data because they had not yet matured enough for mapping methods to classify these areas as forest, or reforestation to meet FRO requirements occurred off-site and was therefore not captured in the land use polygon analysis. In addition, in many areas within the County, the predevelopment land use is agriculture, which experienced forest clearing prior to development. Unless on-site reforestation were to occur as part of the development process, these areas would continue to have low percentages of forest cover. Note that the County updated the FRO in 2007 to encourage greater forest conservation on-site. All of the land use polygons delineated for determining the FCCs were built prior to 2007; therefore, the FCCs may not be representative of expected forest conservation on future development sites in Frederick County.

One of the assumptions of the leaf-out analysis was that land outside of the community growth areas would not be rezoned. However, rezoning is a real possibility, especially considering the population increase projected for the watershed. In fact, future forest loss may actually be greater than predicted. In addition, even with the concentration of growth caused by the community growth areas, development within the watershed continues to place added stress on the drinking water supply reservoir in terms of water quality. While the WTM-estimated increase in pollutants and runoff in the Future Build Out scenario relative to the Predevelopment scenario cannot be ascribed solely to the loss of forest cover (since forest loss is always associated with the addition of a new land cover), the results imply that forest conservation and reforestation measures have great potential in helping the County meet regulatory requirements for pollution reduction in the watershed.

The leaf-out analysis results presented in this study will aid the development of (1) a forest cover goal for the Linganore Creek watershed and (2) recommendations to achieve this goal and to analyze the impacts of these actions. Recommended actions may include reforestation, the protection of forests with high value for water quality and habitat protection, and the implementation of outreach or incentive programs to encourage tree planting on private land. The GIS data derived for input to the leaf-out analysis can be used to target actions to specific land use types. For example, if most of the forest loss will occur on LDR lands, forest tracts on these lands can be evaluated, prioritized, and targeted for conservation. Similarly, one could use the land use distribution and associated land use coefficients in the watershed to identify the land use types with the greatest reforestation potential and target outreach programs accordingly. Once identified, these actions can be incorporated into the leaf-out analysis to determine how their implementation will impact future forest land use in the watershed.

Summary

Urban watershed forestry acknowledges the importance of trees and forests in protecting water resources. The development of FCCs facilitates the ability of local governments to anticipate and manage the forest loss that accompanies urban growth. The FCCs and leaf-out analysis can be useful tools for estimating future changes in forest land use, defining watershed forestry goals, and informing local government strategies on forest conservation and afforestation. The FCCs presented in this analysis can be used in similar Maryland communities (i.e., watersheds with a mix of urban, suburban, and rural land with development pressure), but have limited application outside of the state because of variations in forest management regulations and watershed conditions. When applying the methodology presented in this study to derive FCCs and conduct the leaf-out analysis in other communities, the methods should be adjusted based on available data and local conditions.

Acknowledgments

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Forecasting Future Land Use and Its Hydrologic Implications:

A Case Study of the Upper Delaware River Watershed

*Scott Goetz,[a] * Claire A. Jantz,[b] and Mindy Sun[c]*

Abstract

We mapped recent land use change patterns in an approximately 8,000-km^2 area encompassing the Upper Delaware River watershed with satellite imagery and used these data to calibrate a predictive spatial model of urban growth rates and patterns. With local stakeholders, we developed various future scenarios of growth to simulate the influence of different land use policies and land management practices, incorporating a variety of environmental, transportation, and other data sources. We generated forecasts of future urban growth patterns, including low-density residential development, under scenarios featuring current growth trends, increased growth, and increased conservation. These future scenarios form the basis for a number of environmental assessments of urbanization in the region. We incorporated the forecasts into a hydrologic model to examine the implications of urbanization on hydrologic factors—runoff, baseflow, and sediment loads—that are linked to water quality and aquatic biota management priorities for the watershed. The outcomes demonstrate how the spatial patterns of urbanization are likely to influence hydrologic dynamics in the future, notably by increasing runoff and sediment loads while decreasing baseflow under scenarios with greater development and associated impervious cover. The approaches, tools, and data sets employed here are useful not only because they produce forecasts in easily understood map form, but also because they are well documented and widely available to resource managers, policymakers, and a range of other stakeholders for diverse watershed applications, including mitigation, restoration, and adaptation objectives.

Introduction

Increased urbanization is well known to result in greater impervious cover, which modifies hydrologic processes such as the timing and magnitude of flow volume and peak discharge rates (e.g., "flashiness;" Ackerman and Stein 2008; Jacobson 2011; O'Driscoll et al. 2010; Schueler et al. 2009). These hydrologic changes, which occur even with low-density residential development, also modify water quality, instream habitat, and aquatic diversity (Booth et al. 2002; Goetz and Fiske 2008; King et al. 2005; Snyder et al. 2005). Estimating the magnitude of these changes and forecasting them into the future would allow land planners and managers to tailor development activities to effectively mitigate the negative consequences of urbanization, specifically those associated with commercial, industrial, and residential development.

Greater impervious cover hinders the infiltration of precipitation into the soil and ground water; thus, the overall expectation is one of reduced baseflow and increased overland flow (runoff). To better understand the various impacts of impervious cover, however, one may need to establish how the spatial distribution of new development influences hydrologic dynamics. For example, the placement of housing and commercial development will alter flow patterns within watersheds, changing both the timing and location of peak flows. Hydrologic models that incorporate spatial information (i.e., map data), particularly regarding land cover change, can be used to predict these dynamics (Beighley et al. 2009; Brabec et al. 2002), but future land cover information is not generally available for most areas.

This paper describes a unique case linking a spatially explicit urban land cover change model to a hydrologic model to investigate the expected future hydrologic impacts of impervious cover associated with exurban development in an environmentally sensitive landscape, the Upper Delaware River watershed. We first describe the land cover change component of the analysis, followed by the hydrologic modeling component that incorporated the land cover change results. For the land cover change predictions, we simulated several possible scenarios of development out to the year 2030 using version 3r of the Slope, Land cover, Exclusion, Urbanization, Transportation, and Hillshade (SLEUTH) model (Jantz et al. 2010), a widely available model with an active group of users (Clarke et al. 2007; National Center for Geographic Information and Analysis n.d.). We used land cover change data, mapped by satellite imagery, to calibrate SLEUTH for the simulation of historic rates and patterns of development

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Table 1. Population growth rates in the four counties included in this study.

County	2010 Population	2000–2010 Growth Rate	Developed Area 2005 (km²)[a]	1984–2005 Growth Rate
Pike Co., PA	57,369	24.0%	23.75	191%
Wayne Co., PA	52,822	10.7%	32.80	260%
Sullivan Co., NY	77,547	4.9%	42.12	131%
Delaware Co., NY	47,980	–0.16%	33.02	206%

[a] *Does not include road area.*

and to create forecasts of future urban land cover change for a range of scenarios developed with a group of local stakeholders. We then used the Soil and Water Assessment Tool (SWAT2000), which is part of the Automated Geospatial Watershed Assessment (AGWA) program, to predict the impact of land use on water and sediment yields.

The Study Area

The Upper Delaware River basin (Figure 1) is located at the intersection of New York, New Jersey, and Pennsylvania, within 161 km (100 miles) of the New York Metropolitan Area (some southern counties within the basin form the northwestern extent of the metropolitan area). The watershed contains some important natural, scenic, and recreational resources, including two National Park Service (NPS) units—the Upper Delaware Scenic and Recreational River (UPDE) and the Delaware Water Gap National Recreation Area. Along with watersheds in the adjacent Catskill Park and Catskill Forest Preserve, the Upper Delaware River watershed provides source water protection and reservoirs for New York City's water supply. The watershed also includes the New Jersey Highlands, an environmentally sensitive region of source water protection for millions of residents in New Jersey.

Despite the designation of UPDE as a scenic and recreational river, NPS has little direct control over land use in the river corridor and thus works closely with adjacent municipalities to encourage land preservation and land use practices that will not threaten the park's resources. Given the growth pressures that originate primarily from the New York Metropolitan Area, many of the counties in the southern and central part of the study area have experienced sustained exurbanization over the past few decades. Recent growth rates continue to be high; many counties in the study area are among the highest-ranked counties within their states in terms of growth rates between 2000 and 2010. The 2000–2010 growth

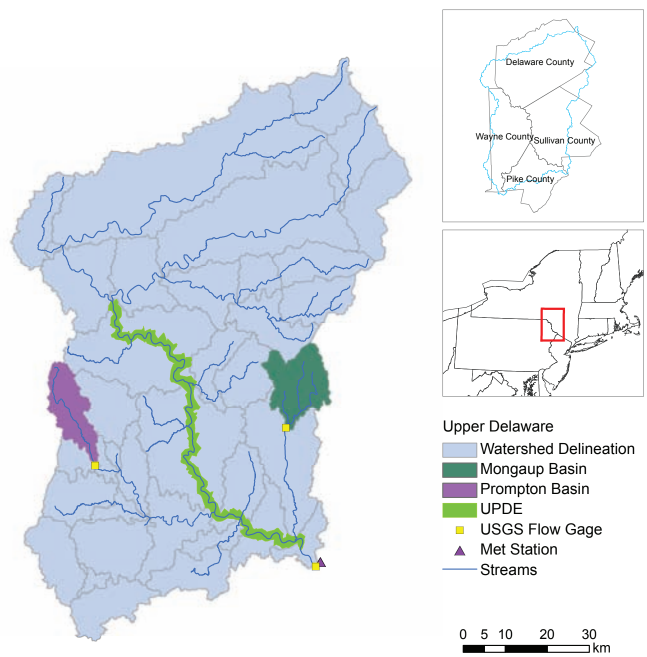

Figure 1. The study area showing boundaries of the Upper Delaware River watershed, as defined by the 7,960-km² area draining to the US Geological Survey river gauge at Port Jervis. Sub-basins within the watershed (gray lines) were used as the basis for hydrologic modeling. The locations of the Prompton and Mongaup sub-basins, used for model testing, are shown for reference. The UPDE is also highlighted, and the location of the Port Jervis meteorological station is indicated.

rate in Pike County, Pennsylvania, was 24.0%, compared to a statewide growth rate of 3.4% (Table 1). Monroe County, Pennsylvania, grew at a rate of 22.5%; together Monroe and Pike Counties ranked second and third, respectively, of the 67 counties in Pennsylvania. The population in Orange County, New York, increased 9.2% compared to a statewide growth rate of 2.1%; this county ranks second of the 62 counties in the state. Thus, the question of how development in the surrounding communities might affect hydrology and other ecosystem processes in the Upper Delaware River basin has generated considerable interest.

To address these concerns, we coordinated an effort with NPS and the four counties that account for most of the land area within the Upper Delaware River basin (Pike and Wayne Counties in Pennsylvania and Sullivan and Delaware Counties in New York) to simulate and forecast urban land cover patterns. These stakeholder groups are now using the forecasts as a basis for ecosystem assessment studies, including the hydrology discussed here.

Mapping Current and Future Urbanization Patterns and Rates

We defined current land cover using the widely available National Land Cover Database (NLCD) combined with more detailed impervious cover maps derived specifically for the study area for 1986, 1996, and 2005 (Jantz and Goetz 2007; Jantz et al. 2009). The NLCD provides a nationwide impervious cover layer (circa 2001), but calibration of the urban change model requires a time series of maps that identify land cover change. This study used a time series of change derived from the Landsat series of satellites at a nominal spatial resolution (grain size) of 30 m (900 m^2). We assessed the products for accuracy using aerial photographs and corrected them for false positive change detections (e.g., bare agricultural fields, bare rock outcrops, quarries, and landfills) through visual editing of the digital maps (Jantz et al. 2009). We then used the land cover products to calibrate the urbanization model as described below; this permitted predictions into the future.

Scenario Development

We generated future urbanization predictions using SLEUTH, a probabilistic, cell-based model that we applied separately for each county in the study area. Inputs to SLEUTH include a slope layer derived from the US Geological Survey (USGS) 30-m National Elevation Dataset (NED), a transportation layer reflecting primary roads, and a layer describing areas that will either attract or not attract (or, one might say, *exclude*) development. The exclusion–attraction layer is particularly important for SLEUTH predictions because it provides what is essentially a weighted surface to guide the spatial allocation of growth. Here, exclusion and attraction refer to urbanization, specifically the urban land cover categories on which the model was calibrated (described in the next section), and various data layers developed for each county-specific future scenario.

We worked with county planners and other stakeholders within each county to (1) identify primary "attractors" of growth (e.g., proximity to the New York Metropolitan Area, proximity to natural amenities, and local land use policies), and (2) define future land use scenarios expressed through the exclusion–attraction layers. For forecasting, stakeholders from each county developed a county-specific set of narratives representing a range of relevant land use policy and land use change scenarios. We translated each narrative into a map representing the areas that would attract or repel development. Pike County stakeholders, for example, developed a total of six narratives ranging from a scenario with strict spatial controls on growth and high levels of protection for lands rich in natural resources, to a scenario that allowed dispersed growth patterns with minimal protection of natural lands. This resulted in scenarios that represented a range of realistic future policies and drivers (attractors) relevant for the specific planning needs of each county.

In addition to county-specific land use policy and land use change scenarios, which essentially enabled the application of spatial weights to areas where growth is more or less likely to occur, we also modeled different rates of growth for each scenario: a linear growth rate based on the 1984–2005 growth rate, a "boom" growth rate that was roughly 25% higher than the 1984–2005 trend, and a "bust" growth rate that was roughly 25% lower than that trend. In the case of Pike County, this resulted in a total of 18 different forecasts (3 growth rates for each of 6 land use change scenarios).

Even though the scenarios were county-specific, the emergence of common themes (e.g., "smart growth" vs. "sprawl") across counties allowed us to group common scenarios together. For this study, we were therefore able to combine results across counties to develop three watershed-wide scenarios that essentially reflect low, moderate, and high expectations for future urban growth (including low-density residential development). These scenarios include (1) a Conservation scenario reflecting land use policies that require strong protections on natural lands, spatially clustered development, and a low rate of growth; (2) a Trend scenario with policies reflecting the status quo of moderately focused development and moderate protection of natural lands with a linear growth rate; and (3) a Growth scenario reflecting limited protection of natural lands, dispersed development patterns, and a high growth rate. The spatial extent of the output simulations (probability maps) is 10,796 km^2 at a spatial resolution of 30 m, with each cell assigned a probability representing the likelihood that the cell will be transformed to impervious cover (i.e., developed) by 2030.

Table 2. Accuracy results for each county at the municipal scale and for the 1 km x 1 km array.

County	Municipal-scale accuracy		1 km x 1 km–scale accuracy	
	Municipalities (N)	r^2	Cells (N)	r^2
Pike	13	0.99	1,464	0.83
Wayne	28	0.98	1,938	0.82
Delaware	19	0.90	3,802	0.88
Sullivan	15	0.95	2,579	0.92

Calibrating the SLEUTH Model

The calibration phase used an expert-weighting approach in which county planners identified a set of factors that had acted to either exclude or attract development between 1986 and 2005 (Jantz et al. 2009). For example, stakeholders in both Pike County, Pennsylvania, and Sullivan County, New York, identified their proximity to the New York urban core as a driver for growth pressure, so we weighted factors to reflect higher growth pressure in the southeastern part of the watershed and lower growth pressure in the northwestern part (our assumption is that this growth pressure will persist through the 2030 forecast period). In contrast, in central and southeastern Delaware County, New York, growth is largely restricted to reflect the protection of watersheds that supply water to the New York Metropolitan Area. For each county, we assigned each factor a weight and combined all factors into a single map that reflected growth pressures over the time period used for calibration. Based on tests of the model's performance both with and without the use of the expert-weighted exclusion–attraction layer, we note that the exclusion–attraction map developed in conjunction with county planners significantly improved model performance (Jantz et al. 2009).

While the exclusion–attraction layer weights areas differentially for potential development, whether an area undergoes nonurban-to-urban change is determined through the application of five growth rules, each of which is associated with a parameter value that can range from 0 to 100 (see Jantz et al. [2010] for specifics). These rules include *diffusion* (the development of single cells), *breed* (the development of a group of cells), *spread* (edge growth around existing urban areas), *slope* (resistance to development on steep slopes), and *road-oriented growth*. During model calibration, we tested multiple possible combinations of growth parameter values over a range of randomized trials, resulting in an optimized parameter set. The particular value derived for a growth parameter describes its influence in generating a particular pattern of development (e.g., dispersed vs. clustered) and also controls the overall amount of growth. Because of this, the SLEUTH model can be adapted to growth rates and patterns that are specific to a study area. By optimizing the model's ability to simulate the amount of development and the patterns of development, we were able to match the amount of growth and the number of urban clusters (a pattern metric) within 5% for all counties.

We measured model fits by comparing rate and pattern metrics for simulated urban growth (averaged over a set of trials) with observed urban growth (mapped from Landsat imagery); this allows one to discern the amount and direction of over- or underestimation produced for each metric and for each set of parameter values being considered and thus to "train" the model. This calibration phase of the SLEUTH modeling showed high accuracy across all scales for simulating rates and patterns of development that occurred between 1984 and 2005 (Table 2). For each county, we matched to within 5% the fractional difference in total urban area between the modeled and observed urban land cover maps and the fractional difference in urban clusters, indicating good performance of the model at the county scale. At the municipal scale, a regression analysis comparing modeled and observed urban extent explained 95% of observed urbanization in Sullivan County, New York, and 99% in Pike County, Pennsylvania (see r^2 values in Table 2). While accuracy declined at the finer scale of 1 km x 1 km cells, explained variance was still high and ranged from 82% to 92%.

Generating Forecasts of Development with SLEUTH

As described above, we developed a set of forecast scenarios that reflect different land use policies, using the same stakeholder-based approach that we used for model calibration. The scenarios essentially modify the exclusion–attraction surfaces to reflect various possible land use objectives or contingencies. These scenarios form the basis for

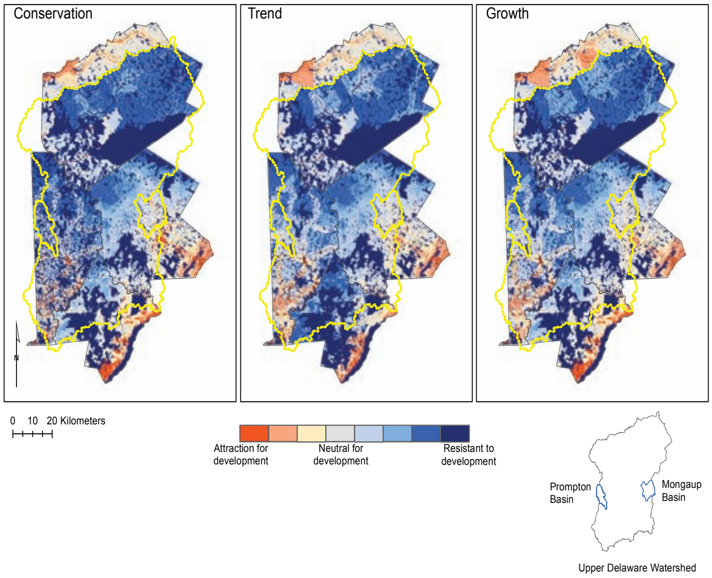

Figure 2. Scenarios depicting the probability of development, as derived from multiple data sources in collaboration with a wide range of stakeholders from four counties within the Upper Delaware River watershed. Areas in blue indicate resistance to development, gray areas are neutral for development, and shades of red indicate areas that are likely to attract development.

forecasts of urban development from 2005 to 2030. The forecasts are the combined result of 100 randomized trials for each scenario, with each cell assigned a probability of development by 2030. While each county generated a unique set of forecasts, they had common elements that allowed us to combine results for all four counties to represent the three watershed-wide scenarios described previously. In Figure 2, areas that are likely to attract growth are shown in shades of red, areas that are likely to repel growth are shown in shades of blue, and areas considered neutral for development are gray. Note that, in general, the Conservation scenario tends to have more areas shaded blue, whereas the Growth scenario tends to have more areas in red.

As noted above, SLEUTH generates maps that show the probability of development. We translated the probability maps into impervious cover maps by classifying any grid cell with a probability of development greater than 50% as developed land. We chose the 50% threshold on the basis of past work calibrating satellite imagery to aerial photos (Goetz and Jantz 2006; Jantz et al. 2005), but variations on this threshold are possible depending upon the user's desired application. We overlaid these development predictions on the 2001 NLCD land cover map and designated all areas that were either "developed" according to the model or "urban" in the NLCD map as developed areas. All other areas maintained their current land cover, as defined by the NLCD. We applied this approach to each of the three future land use scenarios (Table 3).

