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# A geospatial approach for assessing denitrification sinks within lower-order catchments

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ABSTRACT

Local decision makers can influence land use practices that alter N loading and processing within the drainage basin of lower-order stream reaches. Because many practices reduce water retention times and alter the timing and pathways of water flow, local decisions regarding land use can potentially exert a major influence on watershed N export. We illustrate a geospatial approach for assessing the role of denitrification sinks in watershed N delivery at the local level using: (a) widely available geospatial data, (b) current findings from peer-reviewed literature, (c) USGS stream gage data, and (d) locally based data on selected stream attributes. With high resolution, high quality GIS data increasingly available to local communities, they are now in a position to guide local management of watershed N by targeting upland source controls and by identifying landscape sinks for protection and/or restoration. We characterize riparian wetlands, lentic water bodies, and stream reaches as N sinks in the landscape and use geospatial particle tracking to estimate flow paths from N sources and evaluate N removal within sinks. We present an example analysis of the Chickasheen drainage basin, RI, USA, comparing N flux from three equivalent hypothetical N source areas situated in different regions of the watershed and illustrating the role of each N sink type in mediating N flux. Because our goal is to generate a tool that is used by and useful to decision makers we are exploring methods to better understand how decision makers understand and respond to the manner in which information is presented.



## 1. Introduction

Nitrogen (N) export from coastal watersheds exerts profound effects on the function and value of coastal estuaries. Harmful algal blooms, hypoxia, and destruction of critical spawning habitat are among the many problems linked to elevated N contributions to coastal waters (Howarth et al., 2000; Diaz, 2001; Goolsby et al., 2001; Nixon et al., 2001; Rabalais et al., 2001; Diaz and Rosenberg, 2008). The annual N loading to the biosphere has more than doubled in the past 50 years, and estuaries are receiving substantially more N from terrestrial sources than in the past (Vitousek et al., 1997). High concentrations of nitrate in shallow groundwater and streams are correlated with agricultural land use and unsewered residential developments (Gold et al., 1990; Nolan et al., 2002; Nowicki and Gold, 2008). However, watershed processes can mitigate N delivery to coastal waters. Mass balance studies conducted across a wide range of geographic scales consistently find that watersheds retain 60–90% of total watershed N inputs (Howarth et al., 1996; Jordan et al., 1997).

One of the major advances in watershed science over the last 25 years has been the realization that certain areas of the landscape have a capacity to function as "sinks" for N. Areas of high N sink capacity can include riparian wetlands, reservoirs, and lower-order streams where particular features, such as pools or organic debris dams play an important role in N removal (Mitsch et al., 2001; Peterson et al., 2001; Groffman et al., 2003; Mitsch and Day, 2004; Seitzinger et al., 2006). Seitzinger et al. (2006) suggested that water residence time was a controlling factor for reducing N loading in all these settings and that hydrology and geomorphology strongly influences residence time. In sink areas, biogeochemical processes transform inorganic N, especially nitrate, into organic N in plant and/or microbial biomass, or into N gases via denitrification (Gilliam, 1994; Hill, 1996; Gold et al., 2001; McClain et al., 2003), preventing movement of N into receiving waters. In contrast, where landscape sinks are absent or are bypassed by land management practices (e.g., tile drainage or storm water conveyance systems), activities generating N losses (sources) pose a greater risk of watershed N export (Gold et al., 2001; Paul and Meyer, 2001; Dinnes et al., 2002).

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In a synthesis of landscape denitrification, Seitzinger et al. (2006) found that the amount of land-based N denitrified generally declines as N flows through watersheds. From a mass balance perspective, terrestrial soils account for the greatest loss of N via denitrification with lesser amounts in groundwater, followed by rivers, then lakes/reservoirs, and finally estuaries. However, on a per-area basis, denitrification rates in rivers or lakes/reservoirs are ten times greater than in groundwater or soils. McClain et al. (2003) argued that interfaces of terrestrial and aquatic systems, e.g., riparian areas and hyporheic zones, represent hot spots of denitrification where N-enriched water mixes with complementary water sources or substrates. Alexander et al. (2007) found within the glaciated Northeast that lower-order stream watersheds greatly influence downstream water quantity and N loading. Since lower-order "headwater" streams generally comprise 70–85% of total stream length within a watershed (Peterson et al., 2001; Rosenblatt et al., 2001), and contribute 80% of stream flow and >50% of the total N mass delivered to all stream reaches, effectively managing these drainage areas for N sinks will allow local decision makers to make the most effective use of restoration and protection dollars and efforts. In a recent review of denitrification modeling approaches, Boyer et al. (2006) call for more linkages between field measurements, spatial databases, and model structures. We suggest that decision support tools that target local audiences can help promote voluntary and locally based efforts to manage denitrification at the scale of lower-order catchments.

Local decision makers can influence land use practices that alter N loading and processing within the drainage basin of lower-order stream reaches. Because many land use practices reduce water retention times and alter the timing and pathways of water flow, decisions regarding the pattern and extent of open space can influence watershed N export. New subdivisions and roads restrict groundwater recharge and stream baseflow and limit the amount of N that can undergo riparian groundwater denitrification. Urbanization creates flashy streams that reduce retention times and deepen stream channels (Paul and Meyer, 2001; Groffman et al., 2003) diminishing the capacity for stream N processing. Storm water drainage systems – in agricultural or suburban development – alter flow paths within natural swales and ephemeral streams that constitute the expanded stream network during rain events, thereby bypassing potential N removal (Wigington et al., 2005).

The objective of this paper is to illustrate a geospatial approach for assessing the role of denitrification sinks in watershed N delivery at the local level using: (a) widely available geospatial data, (b) current findings from peer-reviewed literature, (c) USGS stream gage data, and (d) locally based data on selected stream attributes. With high resolution, high quality GIS data increasingly available to local communities, the time is ripe to make use of those data to guide local management of watershed N by targeting upland source controls and by identifying landscape sinks for protection and/or restoration (Last, 1995; Lovejoy, 1997; Groffman et al., 2009). Source controls may include wastewater treatment options (sewers vs. conventional septic systems vs. alternative on-site systems; Oakley et al., this issue), housing density specifications, stormwater management (Collins et al., this issue) or agricultural Best Management Practices (BMPs), including carbon bioreactors (Schipper et al., this issue). Sustaining and improving N sinks can be accomplished through riparian buffer regulations or preservation efforts and through restoration of streams (Kaushal et al., 2008), and wetlands (Mitsch and Day, 2005).

