Ecosystem Modeling and Restoration Strategies to Improve Water Degradation and Social Equity in Urban Watersheds

Ruoyu Zhang
Yangquan, Shanxi, China

M.S. Data Science, University of Virginia, 2021
M.S. Geography, Virginia Tech, 2018
B.S. Geography and Meteorology, Virginia Tech, 2016

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Committee Members:
Lawrence E. Band (advisor)
Howard E. Epstein
Xi Yang
Jonathan L. Goodall

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ABSTRACT

Urban ecosystems are vulnerable to extreme weather events due to environmental degradation induced by human activities. Massive deforestation and build-up of impervious areas in urban areas block infiltration of precipitation which now becomes surface runoff in urban watersheds. The rapid release of huge quantities of water after storms not only cause erosion of stream channels and floodplains but also dimmish a series of ecosystem functions (e.g., denitrification, plant nitrogen uptake, etc.) that serve as important roles to mitigate nutrient sources in uplands released to streams. Consequently, urban watersheds are suffering from many environmental issues locally, such as flash floods and excessive in-stream nutrient loads and sediment, and these upstream degradations propagate to downstream and coastal waterbodies and cause eutrophication and elevated levels of stream bed, threatening not only ecosystem health but also human lives there. To address these water-related issues, various efforts (e.g., low-impact development or LID) are introduced to manage water quality, restore the pre-urbanization flow regime, and mitigate risks from previous destructions in urban watersheds, receiving attentions of urban water managers these days. However, the effectiveness of these ecosystem restorations is heterogeneous among watersheds. Quantifying local outcomes and improvements could not been done without universally applied and reliable analytical frameworks.

Therefore, this dissertation focuses on building frameworks for analyzing and quantifying current states and upcoming improvements brought by ecosystem restorations in urban watersheds. Specifically, these frameworks allow urban managers to project

nitrate reduction from stream restoration projects and associated socio-economic benefits for every 1,000-ft stream reaches in Baltimore metro areas (Chapter 2), quantify the possible changes in streamflow and upland ecological responses from each scenario of LID implementations in Scotts Level Branch, Baltimore (Chapter 3), and include human-induced nitrogen load from fertilization and septic wastewater to improve water chemistry analysis using our hydro-ecological model, RHESSys, in Baisman Run, Baltimore (Chapter 4).

Overall, the results of each chapter in this dissertation show a synthetic theme: Shifting ecohydrological conditions of urban watersheds back to pre-urbanization conditions is unattainable, no matter how massively ecological restoration practices are implemented. Human-induced land cover changes and nutrient inputs permanently alter hydrological flow regimes and nutrient cycles of urban ecosystems. Relying solely on green infrastructures would be insufficient to reverse current issues and, in some cases, may even exacerbate them. Coupling of grey and green infrastructures with regulating nutrient inputs to urban ecosystems is essential for future management practices and protection of local and downstream aquatic ecosystem health. The statistical and process-based models in this dissertation provide valuable and easy-to-use tools for decision makers to plan possible restoration scenarios spatially and evaluate corresponding responses of upland and streams in urban ecosystems systematically.

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Chapter 1: INTRODUCTION

1.1 OVERVIEW

1.1.1 Environmental challenges in urban ecosystems

Urbanization is rapidly expanding in both developed and developing world, which brings dense populations with high consumption of imported food, water, energy, and new infrastructure. In the United States, many metropolitan regions are rapidly expanding outward to accommodate growing populations, during which many forest and agricultural areas are converted to residential and commercial land use, with expanded transportation and piped drainage infrastructure.

With the massive land cover change from vegetated to impervious areas, the hydrology and ecosystem undergo large changes in urban watersheds. In a forested watershed, most rainfall infiltrates soils and flows through subsurface pathways. Some water travels vertically to deeper soil layers and aquafers, some evaporates and transpires, and the rest is slowly released into streams over several days through months and years. Infiltration excess runoff is rare in forest, though it may occur when rainfall is exceptionally intense and higher than the infiltration rate. When soils are fully saturated, surface runoff can occur as well. The mechanism of water flowing through soils in forested watersheds controls both peak and total volume of streamflow (Horton, 1945). On the other hand, urban watersheds have greater coverage of impervious areas blocking the infiltration process, and rainfall is either transported into storm sewers and streams directly as overland flow in a short period of time with little loss in evapotranspiration or soil storage, or can run-on to pervious areas for potential infiltration. Compared to flow mechanism in forested watersheds which releases rainfall into streams over much longer time, urban streamflow tends to have a higher peak and steep recession after storms. The rapid rise and fall of streamflow in response to storm events in urban watersheds is commonly referred as urban flashy flow regime. The direct connection between rainfall and streams through impervious surfaces and storm sewers significantly increases the flow velocity and the amount of water in streams during storms, leading to stream erosion and excessive sediment and nutrient loadings to downstream waterbodies, commonly known as urban stream syndrome (Walsh et al., 2005).

The flashy urban flow regime introduces several environmental issues. High flow speed erodes stream beds and banks, cuts stream channels deeper, and transports large amounts of sediment downstream. Incised stream channels disconnect streamflow from riparian and floodplain areas, where the vegetation and microbes are unable to absorb or uptake in-stream nutrients (e.g., nitrogen, phosphorus, etc.) and reduce nutrient retention and temporary storage of flood waters. Eventually, downstream ecosystems are under higher risk of environmental hazards such frequent algae blooms and elevated river beds due to sediment accumulations.

Downstream water bodies suffer eutrophication due to the increased loading of nutrients and sediment from urban watersheds and reduced capacity for ecosystem retention. Studies (Hagy et al., 2004; Li et al., 2016) have shown that the Chesapeake Bay has experienced more frequent algae blooms in the last 20 years because of increasing nutrient input from the watershed. Meanwhile, climate change is projected to increase storm frequency and intensity in the future (Trenberth, 2011), introducing higher-level uncertainty and risk of environmental hazards to urban watersheds where population are dense and environmentally vulnerable and downstream water bodies. Therefore, there is an urgent need to mitigate the flashiness of urban streamflows, reverse the ecological degradation of urban watersheds, and promote the resilience of urban ecosystems.

The complexity of urban ecohydrology is not only about the geophysical processes and engineering management, but also interactions with humans. Human activities introduce additional pollutants and complicate the ecohydrological processes of the ecosystem by introducing strong heterogeneity (Band et al., 2005; Groffman et al., 2016; Groffman et al., 2004; Kaye et al., 2006). For example, fertilization, septic effluents, and sanitary sewer leakage add excessive, localized nitrogen and phosphorus to urban ecosystems, which can then be transported and degrade local and downstream water quality. The unequal distribution of resources within many urban areas, causes different distribution of hazard and pollutant exposure by demographic characteristics including wealth and race (Boone et al., 2009; Boone et al., 2014).

1.1.2 Benefits of urban ecosystem restoration

Ecosystem restoration provides an opportunity that can not only restore urban ecosystem health and but also fill the gaps of equity issues such as inequitable living environments between wealthy and low-income communities. Specifically, there are many low-income neighborhoods in intra-urban areas of the United States metropolitan cities having little vegetation and green recreational space which is considered crucial to people’s quality of living environment. In contrast, when moving outward from city centers to rural and ex-urban neighborhoods, residents generally have easy access to forests, parks, and other green amenities nearby. Terrestrial and aquatic ecosystem restoration (i.e., green infrastructures and stream restoration) practices in urban watersheds could bring green spaces to low-income neighborhoods, address the unequal distribution of resources in the past urbanization, and improve the living environment for disadvantaged communities. This approach to abate environmental justice issues in metropolitan regions could not only reduce the "green" inequality but also promote future discussion about other co-benefits that urban ecosystem restoration can bring to the society, such as mitigation of urban heat island and improve air quality, beyond mitigations of stormflow and in-stream nitrate loads.

As discussed above, growing imperviousness and concentrated flow (e.g., storm sewers, curbs) are major factors contributing to the series of ecological and environmental issues in urban watersheds. Therefore, restoring the ecohydrological processes and resulting ecosystem services in urban watersheds so that they behave more like natural watersheds could reverse the accelerated stormwater derived flood hazard and water quality degradation in sustainable manners. Currently, many stormwater mitigation facilities, both grey and green infrastructure or low impact development (GI or LID, Figure 1.1), are adapted to control water quantity and quality in urban watersheds. Typical grey infrastructures are generally non-natural and engineered facilities, including stormwater and sanitary sewers and wastewater treatment plant. GI or LID are small-scale and try to mimic nature-based systems of vegetation clusters and ponds, including reforestation, rain gardens, and bioswales. Grey infrastructures are typically designed to drain stormwater out of streets rapidly, either directly into nearby streams, or into detention storage features to temporarily store and reduce peak flows. Wastewater Treatment Plants (WWTP) are centralized facilities to treat and eliminate contaminants in wastewater through series of physical and chemical treatments. The focus of GI or LID is to promote infiltration, evapotranspiration, and ecological processes to sustainably mitigate both stormflow and in-stream nutrients load. Therefore, they have the potential to restore terrestrial ecosystem functions in urban watersheds, and these approaches are referred as terrestrial ecosystem restoration in this dissertation. Aquatic restoration (i.e., stream channel restoration, discussed in the next section) is also widely used in urban watersheds to mitigate eroded stream channels, reduce sediment export, and reestablish riparian ecosystems.

Many studies have evaluated the effectiveness of green infrastructure to reduce storm peak volumes and timing from field measurements or model simulations, and suggest that GI is effective for small storm events. For example, Damodaram et al. (2010) found LID can effectively mitigate peak discharge from small and frequent storm events (i.e., 1-year, 24 hour storms) while detention ponds are more effective to control peak discharge for larger storm events (e.g., 2-year 24 hour storms and larger). Page et al. (2015) found permeable pavement and bioretention cells along streets can significantly reduce the runoff coefficient (i.e., runoff volume / total precipitation) and increase storm runoff thresholds (i.e., precipitation that triggers runoff occurrence). Similarly, Gilroy & McCuen (2009), Jarden et al. (2016) and Fiori & Volpi (2020) all found LID can mitigate peak discharge volumes and extend lag times of peak discharge after small storm events. However, the effectiveness of GI to reduce peak discharge volumes and delay lag times is inconsistent for large storm events. Some studies evaluated the on-site performance of infiltration GI show these facilities' ability to remove inflow runoff. Lewellyn et al. (2015) found an infiltration trench system with vegetated pretreatment and designed to capture storms below 2.5 cm built at Villanova University was capable to significantly reduce runoff volumes for storm events greater than 2.5 cm. Lord et al. (2013) showed similar results, that a bioinfiltration system reduced 50% of runoff volumes for storms above the designed rainfall amounts. On the other hand, studies evaluating GI performance at watershed scale argue that implemented GI in their watersheds could not alleviate peak discharges for large and intense storms. Damodaram et al. (2010) found no significant reduction in peak discharge volumes under LID or detention ponds only or combined scenarios, and similar results are found in Gilroy & McCuen (2009), Qin et al. (2013), Jefferson et al. (2017), and Hopkins et al. (2022). These findings suggest that individual GI facilities could potentially retain significant amount of runoff into the systems, but at the watershed scale, insufficient GI would not alter the flash flow regime dominated by impervious areas and piped drainage systems in urban areas.

Meanwhile, GI/LID's plants and microbes in soils can assimilate nutrients by vegetation uptake, and denitrification. Both of these processes can be promoted to potentially improve water quality and reduce excessive nitrate loadings to downstream receiving waterbodies. In-stream NO_3 can also be transformed and stored in other forms (e.g., organic nitrogen) in algal biomass, but may be mineralized to ammonium and mobilized at storm flows (Lin et al., 2021). Comparing the number of studies evaluating the effectiveness of GI to control water volume, there are fewer studies examining GI's biogeochemical functions. One on-site example (Carpenter et al., 2016) found a green roof project was able to reduce the total nitrogen (TN) and phosphorous (TP) by 69 ± 56% and 84 ± 30% from roof drainage from wet deposition, and the retention was higher in non-growing season than the growing season. Collins et al. (2009) showed subsurface drainage from permeable concrete grid pavers filled with sands has lower N concentration than the drainage from standard asphalt pavement. At watershed scale, Reisinger, Woytowitz et al. (2019) showed, with the increase of cumulative LID areas, PO_4^{3-} and TP loads decrease at the watershed outlet of Gwynns Falls in Baltimore. Pennino et al. (2016) found a positive relationship between the size of stormwater GI and NO_3^- and TN retention in several watersheds in Baltimore and Washington DC. Bettez and Groffman (2012) evaluated potential denitrification on several GI (e.g., wet ponds, infiltration basin, etc.) facilities and found they have three times the potential denitrification rate than in riparian areas, indicating the potential to implement GI within urban watersheds to reduce nitrate input and to streams. Wetland restoration, a much larger-scale restoration and GI project, could effectively reduce nitrate in water from agricultural or urban watersheds to approach water quality goals (Evenson et al., 2021; Hansen et al., 2018).

However, there are several limitations of current LID/GI studies. Firstly, most of these studies evaluated GI/LID's effectiveness to stormwater or nutrient retention at facilities themselves only. Secondly, many studies used an artificial and topographically uniform drainage area to quantify changes in infiltration, evapotranspiration, and streamflow, but not a real watershed. Next, LID/GI induced watershed-level changes of streamflow and nutrient load export are rarely evaluated empirically or modelled systematically. Finally, few frameworks are available so far to allow spatially explicit planning of LID/GI to systematically evaluate effectiveness or impacts of different design scenarios of LID/GI on directly coupled streamflow and ecological processes in urban watersheds.

1.1.3 Aquatic or stream restoration

Stream channel restoration has been implemented in the United States with the goals of mitigating of series of environmental issues from urban stream syndrome and protecting adjacent infrastructure (Bernhardt et al., 2005). Stream channel restoration typically includes regrading steep banks to more stable angles, removal of exposed sediment, and introducing sinuous channel planform and/or pool and riffle longitudinal profile with greater hydraulic roughness (Yochum & Reynolds, 2018). These changes are expected to reduce sediment and nutrient loads, emphasizing reactive nitrogen and phosphorous (Craig et al., 2008; Reisinger, Doody, et al., 2019). McMillan and Noe (2017) found reconnecting floodplain and stream and increasing the frequency and time of overbank flooding could reduce sediment and nutrients loads. Introducing new vegetation in the restored riparian areas is also a goal of stream restoration that brings green space and associated ecosystem services back to urban streams. New vegetation provides not only recreational opportunities for urban residents but also climate benefits such as mitigation of urban heat islands, and ecological benefits to local and downstream aquatic system by increasing channel roughness, slowing flow velocities, and promoting sediment and nutrient deposition (Noe et al., 2013; Surridge et al., 2012). Unlike many LID practices which take little space to implement, stream restoration is considered a larger-scale project that impacts nearby residents and involves many stakeholders. Restoration projects are quite expensive and its success in achieving nutrient reduction goals appears to vary substantially with contributing watershed conditions, mediated by stormwater and nutrient loading regimes, and the distribution of nutrient load by stream discharge (Groffman et al., 2004; Shields et al., 2008). Therefore, there is a need to identify stream reaches that restoration can produce both environmental and socioeconomic benefits. For example, a stream that has high potential of nutrient retention and support from nearby residents would be ideal to achieve environmental improvement goals and provide riparian green space to nearby residents, and easier for the restoration project to initiate because of the lower risk of conflict of interest among stakeholders.

However, most urban stream restoration studies are restricted to on-site evaluations and inapplicable to other streams to predict the potential sediment and nutrient reductions after stream restoration. The rich urban hydrological and ecological dataset from the Baltimore Ecosystem Study (BES), an urban long-term ecological research (LTER) site, makes spatial extrapolation of stream restoration effectiveness possible. There are several urban stream restoration projects finished in the Baltimore region, and Scotts Level Branch, an urban headwater watershed with much of its streams restored, is an excellent study watershed to evaluate how stream restoration would affect the aquatic nutrient retention (Lin et al., 2021; Reisinger, Doody, et al., 2019).

1.1.4 Modification of nitrogen cycling by human activities

Urban ecosystems experience rapid increase of impervious areas and infrastructures and receive massive nutrient loads from human activities. The additional loads could not only increase in-stream nutrient export which threatens aquatic ecosystem health downstream, but also alter the on-site hydrological and biogeochemical processes (e.g., evapotranspiration, soil moisture, vegetation uptake, denitrification, etc.) which could significantly altered compared to unmanaged watersheds. Understanding how human activities can impair environmental conditions in urban ecosystem lays the foundation to plan restoration practices.

In an urban watershed, additional inputs of water (e.g., lawn irrigation and septic effluent), carbon (e.g., mulch), and nitrogen (e.g., septic system, lawn and garden fertilization, sanitary sewer leakage and combined sewer overflow) are commonly introduced through human activities. In monitored urban watersheds in Baltimore, lawn fertilization could contribute more than half of the total N input (Groffman et al., 2004). In residential areas beyond the urban service boundary, septic N input, though a large input, may be only concentrated on a tiny portion of the landscape (Band et al., 2005). At the watershed scale, septic N input could be comparable to atmospheric deposition, but the concentrated N inputs at only a small portion of landscape by fertilization and septic systems could create N hot spots (McClain et al., 2003) that should be targeted for meeting N mitigation goals (Bernhardt et al., 2017).

To comprehensively understand interactions between human activities and ecosystem responses, both empirical and modeling efforts are required. The empirical datasets would provide sufficient information that quantify the frequency and amount of N inputs at household and watershed level, and the modeling framework could all significant inputs of N load into integrated hydrological and biogeochemical cycling of urban ecosystems. To accomplish this, the modeling framework needs to be spatially explicit to route water and nutrients from spatially distributed inputs through detailed hydrologic flowpaths and simulate ecological processes at high spatial resolution. A gully distributed ecohydrological model, Regional Hydro-Ecological Simulation System (RHESSys, Figure 1.2) meets all the requirements discussed above. It can not only simulate urban ecohydrological processes at the user-defined resolution, but also allows the incorporation of spatially heterogeneous inputs of N and evaluate the corresponding impacts to the upland ecosystem features, and cumulatively to the watershed outlet. Once RHESSys can reasonably simulate the current states of a watershed, it can be further used to evaluate the effectiveness of potential restoration management (e.g., LID) plans to mitigate N loads. In summary, RHESSys has the ability to represent important ecological processes at both human perception and watershed scales and provide insightful information for decision making and community involvement of future restoration planning.

1.2 RESEARCH QUESTIONS

This dissertation evaluates several research questions related to urban ecohydrology in three papers:

1. How does the pattern of potential nitrogen reduction from aquatic restoration spatially align with associated socioeconomic benefits with demographic characteristics along the urban-rural gradient?
    a. What are the spatial distributions of environmental and socioeconomic benefits and their alignments with demographics in Baltimore?
    b. Are water quality gains and socioeconomic benefits across Baltimore synchronous?
    c. What are the implications for future aquatic restoration siting?
2. How can certain types of terrestrial restoration practices (i.e., LID/GI) help mitigate urban stream syndrome?
    a. How do LID/GI practices produce "on-site" and "off-site" effects on hydrologic and biogeochemical processes, including biogeochemically critical offsite regions such as downstream restored stream and riparian areas?

b. What is the effectiveness of LID/GI restoration practices on mitigating urban runoff and nitrate export regimes towards pre-urbanization conditions and increasing nitrogen retention from stream restoration?