Figure 3 shows basin-wide forecasts of future urbanization for each scenario compared to current conditions (2005). As expected, the Conservation scenario shows the least overall growth compared to the other two scenarios (Table 2). Under the Trend scenario, low-density development expands significantly across the central watershed, and this outcome is enhanced under the Growth scenario. The Conservation scenario shows urbanization mostly intensifying in and around existing developed areas but also shows some dispersed, low-density development.

We then used each of these scenarios, and the differences between them, to explore the hydrologic implications of increasing urbanization and associated potential land management policies.

Hydrologic Modeling and Outcomes

We incorporated the land cover change forecasts described above into the SWAT model (within AGWA). SWAT is a quasi-spatial (distributed) model developed by the US Department of Agriculture Agricultural Research Service to predict the impact of land management practices on water, sediment, and agricultural

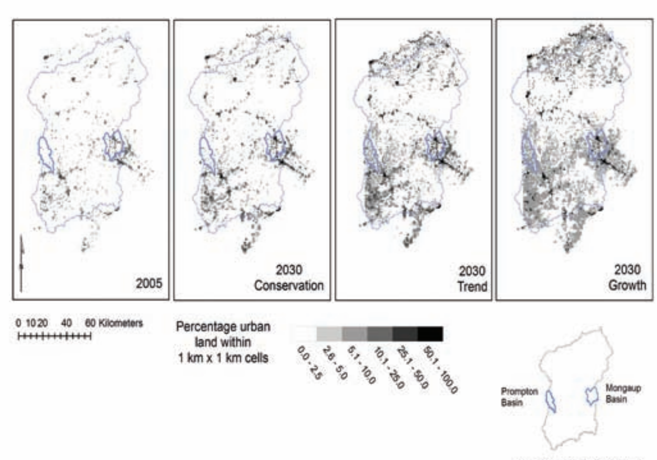

Figure 3. Forecasts of future impervious cover associated with different scenarios of urban development (three right panels) compared to impervious cover mapped in 2005 (left panel). The original 30-m resolution data have been rescaled to a 1-km resolution so that basin-wide patterns can be discerned.

Table 3. Area of land (in km^2) devoted to each land cover type for current (2005) conditions and under each development scenario. Although the percentage of developed land in the entire basin increased from 4.5% currently to 6% for the Growth scenario, forest was the predominant land cover for every scenario.

Land Use	Current	Conservation	Trend	Growth
Developed	363	422	448	479
% Change from Current	—	16	23	32
Undeveloped	7,423	7,365	7,339	7,308
Forest	6,212	6,181	6,168	6,152
Agricultural	927	904	892	880
Wetlands	189	187	186	185
Other	95	93	93	92
% Change from Current	—	–0.8	–1.1	–1.5

chemical yields in complex watersheds with varying soils, land uses, and management conditions (Gassman et al. 2007). It is a widely used model, partly because it was designed to operate within commonly available geographic information system (GIS) software. SWAT is simpler than fully spatial hydrologic routing models, but it requires fewer, less detailed inputs to produce results useful for assessing general hydrologic trends resulting from land use change.

As noted above, we delineated the Upper Delaware River watershed into sub-basins (shown in Figure 1), each parameterized by its hydraulic geometry, flow length, land cover, and soil properties. We ran the SWAT model to include each of these small watersheds, which experienced different rates of urbanization (shown in Figure 3). We used the Prompton sub-basin (1,954 km^2), located near the southwestern corner of the greater watershed, and the Mongaup sub-basin (2,576 km^2), located toward the southeastern corner of the basin, to assess the model calibration based on USGS river gauge measurements. The hydrologic model used daily meteorological data, specifically minimum and maximum temperature and precipitation. We assumed homogeneous climatic conditions throughout the study area, based on the meteorological station data collected at Liberty, New York, about 50 km from the Port Jervis river gauge in Pennsylvania. Although one could produce a more accurate model calibration using meteorological data from multiple stations, the present study instead focused primarily on the effects of land use change rather than, for example, spatial variability in precipitation. Additional model inputs were based on data sets freely available to any user, including 30-m NED data, a national hydrologic database of the stream network (flowlines), State Soil Geographic Database soils data, NLCD 2001 land cover data, and roads.

Hydrologic Model Calibration and Assessment

For model calibration, we refined the curve numbers for land cover parameterization to determine the best match between the modeled flow volume and the observed river gauge measurements. We parameterized each land cover class within SWAT using a number of factors (the curve numbers for each hydrologic group, percentage impervious cover, interception, and Manning's N), with impervious cover as a particularly important calibration parameter. We ran the model calibration using precipitation data from the Port Jervis gauge (downloaded from the USGS website, station 01434000) for the three-year period 2000–2002 at a monthly time step. The calibrated parameters produced a match (Nash–Sutcliffe coefficient) of 0.4, which is considered good (Moriasi et al. 2007). Although the model tended to slightly underpredict water yield in periods of low flow and slightly overpredict it in periods of high flow, the timing of the minima and maxima were close to the gauge-measured values.

Using the calibrated parameters, we then ran a 25-year simulation for the entire watershed, obtaining monthly averages for baseflow, runoff, and total water yield. In repeating this process for each future land use scenario we captured changes in impervious cover associated with development. We used the same parameter set for all model runs for each scenario to ensure that results would be consistent and comparable.

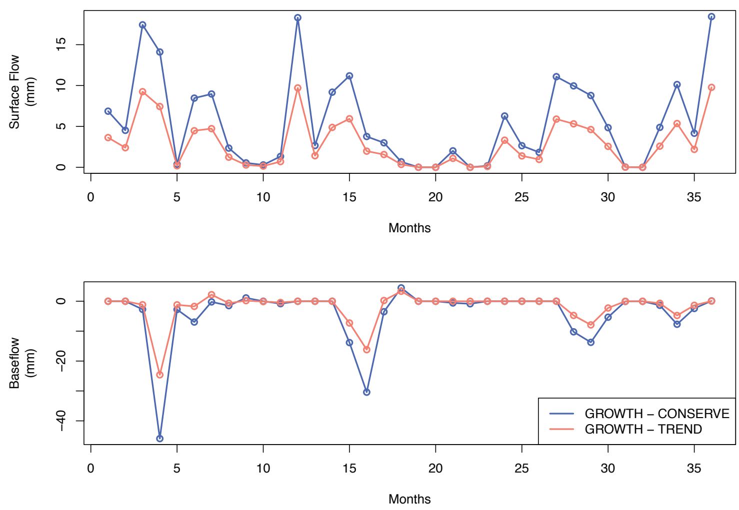

Figure 4a. A comparison of differences in monthly surface flow (top) and baseflow (bottom) for the Growth scenario relative to the Conservation scenario (blue line) and the Growth scenario relative to the Trend scenario (red line). The values reflect the differences between the Growth scenario values for surface flow and baseflow and the values from the other two scenarios. Results show greater amounts of surface flow and reduced amounts of baseflow when and where urban development increases.

Hydrologic Implications of Future Urbanization

The model for the land use change scenario forecasts out to 2030, baseflow would decrease and surface runoff would increase as the level of development intensified, with the Growth scenario showing the greatest changes and the Conservation scenario the least. These contrasts were emphasized when we examined the difference between the possible future land use scenarios, that is, the difference between the Conservation and Growth scenarios or between the Trend and Growth scenarios (Figure 4). The hydrologic implications of these comparisons were most pronounced for high-runoff events (Figure 4a), indicating that peak flows would be much greater if the stakeholder-identified conservation measures were not considered. Conversely, baseflows would be substantially reduced without conservation-oriented land management policies, meaning that headwater streams would be more likely to "run dry" or flow at very low levels at some point during the year. Changes of this magnitude would thus not only negatively impact stream biota (including native trout populations), but would also increase the likelihood of potentially damaging and expensive flood events in downstream communities. Spatially, sub-basins with more highly developed areas, particularly those in the northern and southwestern parts of the greater watershed, would experience the greatest changes (Figure 4b).

We also simulated erosion from the watershed (Figure 5) using the Modified Universal Soil Loss Equation, which is part of the SWAT model. We derived sediment yield using some of the other hydrologic variables produced by the model, such as surface runoff volume (Figure 4b) and the peak runoff rate, but also including soil type erodibility and factors related to management and topography that influence the sediment lag time in surface runoff. As with runoff and baseflow, changes in sediment load were clearly associated with the differences between the land use forecast scenarios. Sediment loading was greater where impervious cover and associated surface flows increased. We expected this since greater flow volumes from more impervious areas would have greater capacity to produce erosion and to transport greater loads and larger particle sizes. Sediment loading could be reduced in some areas of greater impervious cover if the associated urbanization process replaced agricultural lands, as opposed to, for example, forested lands. The results shown in Figure 5 thus represent the net effect of changes in urbanization among multiple land cover type transitions.

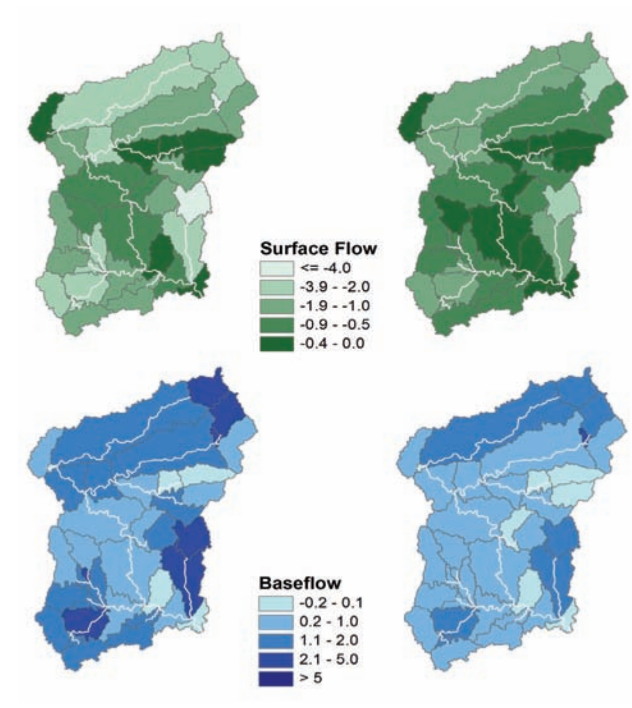

Figure 4b. Map depiction of the results from Figure 4a, showing differences in average annual surface streamflow (top) and baseflow (bottom) for the Conservation (left) and Trend (right) scenarios. As in Figure 4a, each is differenced with (subtracted from) the Growth scenario. These SWAT model outcomes are for a 25-year simulation period (2005 to 2030) but do not incorporate future climate (i.e., they are based on 1981–2006 precipitation records). Results clearly show reduced surface runoff and greater baseflow when and where increases in impervious cover are constrained.

The outcomes of the hydrologic analyses highlight the importance of spatial information in modeling the implications of impervious cover changes associated with urbanization. That said, we recognize that this analysis could be improved in a number of ways. For example, one could illuminate mechanisms of flow through areas of varying impervious cover by using more specific and detailed information on flow connectivity, or perhaps by distinguishing among different types of development (industrial, commercial, or residential). Similarly, available data did not allow for a consideration of the location of retention ponds or for the use of specific low-impact development techniques or best management practices that may mitigate some of the negative impacts of increased urbanization and associated impervious cover on hydrologic dynamics, such as peak runoff volume and increased flashiness of streams (Booth et al. 2002; Dietz 2007). This is not to say that site design can be expected to fully mitigate the impacts of land use change, but rather simply to note that, where the effectiveness of such efforts has been quantified, it may be possible to incorporate those outcomes into spatial scenarios, such as the ones presented here. Notably, the location and situation of new development, such as proximity to stream networks and surrounding topology, could be incorporated, via spatial distance–weighting schemes to assess the influence on streamflow patterns associated with flow routing across areas of interest (e.g., where mitigation efforts are planned). Related efforts may include simple metrics of housing density per square kilometer (Jacob and Lopez 2009) or other spatial metrics capturing more dispersed or clustered development (Steuer et al. 2010).

Even without explicitly modeling flow paths, the statistical averages for each sub-basin used in our SWAT modeling were able to capture the hydrologic implications of changing urbanization and how spatial changes in impervious cover accumulate across watersheds (see Figures 4a, 4b, and 5). The model captured significant changes in hydrologic dynamics and demonstrated the potential implications of urbanization associated with the various forecast land use change scenarios.

Conclusion

The findings presented here underscore the relevance of policies that broadly support growth strategies emphasizing resource protection and the positive benefits of reducing impervious cover at the landscape scale. Clearly lower growth levels, specifically in terms of minimizing impervious cover associated with development, will also minimize impacts on water resources. These results are perhaps not

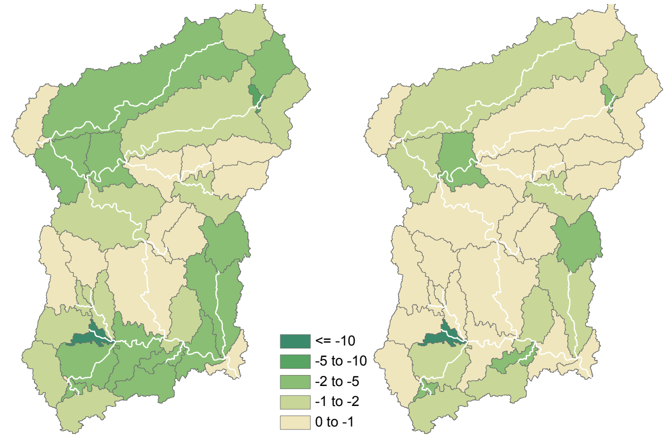

Figure 5. Percentage differences in average annual sediment yields for Conservation (left) and Trend (right) scenarios. As in Figures 4a and 4b, each map illustrates values that represent differences from the Growth scenario. Under the Conservation scenario, sediment loading was lower than under the Trend scenario, and sediment loading in both of these scenarios were lower than that under the Growth scenario.

surprising, but they highlight the importance of limiting the footprint of urban land cover if protection of water resources is a priority, and they demonstrate that this can be accomplished in a readily conveyed map form incorporating future land development and conservation scenarios. This case study of the Upper Delaware River watershed did not incorporate population or employment forecasts, so one should keep in mind that the three development levels we forecast might accommodate similar levels of population and employment growth, assuming a higher density (smart growth) in the Conservation scenario and a lower density (sprawl) in the Growth scenario. Proactive land use planning therefore remains paramount in this, and undoubtedly other, environmentally sensitive regions.

The involvement of stakeholders, especially county planners, in both the model calibration phase and the development of forecast scenarios greatly enhanced this modeling effort. First, the local knowledge of the study area provided by stakeholders improved the performance of the land use change model. Second, the forecast scenarios reflect what planners perceived to be realistic future alternatives. Our findings regarding the importance of land use policies that encourage spatially clustered development, higher densities, and the protection of natural lands provide support for the adoption of such policies.

All analyses and outcomes reported here were based on tools and data sets available to the land planning and watershed management communities, among other stakeholders.

They reflect the fact that most development routes surface flow across impervious surfaces to storm drain systems that effectively connect development to the hydrologic network (i.e., streams and rivers). To the extent necessary, different types of hydrologic models may allow one to incorporate various types of impervious cover and explicitly route flow by coupling them to realistic representations of storm drain networks, but our results show that users can reasonably and realistically predict the hydrologic implications of future development using an approach like the one we describe, which is intuitive, effective, and readily available.

Acknowledgments

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Land Cover Change in the Riparian Corridors of Connecticut

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Abstract

Riparian, or streamside, corridors are environmentally important areas critical to shoreline stability, pollutant removal, and both aquatic and terrestrial wildlife habitat. The University of Connecticut's Center for Land Use Education and Research recently conducted a statewide study of land cover change in riparian zones in Connecticut in an attempt to (1) characterize change in these areas and compare it to overall land cover change, (2) gain insight into what factors drive this change, and (3) determine priority areas for outreach to local land use decision makers. The amount of developed land, and increases in developed land during the study period (1985–2006), were lower in the riparian corridors than in the state as a whole. However, increases in riparian zone development within any particular town were closely correlated with overall increases in development in that town. These results suggest that overall development pressure is the primary driver of new development in riparian areas, though the effects of this pressure are mitigated to some extent by local zoning codes related to building suitability and by wetlands and watercourses regulations related to the protection of water resources. In addition to the town-level study, we studied riparian forest loss by watershed to help prioritize locations for targeted educational programs on riparian zone protection and restoration. This targeted outreach has generated considerable interest by town land use boards, and several restoration projects have already resulted. Land cover change information can be a powerful catalyst to watershed protection at both the local and statewide levels.

Riparian Corridors

The ecological and environmental importance of riparian areas is well documented. Often referred to as a transition zone, or ecotone, between two systems (Mitsch and Gosselink 1986; Naiman and Decamps 1997), riparian areas are biologically rich and provide numerous ecological functions. As the interface between aquatic and terrestrial communities, riparian areas are influenced by geomorphology and hydrology. These areas can harbor high biodiversity and provide ecological corridors (Naiman and Decamps 1997; Wenger 1999); they can also perform such functions as stormwater infiltration and filtration, stormwater management, flood water management, streambank stabilization, and sediment trapping (Bentrup 2008; Lowrance et al. 1997; Naiman and Decamps 1997; Wenger 1999). In addition, the combination of surface filtering of sediments, plant and microbial nitrogen uptake, and subsurface denitrification in these areas often makes riparian zones a sink for nitrogen, albeit with tremendous variability resulting from differences in soils, vegetation, buffer width, and other factors (Mayer et al. 2007; Gold et al. 2001). Studies in both urbanizing (Kaushal et al. 2008) and agricultural (Clausen et al. 2000) watersheds have demonstrated that riparian restoration can reduce the delivery of nitrogen to streams.

Because of the many beneficial functions of healthy riparian areas, land cover change in the riparian zone has become a topic of interest. Although the literature is not as robust as that on impervious cover, studies relating stream health to riparian forest cover—sometimes in combination with other land cover metrics—have begun to emerge (Goetz 2006; Goetz and Fiske 2008; Sawyer et al. 2004; Van Sickle et al. 2004; Snyder et al. 2003). For instance, Goetz et al. (2003) found that the best predictor of stream health, as determined by intensive multiparameter chemical and biological stream sampling, was a land cover index that combines watershed impervious cover and riparian area forest cover. Studies such as these typically focus on the site or stream reach level, using detailed data to look at the complex interplay of factors influencing stream health. The present study takes a broader view, making use of a unique, ongoing multitemporal land cover mapping project to focus on riparian corridors throughout Connecticut and to (1) document change in these critical areas over a long period of time and (2) help identify the factors influencing that change.

Methods

This study is an offshoot of Connecticut's Changing Landscape (CCL), an ongoing project of the University of Connecticut's Center for Land Use Education and Research (CLEAR) that

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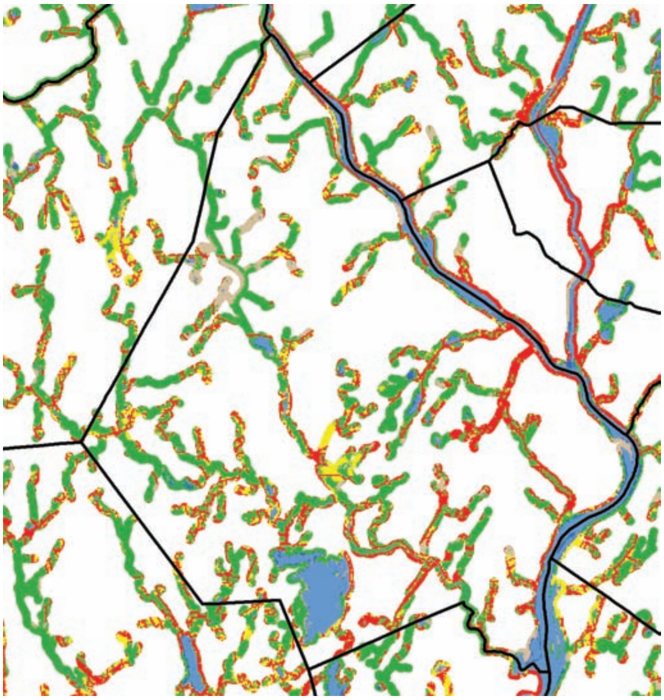

Figure 1. A town-level subset of the study area, showing land cover within a 300-foot (90-m) riparian zone. The area shown is about 200 km^2. Black lines are town boundaries, green areas are forested, red areas are developed land, yellow areas are turf and grass, and blue areas are water.