We use hydrologic, particle tracking GIS tools to estimate the flow path from a known N source, such as an unsewered residential development, croplands or livestock area, to a watershed outlet. In this way we follow the movement of N through a watershed and estimate the reduction in N from denitrification and other N cycling processes as flow moves through (1) riparian wetlands, (2) lakes or reservoirs, and (3) stream reaches. We recognize that N cycling occurs in a variety of settings throughout a watershed, but here we focus on those sinks that have demonstrated high denitrification potential, and that are directly relevant to local decision makers. In this paper we will refer to all processes that remove N from ground and surface water, e.g., denitrification, soil immobilization, plant uptake and accumulation in sediments, and other microbially mediated processes such as dissimilatory nitrate reduction to ammonium (DNRA) and anaerobic ammonium oxidation (anammox) (Burgin and Hamilton, 2007) as N removal.

## 2. Approach

The formulation of our approach to assessing N sinks at the local level has been guided by the ever increasing availability of GIS data, with the provision that as more refined or accurate data become available they can be incorporated. We developed estimates of N removal efficiency based on peer-reviewed literature available on each of the three types of watershed N sinks. To facilitate use of this tool by local decision makers, most of whom have limited time and resources to devote to GIS tasks, we developed relatively simple relationships that make use of widely available data.

### 2.1. Setting

Our modeling efforts are focused on the 17.4 km$^2$ Chickasheen watershed in southern New England. Land use in the watershed is a mixture of forest, croplands (commercial turf and silage corn) and unsewered, low density residential developments. The watershed is glaciated, with several naturally occurring lakes and ponds. The surficial geology is composed of till in the headwater uplands and glacio-fluvial deposits overlain by loess in the valleys (Rector, 1981). The climate is classified as humid continental (Ward and Trimble, 2004) with average precipitation and pan evaporation of 1300 mm y$^{-1}$ and 550 mm y$^{-1}$, respectively (Dickerman et al., 1997). Elevations within the watershed range from 28 to 98 m.

The Chickasheen watershed drains into the larger Pawcatuck watershed which discharges to Little Narragansett Bay, an estuary under consideration by the State of RI for management to control N inputs due to anthropogenic eutrophication. New England estuaries are sensitive to accelerated N inputs. Eelgrass beds, prime shellfish habitat, have undergone severe declines in shallow embayments throughout New England. These declines have been attributed to shading from algal blooms and accelerated growth of algae stimulated by excess N loading (RI DEM, 2003). Local communities in concert with State regulators actively promote nonpoint source controls in the vicinity of this study (Cox, 2002), including N removal technologies for on-site wastewater disposal (see Oakley et al., this issue) and carbon bioreactors (see Schipper et al., this issue). In addition The Nature Conservancy, a nongovernmental organization dedicated to land preservation, is now including the N removal function of land parcels in its decisions related to land acquisition. Sources and sinks of N within the land uses, soils and physiography found in the Chickasheen watershed have been the focus of considerable research (Gold et al., 1990, 2001; Nelson et al., 1995; Groffman et al., 1996; Kellogg et al., 2005).

### 2.2. Data sources and flowpath processing

Topography in the form of digital elevation models (DEMs), soils, hydrography, land use and stream discharge was used to develop our estimates of N removal. We used the widely available National Elevation Dataset (http://ned.usgs.gov/) with 10 m resolution for the DEMs and the National Hydrography

**Table 1**
Area-normalized long-term annual average flow rates of selected lower-order catchments (<160 km$^2$) on the Atlantic Coast, U.S.

| Gaging station | Drainage area (km$^2$) | Location (Lat, Long) | Area-normalized average annual flow[a] (m$^3$ s$^{-1}$ km$^{-2}$) |
|---|---|---|---|
| Oyster River near Durham, NH | 31 | 43°09′, 70°57′ | 0.018 |
| Indian Head River at Hanover, MA | 78 | 42°06′, 70°49′ | 0.023 |
| Chipuxet River at West Kingston, RI[b] | 25 | 41°29′, 71°33′ | 0.025 |
| Manasquam River at Squankum, NH | 114 | 40°10′, 74°09′ | 0.018 |
| Pocomoke River near Willards, MD | 157 | 38°23′, 75°19′ | 0.013 |
| Piscataway Creek near Tappahannock, VA | 72 | 37°52′, 76°54′ | 0.012 |
| Hood Creek near Leland, NC | 56 | 34°17′, 78°08′ | 0.018 |

[a] Flow data from long-term gaging stations (USGS, 2009).
[b] The catchment of the Chipuxet River at West Kingston, RI is immediately adjacent to the ungaged Chickasheen catchment (17.4 km$^2$), also located in West Kingston, which is used in this study.

Dataset (http://nhd.usgs.gov/) at a scale of 1:24,000 for hydrography. Soils data, specifically hydric/non-hydric status, were obtained from the Soil Survey Geographic (SSURGO) Database (http://soils.usda.gov/survey/geography/ssurgo/) at a scale of 1:15,840. Land use was obtained from 1995 Anderson Level III coverages (1:24,000). We used an extension to ArcGIS Version 9.2 (Environmental Systems Research Institute, Inc., Redlands, CA), ArcHydro (Maidment, 2002), to model water flow at the catchment scale. We extracted a flow accumulation grid that allows for particle tracking from any point (e.g., source location) in the landscape to the watershed outlet as well as the delineation of the drainage area to any point (e.g., sink location) in the landscape.

For discharge data, a key factor in retention time and N sink effectiveness, we relied on long-term records of average monthly and annual area-normalized stream flow (volume per time per area), derived from the extensive USGS gaging network, to provide estimates of flow at every sink location within the study (USGS, 2009). These values reflect land use/land cover and water extraction but seasonally and annually are largely driven by differences between precipitation and evapotranspiration. Area-normalized flows in our study area are at the high end for small, coastal watersheds along the Atlantic coast (Table 1), suggesting that water residence time and corresponding N removal in sinks may be higher in other areas.

### 2.3. Riparian wetlands

There is an extensive body of work spanning two decades focusing on denitrification in riparian areas. Investigators have explored influences on riparian zone N removal efficiency such as riparian buffer width (e.g., Vidon and Hill, 2004), vegetation type (e.g., Addy et al., 1999), geomorphic setting (e.g., Lowrance et al., 1997; Pinay et al., 2000), soils (e.g., Simmons et al., 1992; Young and Briggs, 2007), hydrological flow path (e.g., Hill et al., 2000; Maître et al., 2003), climate (e.g., Sabater et al., 2003) and season (e.g., Pinay et al., 1993; Nelson et al., 1995), as well as adjacent land use (e.g., Hanson et al., 1994; Baker et al., 2006). Mayer et al. (2007) performed a meta-analysis on data available in the scientific literature to identify trends between riparian N removal efficiency and riparian buffer width, surface vs. subsurface flow, and vegetation.

Using widely available GIS data we cannot readily identify vegetation type or whether flow through a riparian area is dominated by surface or subsurface processes. We can, however, characterize riparian land use and soils. It has been well documented that riparian zones are most effective as N sinks when undeveloped and vegetated, and relatively ineffective if hydrologically altered to bypass the riparian ecosystem through residential, agricultural or other types of development (e.g., Carpenter et al., 1998). Research also suggests that riparian wetlands, characterized by hydric soils, act as effective N sinks while riparian areas with non-hydric soils are less reliable N sinks (e.g., Lowrance et al., 1997; Gold et al., 2001; Groffman et al., 2009). Hydric soils provide conditions that favor microbial denitrification: high water table, low dissolved oxygen, and high soil organic matter to provide carbon as an electron donor.