3. How do human-induced N loads contribute to the N cycling of urban ecosystem?
   a. What are the individual and interacting contributions of different watershed nitrogen sources to streamwater nitrogen export?
   b. How do the spatially nested patterns of water and nitrogen inputs from human activities alter spatial patterns of a set of key ecohydrological processes (e.g., nitrogen retention, evapotranspiration, soil and groundwater levels and flows)?
   c. What are the emergent patterns of nitrogen cycling and retention, including "hot spots" and "control points" at sites receiving direct additional N and downslope, offsite locations receiving transported N?

1.3 STUDY AREA: BALTIMORE, MD

Baltimore, MD, is an excellent study area to understand and address urban ecohydrological challenges and the above research questions. Baltimore has experienced negative effects from impervious development, infrastructure decay, climate change, and pronounced inequity in impact on different demographic neighborhoods. The region has recorded several catastrophic flash floods in recent years, and the City of Baltimore and Baltimore County are seeking to reduce stormwater, flooding, and nutrient loadings to the Chesapeake Bay and improve local water quality. There are three major watersheds (Figure 1.3) Gwynns Falls, Jones Falls, and Herring Run, spanning from the urban Baltimore City to its surrounding Baltimore County. Based on the latest 2020 census data, Baltimore city has population of 585,708 (2789 per km^2) with the median household income \$54,124. Baltimore County has population of 854,535 (551 per km^2) with the median household income \$81,846. Within Baltimore city, 20.3% of population live in poverty, which is above the national average of 11.6%. Baltimore county's poverty rate, 9.8%, is significantly lower than the city's rate and the national average rate.

The Baltimore Ecosystem Study (BES) has been monitoring the water chemistry of eight watersheds (Figure 1.3) in Baltimore over a rural-urban gradient for more than twenty years. Most sites locate within Gwynns Falls, and two reference watersheds with minimal human impacts and sufficient forest coverage, Baisman Run and Pond Branch locating in the Gunpowder watershed north of Baltimore. With detailed streamflow records collected by United States Geological Survey (USGS) and high-resolution land cover and use data from the Chesapeake Bay Conservancy, we are able to explore the wide-range urban ecohydrological research questions related to water quantity, land cover change, water pollutant loadings, and socioeconomic impacts of environmental changes to local residents.

1.4 GOALS

This dissertation is designed to establish a framework for various stakeholders to systematically quantify 1) impacts of human activities to water balance and nitrogen cycling of urban watersheds and 2) project changes in water and ecosystem functions with spatially explicit user-defined terrestrial and aquatic restoration scenarios in urban watersheds. With expanding data collection in water quantity and quality in urban hydrology, there are opportunities to better understand the interacting biophysical processes of complicated human-dominated ecosystems and improve process-based models to meet increasing demands for high-accuracy predictions of ecohydrological response to management activities. Specifically, scientists want to quantify how ecosystem restoration practices would affect stream runoff regime, quality and other ecological processes, sociologists want to understand what benefits urban ecosystem restoration could bring to residents nearby, and decision-makers need detailed and accurate predictions to evaluate what restoration scenarios are required to meet environmental regulations from local through national government. Spatially explicit process-based models provide fine-resolution evaluations of hydrologic and ecological processes in urban ecosystems, especially ones that are difficult to measure (e.g., evapotranspiration, water table depth, denitrification, plant nitrogen uptake, etc.). This dissertation contributes to the development of easy-to-deploy frameworks simulating patch through watershed-level ecohydrological processes to meet various needs from scientific, urban planning, and socioeconomic communities and evaluate multidisciplinary questions between human and ecosystem.

In summary, the dissertation contains three research chapters (Chapter 2 to 4), with introduction (Chapter 1) and conclusion (Chapter 5) at the beginning and end. The central theme of the research chapters is to understand interactions between human and watershed ecosystem processes and evaluate effectiveness of several types of LID/GI to mitigate stormwater, improve water quality, and provide socioeconomic benefits. Specifically, **Chapter 2** evaluates the spatial asynchrony of nitrate reduction and socioeconomic benefits to nearby residents after projected stream restoration in Baltimore metropolitan region. **Chapter 3** assesses the quantities of nitrate load reduction and flood mitigation after implementing LID/GI restorations (i.e., urban reforestation, road-side bioretention swales, and permeable road) in Scotts Level Branch watershed, and its effectiveness to improve nitrate retention for stream restoration; **Chapter 4** explores the impact of individual and interacting human mediated N sources (i.e., fertilization, septic, and irrigation) on local to watershed scale ecohydrological processes and nitrogen retention in an exurban watershed. There are several counter intuitive findings that are crucial for future planning of ecosystem restoration practices, summarized in the conclusion chapter (**Chapter 5**) and especially in section 5.1.5.

1.5 FIGURES AND TABLES

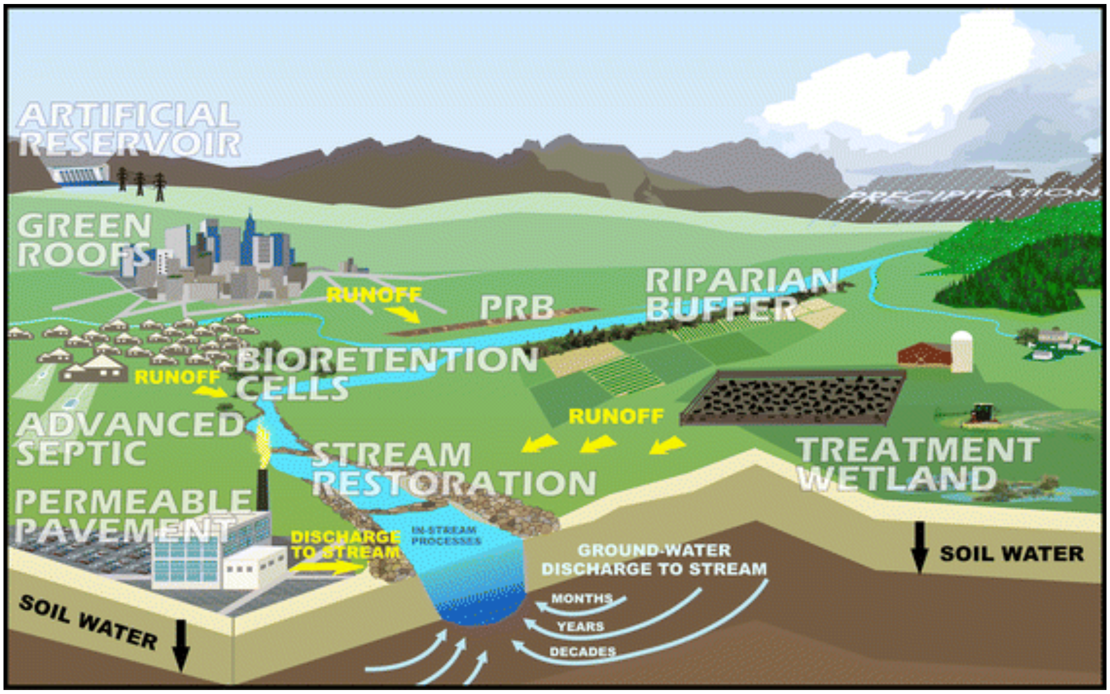

Figure 1.1. Types of green infrastructures and ecosystem restoration methods within watershed (Passeport et al., 2013)

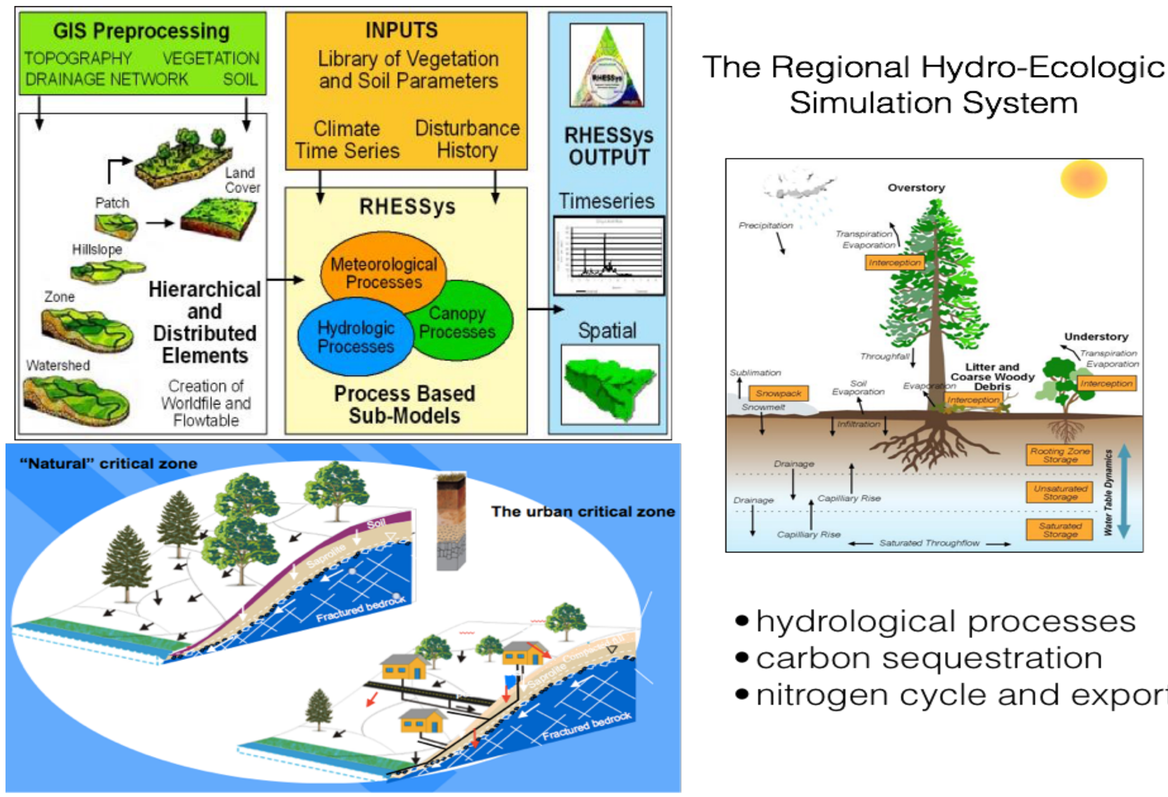

Figure 1.2. Structure of Regional Hydro-Ecological Simulation System (RHESSys) model (Tague and Band, 2004). This spatially explicit model simulates water balance and biogeochemical processes at patch scale and route the flow from patch to hillslope to the whole watershed

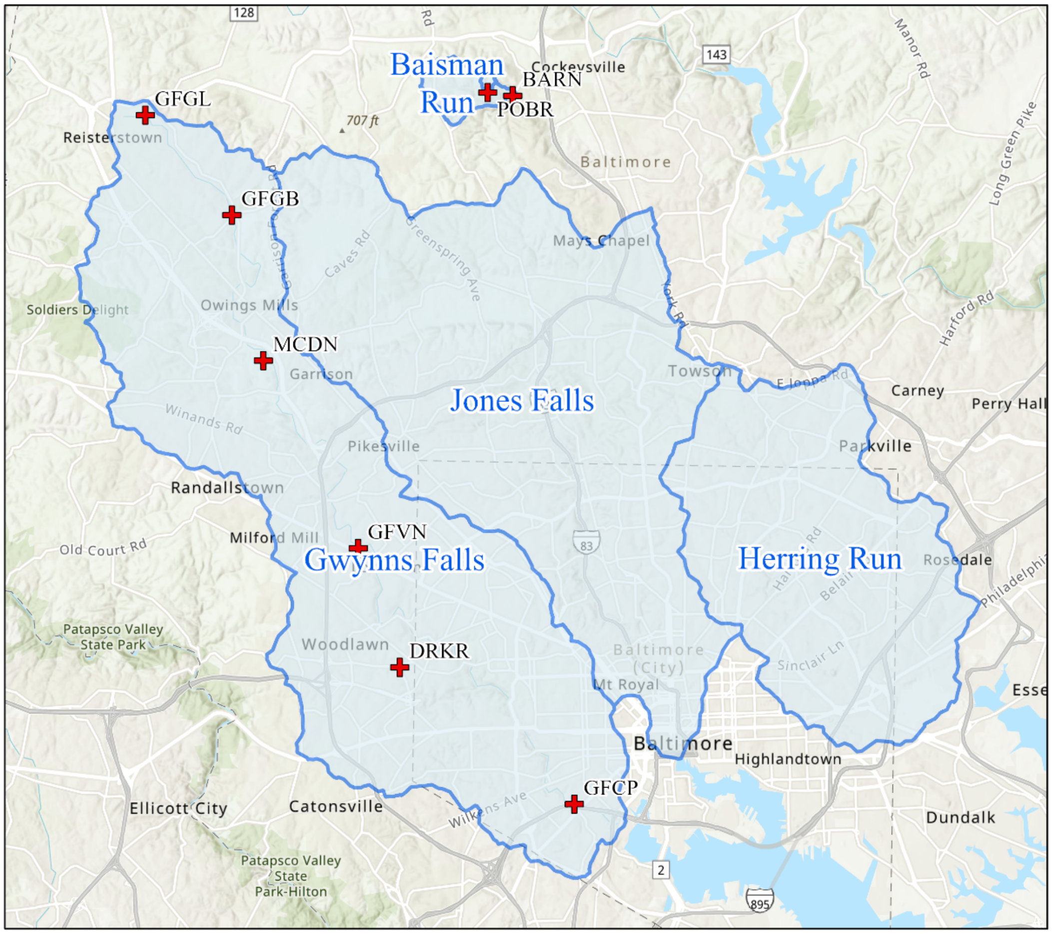

Figure 1.3. Three main watersheds (blue polygons) in Baltimore metro areas and long-term monitored urban watersheds (red cross) by Baltimore Ecosystem Study (BES) since 1998, where discharge and weekly water chemistries are recorded continuously

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Chapter 2: SPATIAL ASYNCHRONY IN ENVIRONMENTAL AND ECONOMIC BENEFITS OF STREAM

2.1 PREFACE

This chapter was authored by myself, Drs. David Newburn (University of Maryland, College Park), Andrew Rosenberg (Economic Research Service, USDA), Laurence Lin (University of Virginia), Peter Groffman (Cary Institute of Ecosystem Research), Jonathan Duncan (Penn State), and my advisor Larry Band (University of Virginia). The assessment of effectiveness of stream restoration in nitrate reduction was based on Laurence's statistical model and Peter's rigorous data collection in Baltimore; The economic model and household level WTP were modeled by Andrew and David; Larry, Jon, and I evaluated synthesized individual components. Together, this chapter was published in Environmental Research Letters in 2022.

2.2 ABSTRACT

Stream restoration is widely used to mitigate the degradation of urban stream channels, protect infrastructure, and reduce sediment and nutrient loadings to receiving waterbodies. Stabilizing and revegetating riparian areas can also provide recreational opportunities and amenities, and improve quality of life for nearby residents. In this project, we developed indices of an environmental benefit (potential nitrate load reduction, a priority in the Chesapeake Bay watershed) and economic benefit (household willingness to pay, WTP) of stream restoration for all low order stream reaches in three main watersheds in the Baltimore metro region. We found spatial asynchrony of these benefits such that their spatial patterns were negatively correlated. Stream restoration in denser

urban, less wealthy neighborhoods have high WTP, but low potential nitrate load reduction, while suburban and exurban, wealthy neighborhoods have the reverse trend. The spatial asynchrony raises challenges for decision makers to balance economic efficiency, social equity, and specific environmental goals of stream restoration programs.

2.3 INTRODUCTION

Rapid urbanization over the last century introduced vast conversion of forest and agricultural land, increasing impervious cover, stormwater runoff and pollutants (Booth and Jackson, 1997, Booth et al., 2002, Walsh et al., 2005a). Increased runoff volume and peaks erode stream beds resulting in incised, steep bank channels, high sediment and nutrient delivery, and reduced ecosystem function, characteristics of the urban stream syndrome (Walsh et al., 2005b). Stream restoration has been implemented as a common strategy with the goals of mitigating these effects and protecting adjacent infrastructure (Bernhardt et al., 2005). However, stream restoration is an active change in resident's local environment, and watershed managers have found varying degrees of support or opposition to stream restoration from local communities (e.g., Wheeler, 2020). Channel restoration typically includes regrading steep banks to more stable angles, removal of exposed sediment, and introducing sinuous channel planform and/or pool and riffle longitudinal profile with greater hydraulic roughness (Yochum and Reynolds, 2020). In the process, existing vegetation, including trees, are removed when present, but may be replanted on completion of the restoration project. These changes are expected to reduce nutrient loads, emphasizing reactive nitrogen and phosphorous (Craig et al., 2008, Reisinger et al., 2019a), but also may have direct impacts or be perceived as improving or degrading local residents' environment.

Restoration success in achieving nutrient reduction goals appears to vary substantially with contributing watershed conditions, mediated by stormwater and nutrient loading regimes, (Groffman et al., 2004; Shields et al., 2008), and the benefits and support of local residents may also vary based on pre-restoration environmental and community characteristics. Many studies have investigated sediment and nutrient load reductions and ecological effects on aquatic and riparian ecosystems (Sudduth and Meyer, 2006; Alexander and Allan, 2007; Tullos et al., 2009; Filoso and Palmer, 2011; Pennino et al., 2016a; McMillan and Noe, 2017), or have addressed economic aspects of stream restoration (Johnston et al., 2005; Kenney et al., 2012; Jarrad et al., 2018). However, the spatial alignment of water quality and economic benefits to nearby residents at the watershed level has not been rigorously evaluated.

To understand the joint benefits of stream restoration, we leverage long-term sampling of watershed, stream form and fluxes, land use, and socioeconomic household survey data for the Baltimore Ecosystem Study (BES) (Pickett et al., 2020). The BES, an urban Long Term Ecological Research project, has been measuring streamflow and nutrient loads since 1998 in a set of catchments ranging from fully forested or agricultural, through highly developed land use, providing one of the most comprehensive urban stream datasets in the world. Long-term ecohydrological data and recently developed terrestrial and aquatic ecosystem models facilitate the estimation and prediction of nitrate export regimes (Groffman et al., 2004; Shields et al., 2008; Lin et al., 2021) across the urban-rural gradient, and nitrate cycling and load reduction from stream restoration. Household willingness to pay (WTP) for ecosystem restoration provides a nonmarket valuation approach to assess the economic benefits and a potential metric for community support. Detailed sampling and analysis of stream restoration WTP provides the basis to estimate patterns of economic benefits and neighborhood/community support for restoration interventions across the watersheds.

In this study, we investigated the potential of stream restoration to jointly achieve the ecosystem goals of reducing reactive nitrogen export to the Chesapeake Bay and provide economic benefits to local residents. We focused on nitrate as long-term data has shown it to be the dominant form of dissolved inorganic nitrogen in our sampled streams, and nutrient load reduction in these watersheds is mandated under the Chesapeake Bay total maximum daily load (TMDL) requirements to improve ecosystem health (EPA, 2010; Hagy et al., 2004; Li et al., 2016). We conducted a study in three watersheds (Figure 2.1a), resolved at stream-reach-level, extending from Baltimore County through the City of Baltimore, to comprehensively evaluate:

1) spatial patterns of potential nitrate load reduction expected from a reference restoration design on all low order stream reaches, conditional on reach specific runoff and nutrient loading regimes and canopy cover,

2) spatial patterns of local residents' WTP for stream restoration in their neighborhoods, conditional on potential nitrate load reduction and canopy cover, and

3) the spatial alignment, or synchrony, between these ecosystem and economic benefits of potential restoration in low order stream reaches over the rural to urban gradient.