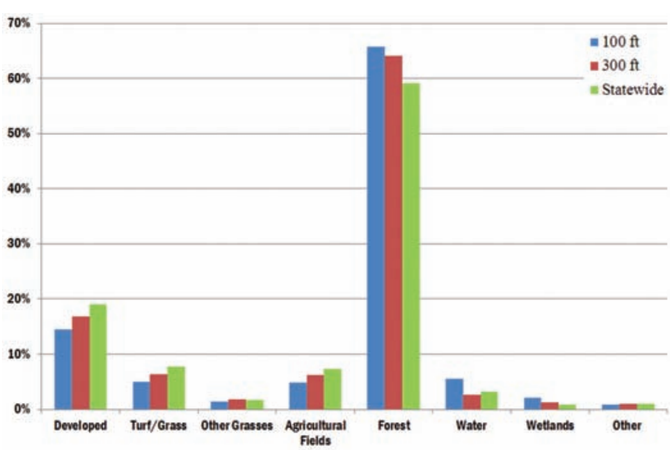

Figure 2. Percentage cover of 2006 land cover categories for the 100-foot riparian zone statewide (blue), the 300-foot riparian zone statewide (red), and the entire state (green).

uses remote sensing technology to chart changes in the state's major land cover categories over time. CLEAR developed the CCL project (see Hurd et al. 2003; CLEAR n.d.[a]) specifically to enable "apples-to-apples" comparisons of multitemporal land cover data sets, all based on 30-m pixel Landsat imagery and dating back to 1985, the first year for which imagery of this resolution is available. Hurd et al. (2003) used cross-correlation analysis—which employs statistical analysis to identify pixels indicating a potential change between images (Koeln and Bissonette 2000)—to produce a consistent set of land cover data sets that one can assess for land cover change over time. They classified the potentially changed pixels and merged them with the 1985 classification to create the 1990 classification; they repeated this process for the 1995, 2002, and 2006 classifications. All five final classification data sets have 12 categories; the major categories of interest are developed land, turf and grass, agricultural field, and deciduous and coniferous forest (Hurd et al. 2003).

In addition to basic land cover change data, CLEAR also has conducted several subsidiary analyses that use the land cover data as the basis for a closer examination of landscape indicators of interest. For instance, CLEAR researchers adapted a landscape fragmentation analysis originally developed by the US Department of Agriculture (USDA) Forest Service (Vogt et al. 2007) for its 30-m data and applied it to Connecticut to go beyond simple forest cover data and provide information on the status of "core forest" areas in the state (Hurd et al. 2010. In another study, researchers analyzed land cover change over areas designated by USDA as having "prime" or "important" agricultural soils and compared it to land cover change statewide (CLEAR n.d.[c]). To this list we add this study, which focuses on land cover change within riparian zones across Connecticut.

We conducted the riparian corridors study by analyzing the CLEAR CCL multitemporal land cover information for areas on both sides of Connecticut streams, lakes, and other water bodies. We created a seamless, continuous GIS data layer of water edges that included not only small stream lines (as determined from statewide hydrography data), but also shorelines of rivers, wetlands, tidal marshes, and water bodies that intersected the stream lines. Thus, rather than follow the stream lines through connected water bodies and wetlands, we used the outside edges of these features as the starting points of the corridor area (Figure 1). To keep the focus on riparian areas and to maintain analytical feasibility, this study did not include inland wetlands and small water bodies that were not directly connected to the stream network. Although the statewide hydrography data can vary from the actual location of smaller streams, the analysis provides a useful overview at the state, town, and watershed levels.

We extracted land cover information for 1985 (T^1) and 2006 (T^2) and the land cover change information for 1985–2006 for this continuous riparian zone. We measured land cover as an area and as a percentage of the unit of interest (the town or watershed), and we measured land cover change as an absolute change (hectares T^2 – hectares T^1), and as a relative change (% area T^2 – % area T^1). The study looked at the riparian zone both 100 feet (30 m) and 300 feet (90 m)[1] to either side of the water features (Figure 1). Since the land cover data have a ground resolution of 100 feet by 100 feet, the 100-foot corridor analysis involves a very small sample size, which we feel approaches the limit of the appropriate use of the land cover data. However, the study included the 100-foot corridor because it encompasses the regulated review zone in many Connecticut towns (see next section). As discussed below, the 100-foot data correlate strongly with the 300-foot data; this raises our confidence in the usefulness of these data.

Results and Discussion

Statewide

We first examined the current (2006) state of land cover for the 100-foot corridor (an area of about 120,700 ha) and the 300-foot corridor (about 343,600 ha) for the state of Connecticut (Figure 2). Statewide, the percentage of forest class increased with proximity to water features. For the 100-foot corridor, forest accounted for more than two-thirds of the area (67.1%); developed land (14.5%) and the closely associated category of turf/grass (5.1%) were the next most prevalent. For the 300-foot corridor, forest was still the most prevalent land cover (64.1%), with developed land (16.8%) and turf/grass (6.3%) again rounding out the top three. By way of comparison, the overall statewide figures from the CCL project were 58.8% forest, 19.0% developed, and 7.7% turf/grass.

We then compared the 2006 data to the 1985 data to evaluate changes in land cover in the riparian zone. Figure 3 shows the change, in hectares, of each major land cover class over the 21-year study period for the 300-foot corridor. The biggest changes were apparent for developed

Figure 3. Absolute change (ha) from 1985 to 2006 in the 300-foot riparian corridor, by land cover class.

land, which increased by more than 7,700 ha, and for forested land, which decreased by more than 10,000 ha. As with the findings in the parent CCL study, the combined increases in the three land cover categories considered by CLEAR researchers to approximate the "urban footprint"—developed, turf/grass, and "other grasses" (13,366 ha)—roughly balance the combined losses to the agricultural field and forest categories (12,940 ha).

A focus on the developed land cover class showed less developed land, and a smaller increase in developed land over the study period, with proximity to water features. Table 1 compares the percentage of the developed class within the 100- and 300-foot corridors with the results for the entire state, as determined by the CCL project. The relative change in developed land was 1.7% for the 100-foot corridor, 2.3% for the 300-foot corridor, and 3.0% for the entire area of the state.

Table 1. Percentage developed land in the 100-foot and 300-foot riparian corridors and for the state as a whole, 1985–2006.

Area of Interest	1985	2006	21-Year Change
100-Foot Corridor	12.7%	14.4%	+1.7%
300-Foot Corridor	14.5%	16.8%	+2.3%
Entire State	16.0%	19.0%	+3.0%

Town-by-Town Assessments

Because land use in riparian areas (as with all areas in Connecticut) is determined at the municipal level, CLEAR also looked at the data by town. One objective was to see if this study could shed any light on the long-term impact of

[1] We use English units for the corridor widths because this is the unit we used in the analysis to better correlate with the regulatory review widths commonly found in town regulations.

inland wetlands and watercourses regulations. Since 1972, Section 22a–42c of the Connecticut General Statutes has required each of the State's 169 municipalities to establish an inland wetlands and watercourses agency. These local bodies are empowered to establish "upland review areas,"[2] within which they may regulate activities based on their impact to wetlands and watercourses. Note, however, that (1) the width of these areas varies from town to town and (2) they are not "no-development" zones but only zones that trigger a review by the local agency. Thus, the consequences of these regulations vary considerably as a result of differences in the local interpretation of a given project's environmental impacts. The Connecticut Department of Environmental Protection estimates that about 80% of the towns have a review zone of 100 feet, and most of the other towns use review areas of 50 to 200 feet; however, a few towns have review zones of up to 600 feet (Connecticut Department of Environmental Protection 2010).

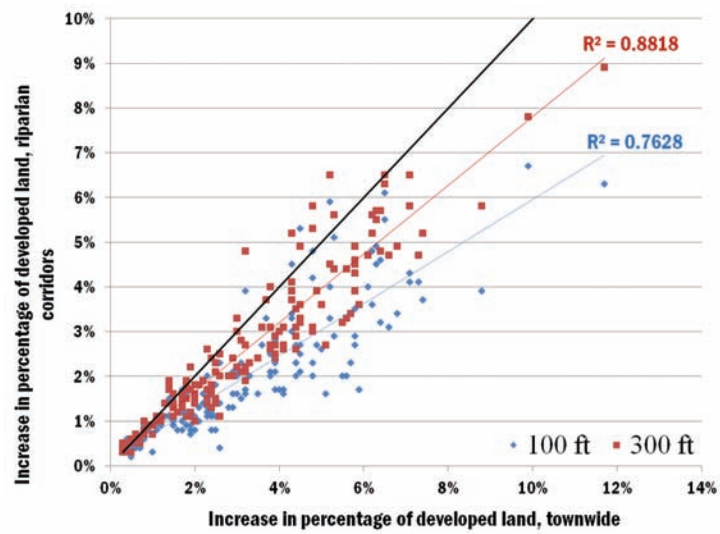

Figure 4. The relationship between the percentage of new development in a town (x-axis) and the percentage of new development in the town's riparian corridors (y-axis) for the 169 towns in Connecticut. Blue points reflect the 100-foot riparian zone, red points the 300-foot zone. Most towns fall below the black line, indicating a higher percentage of development in the town as a whole than in the riparian areas.

This study looked at the relationship between new development in the riparian zones and new development, overall, for each of Connecticut's 169 towns. We plotted the percentage of each town covered by new development during the 1985–2006 period against the same metric for both the 100-foot and 300-foot corridors (Figure 4). The black line in Figure 4 represents a one-to-one relationship between the percentage developed area in the entire town and the percentage developed area in the riparian corridors. That is, a point that falls on the black line denotes a town in which the percentage increase in developed land in the riparian zone is the same as that in the entire town. As Table 1 suggests, most of the data points fall below the black line, indicating that most individual towns had less new development in the riparian areas than in the town as a whole. However, a simple regression analysis shows a very strong correlation between the town and riparian corridor data for both the 100-foot and 300-foot zones. Thus, the greater the amount of new development in a given town, the greater the amount of new development is likely to be in the riparian areas of that town.

The strong, statistically significant correlation between town and riparian development ($p < 0.001$ for both regressions) indicates that local development pressure is a principal factor controlling riparian conversion—not a surprising result. The fact that the relative amount and rate of increase of new development in riparian corridors is lower than in their respective towns, overall, is most likely due to a combination of factors. Local regulation of riparian areas through the state inland wetlands and watercourses program no doubt plays a role in reducing or retarding development near watercourses for particular towns. However, this is surprisingly difficult to prove for several reasons. First, town regulations can change and, if they do, it is highly unlikely that the change will be exactly concurrent with the dates of the land cover data. Also, even with the same review zone regulation, some town commissions are quick to grant a permit while others are more restrictive. Finally, examining town records to track the regulatory history of 169 municipalities is difficult and time consuming. Despite these confounding factors, it seems likely that, if inland wetlands and watercourses regulations were having a widespread effect throughout the state for the past 40 years, one might expect to see additional scatter in Figure 4, created by more uniform low riparian development rates that are independent of local development pressure.

This raises the possibility that lower levels and rates of development in riparian areas may be due more to intrinsic impediments to development than to regulatory factors. To further explore this hypothesis, we used the riparian zone buffer analysis previously applied to our land cover data

[2] Upland review areas are widely known in the state as "buffers." To avoid confusion, we do not use this term when referring to the study area; instead, we use the terms *riparian corridors* or *zones*.

and examined slope and soils within this same zone. Steep slopes (over 20%) and USDA-designated "poorly drained" and "very poorly drained" soils are barriers to development commonly referenced in local zoning codes (B. Hyde, University of Connecticut, pers. comm. March, 2011). As zoning limitations, these are also "regulatory" controls, but they are based on the ability of a given site to support development rather than its potential impact to water or other natural resources.

The 300-foot riparian zone had only a very slightly higher percentage of slopes over 20% (15.8% vs. 15.4% for the state) but contained about twice the amount of poorly or very poorly drained soils (26.5% vs. 13.4%). This lends credence to the supposition that lower levels of development in riparian areas are influenced by building-related zoning restrictions as well as environment-related regulation of wetlands and watercourses.

Ultimately, this statewide view is of insufficient detail to draw firm conclusions on the impact of local regulations. The wide range of upland review zones, combined with the even wider variability in local interpretation of permissible environmental impacts, makes it extremely difficult to tease out the effectiveness of these laws. Detailed town- and site-level work, involving town hall records rather than land cover pixels, are needed to further advance our understanding of the factors driving riparian conversion. We hope to study the record of land use decisions in several of the outlier towns in Figure 4 to try to determine why the riparian rate of development in these towns is so different from the townwide average.

Assessment by Watershed

We also determined land cover status and change in riparian corridors by watershed, with a focus on the forest land cover class. This study examined the 333 subregional watersheds in Connecticut, a state designation that approximates the US Geological Survey hydrologic unit code 12–level of organization with an average size of about 38 km^2. As previously noted, studies suggest that forest cover in riparian zones can be a good indicator of watershed health, particularly if used in combination with overall watershed metrics like impervious cover (Goetz et al. 2003). The current study simply looked at relative change within the 300-foot corridor of these watersheds during the 1985–2006 study period (Figure 5).

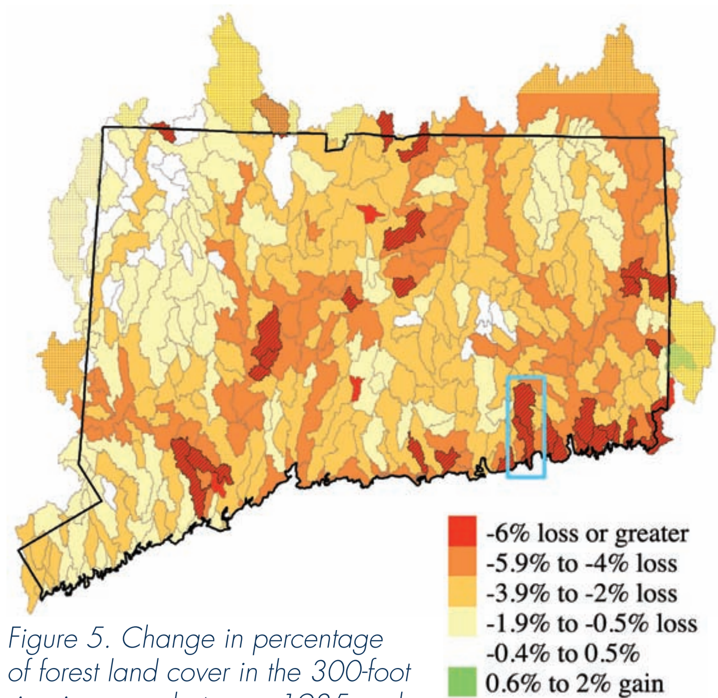

Figure 5. Change in percentage of forest land cover in the 300-foot riparian zone between 1985 and 2006, by watershed. Negative numbers denote a decrease in the percentage of forested land within the riparian corridor of the basin due to the conversion of forest to some other land cover type. The 25 watersheds with the greatest percentage loss in forest in the 300-foot riparian zone are cross-hatched.

The 25 subregional watersheds with the greatest percentage loss of riparian forest land during the study period appear in several parts of the state, with a noticeable concentration along the southeastern coast. Not surprisingly, these areas correlate closely with areas of overall growth, as determined by the parent CCL project. Of concern to smart growth advocates and others is that these areas are not, for the most part, located along the state's traditional urban corridors, which lie along the southwestern coast and through the middle part of the state.

Making Use of the Data

Based on this analysis and its identification of development "hot spots," the Niantic River watershed along the coast in southeastern Connecticut was identified as a priority area for outreach (Figure 5, blue box). The timing was fortuitous in that the Connecticut Department of Environmental Protection had recently completed a watershed plan for the Niantic, which identified nonpoint source pollution as the primary cause of impaired water quality. In addition, the process of developing the plan had attracted the interest and involvement of both town officials and a local nonprofit, and a watershed coordinator to oversee implementation of the plan had recently been hired. Thus, the Niantic River watershed stood out as an excellent location for riparian area management and protection, both from a practical standpoint and as a model for other watersheds.

With funding from the Long Island Sound Study Futures Fund and the National Fish and Wildlife Foundation, Sea Grant and CLEAR researchers and the Niantic River watershed coordinator developed a series of educational workshops for both municipal officials and local landowners. The workshops used the statewide and local results of the CLEAR riparian zone analysis as a jumping off point to discussing protection and restoration issues. The watershed encompasses four towns, two coastal and two inland, providing the opportunity to discuss not only riparian corridor protection, management, and restoration, but also the ecological importance of, and relationship between, sensitive estuarine and riparian habitats.

In coordination with the environmental planners in each of the four towns, we developed customized riparian workshops for land use commissions. In addition, we conducted two workshops for local land owners, one focusing on coastal habitat and the other focusing on the importance of headwater streams within the watershed. Following these presentations, during the spring and summer of 2010, a dozen Connecticut towns have participated in or requested similar workshops, with more than 400 participants to date. The ability to provide municipal officials with town-specific data and trends developed through the land cover analysis is serving as a unique catalyst for the review and revision of municipal comprehensive plans and regulations. In addition to the protection of riparian corridors, these workshops have sparked interest in on-the-ground riparian area management and restoration projects. Four such projects are underway and will serve as templates for other interested groups. Finally, plans are underway to replicate the Niantic effort in other watersheds identified by this analysis as experiencing the most rapid loss of riparian vegetation.

The riparian corridor analysis is the latest of several studies derived from the CCL project. Based on our experience with prior CCL-related studies, we believe that the data will be widely used. CLEAR's goal is to make the data from all of our land cover studies easily accessible and understandable to a broad spectrum of users, through a combination of direct outreach and project websites. The websites contain information in many formats, from simple diagrams to charts, data tables, and maps, including interactive maps (CLEAR n.d.[b]; Rozum et al. 2005). Land cover change information is used in a variety of ways, from enriching local comprehensive plans, to fueling additional research, to informing state policy. For instance, although only recently completed, the riparian corridor study results have already informed debate in the last two state legislative sessions on whether local inland wetlands and watercourses upland review zones should be made more uniform and transformed into "no-development" areas.

Summary

Connecticut is experiencing urban development in upland areas and critical riparian corridors alike. A 21-year record of directly comparable land cover change enables us to evaluate, on a broad scale, what is happening in these areas. More than 7,700 ha of riparian vegetation in the 300-foot zone was converted to the "developed" land cover class between 1985 and 2006, and another 4,000 ha was converted to turf and grass. The percentage of the landscape in the developed category, and its increase over the 21-year period, are lower within the state's riparian zones than for the state as a whole. This is undoubtedly due to the influence of a complex combination of jurisdictional and intrinsic landscape factors. Although this study did not definitely determine the exact interplay of the drivers behind this change, the results show a strong correlation between development rates in riparian zones and those of the towns in which the riparian zones are located; this suggests that local development pressure is chief among the driving factors. Secondary factors that may explain the lower amounts and rates of development in riparian zones include local regulation of development based on suitability for building, probably enhanced by local regulation based on possible impacts to wetlands and watercourses. Detailed town-by-town analysis is needed to determine the true nature of these relationships.

Land cover data generated at a resolution of 30 m may seem almost mundane in a world where high-resolution imagery is readily available on personal computers and mobile devices. However, our 20-year experience at CLEAR, reinforced by our work to date with the riparian study, demonstrates that these data can be very effective at stimulating discussions about sustainable land use plans and regulations and catalyzing changes to those plans and regulations.

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Bottomland Hardwood Forest Influence on Soil Water Consumption in an Urban Floodplain:

Potential To Improve Flood Storage Capacity and Reduce Stormwater Runoff

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Abstract

Despite considerable interest in the rehabilitation of wetland and stream ecosystems, guidance for the restoration of forested urban floodplains is limited. This study provides baseline data describing floodplain vegetation and soil characteristics relationships in Lower Hinkson Creek, a Clean Water Act Section 303(d)–listed impaired stream located in Columbia, Missouri. We quantified the dominant tree species composition, basal area, and leaf area index of a bottomland hardwood forest (BHF). We then estimated differences in soil infiltration, bulk density, porosity, volumetric water content (VWC), and water storage between paired BHF and agricultural sites within the floodplain. Infiltration rates varied but were significantly greater in the BHF site with a 61% difference in mean infiltration between the two sites. Locations of high maximum infiltration rates were associated with locations of large trees (namely, eastern cottonwood). Vegetative influence on soil characteristics is apparent, particularly soil VWC above a soil depth of 50 cm. Results demonstrate the potential benefit of sustaining or reestablishing floodplain forests to enhance storage capacity, attenuation, and consumptive water use, thus reducing flooding and mitigating stormwater runoff problems in rapidly developing urban environments.