Based on these findings we use a series of if/then statements to arrive at estimates of riparian N removal efficiency. If land use is developed, then we assume no removal. If land use is undeveloped and soils are non-hydric, then we assume no removal. If riparian land use is undeveloped and soils are hydric (i.e., wetland soils), then N removal efficiency is based on the width of the undeveloped (vegetated) hydric soils (Table 2). We used regression equations provided by Mayer et al. (2007) to guide our estimates of N removal effectiveness in vegetated riparian areas as a function of buffer width. The selected width classes recognize commonly used regulatory limits. These estimates are comparable to estimates derived from field assessments that are used to direct management efforts in the Neuse River watershed (Osmond et al., 2008).

### 2.4. Lakes and reservoirs

Lakes and reservoirs, i.e., lentic aquatic systems, are potentially large sinks for N loads due to their long retention times (Seitzinger et al., 2002, 2006; Harrison et al., 2009), with hypoxic and anoxic benthic zones providing conditions that favor denitrification. Seitzinger et al. (2006) examined lentic N sink data within a variety of aquatic ecosystems and found that lakes and reservoirs followed the behavior of streams, where N loss has been positively related to retention time. We extracted the lake and reservoir data presented in Seitzinger et al. (2002) and performed a linear regression analysis that yielded the following relationship between N removal and the ratio of reservoir depth, $D$, and residence time, $T$ (Fig. 1):

$$\text{N removal}\,(\%) = 79.24 - 33.26 \times \log_{10}\left(\frac{D}{T}\right) \quad (1)$$

Average reservoir depth can be expressed as volume, $V$ [km$^3$], divided by surface area of the reservoir, $A_r$ [km$^2$]. Residence time, $T$ [y], can be expressed as volume, $V$ [km$^3$], divided by discharge, $Q_{yr}$

**Table 2**
Riparian buffer N removal effectiveness.

| Riparian land use | Hydric soil status | Width (m)[a] | % N removal |
|---|---|---|---|
| Developed | | | 0 |
| Vegetated | Non-hydric | | 0 |
| | Hydric | <5 | 0 |
| | | 5–15 | 40 |
| | | 15–30 | 60 |
| | | >30 | 80 |

[a] Width classes are based on current regulatory practices and are relevant to local decision makers.

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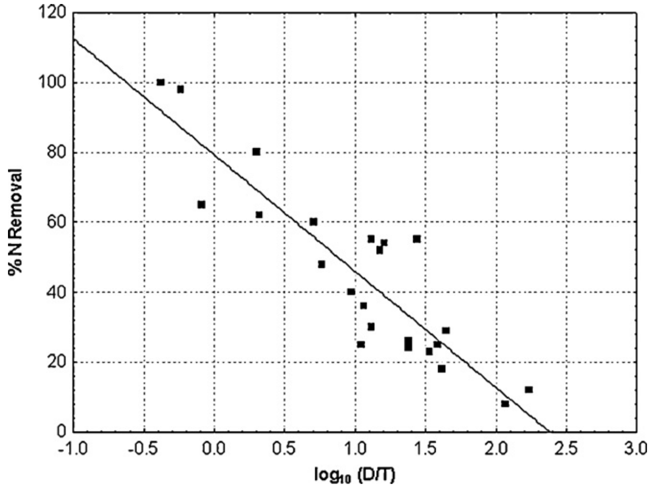


**Fig. 1.** Reservoir N removal (%) as a function of $\log_{10}(D/T)$, where $D$ = depth (m) and $T$ = retention time (y). Data from Seitzinger et al. (2002).

[$km^3\,y^{-1}$]. Thus,

$$\frac{D}{T}\,[\mathrm{m\,y^{-1}}] = \frac{V/A_r}{V/Q_{yr}} \times 1000 = \frac{Q_{yr}}{A_r} \times 1000 \quad (2)$$

We can use available spatial data to estimate discharge at any point in the drainage network as

$$Q_{yr} = A_d \times Q_{norm} \times 0.031536 \quad (3)$$

where $A_d$ [$km^2$] is the drainage area to the point of interest (i.e., the N sink), and $Q_{norm}$ [$m^3\,s^{-1}\,km^{-2}$] is the regional estimate of discharge normalized by drainage area, and converted to [$km^3\,y^{-1}\,km^{-2}$]. As mentioned above, these regionally explicit flow data are widely available and can be found in USGS reports and USGS online databases (e.g., Armstrong et al., 2004; USGS, 2009). Thus we obtain the ratio

$$\frac{D}{T}\,[\mathrm{m\,y^{-1}}] = \frac{Q_{yr}}{A_r} \times 1000 = Q_{norm} \times \left(\frac{A_d}{A_r}\right) \times 31.536 \quad (4)$$

We used this relationship to explore the effects of different discharge levels on N removal in lakes over a range of normalized discharges (Fig. 2). Reservoirs with $A_d/A_r > 10$ would display substantial differences in predicted N removal in response to changes in the rate of normalized discharge. Thus, the selection of normalized discharge warrants careful consideration. In a study of the Pawcatuck watershed, RI, Fulweiler and Nixon (2005) found dissolved inorganic nitrogen (DIN) flux is seasonal and Alexander et al. (1996) reported that as much as 70% of the annual nitrate flux occurs during the winter and spring in Atlantic coastal rivers, when flows are highest (Armstrong et al., 2004). Armstrong et al. (2004) present regionalized medians of the 50-percent monthly discharges for the period 1976–2000, normalized by drainage area. Because high nitrate flux is correlated with high discharge we have chosen to use the average of these normalized discharges for the winter and spring months within the South Coastal region of southern New England (January through May): $Q_{norm} = 0.03\,m^3\,s^{-1}\,km^{-2}$. The mean annual normalized discharge for the South Coastal region of southern New England is $Q_{norm} = 0.024\,m^3\,s^{-1}\,km^{-2}$, about 80% of the winter/spring normalized discharge, indicating that high flows dominate annual discharge. The fraction of incoming N removed by lakes in the study area is likely to be much higher during summer, given that the average of the summer (June through October) monthly normalized discharges at $Q_{norm} = 0.006\,m^3\,s^{-1}\,km^{-2}$ is approximately one-fifth of the high flow discharges (Armstrong et al., 2004). We recognize that lakes and reservoirs can be viewed as hydrologically distinct. Outflow from reservoirs can be regulated by dam(s), thereby shifting residence time from that expected for a natural lake based on drainage area. Our study area does not contain any reservoirs, but we expect that in future applications reservoirs will be treated as lakes for the purpose of N sink characterization, due the uncertainty associated with dam manipulation schedules.