2.4 METHODS

2.4.1 Economic WTP analysis of stream restoration scenarios

Household WTP for stream restoration was estimated based on a stated preference approach using data from a household survey, conducted in the fall of 2017 in the Baltimore metro region. See Rosenberg et al. (2018) for further details on the survey data, restoration design attributes, and model estimation methods. The survey sample, drawn randomly from the complete tax assessor database from the Maryland Department of Planning, included 11,000 households in the Baltimore metropolitan region, including owner-occupied townhomes and single-family households on parcels less than five acres in size. There were 1,011 survey respondents and 3,980 choice questions (9.2% response rate), which is a similar response rate to other household surveys (Kenney et al., 2012, Cadavid and Ando, 2013, Newburn and Alberini, 2016) on urban stormwater BMPs. Survey respondents were presented with four different choice experiment questions where they were asked to choose between the status quo for a 1000-foot segment of a degraded local stream and two restoration options with different design attributes and costs. Restoration design attributes varied based on the riparian vegetation type, streambank stabilization approach, nutrient pollution reduction in local streams and the Chesapeake Bay, and cost to the household. Nutrient pollution reduction was valued based on a metric capturing the extent to which a stream restoration was able to meet regional goals for satisfying watershed implementation plans and TMDL requirements. Respondents were told that stream restorations in choice questions addressed a specified percentage of the annual reduction goal for the 11-digit watershed where the stream restoration was located. Restoration location in choice experiment scenarios also varied based on the distance from the household and whether the restoration occurred on public or private land. Distances included streams that were located within 1 mile of the household (walking distance) or located farther than 1 mile from the household (driving distance) but still within our study region for Baltimore County and City. The local neighborhood is defined as a stream restoration located within 1 mile from the household, which may be compared to a stream restoration farther than 1 mile from the household but within the study region.

Using the survey responses, household willingness to pay (WTP) is estimated using discrete choice modeling and best practices for stated preference analysis (Hanemann 1984; Johnston et al., 2017). A weighted conditional logit model is used to account for non-response bias based on observable characteristics of survey sample respondents and non-respondents (see Table 2.2). Household WTP for restoration scenarios depends on restoration attributes, land ownership of a stream location, distance from the household to the restored stream, in addition to several household and neighborhood-level variables. Household-level variables include parcel-level data from the complete tax assessor data on housing and property characteristics. Neighborhood-level variables include surrounding land use in the vicinity of each household (using the 1-meter resolution Chesapeake Bay Program Land Cover Data, https://www.chesapeakeconservancy.org) and neighborhood demographic characteristics from the American Community Survey at the census tract level. Principle component analysis (PCA) was used to create two factors, based on the large set of household and neighborhood-level variables, that can be interpreted intuitively in the post-PCA analysis as related to household wealth and how rural is the neighborhood. Household and neighborhood-level characteristics are incorporated into the discrete choice models using the two PCA factors interacted with the restoration design attributes, distance from restoration project to household, and land ownership.

Household- and neighborhood-level variables are available for each household in Baltimore, and estimation results are used to predict WTP for each household in the study region. All reported WTP values are based on local premiums, which measure the additional amount a household is willing to pay for a restoration project scenario that is within 1 mile distance of the household, relative to the baseline for the same project located in the study region but farther than 1 mile from the household. In this study, the econometric models from Rosenberg et al. (2018) are adapted to estimate household WTP for restoring degraded streams with a riparian area with two land cover types, forest and grass, and boulders to stabilize channel banks designs (FB and GB) for each 1000-foot stream reach in the study area. The WTP estimates are allowed to vary based on household-level wealth and rural PCA factors, percent of the restored stream segment going through public land, and predicted impact of each stream restoration on potential nitrate reduction. Both the FB and GB designs result in closed- or open-canopy stream channel conditions yielding different projected nitrate reduction amounts, respectively. Both scenarios can potentially provide various levels of recreational and visual amenity opportunities for local residents, in addition to stabilizing streambanks to improve environmental outcomes and protect public infrastructure such as sewer lines.

2.4.2 Determination of potential nitrate reduction in restored streams

We developed estimates of potential nitrate reduction in low order, restored streams throughout the three-watershed region by extrapolating a process-based aquatic ecosystem model (Lin et al., 2021). The aquatic ecosystem model leveraged long-term weekly stream chemistry from 1998 to 2021 in eight watersheds at the BES, and measurements of nitrogen spiraling, gross primary productivity (GPP) and ecosystem respiration (ER) in several restored and unrestored stream reaches in the area (Reisinger et al., 2019a, Reisinger et al., 2019b). Lin et al.'s (2021) model simulates nitrogen inputs, transformations, transport, and storage in a paired restored/unrestored channel sequence in a suburban headwater stream, Scotts Level Branch, one of the sites for which Reisinger et al. (2019a) estimated stream spiraling metrics for nitrate, GPP and ER in the Gwynns Falls watershed. The advantage of the model is its ability to extrapolate these variables from low to medium flow conditions under which field measurements are typically made, over the full flow regime including high flows. The model is sensitive to channel geomorphology, hydraulics, canopy cover, and upstream and lateral water and nutrient inflows.

We used BES measured streamflow and chemistry time series in our long-term watershed sites as input to Lin's model to estimate quantities of potential annual nitrate reduction resulting from stream restoration over the BES urban-rural gradient, with two possible scenarios, open (GB) and closed (FB) canopy, for post-restoration conditions. The potential nitrate load reduction quantifies is the difference before and after restoration. As it is not feasible to forecast the precise form of channel morphology and restoration design for all stream reaches, we used the channel dimensions and restoration design in the Scotts Level Branch study reaches with the distinct watershed runoff and nutrient loading regimes over the rural to urban gradient derived from six low order BES monitored catchments. The restoration method at this site included pool/riffle construction with stone and boulder placement, and reforested and re-grassed slopes, consistent with the FB and GB designs. We therefore interpret the potential nitrate load reduction as an index to characterize the watershed scale trend, rather than a site specific quantification, of nitrate reduction, acknowledging that locally adapted methods may yield different reduction levels.

The efficacy of nitrate reduction declines with increasing flow levels. We estimated the cumulative nitrate load by streamflow rate, adapting methods from Shields et al. (2008) which used the streamflow rate corresponding to the 75th percentile of the nitrate load (F75, mm/day) as a metric to characterize the nitrate load-streamflow distribution. Shields et al (2008) showed the F75 ($= 2.01e^{0.06\,IMP}$, $R^2 = 0.89$) was well correlated to percent upstream impervious area (IMP), determined from the National Land Cover Dataset (NLCD). We updated the geospatial analysis with NLCD 2011 impervious surfaces product and mapped the % upstream impervious cover for each stream reach. We simulated the restored and unrestored Scotts Level Branch models with the runoff and nutrient regimes determined for the long-term BES monitoring sites, and built regression models for the simulated nitrate uptake against the F75 for hypothetical open and closed canopy conditions and used this relation to assign potential uptake rates for each reach. The difference of potential nitrate reduction between open and closed canopy conditions is due to differences in simulated photosynthetically active radiation reaching the stream surface.

2.4.3 Spatial aggregation of estimated WTP and census data

To build the stream reach network in the three watersheds, we delineated the watershed drainage system from a 10-meter digital elevation model data downloaded from USGS (https://earthexplorer.usgs.gov/) using the D-infinity method (Tarboton, 1997). We chose a threshold drainage area to define first order streams to approximate the National Hydrography Dataset High Resolution (https://www.usgs.gov/core-science-systems/ngp/national-hydrography/nhdplus-high-resolution) headwaters and excluded analysis of high-order streams (i.e., Strahler order > 3) as measured and modeled nitrate reduction after stream restoration was conducted on low-order streams only. We split each reach between stream junctions into approximately 1000-ft segments to match the length of restoration projects uniformly specified in the household survey. Many streams within downtown Baltimore City are in stormwater pipes that are not suitable for restoration and thus were removed from our analysis referencing the USGS NHD High-Resolution streamline. After removing buried stream segments (largely in downtown Baltimore City), there were 1,512 low-order stream reaches in our study watersheds.

We calculated the estimated WTP for all households within the 1-mile radius of each stream reach midpoint representing the local neighborhood. The proportion of public and private land for each restoration site was derived from the land ownership within a 100-ft buffer zone from each stream segment. To compare how the WTP varies with socioeconomic status, area-weighted averaged metrics for median household income and population were estimated from block-group level U.S. Census Data in 2018 for each neighborhood within the 1-mile radius. Similarly, total neighborhood WTP was calculated as the sum of household WTP for all households within the neighborhood, which is interpreted as the additional economic value the restoration scenario provides when done in the local neighborhood compared to restoring a stream farther away within the study region on private land. We used ordinary least-squared linear regression to assess the existence of a positive or negative trend between nitrate reduction and WTP.

2.5 RESULTS

2.5.1 Demographic distribution in stream reach neighborhoods

Population density and median household income for stream reach neighborhoods (one mile radius) were classified in quintiles (Figure 2.1b & 2.1c). Higher median income neighborhoods at lower density development were clustered in the north of Baltimore City and the adjacent region of Baltimore County, with 87% (264/303) of the wealthiest neighborhoods in Jones Falls watershed. Lower income neighborhoods at higher density were located in the west and east of Baltimore City, with 47% (143/303) of the least wealthy neighborhoods within the city. Additional lower income, denser neighborhoods extend into Baltimore County in development corridors. In summary, the neighborhoods sampled in our study stream reach zones closest to downtown Baltimore City have denser population with lower median household income, while lower density neighborhoods in the suburban and exurban areas are generally wealthier. Note that higher income neighborhoods around Baltimore Harbor are often excluded as all streams are buried in these most urbanized areas.

2.5.2 Spatial variation of nitrogen reduction

We extrapolate annual potential nitrate load reduction in units of mass per unit length of stream (g/m) at the reach level (Figure 2.2a) for restored conditions compared to unrestored conditions using the relationships in Figure 2.2b. In the three study watersheds, many streams in the upper Jones Falls have a higher percentage of annual nitrate export occurring at or below low flows (low F75) and therefore higher potential for stream restoration to reduce nitrate load. About 70% (422/605) of streams in the highest two quintiles of nitrate reduction are found in the low-density upper Jones Falls. A few streams in the upper Gwynns Falls also have high potential nitrate load reduction after stream restoration. In contrast, streams in the lower Gwynns Falls and Back River have a much higher proportion of their total nitrate export during high flows (high F75) and therefore have low potential nitrate load reduction after stream restoration. Over 40% (122/303) of streams in the lowest quintile of nitrate reduction are found in higher density neighborhoods within the city boundary. Therefore, potential nitrate reduction exhibits a rural-urban gradient (Table 2.1), where streams in the low-density upper Jones Falls are more favorable for nitrate reduction goals while much less reduction is expected from stream restoration in denser urban areas.

2.5.3 Spatial pattern of stream restoration WTP

After normalizing for the number of households in each 1-mile radius and controlling for the wide variation of population density in our study watersheds, the spatial pattern of average household WTP for both FB (Figure 2.3a) and GB (Figure 2.3b) scenarios suggests that households in higher density urban areas within and near the city boundary have the highest household WTP, while households in exurban areas have the lowest average household WTP. Average household WTP for FB and GB scenarios was \$39.47 and \$9.80, respectively. For the 442 neighborhoods within Baltimore City predominantly at higher density, the average household WTP for the FB and GB scenarios was \$83.16 and \$25.49 (Figure 2.3a and 2.3b). Outside of Baltimore City, the average household WTP was \$21.48 for the FB scenario and \$-9.91 for the GB scenario. Exurban neighborhoods in the upper Jones Falls outside the urban growth boundary (UGB) had the lowest average household WTP for both restoration scenarios (Figure 2.3a and 2.3b).

Overall, the average WTP for each scenario of stream restoration is high in urban neighborhoods and low in suburban and exurban neighborhoods.

At the total neighborhood level, several neighborhoods with the highest total WTP for the FB scenario are found along Herring Run in the Back River watershed (Figure 2.3c), where the population density is high (Figure 2.1b). The lowest WTP for the FB scenario are found in the upper Jones Falls within low density exurban neighborhoods outside the UGB, with a mean total WTP of $-22,012 which is significantly lower than the mean total WTP of $192,120 for all neighborhoods in our study watersheds. However, these are the neighborhoods where stream restoration has the greatest potential (Figure 2.2a) to reduce nitrate loads. The neighborhood total WTP for the GB scenario (Figure 2.3d), with open canopy potential nitrate reduction, has a similar spatial pattern to the WTP for the FB scenario ($r = 0.95$, $p < 0.05$) but with lower total WTP values (mean of $112,068 for the GB vs. $192,120 for the FB). High density urban neighborhoods in the middle of Back River with the highest total WTP for the FB scenario also have the highest total WTP for the GB scenario, whereas exurban neighborhoods in the upper Jones Falls outside the UGB had the lowest total WTP for both the FB and GB scenarios.

2.5.4 Asynchrony between WTP and nitrate reduction

We show the potential nitrate reduction versus average household WTP and neighborhood total WTP for both FB and GB scenarios in Figure 2.4. For both FB (closed canopy) and GB (open canopy) scenarios, linear regression shows a negative correlation between potential nitrate reduction and average household (Figure 2.4a & 2.4c, with R^2 values 0.57 & 0.52) or neighborhood total WTP (Figure 2.4b & 2.4d, with R^2 values 0.33 & 0.29). Most stream reaches in low density exurban neighborhoods had negative average household WTP for stream restoration and high nitrate reduction (above median) after stream restoration (top left quadrant in Figure 2.4a & 2.4c). In contrast, most stream reaches in high density urban neighborhoods had positive average household WTP and low (below median) expected nitrate reduction following restoration (bottom right quadrant in Figure 2.4a & 2.4c). The lower income neighborhoods in denser urban areas have the highest average household WTP and total neighborhood WTP for community support, but these high density areas have low nitrate reduction (Figures 2.4). We found 71% and 75% of neighborhoods fall in the two asynchronous quadrants (top left and bottom right) for FB and GB scenarios, suggesting asynchrony between environmental and economic benefits of stream restoration in Baltimore. The difference of average household WTP for the GB scenario increases compared to the FB scenario, particularly for households in low density neighborhoods that have lower WTP due in part to tree removal disturbance for the GB scenario.

At the total neighborhood level, several neighborhoods with the highest total WTP for the FB scenario are found along Herring Run in the Back River watershed (Figure 2.3c), where the population density is high (Figure 2.1b). The lowest WTP for the FB scenario are found in the upper Jones Falls within low density exurban neighborhoods outside the UGB, with a mean total WTP of \$-22,012 which is significantly lower than the mean total WTP of \$192,120 for all neighborhoods in our study watersheds. However, these are the neighborhoods where stream restoration has the greatest potential (Figure 2.2a) to reduce nitrate loads. The neighborhood total WTP for the GB scenario (Figure 2.3d), with open canopy potential nitrate reduction, has a similar spatial pattern to the WTP for the FB scenario ($r = 0.95$, $p < 0.05$) but with lower total WTP values (mean of \$112,068 for the

GB vs. \$192,120 for the FB). High density urban neighborhoods in the middle of Back River with the highest total WTP for the FB scenario also have the highest total WTP for the GB scenario, whereas exurban neighborhoods in the upper Jones Falls outside the UGB had the lowest total WTP for both the FB and GB scenarios.

2.6 DISCUSSION AND CONCLUSIONS

Stream restoration has been widely adopted as a method to improve water quality, reduce erosion, provide green amenities and recreational opportunities for residents, and increase ecosystem resilience (Kauffman et al., 1997; Reisinger et al., 2017). Many environmental studies have evaluated the effectiveness of stream restoration in terms of nitrate and sediment load reduction and ecological improvement (Sudduth and Meyer, 2006; Alexandra and Allan, 2007; Tullos et al., 2009; Filoso and Palmer, 2011; Pennino et al., 2016a; McMillan and Noe, 2017). Socioeconomic studies have also assessed residents' awareness and attitudes to stream restoration project (Schwarzmann, 2013; Sarvilinna et al., 2018; Hong and Chang, 2020), and how the siting of future restoration projects could address social justice issues in urban areas (Hoover et al., 2021). To our knowledge, our study is the first to estimate the spatial correlation between potential nitrate reduction and economic benefit patterns from household-level WTP for neighborhood support, and to assess distributions of these benefits according to income and population density. We use the Baltimore metro region to investigate the asynchrony of ecosystem restoration biophysical and economic benefits as the rich data and modeling resources developed by the BES over the past twenty years provide an unparalleled base for this integration. While the results for the estimated F75 and nitrate reduction may be specific to our study region, we expect the general negative correlation of trends and our integrated modeling framework (Appendix 2) may be applicable to other regions where long-term environmental data and socioeconomic household survey data are available.

We find exurban neighborhoods often have negative average WTP (Figure 2.3), suggesting that stream restoration may create disturbance in the local neighborhood that provides lower value than when the restoration occurs farther away in the study region. In addition, the GB scenario has the most negative WTP for the exurban neighborhoods, particularly since the GB scenario has tree removal to create grass buffers during restoration, and tree removal is a local disamenity that is viewed particularly negative in the exurban neighborhoods with high to middle income quintiles in the upper Jones Falls. Our results also suggest a strong spatial asynchrony in the specific indices of ecosystem and economic benefits of stream restoration we quantify. Restoration in low density exurban neighborhoods would contribute significantly towards achieving the nutrient reduction goals in Baltimore County as part of the Chesapeake Bay TMDL regulations, but have lower neighborhood total WTP for both restoration scenarios. In contrast, restoration in denser urban neighborhoods would not contribute as much to reduce nitrate loading but would provide higher economic benefits that current residents are lacking, such as recreational access and environmental amenities, particularly in dense, lower income communities in Baltimore City. The asynchrony of potential nitrate reduction and economic benefits suggests decision-makers need to balance reduction of nutrient loads, economic benefits for local neighborhoods, and social equity factors for stream restoration programs. Between our two restoration scenarios, the FB design is generally more preferred than the GB design, indicating that tree coverage is an important factor that local residents consider to improve their nearby living environment particularly in lower income urban neighborhoods. Though restoration in GB (open canopy) design yields higher nitrate reduction than FB design (Figure 2.2b), we note that much of the nitrate reduction in GB restoration is due to temporary nitrate cycling into organic form, which may recycle back to reactive forms further downstream.

Future research of stream restoration should include and evaluate additional benefits beyond nitrate reduction. For example, the FB restoration in urban areas after restoration can bring other environmental benefits, such as mitigating urban heat islands, reducing sediment load, restoring riparian habitat, and other co-benefits. In addition, we note that there exists substantial variation for the model predictions and neighborhoods demographics in Figure 2.4. Though the linear regression quantified the statistically significant negative trend between nitrate reduction and economic WTP, it cannot fully explain the variation due to the complex spatial distribution of population, income, and other factors in Baltimore. The Baltimore metro region is a complex region with substantial heterogeneity in population density and income as well as other aspects, such as land uses, housing values, employment, and household attitudes and preferences for green infrastructure. While more complex statistical methods exist that can better fit the variation, the main finding would remain that a negative relationship exists between nitrate reduction and economic benefits for average household and neighborhood total WTP for both scenarios.