Introduction

Flood events cause more than $3.5 billion in property damage in the United States every year. Since 2004, more than 40% of annual natural disasters in the United States have been related to flooding (Federal Emergency Management Agency 2009). With increasing local, state, and federal expenditures for flood disaster relief, flood prediction and control have become critically important. A key turning point in floodplain management was the promulgation of the National Flood Insurance Program (NFIP) by the US government in 1968 through the enactment of the National Flood Insurance Act. Subsequent to the establishment of NFIP and the environmental movement of the 1970s, land use planners placed increasing emphasis on floodplain management to minimize flood damage (Mays 2001). In recent years, green infrastructure, including wetlands, ponds, rain gardens, and forested buffer strips, has become an increasingly popular alternative to classic engineered structures for flood and water quality mitigation (Mitsova et al. 2011). Nowhere are green alternatives more desirable than in urban environments, where compounded human disturbance dramatically alters natural resources (Gill et al. 2007). Unfortunately, little guidance is available for the restoration of forested urban floodplains. Methodologies for the assessment and restoration of the ecological functions of forested wetlands in wildland floodplains may be of limited value in urban settings. This is mainly due to the nature and magnitude of the modifications to the hydrologic regime and limitations to the restoration of a more natural regime in urban watersheds (e.g., Simmons et al. 2007; Ravit et al. 2008).

Most floodplain bottomland hardwood forests (BHFs) in the United States were removed in the nineteenth and twentieth centuries in efforts to harvest valuable timber resources and cultivate the rich underlying soils (Abernathy and Turner 1987). In many instances, this required the installation of drainage and flood control structures, such as drainage tiles, ditches, levees, and dams. The channels of many streams and rivers were straightened and enlarged to further reduce flooding. Drainage and flood control structures and channel alterations, coupled with changes in vegetation and soils, drastically altered the hydrology of streams, floodplains, and remnant BHFs (Carter and Biagas 2007). In the lower Midwest, restoration efforts in recent decades have mostly focused on wide alluvial floodplains of large rivers, especially in the lower Mississippi Alluvial Valley (Stanturf et al. 2001). However, riparian and floodplain BHFs along lower-order tributary streams also possess important hydrologic, biogeochemical, and habitat functions and provide ecosystem services. For example, Thomas and Nisbet (2007) showed through modeling that forested riparian zones in southwestern England increased the soil water level by 270 mm and increased flood storage by 15% to 71%; they argued that floodplain

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woodlands could be strategically located to reduce flooding. Additionally, forested stream buffers stabilize banks, improve water quality, maintain soil moisture, and ameliorate local microclimates (Seobi et al. 2005; Wynn and Mostaghimi 2006). Forested floodplain ecosystems are highly productive and often support high plant species richness. A variety of factors influence plant species composition, including soil nutrient status, topographic relief, and soil water flux (Lytle and Merritt 2004; Lytle and Poff 2004) as well as species life history characteristics (Holmes et al. 2005; Unger 2008). Numerous studies indicate that BHF vegetation is sensitive to environmental variability and perturbation (Grell et al. 2005; Tockner et al. 2010, and references therein). The effect of any one of the many floodplain attributes on vegetation structure is difficult to detect given interactions among physical processes, vegetative responses, and the stage of succession.

Urban riparian and floodplain forests of lower-order tributaries may play particularly important roles in the absorption, attenuation, and treatment of storm flows as a result of coupling to urban environments. However, lower-order floodplain capacity to attenuate flow and pollutants from source watersheds is largely unknown, especially in urban settings. An improved quantitative understanding of floodplain processes is critical for restoring the habitat and conditions of the Missouri and Mississippi Rivers as well as the hypoxic zone of the Mississippi Delta in the Gulf of Mexico (Alexander et al. 2008). Because of the tendency toward public ownership, urban floodplains are ideal candidates for restoration. In addition, their location in or near population centers make them well suited for public education and involvement in restoration activities. As a good example of this process, City planners in Portland, Oregon, are restoring the historic floodplain to improve flood storage and attenuation, habitat, and multiple other benefits (for more information, see Portland Bureau of Environmental Services [n.d.]).

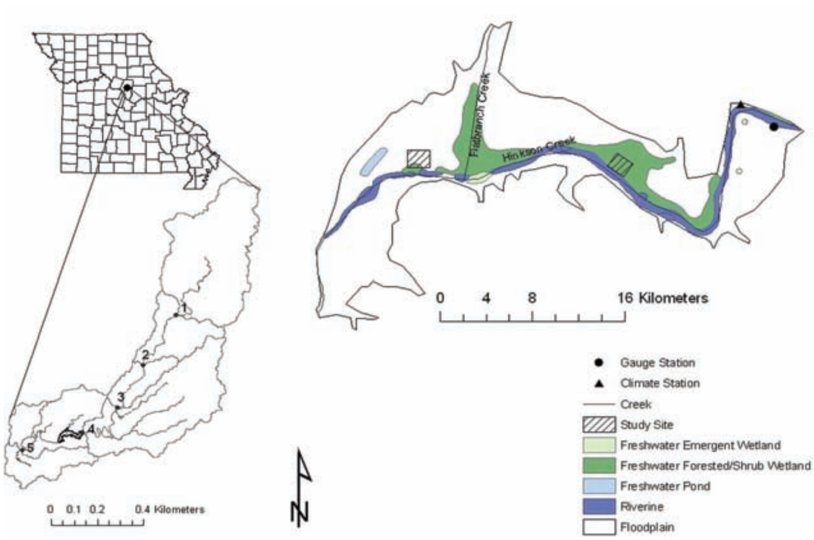


Figure 1. Locations of five nested gauge stations (left) and lower Hinkson Creek floodplain study sites (right, crosshatched boxes) in the Hinkson Creek watershed in central Missouri, USA.

Investigations that provide a quantitative understanding of floodplain and vegetation community process relationships and information for science-based urban land-use decisions are critically needed.

Objectives

The goals of this study were to (1) quantify the species presence, basal area, and canopy density of a section of second-growth BHF; (2) quantify the difference between the BHF site and an agricultural (Ag) site in soil infiltration capacity; and (3) estimate soil water storage differences between the two sites through a soil characteristics analysis. Such data are necessary to estimate the potential role of BHFs in flood attenuation by means of increased infiltration, storage capacity, and transpiration consumptive water use.

Methods

This project focused on an urban floodplain reach of Lower Hinkson Creek located in the city of Columbia (population 108,000; US Census 2011), in central Missouri, USA (Figure 1). The reach is located between two permanent hydroclimate gauge sites on the main channel that are part of an urban watershed study containing five nested gauge sites implemented in 2008 (Hubbart et al. 2010). The watershed contributing to the floodplain reach investigated in this study contains a large portion of the most intensively developed land in the city of Columbia.

The Hinkson Creek watershed (HCW), which is located within the Lower Missouri–Moreau River basin in central Missouri, is classified as a Missouri Ozark border stream located in the Outer Ozark Border Ecological Subsection (Nigh and Schroeder 2002). Average annual temperature and precipitation (from a 30-year record) is approximately 14°C and 980 mm. Soil types range from loamy till with a well-developed clay pan in the uplands (Chapman et al. 2002) to thin cherty clay and silty to sandy clay in the lower reaches. Land use in the watershed is approximately

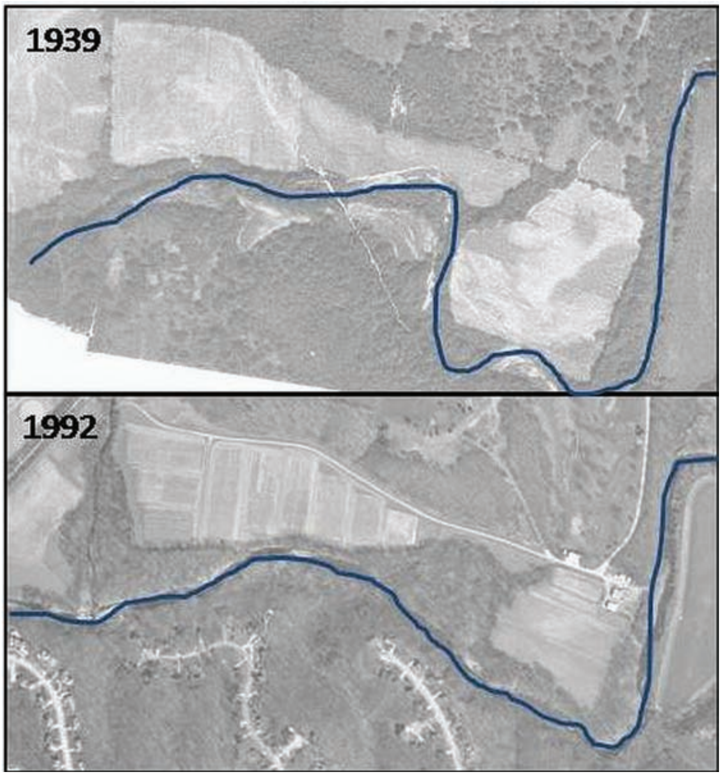


Figure 2. Aerial photographs of Hinkson Creek in 1939 and 1992 flowing through the floodplain study reaches in central Missouri, USA.

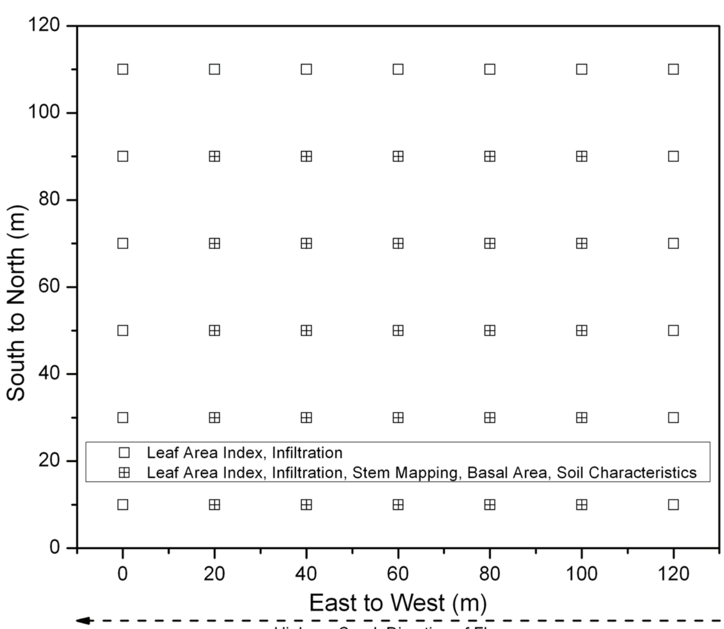


Figure 3. Study grid design of the current floodplain study in central Missouri, USA.

34% forest, 38% pasture or cropland, and 25% urban area; the remaining land area consists of floodplains, wetlands, and open or shrubland/grassland areas.

We located floodplain study grids within one large remnant section of BHF and one Ag section within the study reach (Figure 1, right) during the summer months of 2010 (July–August). The woody species *Acer saccharinum* (silver maple), *Acer negundo* (boxelder), *Ulmus americana* (American elm), *Populus deltoides* (eastern cottonwood), and *Juglans nigra* (black walnut) dominated the BHF site. A woody understory layer was absent, but *Urtica dioca* (stinging nettle) and several grass species of limited occurrence, such as *Elymus virginicus* (Virginia wild rye), dominated herbaceous vegetation. A comparison of historic photographs with photographs from 1992 indicate that Hinkson Creek was more sinuous in the early 1900s (Figure 2). Study grid dimensions included an inner 80 m x 80 m (6,400-m^2) grid equally subdivided for tree diameter measurements and soil sampling (n = 25). We used a larger 120 m (east–west) x 110 m (north–south; 13,200-m^2) grid encompassing the inner grid for canopy cover and soil infiltration work (n = 42; Figure 3). The southern 120-m side of the larger grid was located 10 m from and parallel to Hinkson Creek. We replicated infiltration and soil characteristics work within a grid of equal dimensions approximately 800 m downstream in an area of the floodplain that has been used for agriculture for at least the past century (Figure 1).

Basal Area and Leaf Area Index

This study used basal area and leaf area index (LAI) as a proxy for mass and energy exchange from the forest canopy (Running and Coughlan 1988; Santiago et al. 2000). LAI is defined as the ratio of the total upper leaf surface of vegetation divided by the surface area of the land over which the vegetation grows (Campbell and Norman 1998). Tree transpiration is positively correlated with LAI (Granier et al. 1996), which is a structural determinant of transpiration (Running and Coughlan 1988). Structural composition of forest canopies (i.e., stem diameter and basal area) is also important because of the relationships between boundary layer conductance (Campbell and Norman 1998) and plant water flux from forest soils by transpiration (Meinzer et al. 1997).

We identified each tree within the entire inner 80 m x 80 m BHF area to the species level and measured the diameter at breast height (dbh = 1.3 m) of every tree greater than 10 cm dbh. These data enabled a calculation of basal area (i.e., the cross-sectional area of each individual tree) by species and for the entire study area. To determine the age of dominant species and the establishment year for the study BHF, we used an increment borer to extract one core from each tree on the site with dbh greater than 10 cm (n = 142). We mounted and sanded the increment cores and counted the rings.

We used ceptometer and hemispherical methods to estimate LAI. The ceptometer method required the use of a Decagon Devices LP-80, which measures average photosynthetically active radiation (PAR) along an array of 80 sensors mounted on a 1-m light bar. The amount of PAR transmitted through a vegetative canopy is a direct function of canopy structure and density. We placed one ceptometer on a tripod approximately 1.6 m above ground level within a clearing to log the reference PAR. We compared the reference PAR to the PAR beneath the canopy to calculate the ratio between the two measurements. We collected PAR measurements in four cardinal directions at all 42 sampling locations within the BHF site (Figures 1 and 3; n = 168) and calculated the PAR according to methods described by Decagon Devices (2006). The hemispherical method required a Nikon D60 digital camera with a Sigma 4.5-mm circular fisheye lens. We took photographs at the same time and location as the PAR data with the lens pointing vertically upward and the camera base mounted 1.3 m above ground level. We analyzed hemispherical photographs using Gap Light Analyzer software (Frazer et al. 1999), which relates the gap fraction (the percentage open sky vs. leaf obstruction) to LAI (Stenberg et al. 1994).

Infiltration Capacity

This study used double-ring infiltrometers to measure soil infiltration. The purpose of the double-ring assemblage is to create a one-dimensional flow of water from the inner ring. An inner ring is driven into the soil and a second, larger-diameter concentric ring helps control the flow of water through the inner ring. Water is supplied either with a constant or falling head condition, and the infiltration rate of the inner ring is recorded over a given time period. This arrangement accounts for the lateral movement of water around the infiltration ring blades, thus improving the accuracy of infiltration estimates (Bodhinayake et al. 2004). For more information, please see ASTM International (n.d.).

Soil Water Storage

We estimated soil water storage based on analyses of soil characteristics (e.g., bulk density, porosity, and volumetric water content [VWC]), using the soil core method (Hillel 2003). The soil core method requires a cylindrical metal sampler to be driven into the soil to remove a known volume of soil (a core), which is weighed and then oven-dried at 105°C to remove nonstructural soil water. Oven-drying typically takes 24 to 48 hours, or until the sample mass no longer changes with additional drying. The oven-dried mass of the soil sample is then determined by weighing, and the indices listed above can be calculated (Hillel 2003). We retrieved soil cores from depths of 0, 15, 30, 50, 75, and 100 cm, enabling the computation of soil water to a depth of 1 m.

Analyses

Data analyses consisted of graphical, descriptive, and statistical comparisons. We conducted a two-sample *t*-test to identify statistically significant differences between the BHF and Ag floodplain sites in independent infiltration capacity samples (n = 25, each site; Figure 3; Zar 1999). Two-way analyses of variance (ANOVA) on soil characteristics tested for significant differences between population (site) means and independent soil depths (Zar 1999). After each two-way ANOVA, a Tukey's post hoc multiple-comparison test compared the nominal variables and measurement variable in all possible combinations (Zar 1999).

Results and Discussion

The climate over the period of study was typical for the region. The city of Columbia received 372 mm of precipitation between July 1 and August 31, 2010, and more than 1,346 mm of precipitation throughout 2010. The average temperature during the period of study was 26.3°C.

Silver maple, boxelder, and Amercian elm were the numerically dominant tree species at the BHF site (n = 41, 39, and 35 individuals, respectively). The frequency distributions for both tree dbh and basal area (Figure 4) reveal the negative exponential (i.e., few old or large trees vs. more young or small trees) of a forest with a relatively young cohort. Size (dbh) suggests that the young cohort consisted principally of silver maple and American elm. Eastern cottonwood, however, dominated in terms of basal area (9.01 m^2) in the study plot and was represented by few (n = 9), but large, individuals (Table 1). This observation may be of particular importance in floodplain management since cottonwoods are very successful in shallow groundwater environments

Table 1. Descriptive statistics for stem diameter (cm) and basal area (m^2) in a bottomland hardwood forest floodplain in central Missouri, USA.

Species	Number of Individuals	Average dbh (cm)	Average Basal Area (m^2)	Total Basal Area (m^2)
Acer negundo	39	26.98	0.07	2.55
Acer nigrum	1	21.20	0.04	0.04
Acer saccharinum	41	48.19	0.21	8.66
Aesculus glabra	3	10.90	0.01	0.03
Celtis occidentalis	2	9.25	0.01	0.01
Gleditsia triacanthos	2	23.75	0.05	0.09
Juglans nigra	7	42.49	0.15	1.06
Populus deltoides	12	95.78	0.75	9.01
Ulmus americana	35	19.97	0.04	1.25
Total				22.71

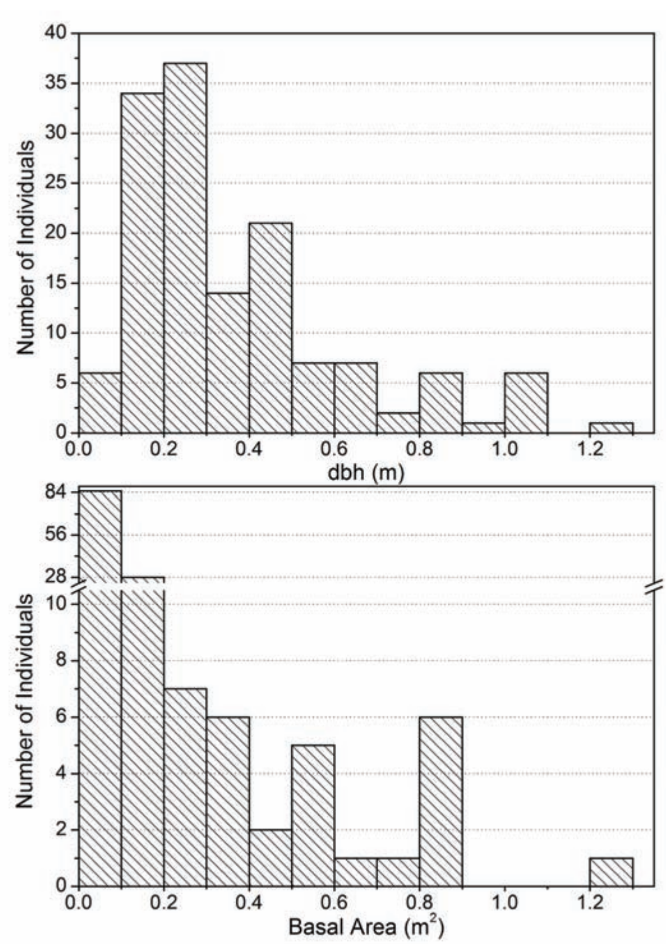


Figure 4. Frequency distribution plots of stem diameter (dbh) and basal area in a bottomland hardwood forest floodplain in central Missouri, USA.

and have high transpiration rates compared to other woody species. Vose et al. (2000) showed through sap flux experiments that cottonwoods transpire as much as 25 cm during the growing season. This holds important implications for floodplain management and possible restoration of BHFs in terms of annual plant water consumption.

Based on tree ring analysis, most tree establishment occurred between 1939 and 1962. One silver maple had an early pith date (center ring) of 1917, and thus represents the oldest individual in the study area. The species established somewhat sequentially, with silver maple establishing between 1939 and 1946, black walnut establishing between 1949 and 1960, and boxelder between 1951 and 1955. Other species were not cored, and the eastern cottonwoods were too hollow for accurate tree ring dating.

Average LAI was 3.11 (SD = 0.69), as calculated from the hemispherical and ceptometer methods (Figure 5), indicating an average of 3.11 canopy layers per unit soil surface area (see Methods). The two LAI estimates were in close agreement (n = 42 each), with an average hemispherical estimate of 3.24 (SD = 0.74) and an average ceptometer estimate of 2.98 (SD = 0.70), thereby increasing our confidence in results. Minimum and maximum LAIs were 1.31 and 4.96 for the hemispherical method and 1.24 and 4.18 for the ceptometer method.