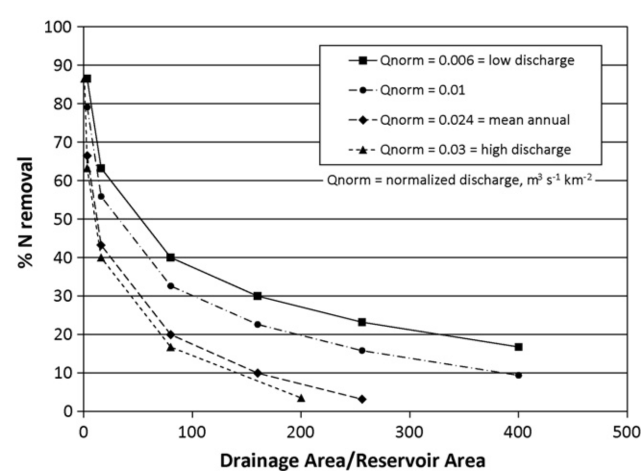


**Fig. 2.** Reservoir N removal (%) as a function of drainage area/reservoir area for a range of area-normalized discharges.

### 2.5. Streams

The role of streams in watershed N dynamics has been the focus of intensive research, with early studies formulating the nutrient spiraling model (e.g., Newbold et al., 1981), which has since been used to assess in-stream denitrification (e.g., Royer et al., 2004). The wide range of observed N loss rates within streams has spurred research using both field experiment techniques and statistical approaches based on spatial data. The Lotic Intersite Nitrogen Experiment (LINX) has used N addition and isotopic analysis to explore the extent to which stream characteristics – hydrodynamic, chemical, and metabolic – might explain the wide variation among streams in N uptake, removal and cycling.

Alexander et al. (2000) developed a hybrid statistical/mechanistic mass-balance model to estimate N flux in the Mississippi basin (SPARROW – SPAtially-Referenced Regression On Watershed attributes), correlating observations of stream N flux with spatially referenced N sources and physical characteristics of the landscape and water bodies. Regression results showed that N loss rates were inversely related to stream depth and that much of the N removal in streams was occurring in lower-order reaches. They concluded that the proximity of N sources to higher-order streams and rivers is an important factor in N delivery to the Mississippi basin outlet. Recognizing the variability of stream function among different regions of the U.S., SPARROW has since been developed for other parts of the country, including New England (Moore et al., 2004). An important result of the New England modeling effort was the lack of statistically significant annual modeled N reduction for streams with flows greater than $2.83\,m^3\,s^{-1}$, highlighting the importance of lower-order streams in mitigating watershed N export.

Alexander et al. (2007) further refined this New England SPARROW model to investigate and quantify the influence of headwater streams of the northeastern U.S. on N delivery to downstream waters. The extent of N removal and cycling in streams, including the permanent removal of N via denitrification, is limited by the extent of interaction with the stream channel and hyporheic

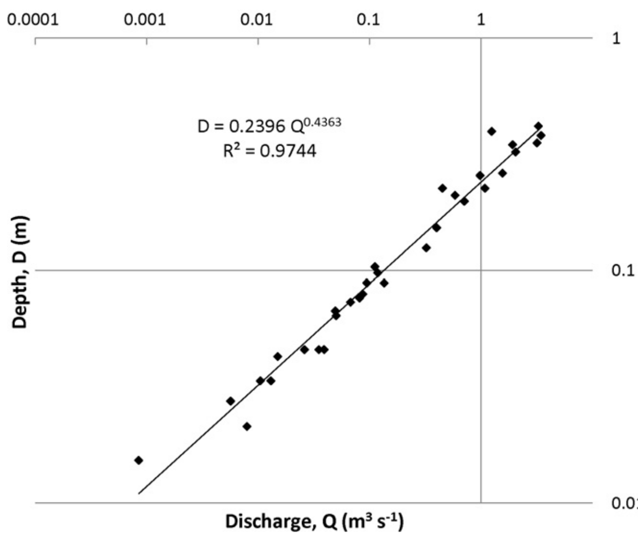


**Fig. 3.** Stream depth-discharge relationship based on data presented in Armstrong and Parker (2003) from nine stream gaging stations in southern Rhode Island, after Leopold and Maddock (1953).

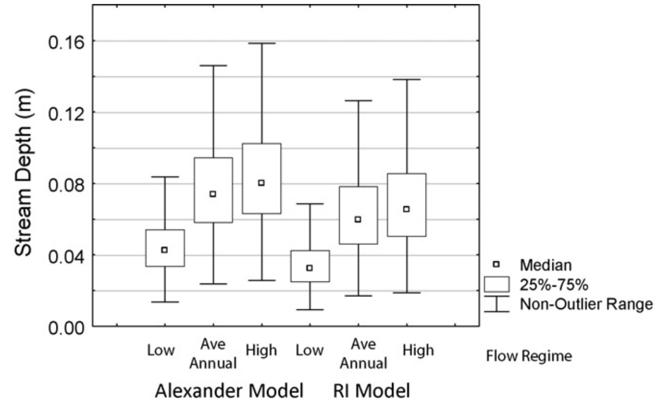


**Fig. 4.** Comparison of stream depths at low, average annual and high normalized discharge, between a model presented by Alexander et al. (2000) and the relationship derived from local data, based on the method presented by Leopold and Maddock (1953). Discharge is the product of drainage area and normalized discharge; drainage areas derived from ArcHydro for 46 first and second order streams in southern RI.

zone, both of which decrease with increasing stream order and drainage area. Alexander et al. (2007) modeled N loss in streams as a first-order decay process, reflecting aggregate net effects of physical, hydrological and biochemical properties of the stream channel and hyporheic zone, to arrive at an expression for the fraction of N transported along a stream reach as a function of stream depth and time of travel. We used this expression as follows:

$$\text{N (\%) removed along stream reach} = (1 - \exp(-\theta_{S1} D^{\theta S2} T)) \times 100 \quad (5)$$

where $\theta_{S1} = 0.0513\,\text{m}\,\text{d}^{-1}$, $\theta_{S2} = -1.319$, $D$ = mean water depth [m], and $T$ = time of travel [d]. Coefficients were derived by Alexander et al. (2007) by an iterative process to best fit model predictions with long-term streamflow and water quality data gathered from gaged streams in the northeastern United States. Water depth, $D$, is expressed as a function of mean annual stream flow, $Q$ [$\text{m}^3\,\text{s}^{-1}$] (Alexander et al., 2000):

$$D = 0.2612Q^{0.3966} \quad (6)$$

This expression originates from Leopold and Maddock (1953) with flow data from 112 streams in the South and MidWestern U.S. We examined the appropriateness of this expression to estimate stream reach depths in our study area by following the method of Leopold and Maddock (1953), using data provided from multiple depth/flow observations on nine streams in Southern New England (Armstrong and Parker, 2003) (Fig. 3). We obtained the following relationship:

$$D_{\text{RI}} = 0.2396Q^{0.4363} \quad (7)$$

where stream discharge, $Q$, is again estimated using drainage area to the downstream end of the stream reach multiplied by discharge normalized by drainage area, $Q_{\text{norm}}$. The ratio of $D/D_{\text{RI}}$ was found to be $1.0901Q^{-0.0397}$ which generates close agreement (within 10%) between the two data sets when $Q$ is less than $10\,\text{m}^3\,\text{s}^{-1}$. For the range of discharges in our region from lower-order streams, depths derived using Eq. (6) are slightly deeper than those derived using Eq. (7) (Fig. 4). We use Eq. (6) in our model because it represents a slightly more conservative (low) estimate of N removal from these streams, is based on a larger sample of streams that encompasses a wide range of geographic settings and places our approach in line with widely-used relationships.