From our analysis, neighborhoods of different population density and income in the Baltimore region differentially value the economic benefits of stream restoration. Exurban neighborhoods (e.g., upper Jones Falls) appear to have low WTP to support restoring riparian areas of local stream reaches, with lower preference for the GB scenario.

Conversely, urban low-income neighborhoods having less access to green amenities and recreational areas show the highest WTP for both FB and GB designs for their local stream reaches. This asynchrony complicates efforts to find and restore streams that maximize both economic benefits to nearby residents and improve water quality in local streams and the Chesapeake Bay. The asynchrony also raises challenges for decision-makers to balance the provision of green spaces to "green-deficit" communities while also reducing in-stream nutrient pollution in the future siting of stream restoration projects. Nonetheless, some neighborhoods have both high average household WTP and high nitrate reduction after stream restoration (top right quadrant of Figure 2.4a and 2.4c), typically suburban neighborhoods with moderate population density rather than those at the extremes. Stream restoration in these neighborhoods may be prioritized for more detailed analysis and consideration for the joint goals of nutrient reduction and economic benefits for community support. We note that different weighting of TMDL goals and economic benefits would result in different restoration prioritization.

Our study found a general spatial asynchrony of nitrate reduction and economic WTP for community support. We note that these results follow from the current segregation of green space and impervious area with population density and income class. Investment in increased tree cover and green infrastructure benefit residents in denser urban areas directly, which would potentially shift nutrient flows to lower discharge levels (lower F75, Figure 2.2b), reduce total stormwater nutrient loads (Kaushal et al., 2008, Pennino et al., 2016) and improve aquatic ecosystem uptake and stream restoration efficiency. This study indicates urban environmental restoration programs need to evaluate systematic frameworks balancing upland and in-stream restoration to benefit both ecosystem and economic efficiency goals, while also addressing systemic inequity and environmental justice issues.

2.7 FIGURES AND TABLES

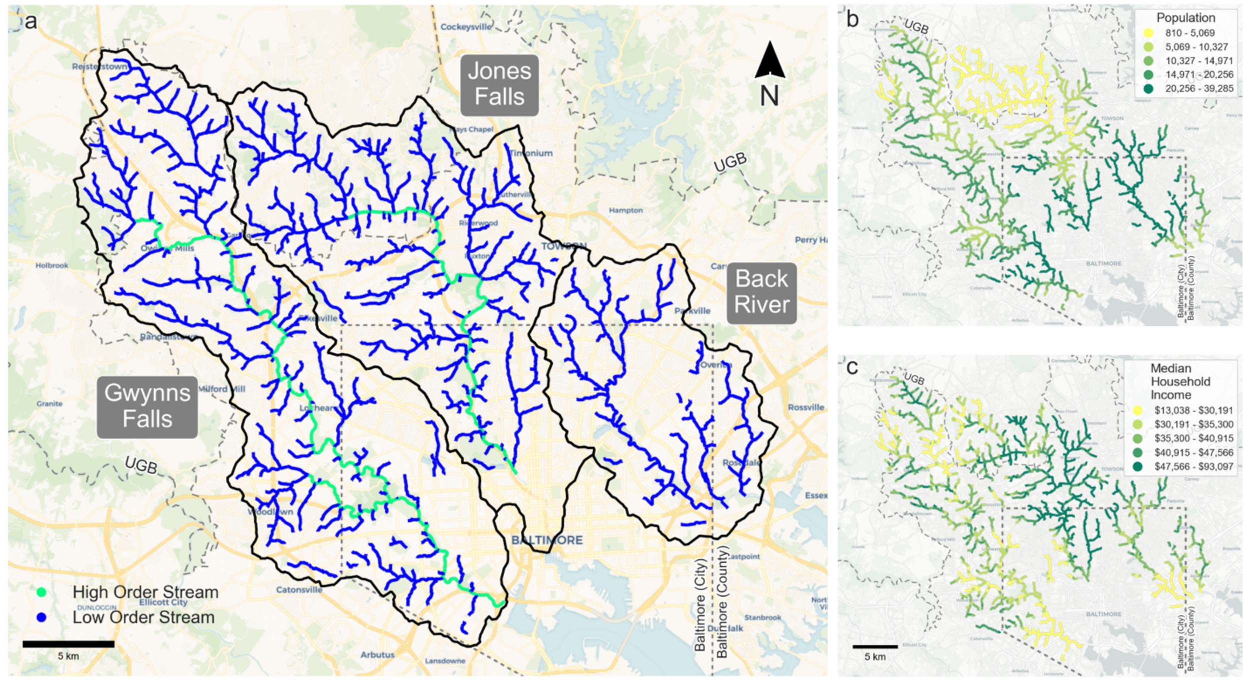

Figure 2.1. Three study watersheds (black solid lines) and area-averaged demographic information for each stream reach neighborhood in Baltimore. MD. (a) Extracted stream network in three watersheds (with buried streams removed, see Methods 2.3), (b) total population, and (c) median household income at 1-mile radius of each stream reach. The urban growth boundary (UGB) and city boundary is shown in grey dash lines

Table 2.1. Two-side t-test of projected nitrate reduction (gN/m/year) at aggregate community level in population density quintiles

Population Quintile	Mean	Standard Deviation	T-Test Scores for Population Quintile 2	3	4	5
1	114.96	8.79	20.5**	36.4**	40.4**	51.2**
2	89.83	19.40		9.2**	-10.6**	14.2**
3	76.38	16.22			1.2	4.4**
4	74.90	14.83				3.4**
5	71.20	12.00				

Note: p-value * (<0.05), ** (<0.001)

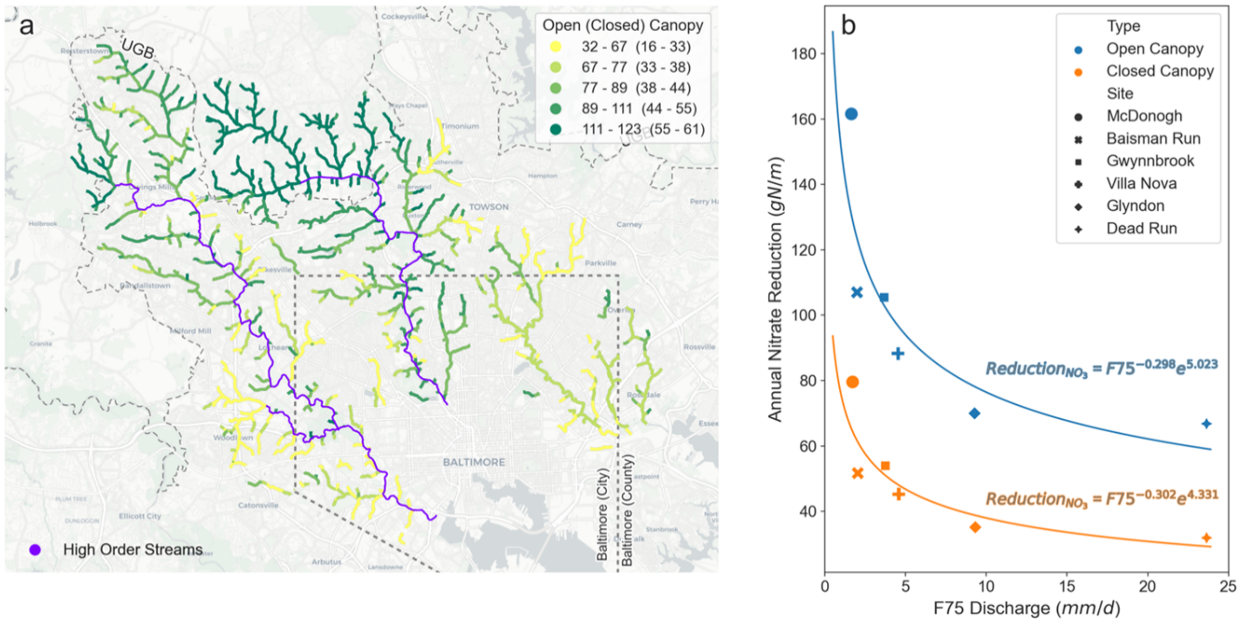

Figure 2.2. (a) Estimated stream reach potential nitrate reduction (g N/m) under closed (open) canopy scenario of stream restoration; (b) Relationship between annual nitrate reduction and F75 discharge (see Methods) under open (blue line, reduction = $F75^{-0.2982}e^{5.0230}$, $R^2 = 0.767$) and closed (orange line, reduction = $F75^{-0.3015}e^{4.3306}$, $R^2 = 0.768$) canopy

Table 2.2. Estimated household WTP from Weighted Conditional Model

Variables	Coefficient	Std error
Grass	-0.425**	(0.159)
Forest	-0.046	(0.144)
Boulders	1.679**	(0.160)
Wetland	1.456**	(0.161)
Nutrients	0.023**	(0.004)
Cost	-0.010**	(0.001)
Public X Grass	0.569*	(0.242)
Public X Forest	0.291	(0.193)
Public X Boulders	-0.088	(0.217)
Public X Wetland	0.020	(0.215)
Public X Nutrients	0.001	(0.005)
Wealth X Grass	-0.131	(0.145)
Wealth X Forest	0.034	(0.131)
Wealth X Boulders	-0.412**	(0.141)
Wealth X Wetland	-0.243	(0.140)
Wealth X Nutrients	0.008*	(0.003)
Wealth X Public X Grass	0.103	(0.205)
Wealth X Public X Forest	-0.154	(0.173)
Wealth X Public X Boulders	0.316	(0.190)
Wealth X Public X Wetland	0.098	(0.172)
Wealth X Public X Nutrients	-0.001	(0.004)
Rural X Grass	-0.006	(0.084)
Rural X Forest	-0.061	(0.076)
Rural X Boulders	-0.237**	(0.085)
Rural X Wetland	-0.213*	(0.084)
Rural X Nutrients	0.000	(0.002)
Rural X Public X Grass	-0.122	(0.125)
Rural X Public X Forest	-0.031	(0.112)
Rural X Public X Boulders	0.207	(0.117)
Rural X Public X Wetland	0.177	(0.115)
Rural X Public X Nutrients	-0.001	(0.003)
Drive X Grass	0.506**	(0.194)
Drive X Forest	0.098	(0.175)
Drive X Boulders	-0.653**	(0.187)
Drive X Wetland	-0.481*	(0.188)
Drive X Nutrients	-0.003	(0.005)
Drive X Public X Grass	-0.541	(0.298)
Drive X Public X Forest	-0.040	(0.253)
Drive X Public X Boulders	0.556*	(0.263)
Drive X Public X Wetland	0.377	(0.266)
Drive X Public X Nutrients	0.008	(0.007)

Drive X Wealth X Grass	0.244	(0.173)
Drive X Wealth X Forest	0.105	(0.157)
Drive X Wealth X Boulders	0.384*	(0.160)
Drive X Wealth X Wetland	0.311	(0.162)
Drive X Wealth X Nutrients	-0.009*	(0.004)
Drive X Wealth X Public X Grass	-0.048	(0.251)
Drive X Wealth X Public X Forest	0.108	(0.234)
Drive X Wealth X Public X Boulders	-0.414	(0.224)
Drive X Wealth X Public X Wetland	-0.317	(0.214)
Drive X Wealth X Public X Nutrients	0.007	(0.005)
Drive X Rural X Grass	0.013	(0.104)
Drive X Rural X Forest	0.088	(0.093)
Drive X Rural X Boulders	0.210*	(0.097)
Drive X Rural X Wetland	0.176	(0.099)
Drive X Rural X Nutrients	-0.001	(0.003)
Drive X Rural X Public X Grass	0.133	(0.157)
Drive X Rural X Public X Forest	-0.099	(0.137)
Drive X Rural X Public X Boulders	-0.274*	(0.137)
Drive X Rural X Public X Wetland	-0.203	(0.138)
Drive X Rural X Public X Nutrients	0.002	(0.003)
Observations	3,962	
Wald test p-value	0.000	
Pseudo R^2	0.250	

Robust standard errors in parentheses
** $p<0.01$, * $p<0.05$

Note: Survey respondents were shown choice experiments and asked to choose between restoration designs that varied based on riparian vegetation type (with riparian buffer types labelled “Grass” and “Forest” in the table), streambank stabilization approach for hard features or floodplain reconnection (labeled “Boulders” or “Wetland” in the table), and levels of nutrient pollution reduction to meet local watershed TMDL goal (“Nutrients”). Each restoration also had a one-time cost to the household (“Cost”). Each restoration was located on either public or private land (with streams on public land indicated by “Public” and “Private” as the baseline) and either within walking (≤ 1 mile) or driving (> 1 mile) distance from the household (with those within driving distance indicated by “Drive” and

“Walk as the baseline). Finally, heterogeneity in WTP was allowed by interacting respondent-specific factor variables derived from principal component analysis with each other variable (where factors are indicated by “Rural” and “Wealth”). The PCA factor for “wealth” incorporated variables for census-level median household income, unemployment rate, college education as well as parcel-level housing value and building quality. The PCA factor for “rural” incorporated variables for population density and surrounding land uses in impervious, forest, and farmland.

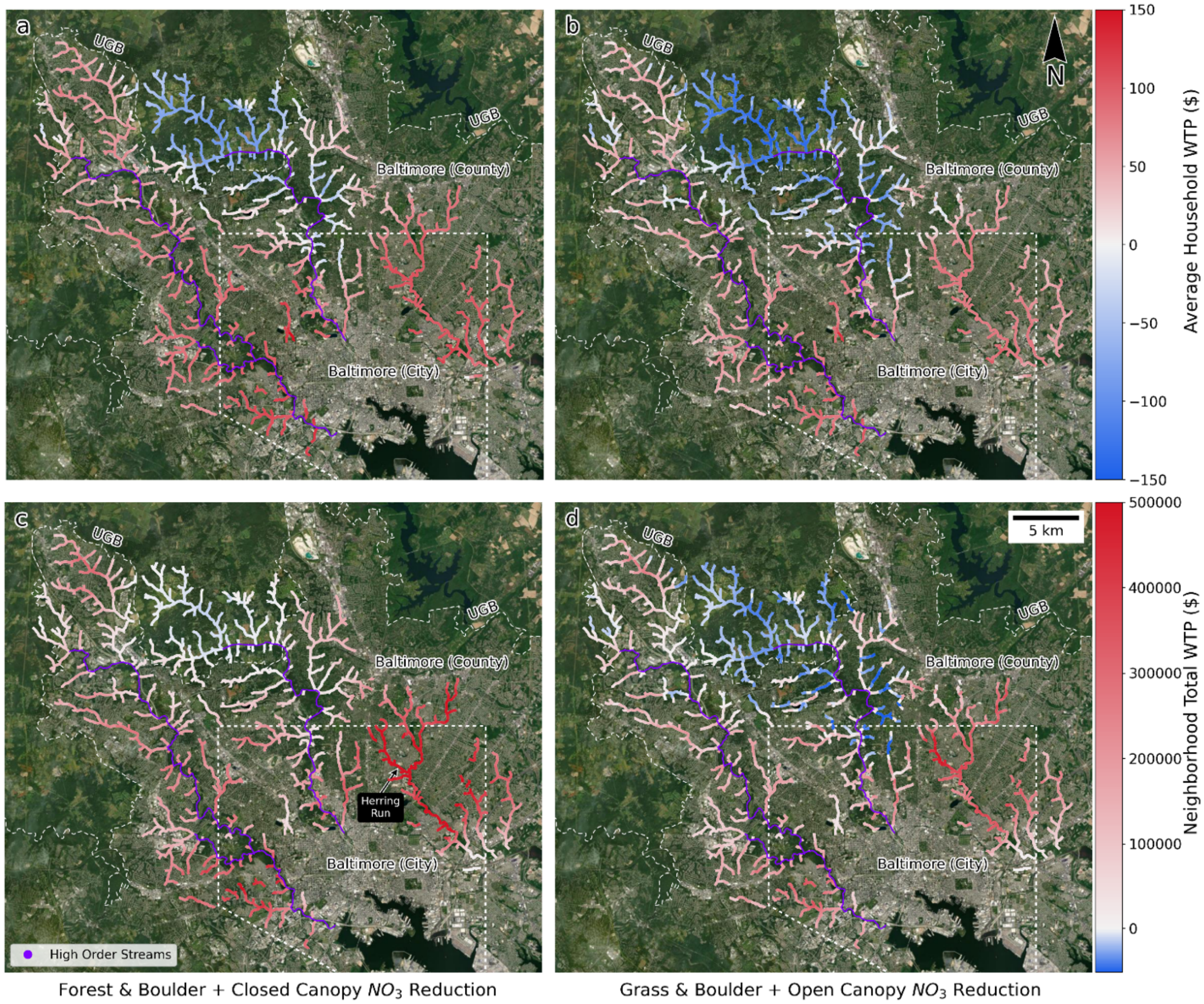

Figure 2.3. Willingness to pay (WTP) of stream restoration at stream reach level in the three study watersheds. Maps are for average household (a & b) and neighborhood total (c & d, greater than \$500,000 in dark red and less than \$50,000 in dark blue to highlight outliers) WTP for the FB and GB scenarios.