Infiltration

Results of infiltration tests comparing the BHF and Ag sites (n = 42) indicated an average infiltration capacity (maximum steady state infiltration under saturated conditions) of 23 (SD = 21.0) and 38 (SD = 29.0) cm hour^{-1} in the Ag and BHF sites, respectively (Figure 6). Minimum and maximum infiltration values were 0.1 and 69.0 mm hour^{-1} for the Ag site and 3.0 and 126.0 cm hour^{-1} for the BHF site. Infiltration rates of the BHF and Ag sites differed significantly ($p < 0.05$). Maximum infiltration rates measured in the BHF site created a dramatic difference between the two sites (61%) in mean infiltration (Figure 6). Based on field observations, locations of high maximum infiltration rates were associated with locations of large trees (eastern cottonwoods). We therefore assumed that the associated root systems of those larger trees, which also corresponded to higher LAIs (Figure 5), were responsible for greatly increased soil infiltration rates. This conclusion is similar to previous work that showed forested floodplain infiltration rates 2 to 17 times greater around trees relative to bare ground (Bramely et al. 2003). Infiltration is an important benefit in urban floodplains since high infiltration rates of forested floodplains have been shown to normally exceed the highest rainfall intensities, thus preventing infiltration excess overland flow (Krause et al. 2007). On this basis, forested floodplains may not only increase the attenuation of urban stormwater flows and flooding, but also may provide a buffer for surface runoff by providing surface area for stormwater flows to infiltrate prior to reaching the stream.

Soil Characteristics

We extracted soil cores at depths of 0, 15, 30, 50, 75, and 100 cm every 20 m within the 80 x 80 study grid (Figure 2), for a total sample size of 150 soil cores from each floodplain study site. When we averaged soil core results from each depth over the total depth (100 cm), we found that average bulk density, porosity, and VWC were 1.3, 0.5, and 0.33 g cm^{-3}, respectively, in the Ag site and 1.31, 0.51, and 0.37 g cm^{-3} in the BHF site. Those averages equate to a 2% difference in bulk density and porosity, and an 11% difference in VWC between the sites (Table 2). Based on the results of a two-way ANOVA, the sites did not differ significantly in bulk density (n = 150, each site; $p > 0.05$). However, a comparison of the sites at all sampled soil depths (via a Tukey's post hoc multiple comparison) showed that bulk density was significantly lower in the BHF site at the 30-cm soil depth (n = 25, each site; $p = 0.01$). Similarly, porosity (n = 150, all depths, each site) did not differ significantly between sites ($p > 0.05$), but was greater in the BHF at the 30-cm depth (n = 25, each site; $p = 0.01$). ANOVA results indicated that

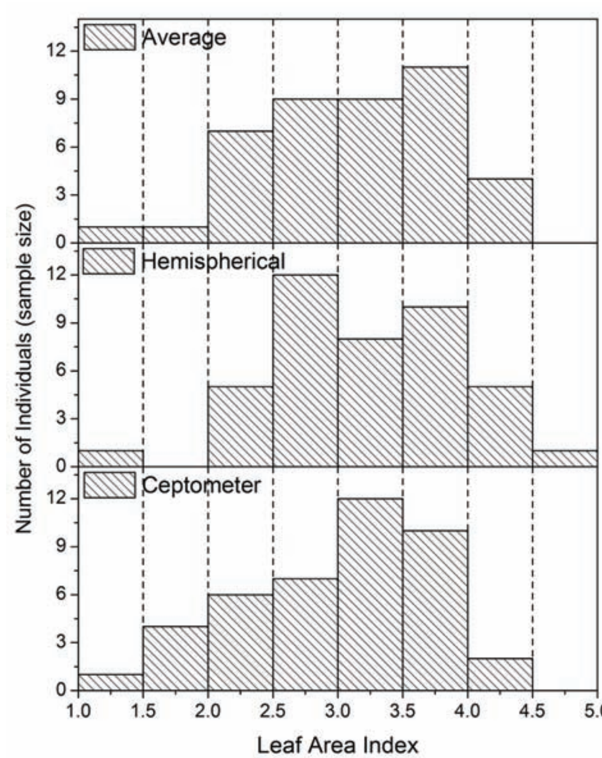


Figure 5. Frequency distribution plots of LAI in a bottomland hardwood forest of Lower Hinkson Creek in Columbia, Missouri, USA.

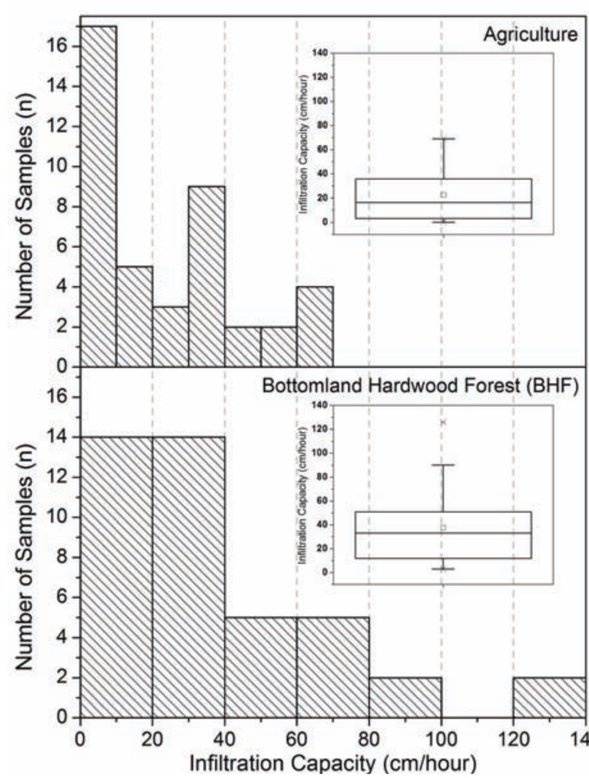


Figure 6. Infiltration capacity frequency distributions and nested box plots (n = 42) comparing bottomland hardwood forest and agricultural floodplain sites in the lower reaches of Hinkson Creek in the city of Columbia, Missouri, USA.

Table 2. Soil characteristics comparisons averaged from five soil depths between bottomland hardwood forest and agricultural floodplain sites in the lower reaches of Hinkson Creek in Columbia, Missouri, USA.

Site	Variable	Maximum	Mean	Minimum	Standard Deviation
Ag	ρbdry	1.71	1.33	1.03	0.08
	Porosity	0.61	0.50	0.36	0.03
	VWC	0.48	0.33	0.13	0.03
BHF	ρbdry	1.60	1.31	0.95	0.08
	Porosity	0.64	0.51	0.40	0.03
	VWC	0.52	0.37	0.10	0.04
% Difference	ρbdry	–6	–2	–8	–5
	Porosity	5	2	12	–5
	VWC	9	11	–21	7

Notes: Ag, agricultural; BHF, bottomland hardwood forest; % Difference, (BHF–Ag/Ag) x 100; ρbdry, dry bulk density (g cm^{-3}); Porosity (%); VWC, volumetric water content (g cm^{-3}).

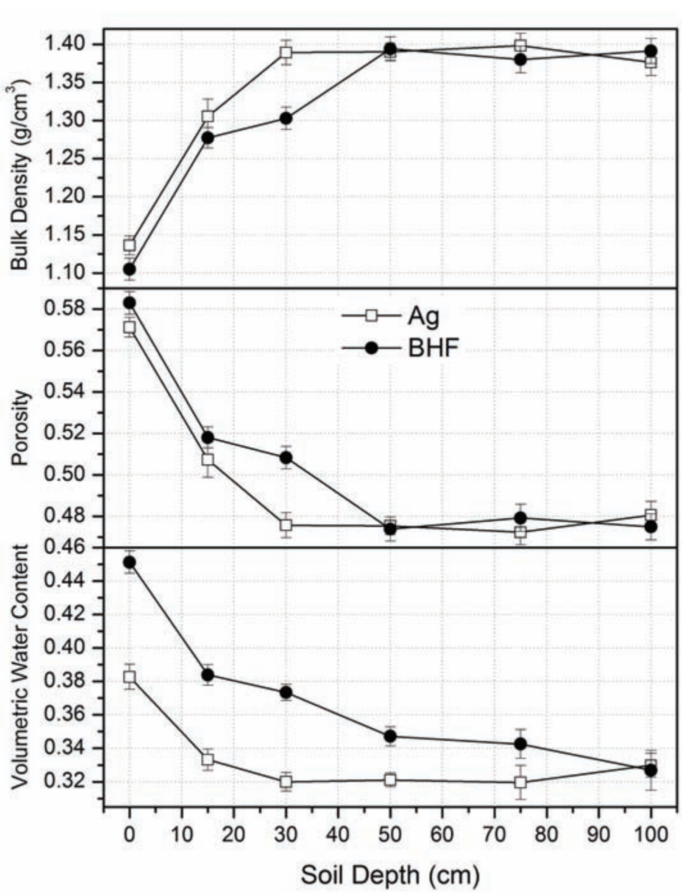


Figure 7. Comparison of soil characteristics (by depth with standard error bars) in an agricultural site and a bottomland hardwood forest site in Columbia, Missouri, USA. Porosity is expressed as a percentage and VWC units are g cm^{-3}.

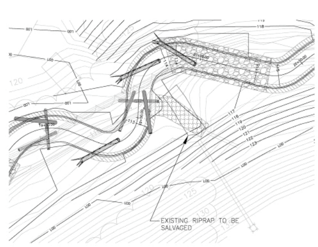
soil VWC (n = 150, all depths, each site) averaged over the 100-cm profile differed significantly ($p < 0.05$) between the sites. However, the post hoc multiple-comparison test showed that specific significant differences were restricted to the 0-, 15-, and 30-cm depths (n = 25, each site; $p < 0.001$; Figure 7).

These results provide a strong argument for a vegetative effect on soil characteristics and, in particular, soil VWC above a soil depth of 50 cm (Figure 7). All soil characteristics diverge from 50 to 0 cm, and the standard error is negligible at depths less than 50 cm. Studies have shown that the majority of plant roots are found in the top 50 cm of soil (Barbour et al. 1999). Given that root systems increase soil infiltration by providing preferential flow paths for water, reducing bulk density and increasing soil VWC (Hillel 2003), these results may not be surprising. However, given the paucity of information pertaining to urban floodplain hydrology and forest relationships, these data are novel and provide support for efforts to restore BHF in urban areas to reduce stormwater runoff and flooding.

One of the most important results of this work pertains to the differences between the BHF and Ag sites in VWC, which equate to a nearly 11% difference in soil water over the 100-cm profile. An 11% difference could be substantial in terms of annual flood control. If one considers only the top 30 cm (as per statistical findings), the average difference between the BHF and Ag sites in VWC is 5%, or 57.6 mm (data not shown). However, tree roots penetrate soils deeper than 30 cm, and the rooting depths of many BHF species surpass 50 cm (Burke and Chambers 2003). On that basis, it is reasonable to assess the absolute differences between study sites in VWC in the entire profile since the transpiration effects of BHF VWC would translate to a reduced VWC at greater depths over the growing season. Previous researchers have noted that willow species can transpire as much as 4 to 6 mm day^{-1} (Hall et al. 1998; Granier et al. 1996; David et al. 1997). Considering the entire soil profile (100 cm), an 11% greater VWC in the BHF site equates to approximately 40 mm of storage difference over a soil depth of 1 m. Further, assuming a conservative value of 4 mm day^{-1} transpiration, a forested urban floodplain could easily consume (i.e., remove from the watershed) more than 720 mm of water per equivalent forested floodplain area, over only a six-month growing period. This could be substantial in urban watersheds like the HCW, where 700 mm is approximately two-thirds the long-term average annual precipitation (980 mm year^{-1}).

The baseline results presented in this article quantify the potential benefit of sustaining, or reestablishing, floodplain forests to enhance water storage capacity, attenuation, and consumptive use, thus reducing flooding and mitigating increased stormwater-related runoff and other effects of urbanization. The many other benefits to reestablishing BHF in urban floodplains include improvements in water quality and aquatic ecosystem health,the creation of inner city parks, and an improvement in human health. Carbon sequestration is an additional potential benefit of reestablishing BHFs. Forests play a significant role in carbon sequestration in aboveground woody biomass accumulation and, to an even greater extent, in forest soils (Cason et al. 2006); in this way, forests account for approximately two-thirds of the terrestrial carbon on the planet, excluding rock and sediment (Sedjo 2001). At least one-third of total forest carbon is contained in wetland and floodplain soils (Trettin and Jurgensen 2003). Thus, not only could the afforestation of converted floodplain lands dramatically improve flood safety and reduce losses, it could also significantly increase soil carbon sequestration.





Conclusions

At the watershed scale, the results presented here hold critically important implications for improvements in the attenuation of urban stormwater runoff and floodplain floodwave as well as consumptive water use. Results indicate that, for every hectare of forest reestablished in a floodplain, more than half of the precipitation falling on that hectare could be removed from the watershed by plant transpiration during the growing season. In that sense, successful restoration of urban forested floodplains may achieve a great deal more per unit area than other current upland stormwater mitigation practices (e.g., detention/retention ponds, rain gardens, or constructed wetlands). Investigations comparing BHF restoration to other contemporary flood mitigation practices are warranted. In 1987, Abernathy and Turner estimated that less than 25% of pre-European development BHF remains. Assuming that a majority of the other 75% of former BHF remains under historic floodplain land-use practices (drainage tiles, ditches, levees, dams, and so on), BHF floodplain restoration may be of critical importance as human populations continue to converge and grow in condensed urban centers.

In urbanizing watersheds such as the HCW, comprehensive management approaches, addressing stormwater runoff and streamflow regimes as well as the pollution load being transported, are imperative. For the HCW and other similar Clean Water Act Section 303(d)–listed North American watersheds, the work presented here is timely given legal mandates to provide quantifiable estimates of total maximum daily loads, improve water quality, reduce stormwater runoff, and decrease flood risk (Cappiella 2010). Given the size of the HCW and the scope of land uses therein, the HCW serves as a model urban watershed for similar studies. In continued work, we will seek to (1) quantify annual BHF transpiration and interception rates and (2) establish multiannual vadose and saturated zone water flux data; in this way, we will quantifiably demonstrate the benefits of reestablishing BHF in the urban floodplains of the American Midwest and elsewhere.

Acknowledgments

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Lawns as a Source of Nutrient Runoff in Urban Environments

John C. Stier[a] and Douglas J. Soldat[b]*

Abstract

Many people believe that lawns and fertilizers contribute substantially to urban runoff. However, data from small-plot and watershed-scale studies indicate that runoff is primarily limited to periods of frozen ground or saturated soils. Lawn runoff research studies have typically found that less than 5% of precipitation in a given year runs off-site. Turf development over time can even overcome the effects of compaction resulting from construction or establishment practices. A small amount of soluble nutrients will always leach from any type of vegetation; this, combined with atmospheric deposition, is readily moved in runoff over the interconnected impervious surfaces found in urban environments. But properly developed and managed lawns can reduce overall runoff volume and nutrient losses. Beneficial practices may include (1) using swales in lawns, particularly near impervious surfaces; (2) avoiding runoff from irrigation; (3) forgoing the application of fertilizer to saturated or frozen sites; and (4) applying fertilizers in recommended amounts and only when turf is actively growing.

Lawns and Urbanization

Urbanization leads to increased runoff as interconnected impervious surfaces, such as rooftops, parking lots, and roads, replace pervious ground cover, such as forests and fields (Shields et al. 2008). The increased runoff results in the pollution of surface waters with sediments, nutrients, and anthropogenic compounds. Increased runoff due to the connectedness of impervious surfaces can also result in the scouring of stream and river banks, causing erosion and adding to pollutant loads entering surface waters (Wang et al. 2001).

After buildings, lawns are the most visible type of ground cover in urban environments. The United States contains nearly 70 million detached single-family homes with an average lawn size of 0.1 to 0.13 ha; this adds up to a total of between 7.1 and 9.3 million ha of ground cover (US Census Bureau 2010; Vinlove and Torla 1995). Lawns and roadsides account for the greatest and second-greatest amounts of turf area, respectively, with additional turf covering parks, corporate grounds, schools, athletic fields, airports, sod farms, and golf courses (Wisconsin Agricultural Statistics Service 2001). Estimates using satellite imagery place the total US turf area at approximately 16.3 million ± 3.9 million ha (Milesi et al. 2005), which is about the size of Wisconsin.

The high visibility of lawns keeps them, and their management, in the public eye. Some of the public believe that the contribution of lawns to urban runoff is similar to that of paved or other impervious surfaces. In Olmsted County, Minnesota, the environmental oversight committee considered an ordinance listing turf as having imperviousness similar to that of concrete (Eric Counselman, Olmsted County Environmental Commission member, pers. comm., October 23, 2009). While excessive irrigation that exceeds the soil's infiltration rate and irrigation deposited directly on sidewalks, driveways, roads, and so on certainly causes runoff, these are cases of human error, and should not be attributed to the turfgrass ecosystem. Nonetheless, the perception of lawns as a significant pollution source has led to proposals to reduce lawn inputs and lawn surface area (Marzluff and Ewing 2001; Robbins and Berkholtz 2003).

Various states and municipalities are taking steps to mitigate total suspended solids (TSS) and nutrients in urban runoff by restricting fertilization (Lehman et al. 2009), in part to comply with the US Clean Water Act. In 2005, Minnesota became the first state to ban most turf applications of phosphorus (P)–containing fertilizers (Rosen and Horgan 2005). Other states—including Michigan, North Carolina, Washington, Virginia, and Wisconsin—have enacted, or are considering, similar bans. In 2010, New Jersey enacted the most restrictive turf fertilization law in the United States, restricting both nitrogen (N) and P applications to turf (Jim Murphy, Professor, Rutgers Univ., pers. comm., January 11, 2011). Although the amount of fertilizer used for lawn care probably varies greatly across states, less than 5% of the fertilizer sold in Wisconsin is used for lawns and gardens, while the rest is used for agriculture (Michael Koran, Fertilizer Regulations, Wisconsin Department of Agriculture, Trade, and Consumer Protection pers. comm., 2004). In fact, lawns may actually

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be able to play a role in the reduction of urban runoff if they are properly sized, placed, and managed. This paper presents a review of the literature on lawns as a source of nutrient runoff to better inform the public understanding of lawns as both a source and a mitigator of urban runoff.

The Use of Turfgrasses as Urban Vegetation

Turfgrasses are unique plant species that evolved under grazing pressure to withstand continuous defoliation and traffic while maintaining a contiguous community that can ensure coverage of bare soil (Casler 2006). Only a couple dozen species of plants, including two broadleaf species suitable only for warm climates, have the ability to provide such cover. Turfgrasses provide an ideal vegetative cover in much of the urban environment because they require only moderate care (e.g., weekly mowing and occasional fertilization) and form a dense cover over soil at low growing heights, even under traffic. The benefits of properly managed turfgrasses—including increased property values, increased recreational opportunities, and decreased crime—have been well summarized (Beard and Green 1994; Kuo and Sullivan 2001). From a water quality standpoint, one of the most important functions of lawns may be their ability to mitigate issues associated with urban runoff, provided that they are properly designed, sited, installed, and managed.

In urban settings, atmospheric deposition can be a significant source of nutrients that readily move in runoff when deposited on impervious surfaces. A three-year study of the Baltimore, Maryland, area showed an average atmospheric deposition of 11.2 kg N ha^{-1} compared to 14.4 kg N ha^{-1} from fertilizers as potential inputs to the watershed (Groffman et al. 2004). While urbanized areas had greater N output than forested sites with the same amount of atmospheric N deposition, the authors concluded that impervious surfaces were largely responsible for the difference in N runoff. Mean annual atmospheric P deposition is approximately 0.4 kg ha^{-1} (UN Environment Programme 1999). A conventionally recommended lawn fertilization program applying 146 kg N ha^{-1} year^{-1} using a 27:1.3 (N:P) fertilizer would supply 7 kg P ha^{-1} year^{-1}. Soldat and Petrovic (2008) found a range of 0.0 to 19.1 kg P ha^{-1} year^{-1} reported in turf field plot research projects, with typical losses of approximately 0.5 kg P ha^{-1} year^{-1} from established turf. These values compare to annual losses of approximately 0.2 kg P ha^{-1} from native prairie, 1.9 kg P ha^{-1} from conventionally tilled agricultural systems, and more than 13 kg P ha^{-1} from construction sites (Daniel et al. 1979; Sharpley 1995).

Effect of Vegetative Cover on Runoff

Precipitation on bare soil results in exorbitant amounts of runoff laden with sediments, dissolved nutrients, and particulate nutrients in both inorganic and organic forms. Runoff and nutrient losses dissipate as vegetative cover and other nonplant (e.g., gravel) ground cover is established; and, in turfgrasses, the newly seeded and seedling phases are most prone to runoff and nutrient loss (Easton and Petrovic 2004). Sodding costs substantially more up front than seeding but quickly and effectively reduces runoff and erosion (Krenitsky et al. 1998). Vegetative cover, mulch, rock, and other covers intercept precipitation, preventing it from disturbing the soil and impeding surface runoff with a concomitant reduction of sediment and nutrient transport (Gilbert and Clausen 2006; Gross et al. 1991; Linde et al. 1995). In many cases, the denser the turf, the less runoff occurs because the contiguousness of the turf plants creates a "tortuous pathway" that slows the water and allows greater infiltration (Linde et al. 1995; Kussow 2008). Civil engineers use roughness coefficients to determine the potential of surfaces to contribute to overland flow; higher coefficient values correspond to less runoff. In a simulated rainfall experiment, pavement had a low roughness value of approximately 0.01, short grass prairie was 0.15, and bermudagrass (*Cynodon* spp.) and bluegrass (*Poa* spp.) sod were approximately 0.4 (Engman 1986).