Mean travel time, $T$ [d], along a given stream reach can be expressed as reach length [m]/mean velocity [$\text{m}\,\text{d}^{-1}$]. Reach length can be extracted from the spatial data using simple GIS tools. The mean velocity of a dissolved constituent along a stream reach can then be estimated using available spatial data applied to the following relationship (Jobson, 1996; Eq. (14)):

$$V\,[\text{m}\,\text{s}^{-1}] = 0.020 + \left[0.051 \times (D'_{\text{a}})^{0.821} \times (Q'_{\text{a}})^{-0.469} \times \frac{Q}{D_{\text{a}}}\right] \quad (8)$$

where $D_{\text{a}}$ = drainage area [$\text{m}^2$] to the downstream point of the stream reach under consideration, $D'_{\text{a}}$ = dimensionless drainage area = $(D_{\text{a}}^{1.25} \times \sqrt{g})/Q_{\text{a}}$, $Q_{\text{a}}$ = mean annual discharge from the stream reach [$\text{m}^3\,\text{s}^{-1}$], $Q'_{\text{a}}$ = dimensionless relative discharge = $Q/Q_{\text{a}}$, and $g$ = acceleration of gravity = $9.8\,\text{m}\,\text{s}^{-2}$.

Thus, water depth, $D$, and retention time (i.e., time of travel), $T$, and N removal within each stream reach can be estimated using spatial data combined with regionally explicit USGS stream flow data applied to each reach. Fig. 5 illustrates the effect of stream depth and travel time on N removal within stream reaches.

### 2.6. Examples

We explored the role of sinks in N watershed export by situating three hypothetical equivalent N sources in different locations within the Chickasheen drainage basin (17.4 $\text{km}^2$) (Fig. 6). For the

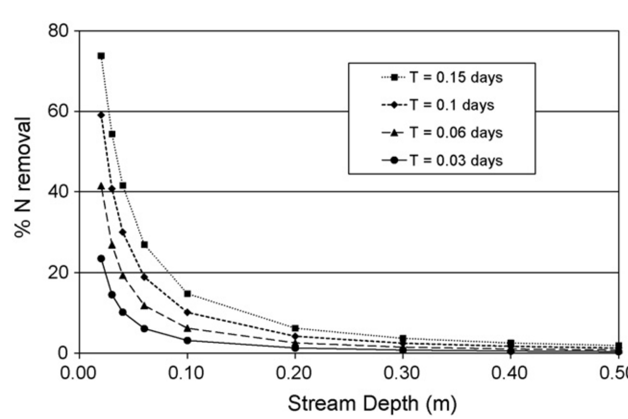


**Fig. 5.** N removal (%) with stream depth over a range of travel times along a stream reach.

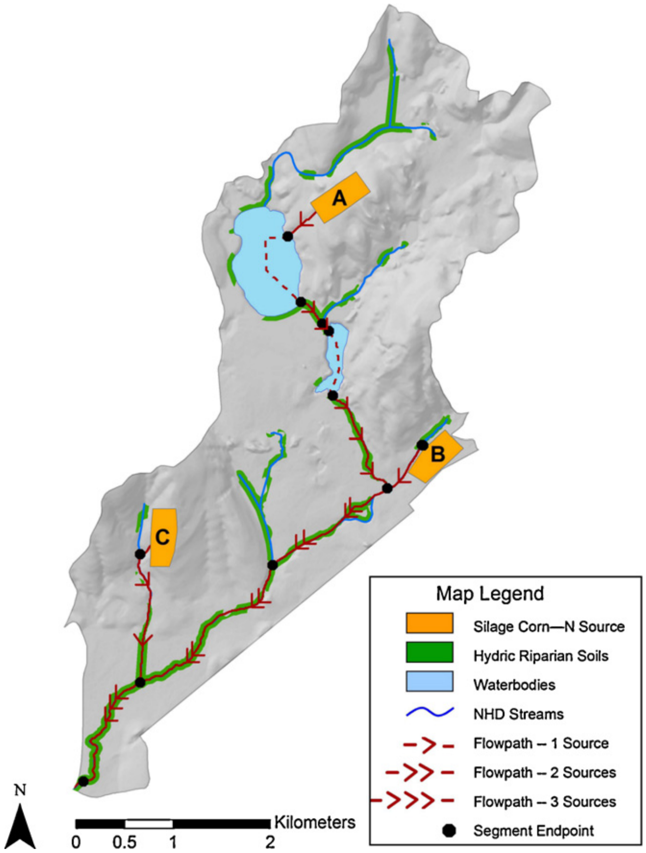


**Fig. 6.** Chickasheen watershed, southern Rhode Island, USA, with three hypothetical N sources. Flow paths to the watershed outlet and N sinks reflect actual watershed characteristics.

purposes of illustration these N sources were configured as 15 ha of silage corn, fertilized with manure and with no cover crop, yielding 60 kg N $ha^{-1}$ $y^{-1}$ (Gold et al., 1990).

Using the particle-tracking capability of ArcHydro, we extracted a flow path from each of the three hypothetical N sources to the watershed outlet, approximating the hydrologic path taken after leaving the source. Each source area was placed within 350 m of surface water. Because flow paths extracted using ArcHydro are derived from elevation data they represent an estimate of surface flow. However, here we assume that ground water flow paths are not substantially different from surface water flow paths. This assumption may not always be valid, especially in regions where subsurface stratigraphy is complex, or surface topography relatively flat, but groundwater flow patterns cannot be verified with widely available geospatial data and requiring their inclusion in the analysis would add substantial complexity and cost that would likely preclude the routine use of this approach by local decision makers. In our study area this assumption is supported by data comparing groundwater and surface water drainage areas for several subbasins within the Pawcatuck watershed (NB: the study area is a subbasin of the Pawcatuck watershed), which shows differences in only two of the six subbasins, and these are less than 10% (Wild and Nimiroski, 2004). By following the hydrologic path from a N source to the watershed outlet we can assess the extent to which the source water may encounter landscape sinks as it moves through the watershed.