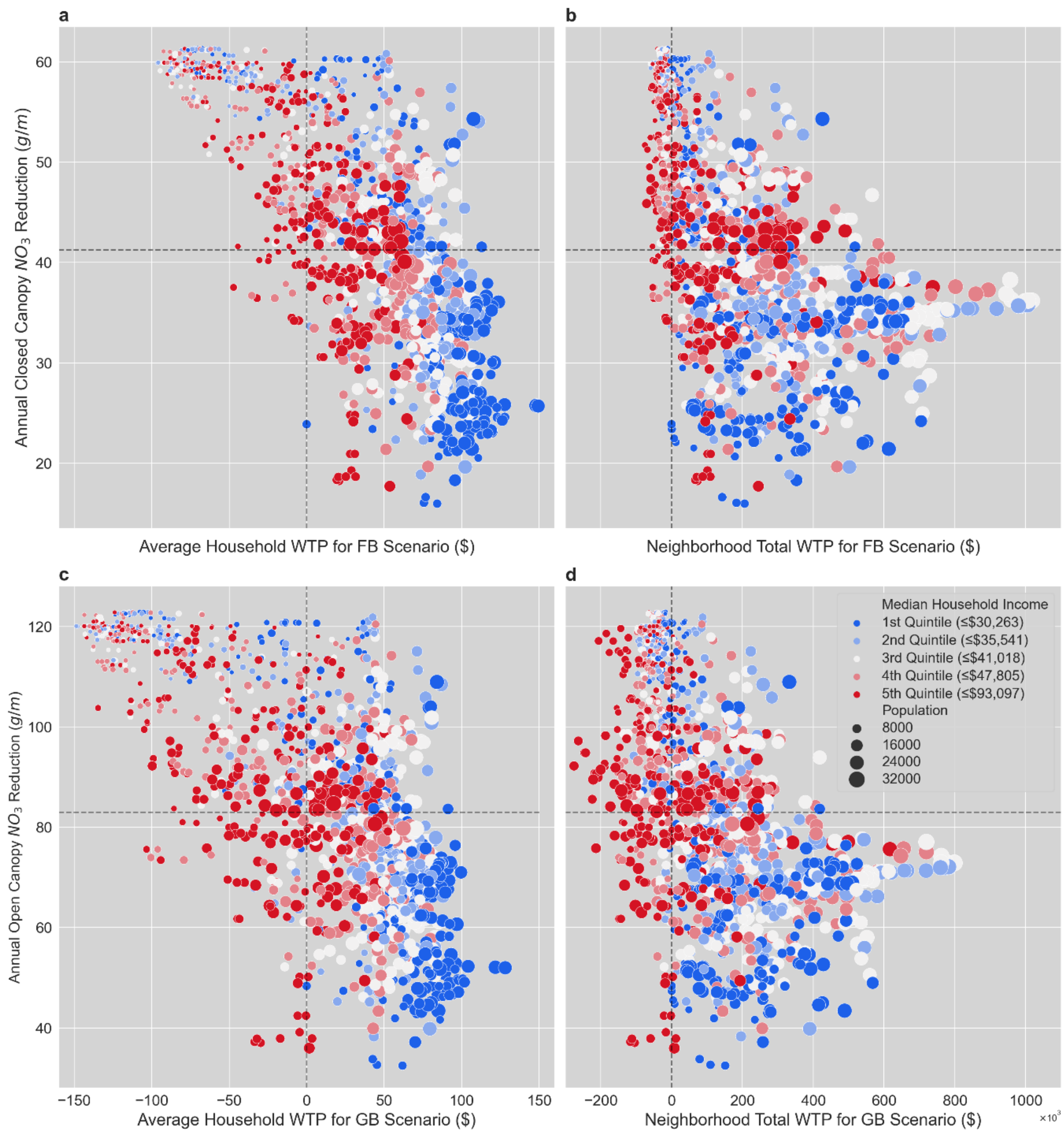

Figure 2.4. Linear regression between WTP and expected annual nitrate reduction (gN/m). Plots are WTP for forest and boulder (FB) design with closed canopy nitrate reduction at average household (a, r = -0.76) and neighborhood total (b, r = -0.72) levels, and WTP for grass and boulder (GB) design with open canopy nitrate reduction at average household (c, r = -0.57) and total neighborhood (d, r = -0.52) levels, with p-value for all smaller than

0.001. Quadrants are defined at zero WTP and median nitrate reduction for FB and GB scenarios

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Chapter 3: BALANCING UPLAND GREEN INFRASTRUCTURE AND STREAM RESTORATION TO RECOVER URBAN STORMWATER AND NITRATE LOAD RETENTION

3.1 PREFACE

This chapter was co-authored with Drs. Larry Band (University of Virginia) and Peter Groffman (Cary Institute of Ecosystem Studies). This study was funded by Baltimore Ecosystem Study, and is currently under review in Journal of Hydrology.

3.2 ABSTRACT

Urban watersheds have experienced ecosystem degradation due to land cover change from vegetation to impervious areas. This transformation results in increased stormwater runoff, stream channel erosion and sedimentation, and both increased inputs and reduced ecosystem retention of nutrients. Ecosystem restoration practices, including terrestrial and aquatic low impact development (LID), are becoming widely implemented in urban watersheds globally. A major question is how “green” and “grey” infrastructure can be optimally balanced to shift ecohydrological behavior towards pre-urbanization conditions. Traditional stormwater engineering typically controls runoff by temporary storage (detention) and release of stormwater, while LID designs are developed to reduce runoff by a combination of infiltrating precipitation and evapotranspiration, while promoting biogeochemical retention of nutrients. These practices and often combined with stream and riparian restoration that increases nutrient retention and reduces in-stream loads. In this study, we simulated the potential impact of three types of terrestrial LID and green

infrastructure (GI) on watershed runoff and nitrate loading to local streams, independent of detention storage effects. The treatments included increased tree canopy, vegetated roadside bioswales, and permeable pavement. We then evaluated the individual and interactive impacts of these practices on the effectiveness of nitrate load reduction provided by stream restoration, which is affected by the altered runoff and nutrient loading caused by the LID and GI. Urban reforestation provided the highest effectiveness in terms of reducing stormflow and nutrient export, while bioswales and permeable pavement unexpectedly increased in-stream nitrate loads. Retrofit of the previously developed watershed by LID/GI alone may not provide sufficient mitigation in stormwater and nutrient loads, and should be balanced with additional grey infrastructure, such as detention ponds, rain cisterns, and sewer system upgrades.

3.3 INTRODUCTION

Urbanization increases impervious area and drainage infrastructure (stormwater and sanitary systems), replacing vegetation and natural streams within watersheds. This development has brought economic growth and reduced water-borne disease, but has elevated stormwater runoff and downstream impacts of flash flooding and stream nutrient loading (Booth et al., 2002; Walsh et al., 2005). In addition, climate change has increased the frequency of intense precipitation (Mishra et al., 2015; Trenberth, 2011), elevating urban watershed vulnerability to stormwater hazards (Ashley et al., 2005; Zhou et al., 2012). Standard stormwater engineering solutions includes installation of temporary detention storage (e.g., wet/dry detention ponds) to capture and slowly release stormwater to reduce peak flows. This detention storage does not reduce flow volume, and there is an interest to use low impact development or green infrastructure (LID/GI) to infiltrate and evapotranspire stormwater. However, there is increasing evidence that increased infiltration and groundwater recharge associated with efforts to reduce surface runoff may mobilize subsurface sources from sanitary sewer leakage and septic systems (Delesantro et al., 2022).

Coupled with impervious surface and engineered drainage expansion, vegetation canopy loss is the most impactful land cover change. Trees are a main source of transpiration and a major influence on urban and peri-urban water, carbon, nutrient end energy balance. Increased tree canopy cover has the potential to mitigate peak flow during storm events by promoting infiltration and water storage capacity of soils (Bartens et al., 2008). Trees also bring co-benefits such as abatement of urban heat islands (Wang et al., 2015), provision of green spaces which can improve quality of life for urban residents (Roe & Sachs, 2022; Wolch et al., 2014), and uptake of carbon dioxide and available nitrogen (Livesley et al., 2016). Other terrestrial LID approaches utilizing vegetation (e.g., rain garden, roadside swale, green roof, etc.) have also been widely used in many metropolitan regions to mitigate stormwater (Davis et al., 2009; Hopkins et al., 2020) and to provide these co-benefits.

Nitrogen export from urban watersheds occurs across the range of stream discharges from baseflow to stormflow. There is great concern about this export in coastal watersheds where this element is a key driver of eutrophication (Conley et al., 2009), and the target of total maximum daily load (TMDL) regulations (Wainger, 2012). In-stream retention processes can reduce these loads, but these processes are only effective at lower flows (Lin et al., 2021; Shields et al., 2008). Therefore, stormflow reduction may improve aquatic nutrient retention, especially where streams have been restored to increase retention processes (Craig et al., 2008). LID/GI promotes increased soil water storage and infiltration to reduce surface stormwater peaks, along with increased evapotranspiration to reduce total outflow. Repartitioning flows from surface to subsurface flows by enhanced infiltration may reduce surface sources of nutrients, but increase baseflows and nutrient loading from subsurface sources (Delesantro et al., 2022; Kaushal et al., 2011). Higher soil moisture and groundwater levels, with enhanced vegetation cover may impact ecosystem retention processes (e.g., plant nitrogen uptake and denitrification) both in "on-site" areas (i.e., where treatments are implemented) of infiltration-based practices and potentially downslope in "off-site" areas (i.e., downslope, downstream, and riparian areas). Therefore, there is an interaction between upland LID/GI and stream restoration in whole watershed retention. Understanding the balance between upland, riparian, and channel restoration is a key gap that needs to be addressed. In addition, deconvolving the effects of increased (temporary) stormwater storage from practices that increase infiltration, evapotranspiration and biogeochemical retention also requires investigation to understand and optimize design.

Ecohydrological functions of LID/GI, have been intensely examined at the facility scale. Studies have evaluated how LID/GI can reduce peak discharge (Damodaram et al., 2010; Hopkins et al., 2020; Li et al., 2017) and affect baseflow levels (Bhaskar et al., 2016). Both the amount and spatial arrangement of LID/GI within a watershed influence streamflow and nutrient regime mitigation, with more distributed implementation potentially providing greater efficiency (Hopkins et al., 2022). Placing LID/GI downstream of impervious areas can disconnect runoff from adjacent impervious areas and store, evapotranspire, and delay surface runoff release to streams (Bell et al., 2016; Fiori & Volpi, 2020; Gilroy & McCuen, 2009; Walsh et al., 2005). However, most studies of single LID/GI facilities are limited to measurement of surface inflows and outflows, and the cumulative performance of LID/GI and corresponding impacts to baseflow at the watershed scale may be much less effective than at the facility scale (Miller et al., 2021). This may be due to insufficient LID/GI volume to mitigate flashy flow regimes from the dominant impervious areas and piped drainage systems. In addition, infiltration-based LID/GI may shift surface impervious runoff to subsurface stormflow or saturation runoff from pervious areas receiving run-on (Miles & Band, 2015) depending on subsurface storage capacity and conductivity.

Stream restoration has gained in popularity as a method to reduce sediment and nutrient loads to downstream waterbodies (Bernhardt et al., 2005). Stream restoration projects reshape stream channels by reshaping steep, eroding banks to create gentle-slope near-stream areas to improve connectivity between riparian ecosystems and the stream channel. The re-engineered channel and riparian zone increases the residence time of water, promoting nutrient retention by plant uptake and denitrification (Craig et al., 2008; McMillan & Noe, 2017; Ward et al., 2011). The reestablished streamside vegetation can provide aesthetic and socioeconomic value for urban residents (Rosenberg et al., 2018). Stream restoration can increase stream-subsurface exchange and hyporheic processing by altering stream hydraulics (Kasahara & Hill, 2008), which is an additional sink for in-stream nutrients. The effectiveness of urban stream restoration is reduced if it is not accompanied by complementary catchment restoration to mitigate runoff and nutrient inputs that are the initial causes of aquatic degradation (Jahnig et al., 2010; Lorenz & Feld, 2013; Palmer et al., 2010). A limitation to evaluation of restoration performance is that most in-stream nutrient retention studies (Newcomer Johnson et al., 2014; Reisinger et al., 2019; Violin et al., 2011) have been conducted at low to moderate stream flow levels, when instream retention is highest, while nutrient loads may be dominated by high flows with minimal retention.

The effectiveness of terrestrial and stream restoration for urban runoff and nutrient regime mitigation has typically been separately studied. Much less is known about how catchment and in-stream restoration can be balanced to optimize long-term changes in terrestrial-aquatic loading at whole-watershed scales. In this study, we developed a spatially-explicit terrestrial-aquatic framework to evaluate the combined effects of upland and instream restoration. We used the Regional Hydro-Ecological Simulation System (RHESSys) to simulate changes in terrestrial ecohydrological processes, the Weighted Regressions on Time, Discharge, and Season (WRTDS) to estimate corresponding changes in nitrate flux for four different watershed restoration scenarios, and an aquatic metabolism model (Lin et al., 2021; Zhang et al., 2022) to estimate subsequent in-stream effects on retention. Though RHESSys is capable of estimating watershed nitrogen cycle and retention processes including vegetation uptake and denitrification, sanitary sewer inflow and infiltration (I&I) and effluent leakage are difficult to parameterize. The data-based WRTDS, on the other hand, implicitly includes the impacts of multiple sources of nitrate but cannot project the impacts of LID/GI implementation on ecosystem retention. Therefore, we used information from both approaches to estimate changes in runoff quantity and changes in water quality in streams after terrestrial restoration scenarios. The aquatic metabolism model was used to depict the effects of stream restoration on nitrogen retention in restored and unrestored downstream reaches (Lin et al., 2021; Zhang et al., 2022). In summary, the framework can provide estimates of hydro-biogeochemical cycling, retention, and export at patch, flowpath, and watershed scales, and the ability to allow stakeholders to evaluate efficiencies of different GI restoration plans on stormwater and nutrient load reductions at "on-site" and "off-site" locations. All numerical experiments were carried out using an urban watershed, Scotts Level Branch in Baltimore.

We addressed the following questions:

1) How do LID/GI practices produce "on-site" and "off-site" effects on hydrologic and biogeochemical processes, including biogeochemically critical offsite regions such as downstream restored stream and riparian areas?
2) What is the effectiveness of LID/GI restoration practices independent of storage detention (i.e., distributed tree canopy and green infrastructure implementation) on mitigating urban runoff and nitrate export regimes towards pre-urbanization conditions and increasing nitrogen retention from stream restoration?

3.4 METHODS

3.4.1 Study area

We evaluated our framework through a case study at Scotts Level Branch (SLB) in Baltimore County, MD (Figure 3.1). SLB has a drainage area of 8.6 km^2, mean elevation of 166 m and mean annual precipitation of 1,153 mm. Residential land use and land cover (LULC) is dominated by single-family houses, driveways and roads, lawn, and tree canopy. According to the Chesapeake Bay 1-m LULC data (Hood et al., 2021), there are 2.4 km^2 of impervious area (structures, impervious surfaces, and roads), 3.3 km^2 of tree canopy, and 2.7 km^2 of lawn, covering 28.4%, 38.8%, and 30.7% of SLB, respectively. A set of stream restoration projects have been completed in the main stream and tributaries (example in Figure 3.2) by Baltimore County, and studies have measured and modeled the efficiency of nutrient reduction from these projects (Lin et al., 2021; Reisinger et al., 2019).

The United States Geological Survey (USGS) has gauged SLB (gage ID: 01589290, 39.36°N, 76.76°W) since Oct 2005. Average discharge from water year 2006 to 2021 is 0.13 m^3/s (1.3 mm/day). We delineated the stream network in SLB using the r.watershed tool from GRASS GIS with a threshold 0.62 km^2, which approximates the length of mapped streams from the NHD High resolution dataset (https://www.usgs.gov/national-hydrography/nhdplus-high-resolution). We approximated riparian areas extent as the areas less than 1.5 meter above delineated streams, using the GIS algorithm HAND (Height Above Nearest Drainage, Nobre et al., 2011).

3.4.2 Terrestrial restoration scenarios

We considered three watershed terrestrial restoration scenarios (Figure 3.2) to compare to status quo (SQ) conditions to represent end-member restoration designs (Table 3.1): 1) replace all lawn more than 10-m from buildings by deciduous tree canopy (urban reforestation, URF), 2) introduce 2-meter shrub bioswale buffers along all roads (roadside bioswale, RBS), 3) retrofit current roads with permeable pavement (permeable road, PR), and 4) combine reforestation and roadside bioswales (URF+RBS). We note that these hypothetical scenarios may not be practical, particularly the extensive replacement of lawns and the use of permeable pavement on all roads. However, we use these to explore what the limits of efforts to increase infiltration and transpiration, reduce road runoff volume and peak, and increase nitrogen retention. We deliberately did not include enhanced detention storage practices in these treatments in order to isolate the impacts of increased evapotranspiration, infiltration, and biogeochemical retention from detention practices. We evaluated impacts of these LID/GI scenarios on watershed-scale runoff and nutrient loading, nitrogen retention efficiency of stream restoration of local stream reaches, and nitrogen retention processes at LID/GI patches and riparian areas. Per unit efficiency of reducing runoff or nitrate load for three types of LID/GI (URF, RBS, and PR) was calculated as total change at the watershed outlet divided by the size of each LID.

3.4.3 Framework and model description

The framework couples an ecohydrological model, the Regional Hydro-ecological Simulation System (RHESSys, Tague & Band, 2004) that simulates the effects of restoration on both on- and off-site changes in surface and subsurface water balance and nitrogen biogeochemistry, with the Weighted-Regression on Time, Discharge, and Season (WRTDS, Hirsch et al., 2010) that estimates the corresponding in-stream nutrient concentration and flux from every new flow regime. We then estimate in-stream nitrate reduction expected for a restored stream compared to an unrestored stream using a stream ecosystem model from Lin et al. (2021) and Zhang et al. (2022).

3.4.3.1 Ecohydrological modeling with RHESSys

RHESSys is a distributed ecohydrological model which simulates water balance (e.g., streamflow, evapotranspiration, soil moisture redistribution, and surface/subsurface flow), plant growth, and biogeochemical (nitrogen and carbon cycles) processes at patch to watershed levels. The model has been applied in many regions with different climates (e.g., semiarid California to humid eastern US) and LULC (e.g., forest and urban) conditions (Bart et al., 2016; Leonard et al., 2019; Lin et al., 2019; Ren et al., 2022).

RHESSys can be set up to simulate at a range of spatial resolutions. The model can incorporate fractional sub-patch LULC as more fine-resolution LULC data are available. We use a computationally efficient resolution (15 m) with fractional land cover generated from a 1m land cover dataset (Hood et al., 2021). A set of model hydrological parameters governing subsurface moisture storage and flux (Table 3.2) were developed from SSURGO mapped soil properties, and further calibrated against USGS streamflow observations at Scotts Level Branch. The parameter set generating highest Nash-Sutcliffe efficiency (NSE, Nash & Sutcliffe, 1970) value was used in this study. Identically calibrated model parameters of RHESSys developed from SQ conditions were applied to RHESSys models of the LID/GI scenarios.

We used a spin-up period of RHESSys from Jan 1, 1993 to Dec 31, 2005 to stabilize state variables, and then simulated watershed processes under different restoration scenarios from Jan 1, 2006, to Dec 31, 2016. Tree species composition was adapted from Lin et al. (2021), which is a mix of red maple (*Acer rubrum*), oak (*Quercus spp*.), and tulip poplar (*Liriodendron tulipifera*) with proportions of 15%, 37%, and 48%, respectively. Trees were assumed mature in our simulation, although RHESSys carbon budgets and allocation results in additional growth. Several "on-site" state, flux, and transformation variables (i.e., saturation deficit, transpiration, denitrification, vegetation uptake, and nitrate fertilization) governing water and nitrogen balance in the study watershed were simulated and used to characterize hydrological and ecosystem cycling and retention.

3.4.3.2 Evaluation of water quality using WRTDS

Altered streamflow regimes produced by the uses of LID/GI were combined with concentration-discharge-season (c-Q-s) relationships derived with the WRTDS model to produce estimates of stream nutrient concentrations and loads. WRTDS is a statistical model built with weekly stream chemistry data (collected by the Baltimore Ecosystem Study (BES) since 1998) for several watersheds gauged by the USGS in Baltimore City and County. Sources of nitrogen from the sanitary sewer system leakage may be significant (e.g., Kaushal et al., 2011), but are very difficult to accurately quantify. We therefore use the WRTDS estimates of nitrate concentration and load, based on discharge and time (seasonality) to encompass all nitrogen sources, including effluent sources (e.g., sanitary sewer leakage) which are not reliably simulated in RHESSys. This approach assumes that the relationships between stream nitrogen concentrations and flow and seasonality are not significantly impacted by alterations in the runoff regime.