Sufficient fertilization is important for maintaining turf cover and reducing runoff. In a cool-season lawn mixture, runoff was reduced three-fold when infiltration increased as a result of greater shoot density in response to fertilization (Easton and Petrovic 2004). Kussow (2008) showed that applying four applications of N- and P-containing fertilizer to a Kentucky bluegrass (*Poa pratensis* L.) turf, with each application providing 49 kg N ha^{-1}, reduced runoff depth by about 25% compared to turf left unfertilized over a two-year period. Fertilized turf had P losses averaging 0.34 kg P ha^{-1} compared to 0.54 kg P ha^{-1} ($P \leq 0.05$) from nonfertilized turf, whereas no difference in N runoff was noted. Bierman et al. (2010) found similar results in Minnesota over a three-year period.

The P lost from dense vegetation is primarily soluble P, much of which leaches from the vegetation, rather than particulate P, which is derived from the soil. In general, ecosystems in which P is lost only as soluble P leaching from vegetation tend to have significantly lower P loss than those in which particulate P also results from substantial soil loss because of insufficient ground cover. Mowing at an appropriate frequency and to an appropriate height helps turfgrasses maintain maximum density. While clippings that fall back into the turf allow for nutrient recycling and reduce fertilizer needs, clippings do not appear to contribute to P runoff from turf (Bierman et al. 2010). Vegetation, turf or nonturf, that overhangs impervious surfaces may actually contribute more to nutrient losses in runoff than shorter vegetation as nutrients are leached from the leaves, particularly following freezing or drying (Bechmann et al. 2005; Kussow 2008).

Compared with turf, nonturf vegetation, such as native prairie plantings, can lose significantly more nutrients from their aboveground biomass during the winter because the prairie plants senesce during the autumn, and precipitation or snowmelt results in runoff over frozen ground (Steinke et al. 2007). Turfgrasses in the northern portions of the country are C_3 plants (cool-season grasses) that often do not die back in winter, thus retaining nutrients in their foliage. In central and southern portions of the United States, C_4 turfgrasses (warm-season grasses) are often used and may senesce with the onset of cool autumn temperatures. However, compared with unmowed native prairie or other plants, the short stature of mowed turfgrasses results in relatively little aboveground biomass and can lead to less overall nutrient losses from the foliage during winter (Steinke et al. 2007).

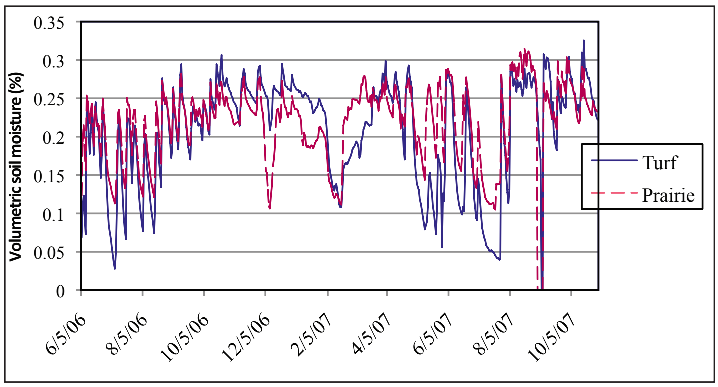

Figure 1. Soil moisture over time at a 6.4-cm depth under mowed Kentucky bluegrass turf or mixed prairie plants (primarily forbs and sedges) in silt loam soil in Madison, Wisconsin. Turf had less soil moisture in the spring and summer periods as a result of greater growth and ET rates; the reduced soil moisture provides space for precipitation to infiltrate and inhibit runoff during rain storms.

Lawns, Compaction, and Impervious Surfaces

Frozen and saturated soils negate the ability of lawns, or other vegetation on smooth ground, to stop runoff. In many cases, most or all annual runoff can occur during frozen soil conditions (Kussow 2008; Steinke et al. 2007). In nonfrozen conditions, runoff occurs when the precipitation rate exceeds the soil's infiltration rate, or when the soil becomes saturated.

Turfgrasses have an evapotranspiration (ET) rate that is similar to or higher than that of many other potential urban ground covers (Ebdon et al. 1999). Using replicated plots of forb-dominated prairie and mowed Kentucky bluegrass turf in a randomized block design, we found that the turf had less soil moisture than the prairie at a 6.4-cm depth during early spring and summer, with spring differences due to a resumption of Kentucky bluegrass growth that is earlier than that of prairie plants (Figure 1; Stier, unpublished data). Thus, the higher ET rates of the turf result in an upper layer of drier soil that allows water infiltration more effectively than would a persistently moist upper layer of soil.

Compaction of turf soils may contribute to runoff, and the extent to which it does so depends on the use of heavy construction equipment during development. A study of 15 lawns in central Pennsylvania assessed the infiltration rates of clay, silt, and loam soils (Hamilton and Waddington 1999). Based on soil characteristics alone, one would expect infiltration rates to be affected by soil type in the following order, from greatest to least infiltration: sand > loam > silt > clay. However, Hamilton and Waddington (1999) found that the soil type of lawns did not correlate with infiltration. Instead, they concluded that the soil's condition, structure, and history are likely to affect lawn infiltration rates, and that these factors are largely a function of construction practices. Preplanting tillage, as recommended for lawn establishment, and core aeration of lawns that have compacted soils can help improve infiltration rates (Partsch et al. 1993; Stier 2000). Over time, pore formation from the development of turfgrass roots, freezing and thawing, and benthic activity (e.g., from earthworms) will improve infiltration (Easton et al. 2005). Thus, in practice, when excessive compaction does occur during construction, properly tilling and establishing turf will negate compaction effects. Most states have extension services that provide guidance for establishing lawns in northern and southern climates (Stier 2000; Waltz 2010).

In some cases, the role played by compaction may be less important than might be perceived. Kussow (2008) simulated home construction site practices by intentionally compacting a silt loam soil with a 5% slope using a vibrating roller. He placed an additional 7.5 cm of the silt loam on top of the compacted area and either mixed it by tilling or left it in a layer. He chisel-plowed another section of the compacted area of each plot prior to the addition of topsoil and then seeded the entire area with Kentucky bluegrass. By year two of the study, the runoff amounts were similar for noncompacted, compacted, and compacted + chisel-plowed treatments, with an annual runoff depth of 30–39 mm from an annual 641 mm of precipitation. Bierman et al. (2010) used a bulldozer to level runoff plots to a 5% slope and then sodded them with Kentucky bluegrass. Over a three-year period, runoff averaged less than 1% of annual precipitation when the ground was not frozen.

The lack of connectedness between impervious and pervious surfaces in urban environments can prevent runoff and snowmelt from infiltrating into the soil. Urban areas are typically designed to channel storm and meltwater quickly away; this may lead to flushes of water, sediments, and pollutants (including nutrients) into surface waters. Properly placed and maintained lawn areas can alleviate runoff and nutrient losses from impervious surfaces (Mueller and Thompson 2009).

While numerous studies have shown the effectiveness of grassed buffers for reducing overland flow and pollutants from crop fields, studies evaluating the effectiveness of grass buffers in urban environments are almost nonexistent, other than perhaps studies of roadside swales. Steinke et al. (2007) studied the effects of nascent prairie and turfgrass buffer strips on runoff from concreted slopes. They developed concrete-to-vegetative buffer ratios of 1:1, 1:2, and 1:4 along a 5% slope on a silt loam soil near Madison, Wisconsin. The vast majority of runoff occurred when soils were frozen, at which times runoff from turfgrass and prairie buffers was similar. During non-frozen conditions, they measured less runoff from the managed turf areas than from than the prairie plantings ($p \leq 0.10$) the year following establishment. A vegetative buffer twice the size of the concrete area reduced annual runoff by more than 60% compared to the 1:1 concrete-to-buffer treatment, though even the 1:1 buffer allowed less than 1.5% of the precipitation to run off during nonfrozen conditions. Mueller and Thompson (2009) conducted 52 stormwater runoff tests on six lawns in Madison, Wisconsin, to determine the ability of lawns to infiltrate rooftop runoff. Using a model to estimate annual lawn runoff as a function of rooftop-to-lawn size ratios, they concluded that lawns could be useful as a stormwater management practice.

Rain Gardens and Lawns

Natural areas typically have a texture that is rough enough to help retain precipitation and reduce runoff. In urban environments, rain gardens have been proposed as a way to trap runoff water from impervious surfaces, such as rooftops and parking lots. Rain gardens are flat-bottomed depressions planted with trees, shrubs, or native vegetation and designed to trap and infiltrate runoff from impervious surfaces (Dietz and Clausen 2005; Wisconsin Department of Natural Resources [WDNR] 2003). While some rain gardens are highly engineered, containing sand-based root zones and drain tiles for high infiltration and exfiltration flow, they can also be carved from existing soil and surrounded by a berm, creating a miniature retention basin. Rain gardens are not usually recommended for clay soils, however, which are prevalent in many US urban areas.

Although rain gardens add texture to a landscape, and the flowering plants can add beauty during the summer, the most effective part of the rain garden for runoff control is the berm. In a two-year study, a student in our laboratory compared bermed and nonbermed rain gardens and lawns for runoff reduction from rooftops (Schneider 2007). A randomized block design with four replications was used to test Kentucky bluegrass turf maintained as lawn, rain gardens with berms, turf with a berm similar to that of a rain garden, and rain gardens without a berm (with the same surface characteristics as a lawn and lacking a depression in the ground). Berms were constructed from soil removed while excavating a 15 cm–deep basin following recommended construction methods (WDNR 2003). Rain gardens were sized to prevent 100% of the potential annual runoff from the rooftops. Given the approximately 5% ground slope and silt loam soil type, a recommended ratio of rooftop-to-rain garden of 2.8:1 was used (WDNR 2003). Transplants of species were used per a recommended design (WDNR 2003). Rooftops (7.6 m long x 2.4 m wide with a 12% slope) equipped with gutters and downspouts channeled water into the plot areas. Runoff was collected following all rainfall events using weirs and collection vessels at the downslope edge of the plots and analyzed the runoff for volume, TSS, N, and P.

In no case did the total amount of runoff exceed 5% of the annual precipitation, showing that pervious surfaces are very good at reducing potential runoff (Table 1). Even the nonbermed lawn area reduced runoff equivalent to the bermed rain garden over the two-year period. The nonbermed rain gardens, using recommended prairie-type plants, allowed significantly more runoff, sediment, P, and, in the first year, nitrate-N than the other three treatments, presumably because of low plant density and exposed soil (additional data in Schneider 2007). Adding mulch around the plants might have reduced runoff, but the study also reinforces the idea that the presence of thatch from turfgrasses

Table 1. Runoff, expressed as amounts and as percentages of precipitation, from lawn-type turf or rain garden plantings receiving rooftop runoff over a 24-month period in Madison, Wisconsin.

Situation	November 2005–October 2006		November 2006–October 2007	
	mm	% Annual precipitation	mm	% Annual precipitation
Turf, no berm	20.4	2.4	9.0	0.9
Turf, with berm	8.9	1.0	10.4	1.0
Rain garden, no berm	42.8	5.0	17.9	1.8
Rain garden, with berm	8.2	1.0	9.0	0.9
$LSD_{0.05}$	12.3		4.0	

Note: LSD, or least significant difference, is used to compare statistical differences of runoff amounts among treatments. One can add or subtract the LSD value to the runoff amount (in mm) from any treatment and compare the result to the value of another treatment. For example, in the first year, the turf with no berm (ordinary lawn) had significantly less runoff than the rain garden vegetation with no berm (20.4 + 12.3 = 32.7; 32.7 < 42.8) and a similar amount of runoff compared to the rain garden with a berm (20.4 – 12.3 = 8.1; 8.1 ≈ 8.2).

provides an ideal cover over soil that effectively reduces runoff.

Schneider (2007) concluded that the depressions and berms, not the type of vegetation, were the effective components of rain gardens. Moreover, bermed plots caused about 1% of the precipitation to run off into collection weirs because the weirs were placed adjacent to the downslope edge of the berms; installing such berms at the edges of impervious surfaces such as sidewalks or roads could actually increase runoff into storm sewers. Runoff from bermed plots would have been reduced if a sufficient buffer area, or swale, had been installed between the berm and the collection weir. In practice, berms or swales placed at some interval in lawns that slope toward impervious surfaces would reduce even the relatively small amount of runoff that occurs from turf or other vegetated areas.

Fertilizers as a Source of Nutrients in Runoff

Turfgrass areas differ greatly from agricultural areas in the manner in which N and P are applied to the vegetation and in the potential for nutrient losses. For example, P losses in agriculture are often highly correlated with soil test P and the amount of sediment loss; however, sediment losses from turfgrass areas are typically very low (Soldat and Petrovic 2008) and are unrelated to soil P levels unless P levels are unusually elevated (Soldat et al. 2009). The small but consistent level of soluble P in runoff from turf probably originates from the plant tissue itself (Soldat et al. 2009).

Although P fertilizer bans enacted in many parts of the country are expected to reduce P runoff, the impact from the bans may not be as large as anticipated. In runoff from nonfrozen ground, Bierman et al. (2010) reported a significantly greater reactive P loss (0.10 kg ha^{-1}) in only the first year of a three-year study from turf fertilized with a high P:N fertilizer (1:2), typical of garden fertilizers and natural or organic fertilizers, compared with a lower P:N fertilizer (1:27), a fertilizer without P (containing N and potassium only), or no-fertilizer treatments (0.03–0.05 kg P ha^{-1}). In the second year and third years, the nonfertilized turf exhibited greater reactive P losses than did any of the fertilizer treatments (year two: 0.11 kg P ha^{-1} for nonfertilized turf vs. 0.04–0.05 kg P ha^{-1} for fertilizer treatments, $p < 0.05$; year three: 0.03 kg P ha^{-1} for nonfertilized turf vs. 0.01–0.02 kg P ha^{-1} for fertilizer treatments; 0.05). The authors attributed the increased P runoff loads to the decreased turf density associated with the nonfertilized treatment, which exhibited higher runoff volumes than the fertilized plots. Kussow (2008) and Easton and Petrovic (2004) similarly found that increased runoff volumes from the less dense turf resulting from nonfertilization led to greater P losses. The use of native plants in lieu of mowed turf may not noticeably reduce P in runoff either, as Steinke et al. (2007) showed that P losses of fertilized turf and nonfertilized prairie plantings were similar, with the majority occurring during frozen conditions. Steinke et al. (2007) examined a relatively young (less than five-year-old) site; further study is needed to compare mature prairie vegetation with turf to develop best management practices for the control of urban runoff.

A recent five-month watershed-scale study found reduced P export from an urban watershed in which a ban on lawn applications of manufactured fertilizers containing P had been enacted compared to a watershed without the ban (Lehman et al. 2009). One would expect that a ban on the use of manufactured fertilizers containing P would reduce dissolved P because the P forms in such turf fertilizers are highly soluble. Instead, however, Lehman et al. (2009) found that "the main effect has been [a] reduction in the particulate P load of the river," not a reduction in dissolved P. Why a P ban would result in a reduction in sediment transport without affecting dissolved P remains an open question. The researchers did not quantify differences in other activities (including construction), and they also point out that, in the watershed with the P ban, public education efforts encouraged citizens to reduce P in other ways, such as through attention to vegetated buffer strips along streams and the reduction of yard waste discharges into storm drains. More research is required to determine the most effective policies and practices for reducing P export from urban areas.

N dynamics in turfgrass systems are also substantially different from those of agricultural systems. In agricultural areas, N leaching is often the result of large applications of soluble N in fall or spring when plant cover and N uptake potential is low. Conversely, turfgrass is a permanent ground cover that has the ability to use N earlier in the spring and later in the fall than forests or agricultural crops (Pickett et al. 2008). Applications of N on turfgrass are usually no more than 45 kg ha^{-1}, often less, and contain some amount of slowly available N, which is not widely used in conventional agriculture. Bowman et al. (1989) reported that cool-season turfgrass was able to absorb 70%–80% of a 45 kg ha^{-1} application of soluble N within 24 hours, and nearly all of it within a 48-hour period. In a Florida study, Erickson et al. (2001) explored the effect of alternative vegetation to manage N export and runoff compared to turfgrass by comparing runoff from a mowed, irrigated, and fertilized St. Augustine grass lawn (*Stenotaphrum secundatum* [Walt.] Kunze) to runoff from a landscape type (containing shrubs, trees, and mulch) recommended for reducing N pollution. The plots were planted on a sandy soil with a 10% slope. Precipitation caused only one runoff event during the study, and both types of plantings had similar concentrations of inorganic N.

Best Management Practices for Turfgrass Fertilization

P is often required to maximize the establishment of turfgrass (Hamel and Heckman 2006). But once turfgrass is established, soil test P levels required to sustain adequate growth are lower than those required for many agricultural crops (Petrovic et al. 2005). Therefore, soil test results should guide the application of P fertilizer to turf, and recommendations for P should be different for established versus newly seeded or sodded turf.

In contrast to P, no soil test can accurately assess requirements for N, which readily converts among various forms and, unlike other nutrients, can convert to gaseous forms (e.g., N_2 and NH_4^+). Therefore, N fertilization should be based on appropriate research-based guidelines that are often highly specific and available from state universities or extension services. For example, recommended N rates will depend on factors like turf species, climate, microclimate (sun vs. shade), level of traffic, and clipping management (bagged vs. mulched). In general, applications should be made only when the turf is actively growing. Most commercially available fertilizers contain 0%–30% slow-release N, while lawn care companies use fertilizers containing anywhere from 0% to 100% slow-release N, depending on the company and situation. The most common types of slow-release sources of N for lawn fertilizers (which must be listed on the packaging) include sulfur-coated urea, polymer-coated urea, methylene ureas, and a generic category listed as water-insoluble N. Research has shown that both N and P nutrient losses can be mitigated by lightly "watering-in" the application (Shuman 2004). Also, avoiding the fertilization of saturated soils is a no-cost, no-effort solution to reducing potential fertilizer runoff and leaching from lawns (Morton et al. 1988; Shuman 2004).

Impacts of Homeowner Lawn Management Practices

Based on sales data, Scotts Miracle Gro estimates that approximately 50% of homeowners in the United States fertilize their lawns (Augustin 2007). Of the 50% who fertilize, the average number of annual fertilizer applications (~45 kg N ha^{-1} per application) was estimated to be 1.8, which includes an estimated 10 million homes treated by professional lawn care companies. Law et al. (2004) independently obtained a very similar estimate in Baltimore County, Maryland. These data indicate that the average homeowner who fertilizes his or her lawn is doing so only 60% as frequently as recommended by most university extension services, which typically recommend three applications per year (with wide variations, as discussed above).

A large-scale, urban watershed study of Baltimore, Maryland, concluded that lawns are useful for retaining nutrients in urban ecosystems (Groffman et al. 2004; Pickett et al. 2008). Conservation subdivisions are designed, among other purposes, to reduce stormwater runoff by ensuring sufficient vegetative cover around buildings (Arendt 2004). Baker et al. (2008) suggested that (1) a very small group of homeowners may be disproportionately skewing runoff and nutrient loading events into urban environments and (2) targeting those homeowners would more effectively reduce nutrient runoff than would general, large-scale efforts to prevent fertilization or encourage lawn replacement. WDNR applied such a philosophy to its technical standards for turf fertilization, stating, for example, that primarily water-soluble N sources should be used on slopes and should be lightly watered-in because solid, nonwater-soluble fertilizers could have a tendency to move as particulates from slopes (WDNR 2006).

Conclusion

Runoff from lawns is typically limited to 5% or less of precipitation. The greatest amount of runoff in northern climates typically occurs during winter when the ground is frozen. At other times of the year, and in nonfreezing climates, runoff occurs when soils become saturated or when sprinkler systems overspray and leak onto impervious surfaces. Lawns with dense turf cover release relatively little TSS. Some fertilization, primarily N, is usually needed to maintain sufficient turf density, which is important to minimize runoff volume and nutrient losses. Research has indicated that fertilizer use per se will not contribute significantly to nutrient losses if applied based on agronomic needs and to actively growing turf with nonsaturated soils. Small amounts of nutrients leach from plant tissues—even nonfertilized, nonturf vegetation—particularly when vegetation is senescent. Nutrient runoff loads tend to be directly related to runoff volume, which can be mitigated by maintaining dense turf and possibly by incorporating swales between vegetated sites and paved areas that concentrate and funnel runoff to storm sewers or surface waters. Based on data and the desirability to have turfgrasses as vegetative ground cover in urban areas for recreation and other activities, the development of practices and regulations that promote the best use of lawns to reduce urban runoff will be beneficial.