Conceptually, every source flow path would typically be comprised of a terrestrial component followed by an aquatic component. The terrestrial component progresses downgradient toward surface water, either a stream or lake/reservoir, and

**Table 3**
Hydrologic characteristics of sinks along the flow path from Source A to the watershed outlet, under low and high flow conditions.

| Segment sink type | Drainage area ($km^2$) | Stream reach length (m) | Reservoir | Low flow | | | | | High flow | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Stream | | | | Reservoir | Stream | | | | Reservoir |
| | | | $A_d/A_r$ ($km^2$ $km^{-2}$) | $Q$ ($m^3$ $s^{-1}$) | $D$ (m) | $V$ (m $s^{-1}$) | $T$ (d) | $D/T$ (m $y^{-1}$) | $Q$ ($m^3$ $s^{-1}$) | $D$ (m) | $V$ (m $s^{-1}$) | $T$ (d) | $D/T$ (m $y^{-1}$) |
| Reservoir | 3.8 | | 6.3 | | | | | 1.2 | | | | | 6.0 |
| Stream | 7.0 | 323 | | 0.042 | 0.07 | 0.09 | 0.042 | | 0.211 | 0.14 | 1.18 | 0.003 | |
| Stream | 7.2 | 93 | | 0.043 | 0.08 | 0.09 | 0.012 | | 0.216 | 0.14 | 1.19 | 0.001 | |
| Reservoir | 8.0 | | 65.6 | | | | | 12.4 | | | | | 62.1 |
| Stream | 8.5 | 1263 | | 0.051 | 0.08 | 0.09 | 0.16 | | 0.256 | 0.15 | 1.23 | 0.012 | |
| Stream | 10.5 | 1523 | | 0.063 | 0.09 | 0.09 | 0.187 | | 0.314 | 0.17 | 1.28 | 0.014 | |
| Stream | 15.5 | 2110 | | 0.093 | 0.10 | 0.10 | 0.243 | | 0.465 | 0.19 | 1.39 | 0.018 | |
| Stream | 17.4 | 1602 | | 0.102 | 0.11 | 0.10 | 0.181 | | 0.512 | 0.20 | 1.41 | 0.013 | |

$A_d$ = drainage area, $A_r$ = reservoir area, $Q$ = discharge, $D$ = depth, $V$ = velocity, $T$ = retention time. Low flow $Q = (0.006\ m^3\ s^{-1}\ km^{-2}) \times A_d$, high flow $Q = (0.03\ m^3\ s^{-1}\ km^{-2}) \times A_d$.

encounters a riparian area whose land use and soil characteristics control the extent of N removal at the land–water interface. Once the flow path encounters surface water it follows surface flow downstream, either along stream reaches or through lakes or reservoirs.

We estimated N removal along each flow path from source to watershed outlet by dividing the path into segments of interest, i.e., potential sinks: (1) riparian buffer to a width of 30 m between the upland and the surface water, (2) lakes or reservoirs, and (3) stream reaches. We define stream reaches as beginning at either (1) a confluence of two or more streams, (2) a pond or lake outlet, or (3) the point at which a flow path enters the stream. The stream reach ends at either (1) a confluence, (2) a pond or lake inlet, or (3) the watershed outlet.

We track the N flux sequentially moving through each landscape feature and calculating the N leaving each sink, $N_{\mathrm{out}}$, based on the site-specific characteristics that control the extent of N removal, and on the mass of N contributed from upgradient, $N_{\mathrm{in}}$. Specifically, $N_{\mathrm{out}} = N_{\mathrm{in}} \times (1 - \%\ \mathrm{N\ removal}/100)$. The extent of cumulative N removal that occurs during movement to the watershed outlet from each N source varies according to the characteristics of the sink ecosystems encountered by the different flow paths.

## 3. Results and discussion

To illustrate our approach, an example of each of the three sink types (riparian zone, reservoir, and stream reach) will be highlighted from the source flow paths shown in Fig. 6. Both high and low flow conditions will be examined to better understand the impact of discharge on N removal.

For example, the flow path from Source A encounters a "reservoir" type sink at Yawgoo Pond (Fig. 7A). Yawgoo Pond is high in the watershed and has a relatively small drainage area, resulting in a drainage area to reservoir area ratio of 6.3 (Table 3). At low flow, % N removal within Yawgoo Pond is estimated at 77%. At high flow, % N removal is estimated at 53% (Table 4). Because the drainage area to reservoir area ratio is relatively low, retention time is expected to be substantial, even at high flow. Barber Pond, the second "reservoir" encountered by the Source A flow path, has a drainage area that is twice that of Yawgoo Pond and a surface area less than one quarter that of Yawgoo, resulting in a drainage area to reservoir area ratio of 65.6, a lower retention time and lower estimated N removal rates of 43% at low flow and 20% at high flow (Tables 3 and 4).

The flow path from Source B encounters a riparian area with 14 m of hydric soils (Fig. 7B). Based on the estimates in Table 2, an estimated 40% of the N entering the riparian zone from Source B is removed. In obtaining the width of hydric riparian soils from the SSURGO database we recognize that we are exceeding the accuracy limits of current SSURGO soil maps. Rosenblatt et al. (2001) assessed our ability to use the SSURGO database to identify hydric riparian soils by groundtruthing the presence and width of hydric soils at 100 riparian sites located on lower-order streams throughout the Pawcatuck River watershed, RI. They found that the SSURGO database was quite reliable in identifying the presence or absence of hydric riparian zones when the field width of the hydric riparian soil was >10 m – but not particularly accurate regarding the actual field width. Hydric riparian soils can occur as narrow corridors within the landscape and soil mappers often chose to denote their presence by expanding the actual size to meet the minimum map unit (Stolt and Baker, 1995). Additional information within the SSURGO database, such as geomorphic setting and slope class can refine our understanding of riparian function. By using ranges of widths to estimate N removal we recognize the uncertainties introduced by limitations of scale inherent in the SSURGO database as well as our level of understanding of the effects of width on N removal function.