As SLB was not monitored for stream chemistry by BES, the WRTDS estimated concentration-discharge-seasonality (c-Q-s) relations for a nearby section of the Gwynns Falls watershed (above the USGS Delight Gauge 01589197) with similar size and land cover, were used. Both Delight and SLB are dominated by residential LULC with similar proportions of impervious areas, lawn, and tree canopy. Long-term BES sampling indicates that while nitrate concentration increases with lower flows, there is a concentration reduction at the lowest flows (<0.1 mm/day). The WRTDS equations did not represent this phenomenon, and therefore, we capped all WRTDS estimated nitrate concentrations that were above the maximum observed concentration value to 8 mg NO_3-N/L in these lowest flows. We note the total loads derived from these very low flows are negligible. Uncertainty analysis for nitrate concentration was performed through bootstrapping.

3.4.3.3 *Aquatic restoration*

Stream channel restoration can reduce nitrate loads by increasing stream water residence times, exchange rates with bed and bank sediments, and inadvertently by increasing solar radiation at the stream surface by reduction of riparian vegetation cover (Reisinger et al., 2019). The latter increases algal nutrient uptake rates. The Lin et al. (2021) stream ecosystem metabolism model was used to simulate restored and unrestored stream reaches in SLB under scenarios with and without riparian vegetation cover, and with runoff regimes from several BES monitored streams with varying degrees of development (Zhang et al., 2022). We defined an exponential relationship between an index of the nitrate-flow distribution, F75 (Shields et al., 2008), and the net stream ecosystem retention (g/m of restoration) of nitrate in restored and unrestored streams, with and without riparian cover (Zhang et al., 2022):

$$Nitrate\ Reduction\ = \begin{cases} F75^{-0.302}e^{4.331}, & closed-canopy \\ F75^{-0.298}e^{5.023}, & open-canopy \end{cases}$$

(Equation 1)

where F75 is the stream discharge level corresponding to the 75th percentile of the cumulative nitrate load, which is highly correlated ($R^2 = 0.89$) to the percent upstream impervious area. Eq. 1 was then used to estimate changes of in-stream nitrate retention following the different watershed LID treatments, based on the altered streamflow and nitrate load distributions that they produced, characterized by F75. The estimated nitrate reduction from stream restoration over all reaches in SLB was combined with streamflow and nitrate flux change from the SQ and each terrestrial restoration scenario to evaluate stream retention and the combined load reduction caused by both terrestrial (LID/GI) and aquatic (stream restoration) efforts.

3.4.4 Mass balance of nitrate in watershed

Nitrogen in a forested watershed is accumulated through atmospheric deposition (ATM) and nitrogen fixation (FIX_{N_2}). The input of nitrogen can be used as vegetation uptake (UPT) and recycled through growing and dormant seasons. Nitrogen leaves watersheds through denitrification (DNF) as nitrogen gas (N_2) or nitrous oxide (N_2O), is immobilized for long periods in stable soil compounds (IMM) and stored, or exported downstream through hydrological flow pathways ($Flux_{stream}$). In urban watersheds, additional nitrogen is added by human activities such lawn fertilization (FRT) and leakage from sanitary sewers (LKG_{sewer}). The annual mass balance for nitrogen in an urban watershed is thus:

$$\Delta \mathrm{N} = ATM + FIX_{N_2} + LKG_{sewer} + FRT - UPT - Flux_{stream} - DNF - IMM \quad \text{(Equation 2)}$$

We assumed that nitrogen allocated for vegetation uptake does not directly leave the watershed unless leaf litter or mowed grass are collected and removed by landscaping practices, although it may mineralize and be transported at a later time. We also assume a fertilization rate (3 g NO_3-N /m^2, twice during the growing season with 90 days interval, equivalent to 60 kg NO_3-N/ha/year) for lawns in our study watershed, based on past household lawncare surveys (Fraser et al., 2013) in the area. Watershed total fertilization (FRT) changed with the area of lawns in different scenarios, while the other three input sources, ATM, FIX_{N_2}, and LKG_{sewer},were assumed to be unaffected by restoration.

As WRTDS does not estimate changes in watershed loads due to altered ecosystem nitrogen retention processes, we estimated changes in plant uptake, denitrification, and fertilization in different scenarios with RHESSys. For seasonal analysis, we defined Spring from March to May, Summer from June to August, Fall from September to November, and Winter from December to February.

3.5 RESULTS

3.5.1 Model accuracy

Calibrated parameters (Table 3.2) for SLB produced a best fit streamflow time-series of SQ conditions with a NSE of 0.79 to USGS observations (Figure 3.3a). Mean simulated streamflow for the SQ was 1.38 mm/day, which overestimated USGS measured flow by 12.8% (0.16 mm/day). Dormant season simulations produced higher baseflow than USGS observations, while growing season simulations underestimated streamflow, particularly in dry periods (Figure 3a). The calibrated WRTDS model predicted NO_3 concentrations at Delight with NSE of 0.57 ± 0.01 (Figure 3.3b).

3.5.2 Changes in water quantity

Mean streamflow for URF, RBS, URF+RBS, and PR, were 1.36, 1.40, 1.37, and 1.43, and mm/day, respectively. Two urban reforestation scenarios, URF and URF + RBS, showed small reductions in mean flow by 0.02 mm/day (-1.8%) and 0.01 mm/day (-1.1%), while other infiltration-based scenarios, RBS and PR, elevated the mean flow slightly by 0.01 mm/day (+0.8%) and 0.04 mm/day (+3.2%), compared to SQ scenario.

To quantify the effectiveness of restoration scenarios during different flow conditions, the SQ streamflow was classified into eight flow percentile groups (Table 3.3), and the mean streamflow produced by each restoration scenario for each percentile group was calculated. URF and URF+RBS scenarios resulted in lower streamflow for all flow groups compared to the SQ scenario, except for the 50-90% groups for URF+RBS. RBS and PR both increased streamflow for percentile groups below 95%. For stormflows above the 99% percentile, all restoration scenarios reduced stormwater quantity: URF+RBS had the highest effectiveness, 0.81 mm/day (3.3% reduction), while URF, RBS and PR scenarios had reduction of flows by 0.54 (2.2%), 0.25 (1.0%), and 0.30 (1.2%) mm/day, respectively, compared to SQ scenario. From three scenarios with only one type of LID, reforestation, roadside bioswale, and permeable road, stormflows (> 99% percentile) were reduced by 3.9, 8.3, and 3.3 mm/day per unit treated area.

During dry, low-flow periods, the RBS and PR scenarios sustained higher streamflow while the URF and URF+RBS scenarios reduced flows compared to the SQ scenario. During wet periods, streamflow responded to our scenarios differently in different seasons. For example, in 2009 (Figure 3.4), all scenarios reduced high flows during the growing season, with the URF+RBS scenario having the highest reduction. In contrast, during the transition from dormant to growing season (e.g., April to May), the URF and URF+RBS scenarios increased streamflow during storm days, while the RBS and PR scenarios reduced streamflow during storm events.

The URF and URF+RBS scenarios decreased depth to water table in the dormant season and increased it in the growing season (Figure 3.5). The RBS and PR scenarios had much smaller effects on water table depth. Over the study period, depth to water table in increased in the URF, URF+RBS, and PR scenarios, by 56.8 and 51.8, and 2.6 mm, respectively. RBS was the only scenario that reduced water table depth, by 5.7 mm.

3.5.3 Changes in nitrate export from terrestrial restoration

The c-Q relationships (Figure 3.6) for all restoration treatment scenarios had similar patterns, as expected using a consistent WRTDS model. Since our model yielded lower low flows in dry growing seasons than USGS gage observations at SLB, WRTDS simulated NO_3 concentrations in these drought days were higher than the highest observed and simulated concentration of WRTDS using USGS streamflow records. Overestimated NO_3 concentrations above 8 mg/L were therefore set to 8 mg NO_3-N/L (see Method 2.3.2). The mean NO_3 flux (Figure 3.7) using USGS flows with WRTDS was 7.86 (± 0.053) kg NO_3-N/ha/year, and the mean NO_3 flux with the SQ scenario flows was 9.01 (± 0.047) kg NO_3-N/ha/year, which is 1.15 kg NO_3-N/ha/year (15.6%) higher. The RBS and PR scenarios resulted in higher NO_3 export than the SQ, with a mean flux of 9.12 (± 0.048) kg NO_3-N/ha/year for RBS and 9.29 (± 0.052) kg NO_3-N/ha/year for PR, increases of 0.11 (1.2%) and 0.28 kg NO_3-N/ha/year (3.1%) respectively. The URF and URF+RBS scenarios reduced NO_3 export to 8.89 (± 0.050) kg NO_3-N/ha/year and 8.97 (± 0.041) kg/ha/year; 0.12 (-1.3%) and 0.04 kg NO_3-N/ha/year (-0.4%) lower than the SQ mean flux, respectively. The cumulative NO_3 export function (Figure 3.8), however, showed minor differences of NO_3 export behavior after tree or LID implementation scenarios.

The per unit area efficiency of nitrate load modification varied among the three types of restoration treatment: one unit area of reforestation in URF reduced nitrate export by .86 kg/ha/year while the two infiltration-promoting methods, roadside bioswale in RBS and permeable road in PR, increased nitrate export by 3.65 and 3.10 kg NO_3-N/ha/year. The small areas affected by RBS and PR led to a small effect at the whole watershed scale.

3.5.4 N change in ecosystem retention processes

Restoration practices altered the biogeochemical processes underlying nitrogen retention. The infiltration-based practices (i.e., RBS and PR) increased denitrification rate (Figure 3.8), both at the LID sites as well as at off-site riparian areas. In contrast, denitrification rate decreased at reforested sites and riparian areas in the URF and URF+RBS scenarios. Annual denitrification (Table 3.4) in SQ scenario was 18.2 kg NO_3-N/ha/year, with the highest rate in spring (24.2 kg NO_3-N/ha/year), followed by summer (17.5 kg NO_3-N/ha/year), fall (17.2 kg NO_3-N/ha/year), and winter (14.0 kg NO_3-N/ha/year). The URF and URF+RBS scenarios lowered annual denitrification to 12.1 (-6.1) and 13.5 (-4.7) kg NO_3-N/ha/year, while the RBS and PR increased rates to 20.2 (+1.9) and 22.7 (+4.5) kg NO_3-N/ha/year. In riparian areas, the RBS and PR scenarios increased denitrification rates to 36.9 (+2.9) and 41.4 (+7.4) kg NO_3-N/ha/year compared to the SQ rate of 34.0 kg NO_3-N/ha/year. Rates in the URF and URF+RBS scenarios decreased to 20.4 (-13.7) and 22.8 (-7.4) kg NO_3-N/ha/year. At patches with the RBS and PR treatments, mean denitrification rate increased to 19.4 (+6.3) and 27.1 (+13.9) kg NO_3-N/ha/year compared to the SQ rates of 13.1 and 13.2 kg NO_3-N/ha/year. At patches with the URF and URF + RBS treatments, the rate decreased to 14.0 (-7.5) and 16.0 (-5.5) kg NO_3-N/ha/year compared to the SQ rate of 21.5 kg NO_3-N/ha/year (Figure 3.8).

Annual vegetation nitrogen uptake (Table 3.4) in SQ scenario was 62.4 kg/ha/year. Uptake varied seasonally, with the highest rates in summer (166.1 kg N/ha/year), followed by fall, spring, and winter (51.5, 30.0, 2.1 kg N/ha/year, respectively). Nitrogen uptake was highly correlated with deciduous tree canopy phenology (Figure 3.9). The URF and URF+RBS scenarios both increased annual uptake to 70.1 (+7.7) and 70.4 (+8.0) kg N/year for the whole watershed. The RBS and PR scenarios had negligible (< 0.5 kg N/ha/year) effects on update. The 1.2-km^2 reforestation treatment in the URF and URF+RBS scenarios increased nitrogen uptake rates in all seasons except for winter. At the reforested patches, the seasonal rates increased to 199.7 (+60.2) and 55.1 (+4.9) kg N/ha/year in summer and fall compared to the SQ rates of 139.5 and 50.2 kg N/ha/year. These treatments decreased uptake in spring and winter to 28.5 (-5.4) and 1.0 (-2.0) kg N/ha/year compared to 33.9 and 3.0 kg N/ha/year in the SQ scenario.

3.5.5 Cumulative nitrate export change through terrestrial and aquatic restoration

The altered flow regimes produced by the restoration scenarios shifted the nitrate export flow distribution (F75), which promoted or reduced estimated in-stream nitrate reduction efficiency from stream restoration, i.e., the efficiency of stream restoration increases if more nitrate is exported at lower flows (Figure 3.10). Compared to the F75 value of the SQ scenario (4.21 mm/day), the F75 of the URF (4.46 mm/day) and URF+RBS (4.30 mm/day) scenarios increased by 5.85% and 1.95%, while the RBS (4.13 mm/day) and PR (3.94 mm/day) scenarios had reduced F75 values by 2.11%, and 6.50%, respectively (Figure 3.11). If all stream reaches were restored with open-canopy design, projected nitrate reduction for the SQ scenario was 313.6 g/ha/year. By comparison, URF, RBS, URF+RBS, and PR scenarios resulted in nitrate reduction of 308.4, 315.5, 311.8, 320.0 g/ha/year, respectively (Figure 3.8). In addition, the reforestation of 1.2-km^2 of lawns reduced watershed level fertilization rates to 10.2 kg N/ha/year from the current rate of 18.3 kg N/ha/year (two applications in growing season).

A mass balance analysis for SLB (Table 3.5) shows that the reforestation (i.e., URF and URF+RBS) scenarios reduced nitrate sources in the watershed by 31.54 and 30.67 kg N/ha/year. In contrast, the infiltration-based (i.e., RBS and PR) scenarios elevated nitrate sources by 0.98 and 2.33 kg N/ha/year, respectively. The reforestation scenarios involved removal of lawns which reduced fertilization by 8.13 kg N/ha/year. For the infiltration scenarios, denitrification rates increased by 1.92 and 4.46 kg N/ha/year in the RBS and PR scenarios, but nitrification rates increased more strongly, by 3.28 and 7.27 kg N/ha/year. For the reforestation scenarios (URF and URF+RBS), denitrification rates decreased by 6.10 and 4.70 kg N/ha/year, but nitrification rate decreased more significantly, by 21.40 and 18.91 kg N/ha/year. Vegetation uptake increased by 7.68, 0.17, 7.98 and 0.44 kg N/ha/year in the URF, RBS, URF+RBS, and PR scenarios. Instream nitrate flux estimated by WRTDS changed very little, with URF and URF+RBS reducing load by 0.12 and 0.04 kg N/ha/year and RBS and PR increasing load by 0.11 and 0.28 kg N/ha/year.

3.6 DISCUSSION

3.6.1 Seasonal responses of hydrological processes to restoration

Our analysis focused on the efficiency of LID/GI practices in mitigating urban runoff and nitrogen regime. We note that each of LID/GI scenarios were modeled without additional built detention storage effects. The LID/GI practices we investigated included both terrestrial green infrastructure to increase infiltration and evapotranspiration, and stream restoration. Both individual and combined effects were considered. As expected, the URF and URF+RBS scenarios that replaced lawn area with tree cover reduced the water yield, especially during the growing season. Over the 11-year simulation period, once forest canopies started to transpire around mid-May, soil saturation deficits in the increased rapidly (Figure 3.4). The drier soil enabled higher water storage capacity and increased the capacity of the watershed to store storm events, reducing stormflow and increasing baseflow in the growing season. However, our results also suggested that replacing lawns with trees created a short, wet period prior to the growing season every year, due to the different phenology of deciduous trees and grass. Converting current lawns to tree canopy thus delayed the onset of transpiration in early spring.

The RBS and PR treatments were able to mitigate stormflows (> 99% quantile), reducing the flashiness of storm hydrographs, but not as effectively as the URF or URF+RBS treatments. The RBS and PR scenarios often had lower streamflow than the SQ scenario during storm days, but higher flows one day after. In other words, they helped to reduce the peak stormflow and create a gentler recession limb after storm events. In summary, the results from all our LID/GI scenarios suggest that large-scale implementation of LID/GI alone in older urbanized areas might be insufficient to meet goals for flood mitigation, without incorporating sufficient detention storage. Other urban LID studies found limited effectiveness of LID/GI to attenuate flood peaks (Bell et al., 2016; Miller et al., 2021) in existing development and that reduction of more moderate flows is dependent on temporary detention storage and release. Our results also highlight the need to include grey infrastructures with LID/GI to increase the capacity of detention storage, reduce flashiness of flow regime, and better regulate stormwater peaks in urban watersheds.

3.6.2 Change in components of nitrate fluxes

The altered flow regimes produced by the URF and URF+RBS scenarios led to reduced stream nitrate export compared to the SQ scenario. In contrast, the RBS and PR scenarios that increased infiltration and baseflows elevated terrestrial export. The higher base flow discharge directly increased nitrate fluxes due to the higher nitrate concentrations in baseflow. The URF scenario reduced baseflows and nitrate flux the most (Table 3.3). The lack of a significant mitigation of nitrate export towards pre-urbanization conditions in our simulations are consistent with empirical results of Hopkins et al. (2022), although our study involved existing urban site restoration, while Hopkins et al. studied new development with varying levels of LID/GI implementation.

The altered flow regimes and nitrate export produced by the upland LID/GI practices, without additional detention storage, had marginal effects on the effectiveness of in-stream restoration. The three scenarios that increased infiltration, baseflow, and nitrate export (PR, RBS, and URF+RBS) increased the efficiency of stream restoration compared to the SQ scenario. This increase was likely driven by the higher nitrate concentrations and loads in the stream at low flows, which is when stream restoration is most able to facilitate nitrate uptake. In contrast to the infiltration scenarios, the URF scenario reduced the efficiency of stream restoration (Figure 3.10). One possible reason for the reduction in efficiency with the URF scenario is that increased tree canopy uptake resulted in lower nitrate delivery to riparian areas and streams, reduction opportunities for aquatic retention, especially during low flows.

The magnitudes of the changes in stormflow and in-stream nitrate load change from LID/GI and stream restoration were small (Figure 3.11). In contrast, internal ecosystem nitrate retention processes (i.e., denitrification and uptake) were much larger, suggesting a potential to significantly reduce nitrogen loads by focusing on these processes (Table 3.5). The simulated denitrification rate in the SQ scenario (18.2 kg N/ha/y) is consistent with measured lawn denitrification, 14.1 kg N/ha/year, in Baltimore (Raciti et al., 2011), which is much higher than any fluxes associated with stream restoration. Similarly, vegetation uptake rate in forested patches, 110.5 kg/ha/year, is also consistent with literature findings (Cole, 1981; Norby & Iversen, 2006) and very high compared to stream restoration effects. The increase in tree canopy uptake and the reduction in input from fertilizer associated with conversion of 1.2-km^2 of lawns to forest in the URF scenarios were also large. The infiltration-based RBS and PR scenarios increased baseflow and induced higher denitrification rates than in the SQ scenario (Figure 3.8), as the higher infiltration rate provided wetter and more anoxic soil conditions that favor denitrification, showing both on-treatment site and off-site impacts. However, the higher moisture level in soil after infiltration-based LID also elevated internal nitrate production by nitrification. The increase in nitrate production by nitrification was larger than the increase in nitrate consumption by denitrification, so the net effect of the RBS and PR scenarios was to make more nitrate available for transport to streams. These treatments increased nitrate retention associated with stream restoration, but this effect was much smaller than the increase in nitrate flux from nitrification. This effect was also much smaller than the increase in plant uptake after the reforestation-based URF scenario. These results highlight the importance of tree coverage, plant uptake, fertilizer application, and denitrification in urban watersheds, which have large effects on nitrate load compared to stream restoration.