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Regional Effects of Land Use Change on Water Supply in the Potomac River Basin

Water system managers face common difficulties in maintaining and providing a safe and abundant water supply for a growing population, including uncertainty about the effects of climate change and continued land use changes brought about by an ever-expanding population. In the Washington, DC, Metropolitan Area (WMA) water supply issues are compounded by the operational logistics of dividing the shared water supply of the Potomac River between Virginia, Maryland, and the District of Columbia. The Interstate Commission on the Potomac River Basin (ICPRB) has been tasked with the coordinated management of WMA's shared water supplies and directing research to improve water supply effectiveness. Like many regional water management and supply organizations, the ICPRB must incorporate the effects of land use change simulations into its water supply forecasts. Virginia Tech's Department of Civil and Environmental Engineering is conducting local- and regional-scale studies to augment this research effort; this vignette describes the results of a regional-scale study in the Potomac River basin.

The Potomac River basin encompasses 38,000 km^2 across four states and is the primary source of water for the WMA. The main stem of the Potomac River remains relatively unregulated, with the largest reservoirs located in the headwaters, approximately 300 km upstream of Washington, DC. Land use patterns over the past two decades in the basin show a steady decline in agriculture and a steady increase in developed land throughout the watershed (Table 1). Agricultural land is being converted to both forest and urban land covers. The most intensely urbanized portions of the watershed are in the furthest downstream reaches, close to Washington, DC, and captured by the Little Falls US Geological Survey (USGS) gauge. The rate of urbanization was highest between 1985 and 1997 (0.6% increase), but continued to increase at a lower rate between 1997 and 2005 (0.2%).

Virginia Tech researchers analyzed the effect of land use change on flows in the Potomac River at four stream gauge locations in the river, beginning in the headwaters at Steyer, MD (USGS 01595000), continuing downstream to Paw Paw, WV (USGS 0161000), and Point of Rocks, MD (USGS 01638500), and ending immediately upstream of the Washington, DC, intakes at Little Falls, DC (USGS 01646500). Key findings from repeated streamflow simulations of the historical 1985–2005 meteorological record—using 1985, 1997, and 2005 land use data within the Chesapeake Bay Program Phase 5.3 Hydrological Simulation Program–Fortran Watershed Model—include the following:

- Land use change between 1985 and 1997 is responsible for a 0.1%–1.1% decrease in low flows, quantified here by the $30Q_{20}$ (Table 2). This suggests that land use change could produce more severe droughts. However, by 2005, this effect became smaller, probably as a result of reforestation in the western portion of the basin.
- Storm peaks, quantified here by the ten-year peak flow, decreased very slightly (0.04%–1.90%) because of land use change.

Table 1. Land use change in the Potomac River watershed from 1985 to 2005 at four USGS gauges.

Gauge Location	1985 Land Use (%)			% Change in Land Use (1997 – 1985)			% Change in Land Use (2005 – 1985)		
	Forest	Agric.	Devel.	Forest	Agric.	Devel.	Forest	Agric.	Devel.
Steyer, MD	81.5	14.2	0.7	0.9	–0.9	0.0	–1.7	1.6	0.0
Paw Paw, WV	78.7	18.3	1.2	0.7	–0.8	0.1	0.9	–1.0	0.1
Point of Rocks, MD	65.8	29.1	3.9	1.4	–1.6	0.2	1.5	–1.8	0.3
Little Falls, DC	61.9	31.7	5.2	1.4	–2.0	0.6	2.1	–2.9	0.8

Source: Land use model input data available from Chesapeake Community Modeling Program. "Chesapeake Bay Watershed Phase 5.3 Model." http://ches.communitymodeling.org/models/CBPhase5/datalibrary/model-input.php.

Table 2. Simulated hydrologic change corresponding to land use change alone between 1985 and 2005 at four USGS gauges.

Gauge Location	Total Volume		Low Flow ($30Q_{20}$)		Peak Flow (Q_{10})	
	% Change (1997 – 1985)	% Change (2005 – 1985)	% Change (1997 – 1985)	% Change (2005 – 1985)	% Change (1997 – 1985)	% Change (2005 – 1985)
Steyer, MD	–0.1	0.0	–1.1	1.7	0.0	0.0
Paw Paw, WV	–0.1	–0.1	–0.5	–0.3	–0.1	–0.1
Point of Rocks, MD	–0.1	0.3	–1.1	–0.7	–0.1	–1.9
Little Falls, DC	–0.1	–0.1	–0.1	2.9	–0.2	–0.2

- Total storm volume decreased slightly with changing land use at nearly all stations.
- Because land use change is integrated over the entire Potomac River basin, its effect on flows is not spatially uniform. Reforestation in the west is competing with increasing imperviousness in the eastern portion of the basin, resulting in modest decreases in both low flows and peak flows. However, simulations showed increased peaks and flashiness in smaller-scale areas of increased development.

The findings presented here show the hydrologic effects of land use change in a large watershed subject to a growth pattern typical of many US cities: expanding imperviousness in urban and suburban impervious areas followed by reconversion of agricultural land to untended forests in more distant rural areas. At the regional scale, these competing effects resulted in more severe low flows, while also decreasing storm peaks and total runoff volume. One would expect more severe effects in smaller watersheds, which are subject to more rapid land use change. Studies such as this are vital for all stages of watershed planning and management in estimating the impact of past development on water resources and forecasting these effects into the future.

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For More Information

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Contributors

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The Curve Number Method in Watershed Management and Watershed Health

Observation and science have demonstrated for many years that land cover and land use impact the runoff generated from a watershed. However, not until 1954 was there a method by which to assess the impacts and inform watershed management practices. The US Department of Agriculture (USDA) Soil Conservation Service (SCS, now the Natural Resources Conservation Service [NRCS]) initiated PL 566, the Small Watershed and Flood Control Act of 1954, which supported upland management and engineering structures for small (<101,171 ha [<250,000-acre]) watersheds. Aimed at upstream flood control and concurrent agricultural conservation, the planning, design, and administration of this act required the routine estimation of runoff depth for a variety of land, soil, and climate conditions. With no suitable existing methods available, SCS developed the curve number (CN) method to meet this agency need. The agency's ability to accomplish this task was limited by available data and by the pencil-and-paper-and-slide-rule techniques of the time.

Today, the CN method continues to be widely used and has been incorporated into simulation models—most notably TR-55—and applied to urban hydrology and stormwater management. The success of this method has enabled watershed and stormwater practitioners to better understand the impacts of development on watershed health and reduce them as needed with best management practices or environmental site design practices. Although ideal and soil-based in concept, the CN method is imperfect in practice: more than 50 years of experience and subsequent comparisons with extensive rainfall–runoff data have generated a series of sobering findings, surprises, cautions, and numerous suggestions for professional users.

What is the CN Method?

The CN method is a simple, empirical equation that provides expected event runoff volume (depth) from event rainfall depth. It does not provide runoff rates and does not require data on rainfall duration or pattern—only depth. Central to its use is the CN coefficient, which is selected on the basis of soils, land cover, and land use. The runoff equation is $Q = (P - 0.2S)^2/(P + 0.8S)$, for $P > 0.2S$, and $Q = 0$ otherwise. And to solve for S, one applies the equation, $\text{CN} = 1{,}000/(10 + S)$, where P (rainfall) and Q (event runoff) and S are in depth units (inches in the English system), and CN is dimensionless. The parameter S is defined as the hypothetical maximum possible difference between P and Q, roughly understood as the potential water retention of the upland drainage area. The parameter $0.2S$ is the *initial abstraction* (*Ia*), or the rainfall required before the initiation of runoff. Land conditions—and the hydrologic response characteristics—are shown by the choice of the CN. Tables in the SCS' *National Engineering Handbook* provide CNs for a variety of land uses according to four different soil classifications. Naturally, land cover is a major issue, but only within the confines of a given soil type.

Experience and Findings

The CN method gave identity, hypotheses, and vocabulary to the processes and concepts of watershed-based runoff. The term *curve number* itself is used as a general description of hydrologic-based land condition and seems well suited as a general descriptor of watershed health. While intended only for internal USDA needs, the method that SCS developed so completely filled a waiting technical niche that it was accepted in much wider settings. Today, it is applied beyond its mere rain-fed agricultural origins and is used and modified internationally. Particularly after the publication of TR-55 in 1975, the CN method has found major application in urban hydrology, stormwater management design, and the analysis of developed watersheds, with natural extension to water quality planning and regulation. Despite its successful applications, watershed and stormwater practitioners must remember the limitations of the CN method to ensure that the integrity of the method and its application are upheld.

- Sensitivity analysis shows that the runoff calculations are more sensitive to the choice of CNs from published tables than to the rainfall depth used. Handbook CN tables are *estimates* given by the author(s) of the tables that are perhaps accepted by approving jurisdictions. However, very few such table entries have been verified by monitoring or other ground-truth data.

- CNs supplied in handbook tables are most reliable in urban situations and in some rain-fed agricultural situations—that is, in areas of high CNs. However, several published comparisons of CNs determined from local data with those from handbook tables show a lack of good universal concordance between the two.

- The best source of valid CNs is through the analysis of local rainfall–runoff data. Some guidance for this is provided in the sources below. Most such analyses show an unexpected secondary drift of CN with the event *P*, approaching a stable value at higher rainfalls. In general, the data show that small storms have runoff volumes consistent with high CNs.

- CNs for forested lands are especially suspect. The problems with these CNs result from misperceptions regarding the role of forests, combined with a set of runoff-controlling processes that differ from those for the agricultural lands and covers on which the method was founded. In particular, the factors controlling runoff in most forest conditions include the presence of multiple, continuous levels of cover, heavy vegetation and litter, absorbent soils with underlying layers, and significant roles for flowing channels as source areas. CN tables in use typically have token entries for "woods," but no entries for commercial forests or for silvicultural treatments analogous to agronomic practices.

- From a general hydrology standpoint, the CN equation is not universally valid. Although not common, distinct exceptions to the CN response pattern are not rare either, as some watersheds do not respond as predicted by the CN equation. Often, but not always, such watersheds are forested.

- Both real CNs and those shown in tables rely heavily on soil properties. NRCS provides authoritative classifications of soil series into hydrologic soil groups, but these classifications are disturbingly inconsistent, especially in the B and C groups.

- Most of the early original documentation and data have been lost, and this method received essentially no technical review in the professional or scientific literature. Its widespread acceptance in spite of the lack of review is based on the authority of NRCS.

- Researchers have found that $0.05S$ approximates the initial abstraction better than the original initial abstraction ratio of $Ia = 0.2S$. However, one should not apply this new value without changing the traditional CN tables, which are based on 0.2.

- The CN method is applied in three different modes: (1) As a runoff calculation for a rainstorm of the same return period (*not* for specific storms). This is its most successful application, and the one most appropriate to the existing CN tables. (2) As a runoff equation with variation attributed to prior moisture and other sources of variation, including error. (3) As a time-based process for infiltration in hydrograph models, or for soil moisture storage in daily time-step models. The CNs for these three different applications are not necessarily congruent: what works best for one application may not be best for another.

Potential for Greater Application

Despite the cautions listed above, the CN method is essentially the only tool of its kind that easily integrates soils, land cover, and practices to describe a watershed's hydrologic response. It is thus well grooved into engineering, design, and impact hydrology. However, the method seems to have a substantial unfulfilled potential for application in land management planning for hydrologic accountability in nonurban venues. Data analysis has shown that some long-established land uses presumed to be benign, such as grazing, have surprisingly strong impacts, even in humid zones. For example, several studies have found meadows (ungrazed) with CNs about 15 units lower than pastures (grazed).

Conclusion

If upland hydrologic responsibility for downstream impacts is an issue, the CN method may be an ideal off-the-shelf tool to appraise it. Hydrologic response is a key element of watershed health. In this respect, a "healthy watershed" would have the lowest possible CN. A lower CN means lower volumes of runoff from a given rainstorm and higher levels of infiltration, interception, evapotranspiration, and plant growth. These characteristics promote a storm runoff regime that creates less stress on downstream banks and channels while improving upland habitat and biological indicators of watershed health. For watershed planners, the CN method is a simple but powerful tool to flag and rate the health and stress at the channel, watershed, and subwatershed levels.

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Contributors

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Metropolitan Portland, Oregon, Urban Growth Boundary:

A Land Use Planning Tool Protecting Farms, Forests, and Natural Landscapes

In the early 1970s, Oregon Governor Tom McCall and a unique coalition of farmers and environmentalists convinced the Oregon State Legislature to adopt the nation's first set of land use planning laws to help protect the state's natural beauty from a rising tide of urban sprawl. The resulting state goals and guidelines require every city and county in Oregon to have a long-range plan addressing future growth that meets both local and statewide goals by using urban land wisely, protecting natural resources, and setting urban growth boundaries (UGBs).

A UGB separates urban land from rural land. It promotes the efficient use of land, public facilities, and urban services, such as roads, water and sanitary sewer systems, parks, and schools, inside the boundary. Land outside the UGB is served by a rural level of roadways, does not allow the development of sanitary sewer systems, and is zoned exclusively for farm and/or forest use or rural residences.

Metro, the regional government created by voters in 1979 for the Portland metropolitan area, is responsible for managing the Portland region's UGB, which contains portions of 3 counties, 25 cities, and more than 60 special service districts. The UGB line is more than 322 km long and includes an area of approximately 103,600 ha. State law requires Metro to have a 20-year supply of land for future residential development inside the boundary. Every 5 years, Metro must complete a 20-year forecast for population and employment growth; conduct a capacity review of the land inside the UGB; and, if necessary, expand the boundary to meet the requirement for a 20-year supply of land. As part of the capacity review, the cities and counties within the Metro UGB also have the opportunity to develop policies, provide incentives, and plan for more intense uses through increased densities or the development of mass transit projects, which can reduce the need to expand the UGB for additional housing.

Two challenges arose with this system as originally implemented. First, landowners near the UGB were under periodic threat of urban expansion with little certainty about where the next expansion would occur. Second, although the identification of areas to preserve was fairly clear-cut, City and regional leaders lacked a method for determining the ideal locations and conditions for urban growth. As a solution, Metro and the three surrounding counties, Clackamas, Multnomah, and Washington, have instituted a regional process for identifying lands suitable for future urban development and for the protection of valuable farms, commercial forests, and other environmentally important natural areas.

In 2007, the Oregon State Legislature passed Senate Bill 1011, 2007 Or. Laws chapter 723, which allows for the designation of lands outside the UGB as urban or rural "reserves," as a way to direct future development while protecting existing rural and/or ecologically significant lands. The legislation prescribes factors for placing land into either reserve category. Lands designated as urban reserves are areas deemed suitable for "city-building," to which future urban development outside the UGB will be directed. Lands recognized for their agricultural or environmental value are placed into a rural reserve and become completely off-limits for all urban development for the next 50 years. By controlling and directing urban expansion through this program, Metro has preserved and enhanced the natural ecological systems of the land while avoiding or minimizing adverse effects on rural or natural lands. For more information on the urban and rural reserves process, see the Metro webpage included in the List of Sources.

The effectiveness of the UGB in concentrating development and limiting the conversion of forest or farmland is evident from examining the data on growth trends. Since the inception of the Portland region's UGB in 1979, it has been expanded to include approximately 11,331 ha of land, an increase in land area of 11%. Between 1980 and 2008, the population of the UGB increased by an estimated 507,000 people, or 35%. This amounts to a conversion of about 0.02 ha of land per capita, which is much lower than the nationwide trend noted in the US Department of Agriculture's 2000 National Resources Inventory (NRI). According to the NRI, developed land in the contiguous United States increased 34% between 1982 and 1997. During the same 15-year period, according to the US Census Bureau, the population grew by about 15%; thus, nationwide, land consumption occurred at more than twice the rate of population growth as a result of modern settlement patterns. The effectiveness of the UGB in limiting the conversion of rural lands for development is described in the Oregon Department of Land Conservation and Development's *2008–09 Farm & Forest Report*, which documented that the rate of farm loss in Oregon is less than one-third the rate of farm loss for the nation as a whole.

The UGB is one of the tools in the Oregon Statewide Planning Program that is used to protect farms and forests by restricting low-density rural development through the promotion of compact urban communities and a balanced transportation system for bicycling, walking, driving, and public transit. This supports the goals of (1) building complete communities by providing jobs and services close to where people live and (2) maintaining a more natural landscape in rural watersheds.

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For More Information

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This vignette was prepared by Tim O'Brien, principal regional planner, Metro Planning and Development, Portland, Oregon.

Grey to Green:

A Watershed Approach to Managing Stormwater Sustainably

The average annual rainfall in Portland, Oregon, a city of approximately 585,000 located near the confluence of the Willamette and Columbia Rivers, generates approximately 38 billion liters (10 billion gallons) of stormwater runoff. The quantity and quality of stormwater runoff resulting from rain falling on impervious surfaces, such as streets, rooftops, and parking lots, is one of Portland's greatest environmental challenges. As an alternative to traditional grey infrastructure that moves stormwater from the point of collection to a centralized treatment area, the City of Portland (City) is incorporating techniques that manage stormwater at its source with facilities that work like natural systems.

Portland is a national leader in green development and sustainable stormwater management. The *Portland Watershed Management Plan* (Plan) uses a scientific foundation to provide a comprehensive, strategic, and integrated approach to the management of stormwater and the improvement of watershed health. This approach addresses the sources and causes of environmental issues, rather than focusing solely on the symptoms or meeting specific regulatory requirements. Recognizing that urban watershed management is complex and requires coordination between City bureaus and community partners, this approach also promotes innovative and cost-effective solutions to stormwater management that meet multiple requirements and provide a range of benefits. The primary goals of the Plan include protecting, restoring, and improving hydrology, water quality, fish and wildlife habitat, and biological communities.

To accelerate implementation of the Plan, the City increased its commitment to funding green infrastructure through the Grey to Green (G2G) initiative. Started in 2008, the G2G initiative invests $55 million over five years in strategies that mimic natural systems to manage stormwater at its source. The purpose of G2G is to expand and enhance the City's green infrastructure using the following strategies, or best management practices: ecoroofs, green streets, tree planting, invasive species removal, revegetation, culvert replacement, and land acquisition (Figure 1). In addition to improving watershed health, integrating stormwater into the landscape saves money in both the short and long term by avoiding investments in grey infrastructure

Figure 1. Ecoroof examples include a backyard community effort (A),an ecoroof overlooking the Willamette River (B) and inclusion of solar panels (C). Photos courtesy of the City of Portland, Bureau of Environmental Services.

for stormwater management, ensuring service longevity for current investments, and providing cost-effective ways to meet water quality regulatory requirements. Measurable co-benefits of green infrastructure include improved public health, energy savings, and enhanced community livability.

The City designed G2G as an opportunity to transform sustainable stormwater management practices from the realm of innovative approach to everyday practice. To accomplish this, the initiative needed to provide value to people and communities, provide opportunities for partnerships and incentives, and include investments in the city's forested ridgelines as well as its urban neighborhoods. The City chose practices that complemented existing efforts but would realize meaningful results within the five-year time-frame.

The City's monitoring approach, which incorporates the best available science and protocols developed by the national Environmental Monitoring and Assessment Program, provides the basis for measuring the effectiveness of G2G strategies. The City convened a group of experts to assess the benefits of G2G beyond stormwater management. The 2010 report, *Portland's Green Infrastructure: Quantifying the Health, Energy and Community Livability Benefits,* presents the panel's findings, which will help guide decisions and funding priorities for future green infrastructure investments.

To date, G2G accomplishments include the following:

- Added 2.7 ha (6.7 acres) of ecoroof (100 roofs) and approved incentive funding for an additional 0.9 ha (2.2 acres). Ecoroof monitoring indicates greater than 50% annual retention of stormwater, meaning that half of the rain runoff from a roof that previously went to the treatment plant or into rivers and streams is now captured on the roof.
- Built 432 green streets, nearly half-way to the five-year goal of 920, at which point the estimated energy saved from avoided pumping and treatment costs will be enough to power 25 Portland homes per year.
- Planted 13,500 street trees and 13,100 yard trees. This was accomplished by partnering with the local nonprofit Friends of Trees and by giving a "treebate" to each city ratepayer who planted a tree on his or her own property.
- Treated 1,214 ha of invasive weeds by working with the Youth Conservation Crew.
- Acquired 106 ha of natural habitat areas in partnership with the City's parks department and other stakeholders.