**Table 4**
N removal (%) within each sink segment and cumulative N removal along the flow path from Source A under both low and high flow conditions.

| Segment sink type | N removal (%) | | N at sink outlet (kg y$^{-1}$) | |
|---|---|---|---|---|
| | Low flow | High flow | Low flow | High flow |
| Source A | | | 900 | 900 |
| Reservoir | 76.7 | 53.4 | 210.4 | 420.4 |
| Stream | 6.4 | 0.2 | 196.9 | 419.1 |
| Stream | 1.9 | 0.1 | 193.2 | 418.9 |
| Reservoir | 42.9 | 19.6 | 110.4 | 336.7 |
| Stream | 20.4 | 0.7 | 87.9 | 334.3 |
| Stream | 21.3 | 0.8 | 69.2 | 331.8 |
| Stream | 22.4 | 0.8 | 53.7 | 329.1 |
| Stream | 16.5 | 0.6 | 44.8 | 327.3 |

Another source of uncertainty is the role of ground water seeps in riparian N removal. Warwick and Hill (1988), working in a glaciated headwater catchment in Ontario, Canada, found that because of low retention time ground water emerging from seeps within riparian zones experienced little N removal whereas O'Driscoll and DeWalle (2010), working in an unglaciated catchment in Pennsylvania, USA, observed N removal in riparian seeps. A second source of uncertainty is the lack of published data on riparian total N budgets. The meta-analysis of riparian buffer N removal effectiveness by Mayer et al. (2007) included data from 88 sites reported in 45 published studies. Of these, only four studies (eight sites) included data on N species other than nitrate. While nitrate may be the dominant form of N carried with inflows to riparian buffers, dissolved organic nitrogen (DON) can be bioavailable (Wiegner et al., 2006) and N transformations in wetlands may result in a net loss of nitrate at a downgradient sampling location, but a net gain in ammonium or DON (e.g., Lowrance et al., 1983; Velledis et al., 2003). Riparian N removal rates based on nitrate alone may therefore be overestimates of total N removal. Our assumption that ground water flow paths are not substantially different from surface water flow paths allows us to use the particle-tracking capability in ArcHydro, but is another potential source of uncertainty in the terrestrial component of the analysis, especially in areas with complex surficial geology.

The flow path from Source C to the watershed outlet is relatively short and includes a lower-order stream reach (Fig. 7C). The reach length is 1.4 km and the drainage area to the downstream end of the reach is 0.97 km$^2$. At low flow the stream depth is estimated at 0.03 m, with a retention, or travel, time of 0.25 d. At high flow, stream depth is found to be 0.06 m, and retention time 0.02 d. Based on these physical characteristics the estimated N removal during low flow is 68% as compared to 4% at high flow. The expression we use to estimate N removal in streams is extremely sensitive to discharge for small catchments, a characteristic that also strongly influences stream reach depth, velocity and retention time. This is consistent with the body of work researching N dynamics in streams that has found retention time and interaction with stream sediments to be key physical factors in the extent of N transformation and removal, with lower-order streams demonstrating the highest N removal potential (e.g., Alexander et al., 2000; Peterson et al., 2001; Bernot and Dodds, 2005). As N is transported within higher-order streams, the opportunities for N removal diminish, as demonstrated by the low removal within the final few segments of flow from Source A (Table 4). The assumption that higher-order streams do not generally play a significant role in the mediation of N delivery to coastal systems is challenged by Tank et al. (2008) who found significant N transformation in a larger river, suggesting the need for further research in this area.

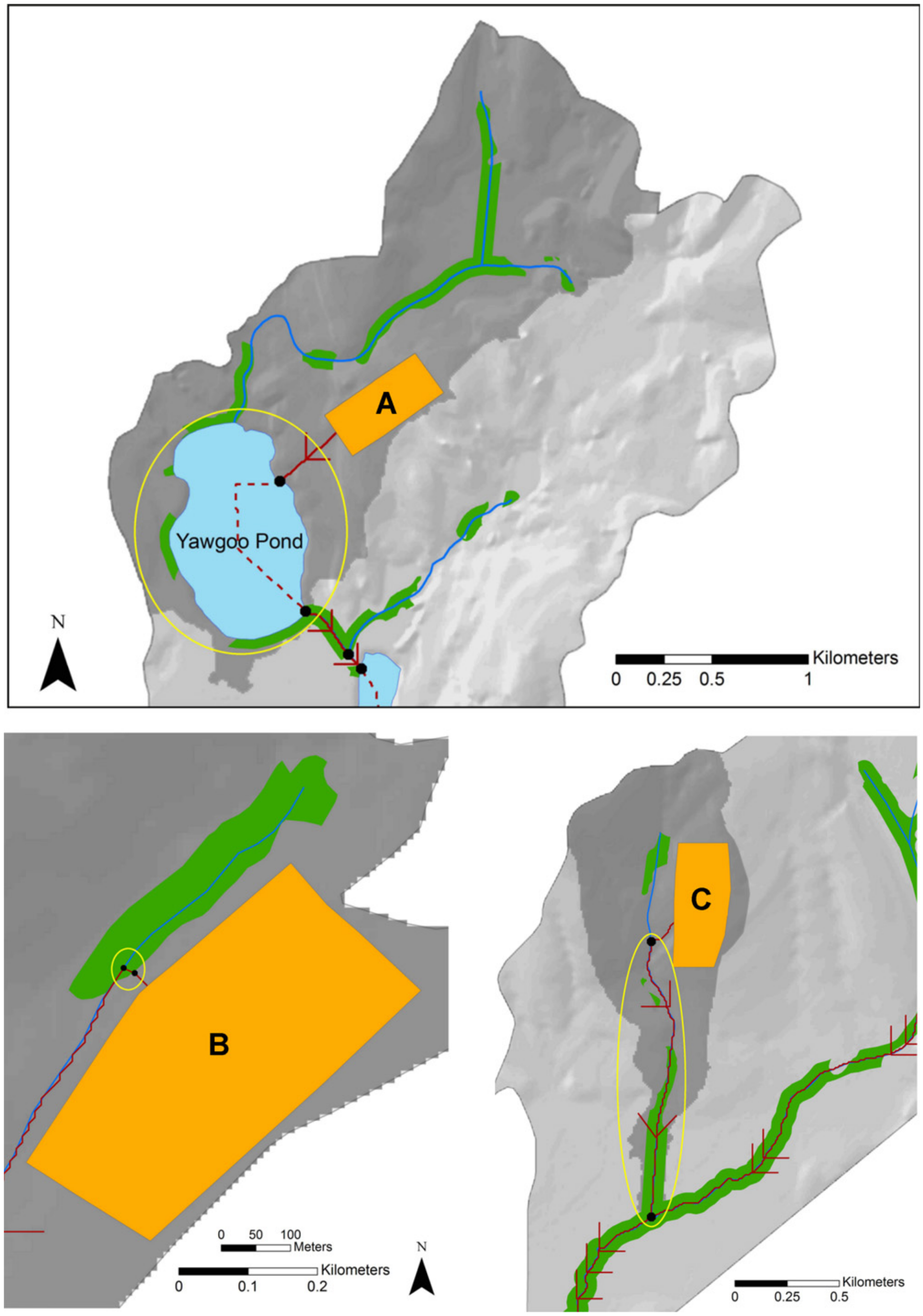


**Fig. 7.** Examples of N sink types under high flow conditions. (A) Reservoir N sink: (drainage area/reservoir area) = 6.3. N removal in reservoir = 53%. (B) Riparian wetland N sink: width of hydric riparian soils = 14 m. N removal in riparian zone = 40%. (C) Stream reach N sink: time of travel in lower-order reach = 0.02 d. Stream depth = 0.06 m. N removal in stream = 4%.