3.6.3 Uncertainties and future improvement to the framework

While our models have been calibrated, there is still bias in our streamflow simulation compared to USGS observations. The RHESSys model overestimated streamflow during low-flow days in the dormant season and underestimated flow during dry days in the growing season. The low flows caused WRTDS to significantly overestimate nitrate concentration at the lowest flows but had little effect on the annual nitrate load (< 0.1 kg/ha/year). The lack of simulated sewer inflow and infiltration (I&I) in our current RHESSys model possibly caused the bias in streamflow estimation. Infiltration into sanitary sewers may be sufficient to cause significant bypass of the stream gauge

(Bhaskar & Welty, 2012), and reduce streamflow when groundwater tables are high. On the other hand, exfiltrating effluent from sanitary sewers may increase groundwater flow and recharge (Delesantro et al., 2022), increasing subsurface nutrient delivery to streams. Using WRTDS to estimate nitrate flux change for each restoration scenario encompasses all sources and retention effects. However, WRTDS is not able to evaluate the effects of LID/GI on soil water and biogeochemical processes, especially plant uptake. Therefore, we used RHESSys to better capture the changes in uptake and denitrification caused by LID/GI, and their effects on watershed nitrate loads. For example, RHESSys and WRTDS both suggested similar increases in instream nitrate load increases in scenarios (i.e., RBS and PR) with infiltration-based LID/GI. This suggested that WRTDS was able to capture changes in-watershed nitrate dynamics due to changes in infiltration and soil moisture. In contrast, the large effects of reforestation on water and nitrate dynamics could not be simulated in WRTDS, so the ecohydrological detail of RHESSys was needed to quantify the effects of the reforestation scenarios. The two models were thus able to complement each other and overcome their individual limitations. The analytical framework would be improved with better understanding and simulation of sewer inflow and effluent. There is a clear need for additional measurement and modeling of sewer I/I and leakage.

We also note various LID/GI scenarios significantly affected groundwater, and that the assumption that nitrogen leakage from sewers is consistent and independent of changes in groundwater and biogeochemical processes induced by LID may not hold. A better understanding of how engineered and natural systems interact would improve our ability to model and managed urban watersheds. The combination of “green” and “grey” solutions could also better address the urban flashy flow and nitrate export regimes.

3.7 CONCLUSIONS

Addressing our research questions, our results indicated that all tested terrestrial restoration methods could mitigate stormflow yield independent of significant detention storage modification, but only modestly. Large scale replacement of lawns with tree cover in URF and URF+RBS scenarios promoted transpiration from increased tree canopy and lowered soil saturation levels, with the exception of periods just before deciduous tree leaf-out. The URF+RBS scenario allowed additional infiltration and evapotranspiration of forest and road drainage, mitigating flood severity the greatest, especially during growing seasons when intense convective storms occur frequently in Baltimore. Infiltration-based LID/GI types (i.e., RBS and PR) increased infiltration which mitigated stormflows but increased baseflow.

Results pertaining to on- and off-site impacts on urban biogeochemical retention of nitrogen varied by LID/GI scenarios. URF and URF+RBS scenarios mitigated nitrate export by lowering streamflow and increasing vegetation uptake. Infiltration-based RBS and PR scenarios increased nitrate loads from enhanced baseflow regimes partially offset the nitrate reduction from stream restoration but promoted nitrification and denitrification significantly. Nitrate retention from denitrification and vegetation uptake were both an order of magnitude greater than the retention from streamflow change and stream restoration. Replacing lawns with forests also displaced fertilization nitrate sources. The significant increase of nitrate uptake from reforestation and infiltration-based LID/GI suggests increasing tree coverage and LID/GI could be efficient in retaining nitrate within watersheds and reducing load export in streams than stream restoration. Sanitary sewer effluents are assumed unchanged after any types of LID/GI restoration and imperfectly modeled in this study, which introduced uncertainties to instream nitrate flux estimation. Further studies are needed to better understand and model these processes to balance nitrogen fluxes within watersheds.

Lastly, we found that retrofit of *existing development* by implementation of increased tree canopy, roadside bioswales, and permeable roads, independent of detention storage, in the medium development intensity watershed did not shift either flow or nitrate export regime regimes significantly. Specifically, their capacity to neutralize urban stream syndromes (e.g., higher peak flow, channel erosion, higher nutrient export, etc.) may be limited. Therefore, other types of facilities and efforts (e.g., detention ponds, reduced septic and sanitary sewer leakage, etc.) that have greater capacity to regulate stormflows from surface runoff and subsurface nitrogen load sources are needed to be coupled with terrestrial LID/GI and aquatic restoration to not only shift the flow and nitrate export regimes toward natural and pre-urbanization conditions and but also increase the resiliency and ecosystem health of urban watersheds in the future.

3.8 FIGURES AND TABLES

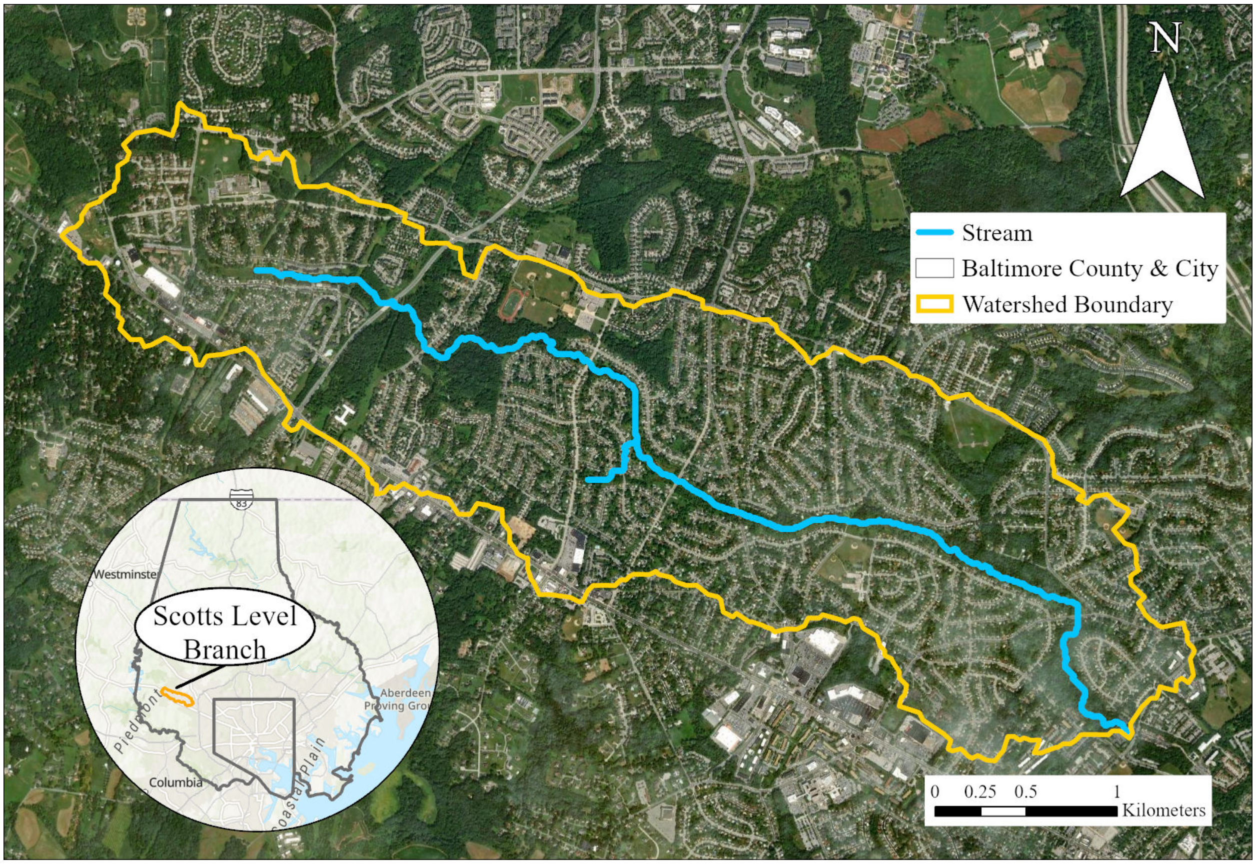

Figure 3.1. Scotts Level Branch (SLB) watershed and its delineated stream (see 3.4.1)

Table 3.1. Description of rules for changing land use and cover and details of sizes converted in SLB

Scenario	**Details**	**Area Converted (km^2)**
Status Quo (SQ)	The current land use and cover (LULC) condition	-
Urban Reforestation (URF)	Change lawns that are more than 10m far from buildings to forest	1.20
Roadside Bioswale (RBS)	Convert the 2m zones (inward) of both sides of current road to bioswale with shrub	0.26
Permeable Roads (PR)	Convert all roads to permeable pavements with K_{sat} = 5	0.78
Urban reforestation and roadside bioswale (URF+RBS)	Combination of urban reforestation and roadside bioswale	1.46

Note: No soil change was considered in RBS. PR increased the saturated hydraulic conductivity (K_{sat}) to 5 for all roads within the study watershed.

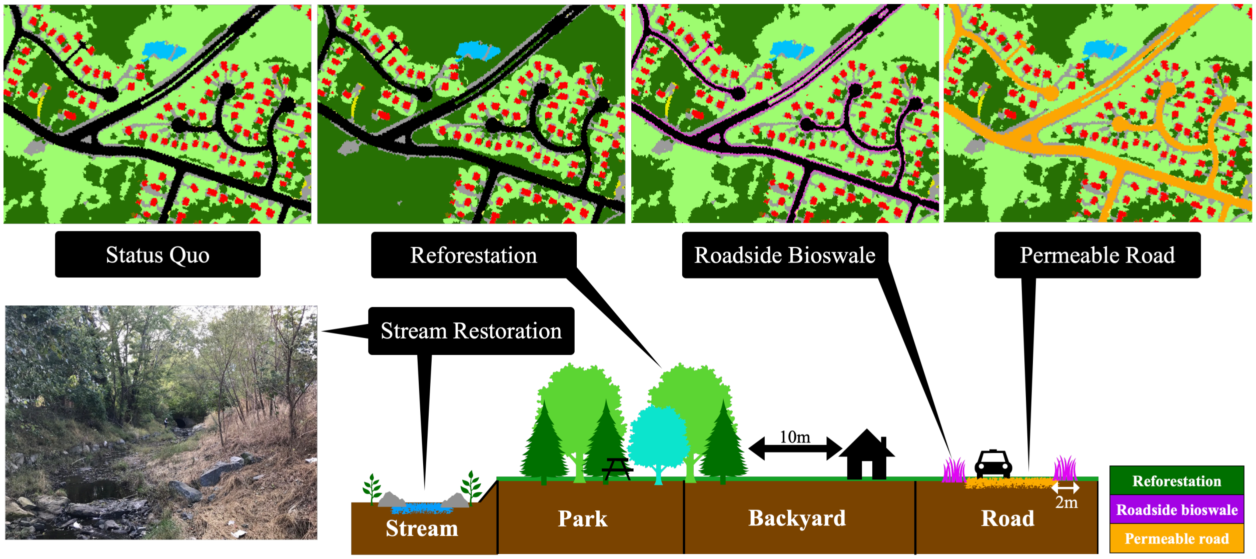

Figure 3.2. Stream restoration and 1-meter LULC implementing low impact development scenarios: reforestation (URF), roadside bioswale (RBS), and permeable road (PR), in Scotts Level Branch. URF+RBS is the combination of URF and RBS. The photo on the lower left corner shows a finished stream restoration project in Scotts Level Branch

Table 3.2. Calibrated multipliers for RHESSys parameters generating the highest NSE for streamflow.

Sensitivity Parameter	**Name**	**Details**	**Value**
s	m	decay of hydraulic conductivity with depth	0.991
	K	hydraulic conductivity at the surface	0.407
	depth	soil depth	2.799
sv	m	vertical decay of hydraulic conductivity with depth	0.751
	K	hydraulic conductivity at the surface	0.835
svalt	po	pore size index	0.905
	pa	air entry pressure	1.427
gw	sat_to_gw_coeff	bypass fraction to deep groundwater	0.052
	gw_loss_coeff	groundwater storage/outflow parameters	0.189

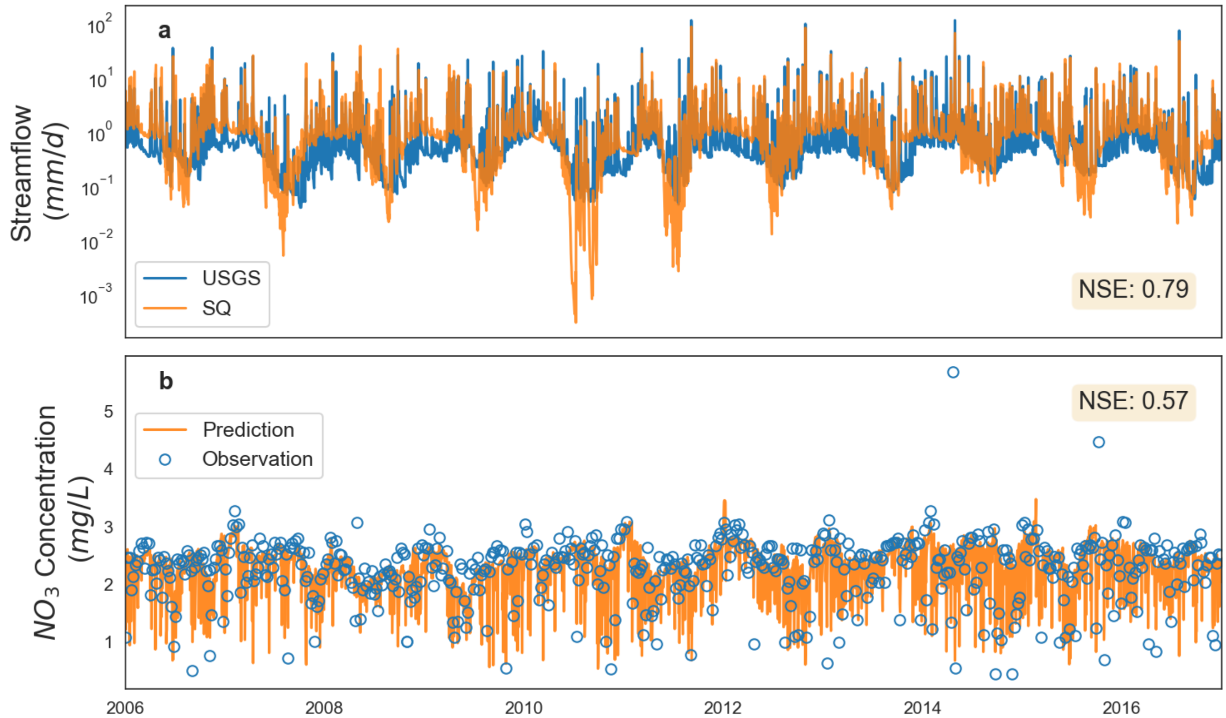

Figure 3.3. Simulation results for RHESSys streamflow at Scotts Level Branch and mean WRTDS NO3 concentration at Delight (bootstrap for 50 times)

Table 3.3. Comparison of mean streamflow (mm/day) at percentile groups from SQ and LID restoration scenarios

Percentile Group	SQ	URF	RS	URF+RBS	PR
0 - 5%	0.08	0.06	0.09	0.06	0.11
5 - 25%	0.25	0.21	0.26	0.22	0.29
25 - 50%	0.66	0.64	0.68	0.65	0.72
50 -75%	1.03	1.02	1.05	1.04	1.10
75 - 90%	1.59	1.57	1.61	1.60	1.67
90 - 95%	2.91	2.89	2.92	2.89	2.92
95 - 99%	7.26	7.25	7.19	7.18	7.12
99 - 100%	24.19	23.65	23.94	23.38	23.89

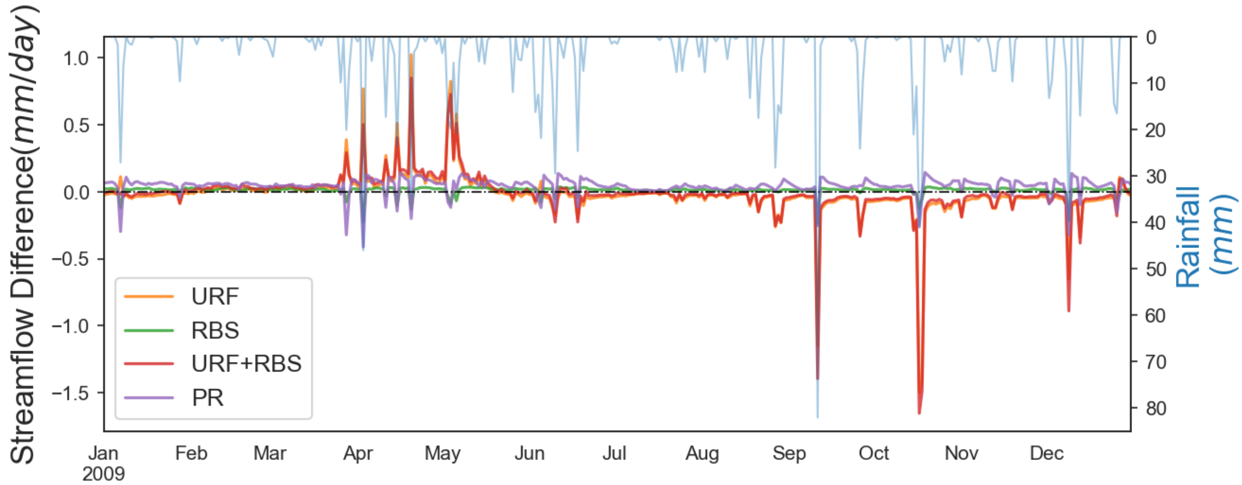

Figure 3.4. Scotts Level Branch's rainfall and difference of streamflow between status quo and other LID scenarios in 2009

Table 3.4. RHESSys derived Denitrification and uptake rate at the whole watershed, riparian, and LID/GI areas before (SQ) and after terrestrial LID/GI restoration.

Nitrogen Pathway	Location	**Scenario Rate (kg N/ha/year)**				
		SQ	URF	RBS	URF+RBS	PR
Denitrification	Watershed	18.2	12.1	20.2	13.5	22.7
	Riparian Area	34	20.4	36.9	22.8	41.4
	URF Patches	21.5	14	-	16	-
	RBS Patches	13.1	-	19.4	15.9	-
	PR Patches	13.2	-	-	-	27.1
Nitrogen Uptake	Watershed	62.4	70.1	62.6	70.4	62.8
	Riparian Area	89.7	96.5	89.9	96.9	90.1
	URF Patches	56.7	71.1	-	71.5	-
	RBS Patches	38.5	-	38.9	46.4	-
	PR Patches	38.5	-	-	-	39.3

Note: Results for scenarios without designed types of LID/GI implemented are masked.