For More Information

For more information, contact Daniela Brod Cargill, City of Portland Bureau of Environmental Services (daniela.cargill@portlandoregon.gov or 503-823-7226), or see www.portlandonline.com/bes/index.cfm?c=47203&.

Contributors

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HAVE A QUESTION YOU'D LIKE US TO ASK OUR EXPERTS? *The upcoming Spring 2012 issue will focus on the application of monitoring and modeling to watershed management. AWSPs members and Bulletin subscribers may email their questions to bulletin@awsps.org. The Bulletin features interviews with experts in the watershed and stormwater professions to discuss the topic of each issue. In this issue, three professionals weigh in with their perspectives on the impacts and management of agricultural land in our watersheds. Here is what our experts had to say….*

Challenges and Successes of Managing Agricultural Impacts

Peter Nowak

PhD, Professor, Nelson Institute for Environmental Studies, University of Wisconsin–Madison

Pete Nowak has spent his career engaging with farmers, managers, and policymakers to advocate for thoughtful farming and conservation practices. He has been an active member of the Soil and Water Conservation Society and has published works and spoken around the country to call attention to ineffective outreach and conservation practices and to challenge researchers and practitioners to best serve farmers, consumers, and the land. Pete founded the Wisconsin Buffer Initiative, which applied an adaptive management approach, based on sound and applicable science, for the implementation of buffers to protect Wisconsin's streams and lakes from nonpoint source pollution.

Q: How are you involved in managing watersheds?

A: I am at the University of Wisconsin–Madison and a professor in the Gaylord Nelson Institute for Environmental Studies, but I also work as a soil and water conservation specialist in the Environmental Resource Center on the agriculture campus. I wear two hats, so this gives me a chance to work outside of the typical agricultural conservation expertise and the opportunity to develop a true interdisciplinary approach. My geographic area is in Wisconsin, but I also provide leadership for the soil and water conservation service at the national level and do so through journal publications (e.g., in the *Journal of Soil and Water Conservation)*.

Q: In this issue, we focus on the connection between watershed land cover (e.g., forest, crops, and impervious) and its connection to water resource condition. What is your perspective on this topic as it relates to the various types of agricultural land covers?

A: Looking at land cover and watersheds is so important. In my experience, watershed scale matters, and the small or subwatershed scale works best for us. These are watersheds that are 10,000 to 25,000 acres. Using this small watershed scale means that farmers are more likely to know their neighbors and work together to coordinate changes in land cover or diversify the landscape. Land cover is important, but attention needs to be given to the scale of these efforts to integrate salient social processes in the watershed.

Q: What programs or assistance are available to advance land management practices to protect stream health? What are the challenges to finding the resources to address the problem?

A: We have over 50 years of research in terms of information sources that farmers use when making a decision. The source varies by the focus of the decision and the grower's stage in the decision process. A key distinction is whether the producer has a problem and is looking for a solution, or whether there are vendors who have solutions and are trying to persuade growers that they have a problem. For example, if the farmer has a production problem (e.g., yellow corn due to a nitrogen deficiency or sick animals) s/he is likely to go to the private sector (a consultant, seller, or vendor) because of expertise or an implied guarantee. Persuading producers that they have a conservation problem is much more complex and difficult as there is a tendency to simply emulate the private sector model. My one piece of advice in this arena is to design your assistance, communication, and marketing based on a robust targeting effort.

Q: Do you envision the application of nutrient-trading or other market-based approaches to achieve the needed nutrient reductions from agricultural land uses?

A: This summer, I am writing an editorial on nutrient trading to introduce a novel idea. Nutrient trading will work, but we have to get the scale or unit correct. My idea is to try nutrient trading with a group of farmers in a watershed where they form a cooperative or a corporation with shares based on the amount of land they manage. A local government or public works agency that needs to show the nutrient reduction would monitor outlets and pay the group based on the results. The payment would be distributed to share holders in proportion to the shares they hold. Farmers would begin to apply nutrient reduction strategies using the money, the local government agency's work load would be reduced, and peer pressure may lead those "last holdout" farmers to join the group in conservation strategies. This is a true market mechanism.

Q: What is the most significant shift you have observed in managing agricultural lands regarding the control of pollutant loads to streams?

A: Our new nonpoint policy in Wisconsin (NR 151) recognizes that a small proportion of land in any agricultural watershed contributes a disproportionate amount of the total degradation. We control pollutant loads by focusing remedial efforts on this small proportion of the land area. In these areas, we need to focus on "inappropriate practices in a vulnerable place or time." Understanding why people are engaging in inappropriate practices is critical to designing the appropriate remedial effort. There are no one-size-fits-all solutions; instead, we must focus on designing solutions based on understanding why the inappropriate practice is occurring. Shifting to this focus is not easy, and it will not happen overnight.

Q: Tell us more about the Wisconsin Buffer Initiative (WBI) and the process/methods by which farmers adopt environmental practices?

A: WBI emerged out of a controversial issue where environmental groups wanted to mandate riparian buffers on all waters of the state and the agriculture industry in the state was opposed to this. The university was asked to step in and assist. We brought all the stakeholders to the table, met regularly, and worked to get both sides to identify unanswered questions regarding the function of riparian buffers. For example, the common idea was that buffer width should be 30 feet. The group asked, "why 30 feet?" and pointed out that, in their experience, concentrated flow exists on the farm. From this observation, the group recognized that efforts to control concentrated flow in the uplands before it reached the riparian area made more sense.

Not all lands contribute equally to pollution in the watershed, and therefore we should focus on landscapes that are major pollution contributors. The idea is to begin in the uplands to try to prevent concentrated flow and then, working down toward the riparian area, only require buffers where upland treatments are insufficient. Using this approach, we did not need buffers everywhere; they were only needed in those situations where they complemented upland conservation designed to address concentrated flow. This was a win–win for the stakeholders involved. In this case, we were able to discuss the cause of the problem, gain an understanding of the "real world" situation, propose solutions, and pool ideas from diverse perspectives. The end result was our state legislature accepting a viable phosphorus management policy (NR 151).

Key Tools and Resources

Wisconsin Buffer Initiative: nelson.wisc.edu/people/nowak/wbi/index.php

Jim Pease

PhD, Professor, Department of Agricultural and Applied Economics, Virginia Polytechnic Institute and State University

Jim Pease conducts extension and applied research programs focused on the profitability of agricultural businesses, the efficiency and impacts of water quality protection policies, and economic development in rural areas of Virginia. Current projects include the development of watershed planning tools, an analysis of a "litter-to-energy" conversion project, and water quality education. Jim serves on the executive board of the Chesapeake Bay Program's Scientific and Technical Advisory Committee and is a member of the US Department of Agriculture (USDA) Mid-Atlantic Water Program. He has received awards for his programs from the Soil and Water Conservation Society, USDA Farm Services Agency, Virginia Cooperative Extension Agents' Association, Southern Extension Forester Resource Specialists Organization, and the Southern Agricultural Economics Association.

Q: How are you involved in managing watersheds?

A: I am both a professor at Virginia Tech and an extension specialist for Virginia Cooperative Extension; I work across the state of Virginia. The water quality education program at Virginia Tech conducts activities seeking to educate a wide range of Virginia citizens and groups about their local water resources and water quality. For example, my colleagues' Center for Watershed Studies conducts research, teaching, and outreach programs to improve watershed planning. My own research has focused on the economics of water quality, particularly in the livestock-dense Shenandoah Valley. In particular, over the last four years I have led a research project in the North Fork watershed of the Shenandoah River. Nearly all my education and research programs seek cost-effective and equitable solutions to water quality challenges of the Chesapeake Bay.

Q: In this issue, we focus on the connection between watershed land cover (e.g., forest, crops, and impervious) and its connection to water resource condition. What is your perspective on this topic as it relates to the various types of agricultural land covers?

A: Many of our most threatened watersheds in rural Virginia are approximately 70% forest and 25% agriculture; the rest of the area is mostly residential. Forest cover has a very benign water resource impact, but provides little in the way of sustainable income for resident families. Our farms are overwhelmingly small and unprofitable—92% of Virginia farms had gross annual incomes less than $100,000, and 62% lost money on the farm in 2007. For a mixed land cover that protects against water quality–damaging dispersed residential housing, agriculture has to be economically viable in these regions.

Q: What strategies seem to be working best for reducing nutrient loads to streams in the watersheds where you work?

A: Riparian buffers are working best, but the strategy that will work the best in the future is to remove poultry litter from nutrient-enriched watersheds instead of applying it to farm land. The Virginia Department of Environmental Quality is currently studying a potential poultry litter incineration plant that would produce 50 megawatts of electricity.

Q: Do you envision the application of nutrient-trading or other market-based approaches to achieve the needed nutrient reductions from agricultural land uses?

A: I'm an economist, so of course I favor a market-based approach. However, the devil is in the details—the effectiveness of nutrient trading depends on how the baseline is defined. It is likely that the agriculture sector could sell credits to the urban stormwater sector, which faces cost-prohibitive retrofits to reduce nutrient pollution. Reasonable baseline requirements for agricultural credit trades and third-party verification could make stewardship a profitable enterprise for agriculture.

Our nutrient budgets website (see below) was developed to provide a regional perspective on agricultural nutrient concentrations.

Q: What is the most significant shift you have observed in managing agricultural lands regarding the control of pollutant loads to streams?

A: Over the past years I have seen the development of an improved, more widespread awareness that what farmers do on the land has impacts on water quality. The awareness needs to be reinforced by feedback mechanisms. For example, Water Stewardship, Inc., is currently conducting a project with Shenandoah Valley producers that provides indicators of pollution-reducing performance based on the implementation of conservation practices, thus providing producers with measures of their impact on water resources and their accomplishments in pollution mitigation.

Q: The agricultural community is varied in terms of the type of agriculture (e.g., livestock vs. horticulture, small family–owned vs. corporate operations). Please characterize how the different types of agricultural operations have responded to regulated mandates to reduce pollutant loads to streams?

A: More than two-thirds of our agriculture sector is poultry and livestock. Additional regulations on animal feeding operations have put many farmers in a bind. One-third of the dairy farmers in the Valley have poultry operations as well, which means that these farms are hot spots for nutrients. Our livestock farmers have a disposal problem since they have an excess of nutrients but little land on which to dispose of these nutrients. Poultry litter incineration is one solution to this problem. Individual farmers have few economically feasible ways to solve the unintended consequences of the intensive livestock feeding system.

On the other hand, cropping farms have little incentive to apply more nutrients than are taken up by crops, and the main challenge is to convince such farmers to apply nutrients consistent with realistic yield goals. Unfortunately, full-time farmers can spend more time implementing conservation practices than can farmers who support the farm with additional jobs, but not 1 farm in 20 provides full-time employment.

Q: What research is still needed for effective watershed management focusing on land use? Is this research and information getting to the practitioners?

A: In addition to groundwater research, there is a need to provide more resources (time and money) for watershed planning conducted by citizens who live in the watershed. We need to create mechanisms whereby people at all levels discuss and reach consensus on future land use and practices that will work best in their watersheds.

Key Tools and Resources

Nutrient Budgets for the Mid-Atlantic States: www.mawaterquality.agecon.vt.edu/

Center for Watershed Studies: www.cws.bse.vt.edu/

Waste Solutions Forum: www2.dasc.vt.edu/faculty/knowlton/07jcds_WasteSolutionsForum_38_4.pdf

Mark Risse

PhD and PE, Professor, Biological and Agricultural Engineering, University of Georgia

Mark Risse is the Georgia Power Professor of Water Policy at the University of Georgia. In this position, he coordinates statewide programs in agricultural pollution prevention, stormwater management, and animal waste management; he also helps the Cooperative Extension program respond to environmental needs in Georgia. Mark serves on the leadership team for the national livestock and poultry environmental learning center, an online resource focused on providing the best science-based information to stakeholders around the country. Mark received his PhD in agricultural engineering from Purdue University, where he worked at the USDA National Soil Erosion Research Laboratory. He was elected fellow in the International Soil and Water Conservation Society in 2009 and is recognized nationally for his expertise and research in the areas of nonpoint source pollutant modeling and control, animal waste management, and outreach programs related to soil and water conservation.

Q: How are you involved in managing watersheds?

A: I work with the University of Georgia Cooperative Extension and have statewide responsibility for water resource issues in Georgia and the Southeast. I also work on national livestock waste efforts as well as international assignments. Primarily, I run an agricultural pollution prevention program, but in the last few years, water conservation has been the big issue because of the droughts in the Atlanta Metropolitan Area. My job is a mix of reactive programming—when a city, county, or the state needs assistance—and efforts to secure funding mechanisms to proactively research target areas identified as important. My work is done in partnership with others, such as the USDA Natural Resources Conservation Service, the Georgia Department of Agriculture and other state agencies, city and county local governments, and local county agents and producers.

Q: In this issue, we focus on the connection between watershed land cover (e.g., forest, crops, and impervious) and its connection to water resource condition. What is your perspective on this topic as it relates to the various types of agricultural land covers?

A: Classifying agriculture as a single land cover is a problem since we know that agricultural systems are highly variable. For example, the water quality impacts of poultry farms are different from those of ornamental nursery farms and cotton fields. Each farming practice has a distinct pollutant type and quantity. There are also differences among urban land covers (e.g., urban core vs. well-designed subdivision). However, in general, land covers, from high pollutant load to low pollutant load, are: urban, agricultural, and forested.

In a recent study conducted in northern Georgia, we wanted to compare agricultural land covers to urban and suburban land covers for nutrient-trading program recommendations, but only received funding to study the agricultural component. When we studied nine poultry farms and compared them to three forested watersheds, we found, first, that the nutrient loads of the farms were, in some cases, hundreds of times greater than those of the forested watersheds. But second, we found wide variation among farms. Comparing farm nutrient loads to values in the literature, the results suggest that the nutrient load of a well-managed farm could be similar to that of a forested area, whereas a poorly managed farm could have loads similar to those of an urban area.

Q: What is the most significant shift you have observed in managing agricultural lands regarding the control of pollutant loads to streams?

A: I have been in this position for 16 years, and the biggest shift I have seen in the agricultural community is a greater recognition of water quality problems and impacts. The environment and water quality are issues that farmers will talk about, and these issues impact their day-to-day decisions. This was not the case 15 years ago.

Q: What research is still needed for effective watershed management focusing on land use management? Is this research and information getting to the practitioners?

A: Improved research is needed to (1) predict nutrient loads for various land uses and conservation practice scenarios at the watershed scale instead of the more narrowly focused existing research at the scale of the best management practice or the farm; (2) develop better targeting methods to determine where our resources will have the most impact and when this makes sense from a funding standpoint and in terms of a nutrient load reduction result so that we don't waste money in watersheds where we are not likely to be able to make a difference; and (3) improve methods for monitoring and modeling pollutant loads from agricultural land use transitioning to urban land use, especially during the construction phase, and for documenting the short- and long-term benefits of better practices, such as low-impact development or the addition of compost to soils.

In terms of getting information out to practitioners, I think we are doing a fairly good job. With today's information technologies, those who want to can easily find the information they need. Where we need to improve is in getting the information to those who are not actively seeking it. Elected officials, the absentee landowner who controls the critical floodplain acreage, or the homeowners who don't even know that they have on-site wastewater treatment systems are the critical links for which we need to develop proactive educational programs.

Q: How have you or other organizations effectively engaged the community and practitioners in the land use management and watershed management discussion?

A: We try to engage multiple groups in almost everything we do. Working as a cooperative extension facilitates this process since we get to work closely with practitioners; do workshops on farms; and interact with erosion and sediment control specialists, city and county staff, and others. Working in the field and having these engaging discussions also allows me to bring the needs of practitioners and community members to the research community.

Key Tools and Resources

Livestock and Poultry Environmental Learning Center: www.extension.org/animal+manure+management

University of Georgia Agricultural Pollution Prevention Program: www.agp2.org

Southern Regional Water Program: srwqis.tamu.edu/

Report on Protecting Water Quality with Incentives for Litter Transfer in Georgia: ftp://ftp.engr.uga.edu/users/mlwilson/Litter%20Transfer%20Final%20306.pdf

AWSPs Photolog Contest

The Association of Watershed and Stormwater Professionals (AWSPs) is accepting photo entries for our next photolog contest. The winning photo will be featured on the AWSPs website and in the Spring 2012 issue of the *Bulletin*.

The photolog contest features the watersheds in which we work, live, and play. The photos can feature any number of subjects including:

- streams, forests, or other natural features;
- stormwater best management practices;
- restoration projects; or
- anything that captures the essence of a watershed.

To submit your photolog, provide one original digital photograph with a 250-word description to photocontest@awsps.org. All photologs must be submitted by **5p.m. November 1st, 2011**. For complete contest rules, see http://www.awsps.org/photolog.html.

Watershed Superstar

The Association of Watershed and Stormwater Professionals (AWSPs) sponsors a Watershed Superstar contest as a way to highlight the achievements of watershed professionals. AWSPs solicited nominations in the Spring 2011 issue of the *Watershed Science Bulletin*. A panel of three watershed professionals from the Center for Watershed Protection, Inc., judged applicants based on their accomplishments as well as the unique qualities that make up a Watershed Superstar, including ambition, innovation, collaboration, and dedication.

The *Bulletin* received an impressive collection of more than 50 applications for Watershed Superstar. Each applicant has made a significant and positive impact on his or her local watershed, and some have done so at national and international levels! The dedication and commitment shown by these applicants demonstrates what can be done to protect and restore our watersheds—one project, one mile, at a time. Congratulations to everyone for their contributions.

The Watershed Superstar for Fall 2011

Carmel Kinsella Brown

Owner, CKB Environmental Consulting, California

Nominated by Kim Fettke, Ecologist, AECOM

Carmel Brown is a professional civil engineer, working most of her 25-year career for municipal government agencies, while dedicating equal time to grassroots watershed protection work in urban communities. Most of Carmel's work is done in the background and, until her award last year by the California Stormwater Quality Association for outstanding volunteer service, she has never been publicly honored for her accomplishments. This award presents an opportunity to applaud a watershed "superstar" who typically works behind the scenes, frequently nominating others for exemplary work, but never shining the light on herself.

Carmel is best known for her work with stakeholders to effect change. As told by Kerry Schmitz, a Sacramento County engineering manager, "Carmel has the unique ability to bring multiple parties with different goals to the table to collaborate in order to find innovative solutions to watershed issues."

Carmel's work began in 1989 as a member of the Woodward–Clyde team tasked with writing the US Environmental Protection Agency (USEPA) guidance manuals for the Phase I National Pollutant Discharge Elimination System regulations. She helped build the early municipal stormwater programs in California and Oregon and successfully nominated the programs for USEPA's municipal excellence awards. She branched out to more holistic watershed work in 1992. She became an avid reader and West Coast promoter of the Center for Watershed Protection's *Watershed Protection Techniques* papers and she solicited the advice and help of Tom Schueler on several projects, including the development of Portland's first stormwater quality design manual in 1996. She founded the Laguna Creek Watershed Council in 2002 because, in her words, "the creek needed a voice." She secured more than $1 million in state grants for the Laguna Creek and Alder Creek watersheds to conduct assessments and prepare management plans in advance of development projects that threatened to modify the creeks. She was a frequent visitor to Elk Grove City Hall, where she presented concerns to elected officials and urged support for the Laguna Creek Watershed Council. She organized and led watershed tours to build awareness and capacity for community action within both watersheds. Today, the Laguna Creek Watershed Council is an active 501(c)(3) organization with a seven-member board of directors, and the Alder Creek watershed is home to the thriving Adopt a Creek/Trail program with hundreds of watershed stewards. Last year, Carmel once again worked behind the scenes to secure the prestigious Sacramento River Watershed Excellence Award for the community leaders of both programs.

Carmel's accomplishments are perhaps best summarized by Eva Butler, founder of the Sacramento Splash! youth environmental education program: "While Carmel … has a deep knowledge of stormwater regulations…, she has not limited herself to working on that part of the watershed protection equation. She has devoted equal time to the pursuit of grassroots solutions involving folks beyond the regulated community. Her on-the-ground efforts have helped to chart a course toward better outcomes for local streams and the communities around them."

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Future Bulletin Issues

Spring 2012
The Application of Monitoring and Modeling in Watershed Management

Fall 2012
Watershed Planning

The deadline for article submission for the Fall 2012 issue is **Friday, April 6, 2012**. For submission requirements, visit www.awsps.org/watershed-science-bulletin.

Upcoming Events

- **October 5, 2011, 12–2 p.m.,** Webcast: Just How Gross Can You Get? Dealing with Gross Solids and Illicit Discharges in your Community (www.cwp.org/our-work/training/webcasts)
- **November 16, 2011, 12–2 p.m.,** Webcast: Stream Restoration (www.cwp.org/our-work/training/webcasts)

WWW.AWSPS.ORG