The cumulative removal of N through the network of ecosystem sinks encountered from a source area to the outlet can be seen by tracking the fate of N from Source A. In this situation, the flow path from Source A to the watershed outlet is divided into nine segments. The first segment from the source to the stream does not encounter hydric soils in the riparian area and thus the riparian area is not factored as a sink. The subsequent eight segments consist of a pond (Yawgoo Pond) (Fig. 7A), followed by two lower-order stream reaches, followed by another pond (Barber Pond), followed by four stream reaches (Tables 3 and 4). Retention time, and therefore N removal effectiveness, in both reservoirs and streams is inversely related to discharge. As illustrated in Table 3, the difference in N sink effectiveness between low flow and high flow conditions can be substantial, with a total of 95% removal during low flow as compared to 64% removal during high flow (Table 5). This is consistent with observations of seasonally elevated N flux that correspond to periods of higher discharge.

Table 5 summarizes the cumulative removal of N as it is transported through the Chickasheen watershed during high flow from Sources A, B, and C, and illustrates the potential capability of each sink type to attenuate N. Riparian zones represent a substantial N sink in the presence of hydric soils. Reservoirs,

**Table 5**
Cumulative N removal during high flow transport from hypothetical N Sources A, B, and C to the outlet of the Chickasheen watershed.

| Source | Edge of field loss ($kg\,N\,y^{-1}$) | N sink removal | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Riparian zones | | Reservoirs | | Streams | | Total | |
| | | $kg\,N\,y^{-1}$ | % | $kg\,N\,y^{-1}$ | % | $kg\,N\,y^{-1}$ | % | $kg\,N\,y^{-1}$ | % |
| A | 900 | – | – | 563.6 | 63 | 10.6 | 1 | 574.2 | 64 |
| B | 900 | 360.6 | 40 | – | – | 20.3 | 2 | 380.9 | 42 |
| C | 900 | – | – | – | – | 40.3 | 5 | 40.3 | 5 |

depending upon their landscape position within a watershed can likewise attenuate N effectively. Streams represent a less effective N sink, especially during high flow and within higher-order stream reaches.

The implications for local watershed management that arise from this exercise help to prioritize N sink protection activities as well as N source control strategies. Because retention time is an important factor for effective N sink function, approaches that slow flow will enhance opportunities for interaction with biologically active zones encountered along hydrologic flow paths, such as wetland (hydric) soils, reservoir sediments, and stream channels and hyporheic zones.

The Source A flow path is the only one that encountered reservoirs (Yawgoo and Barber Ponds), removing 63% of the total N load under high flow conditions, and accounting for almost all of the total N removal (64%) along the flow path. In this case our "reservoirs" were natural lakes not controlled by dams. In New England and throughout the northeast mill ponds are still common (Walter and Merritts, 2009). As communities consider removing dams, among the many factors to be considered are the downstream consequences of removing an effective N sink (Stanley and Doyle, 2003). If a reservoir is eliminated through dam removal, managers may need to consider other types of N removal strategies, e.g., source controls or in-stream bioreactors (Schipper et al., this issue).

The only riparian area where hydric soils were present was along the Source B flow path, removing 40% of the total N, accounting for almost all of the estimated sink N removal (42%). The riparian wetland represents the only significant sink between Source B and the watershed outlet – arguing for the importance of this specific sink for protecting estuarine functions downstream. Communities are routinely asked to grant variances for development within riparian buffers. By understanding the role that a specific riparian zone may play in N removal, communities will have a greater capacity to evaluate the implications of individual requests for variances. Along with riparian wetland protection, other approaches such as low impact development can help maintain or restore natural riparian hydrology that allows for increased interaction of ground and surface water with riparian soils and vegetation (Groffman et al., 2003).

The only N sink encountered by the Source C flow path is a lower-order stream. Discharges in RI are high for the Atlantic coast, so even the lower-order stream reach provided only 4% N removal during high flow. In areas where discharges are lower and retention times are higher, lower-order stream N removal could be substantial. To maintain this potentially significant N sink it is important for communities to sustain stream flow patterns and prevent the development of "urban stream syndrome" (Walsh et al., 2005). Urban settings, characterized by extensive impervious surfaces, present particular challenges for N sink function as a result of conveyance systems traditionally focused on swift and efficient stormwater removal. Riparian ecosystems are generally bypassed with stormwater directly discharging to urban streams. These streams are characterized by lower baseflow between storms and higher, more intense discharge during storm events, reducing retention time and N removal capacity (Groffman et al., 2005). Therefore restoration priorities in more urbanized areas may focus on increasing residence time and opportunities for denitrification by reestablishing hydrologic connections between streams and riparian areas (Kaushal et al., 2008) and through a combination of enhanced infiltration techniques such as bioretention systems and in-stream sinks such as constructed wetlands (Collins et al., this issue).

Source controls may be the most effective option in situations where N sinks are not present and may not be feasible to restore. In the example presented here, Source C represents the highest priority for source controls because of the limited opportunities for N attenuation during transport through the watershed. Close proximity to surface waters, particularly to higher-order streams, also warrants consideration for source controls. Source control strategies include optimizing fertilizer use through crop nutrient management and on-site wastewater treatment that incorporates N removal technology (Oakley et al., this issue). Additionally, ecological engineering techniques such as bioreactors have the potential to serve as sinks in areas where N laden ground water converges, e.g., tile drains or riparian zones (Schipper et al., this issue).

## 4. Conclusions and next steps

We have presented an approach that is intended to constitute a first-step in the development of a decision support tool for local decision makers and regulators in prioritizing N source controls and recognizing and managing N sinks within localized catchments. Because our goal is to generate a tool that is used by and useful to decision makers in exploring "what if" scenarios we are incorporating social science insights that elucidate how decision makers understand and respond to the manner in which information is presented (Carver, 1991; Simonovic and Bender, 1996; Zigurs et al., 1999).

We recognize that we did not incorporate and communicate uncertainty in our estimates of N sinks. The usefulness and method of expressing uncertainty must be explored through interactions with the intended users of these tools. We have mentioned several examples of uncertainty in our approach, such as riparian zone width estimates. While this particular source of uncertainty is a limitation in the SSURGO database, site-specific wetland mapping is widespread and available to local communities and could be used to refine local databases. Future collaboration between natural scientists, social scientists and decision makers is essential to develop approaches that convey our understanding of N dynamics and address the capabilities and needs of local communities.

Some uncertainty arises from our limited understanding of N removal processes. Future research that would help to reduce this uncertainty could focus on (a) total N budgets for riparian wetlands; (b) N removal effectiveness of higher-order streams; (c) the relative importance of different N removal pathways within streams, reservoirs and riparian wetlands; and (d) the role of seeps in riparian N removal.

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