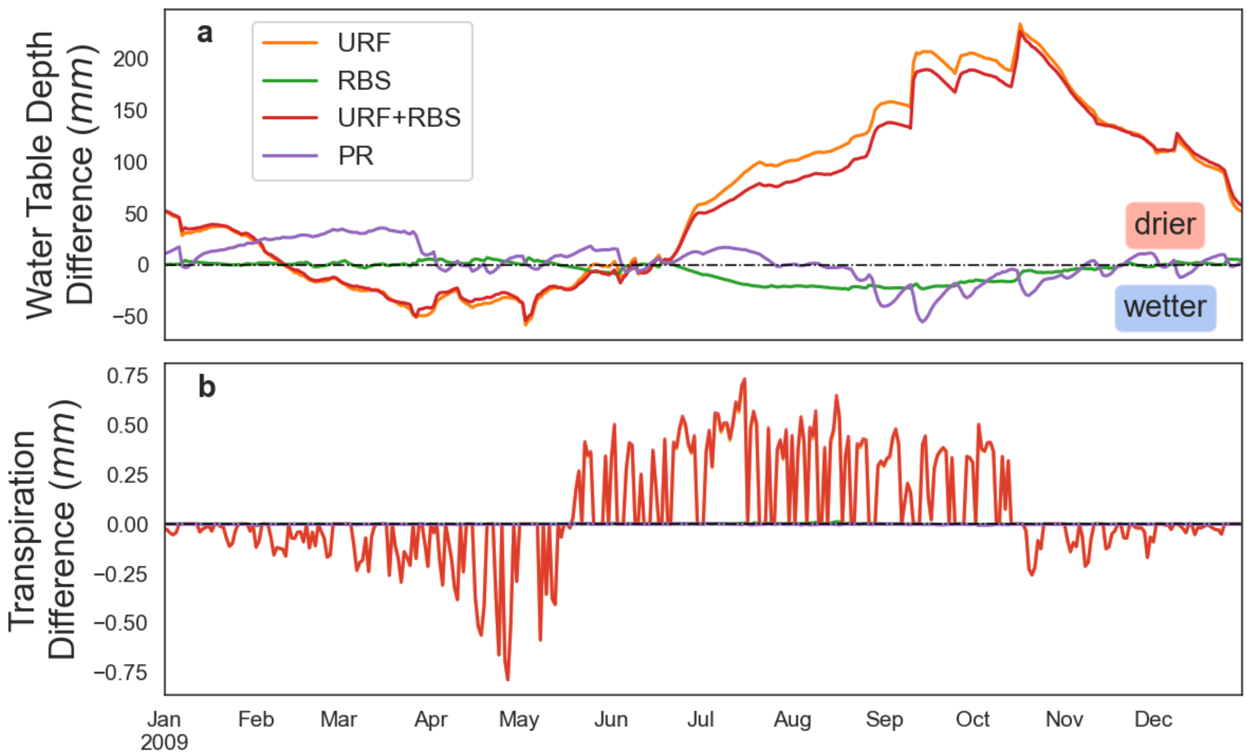

Figure 3.5. Difference of (a) water table depth and (b) transpiration for all scenarios in 2009 at Scotts Level Branch. Transpiration difference for RBS and PR are almost negligible compared to URF and URF+RBS

Table 3.5. NO_3 budget component fluxes and load reduction (kg N/ha/year) from current condition (SQ) and corresponding load changes in four pathways for nitrate retention in restoration scenarios.

Model simulation from	**Pathways**	**NO_3 load**	**Annual NO3 load reduction (kg N/ha/year) of restoration scenarios from SQ**			
		SQ	URF	RBS	URF+RBS	PR
WRTDS	Flux	9.01	0.12	-0.11	0.04	-0.28
RHESSys	Fertilization	18.29	8.13	0.00	8.13	0.00
	Denitrification	18.20	-6.10	1.92	-4.70	4.46
	Nitrification	51.87	21.40	-3.28	18.91	-7.27
	Uptake	62.35	7.68	0.17	7.98	0.44
	Stream Restoration*	0.31	0.31	0.32	0.31	0.32
	Total Change	-	31.54	-0.98	30.67	-2.33

Note: Expected absolute reductions of NO_3 load by restoring all streams in SLB are reported in "Stream Restoration*" row. Full-watershed stream restoration has not yet achieved in SQ, and we reported reductions with corresponding F75 for LID/GI scenarios assuming all streams are restored with open-canopy condition (Eq. 1, Zhang et al. 2022).

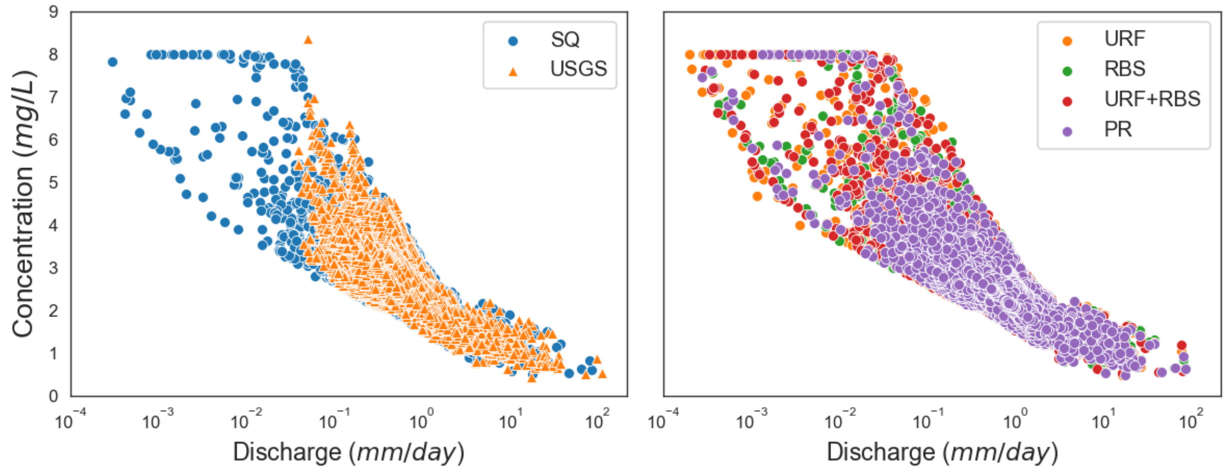

Figure 3.6. Simulated relationship between concentration and discharge (c-Q) from WRTDS using USGS streamflow records and SQ streamflow simulations (left) and restoration scenarios simulations (right)

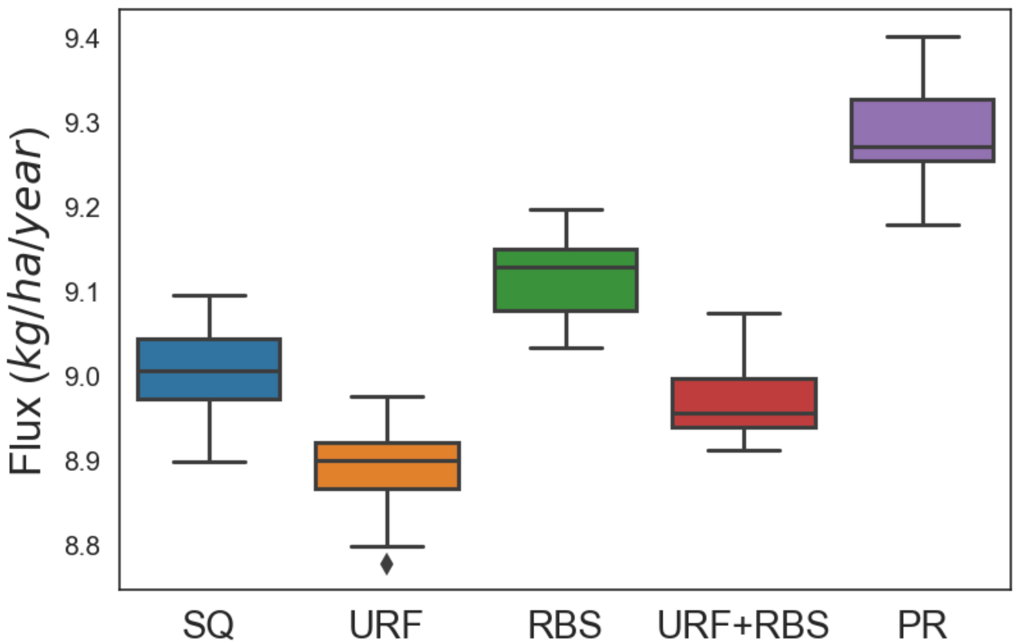

Figure 3.7. Estimation of mean annual flux load of NO_3 (kg N/ha/year) for each scenario, simulated from WRTDS and corresponding streamflow simulation of RHESSys

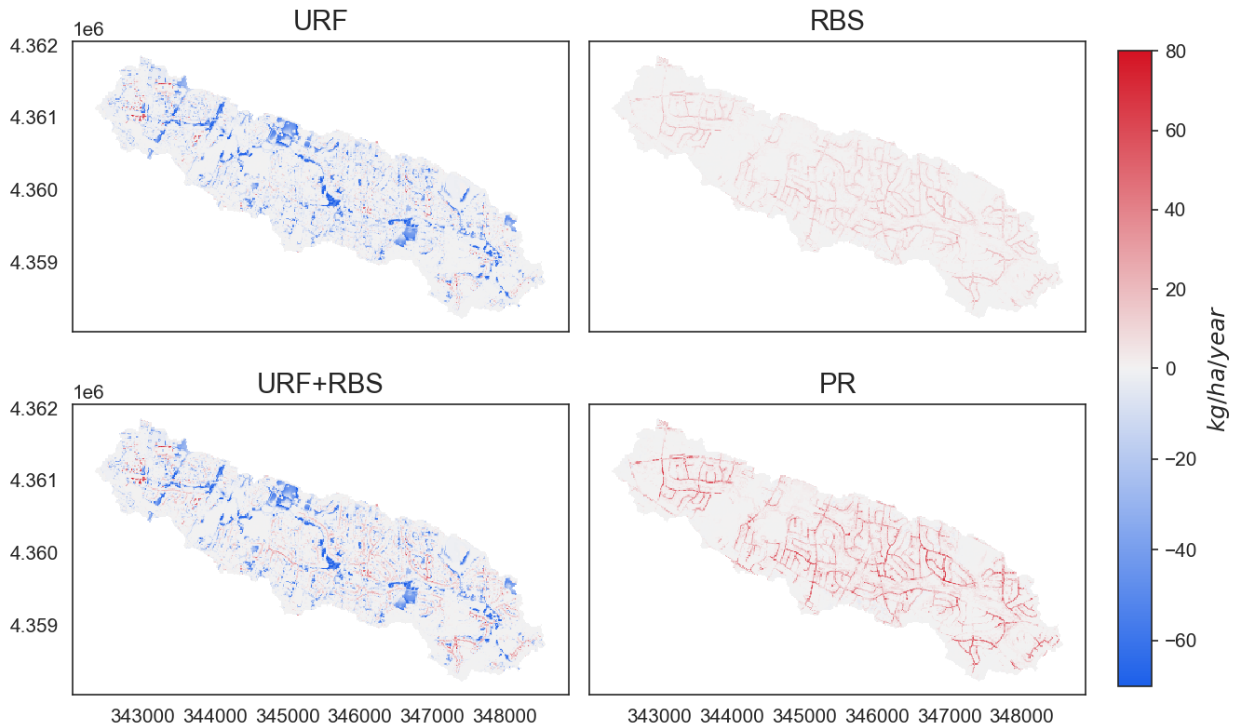

Figure 3.8. Denitrification change (kg N/ha/year) from SQ at each patch (15 m) in Scotts Level Branch

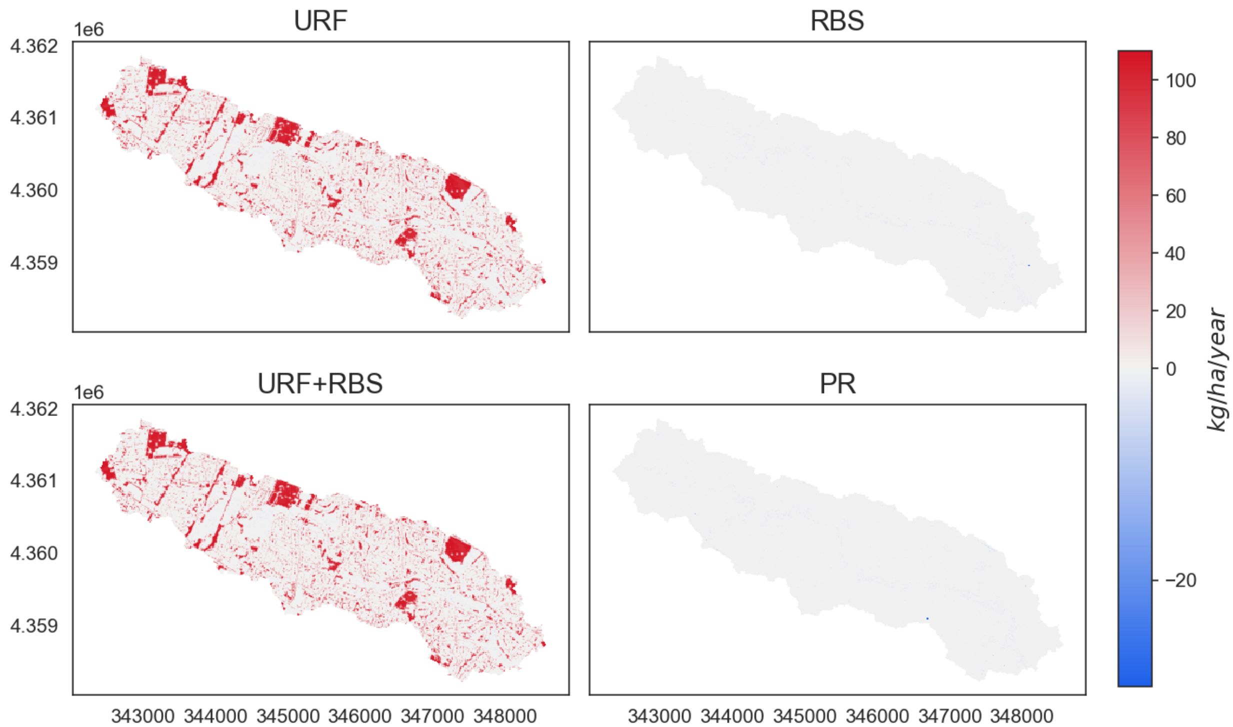

Figure 3.9. Uptake change (kg N/ha/year) from SQ at each patch (15 m) in Scotts Level Branch

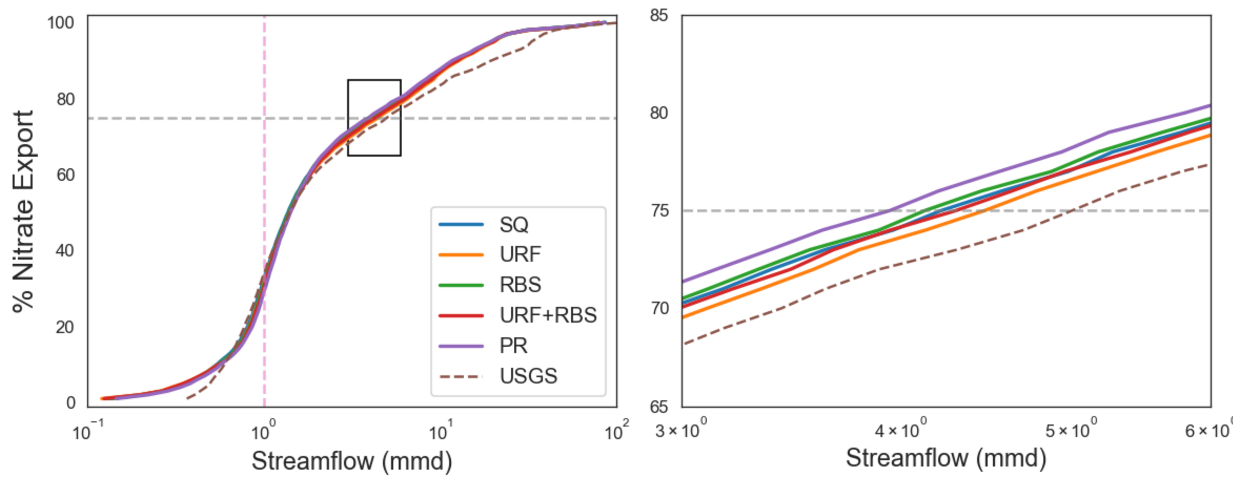

Figure 3.10. Nitrate load duration curve for all scenarios (left) and detailed zoom-in for F65 to F85 (right)

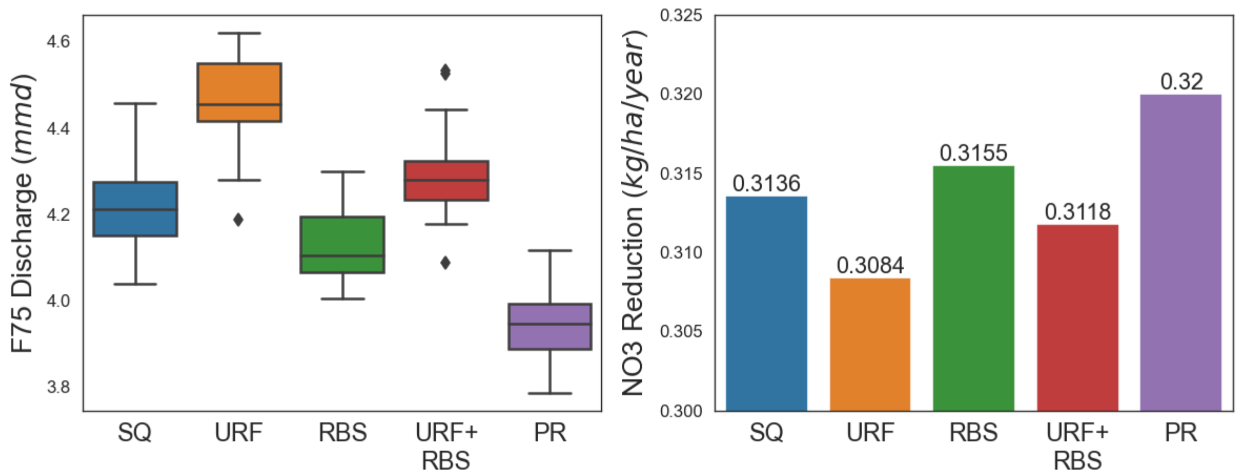

Figure 3.11. Estimation of F75 (left) and corresponding nitrate reduction (kg N/ha/year, right) from stream restoration under open-canopy condition

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Chapter 4: SIMULATION OF ECOHYDROLOGICAL HOT SPOTS OF NITROGEN IN A SUBURBAN WATERSHED

4.1 PREFACE

This chapter was jointly authored by myself, Drs. Larry Band (University of Virginia), Peter Groffman (Cary Institute of Ecosystem Studies), Amanda Suchy (Central Michigan University), Jonathan Duncan (Penn State), and Arthur Gold (University of Rhode Island). The data used in the study are acquired from Baltimore Ecosystem Study funded by National Science Foundation. The manuscript will be submitted for publication soon.

4.2 ABSTRACT

Excess nitrate (NO_3^-) export from urban watersheds is a major source of water quality degradation and threatens the health of downstream and coastal waterbodies. Ecosystem restoration and best management practices (BMPs) can be introduced to reduce in-stream NO_3^- loads by promoting vegetation uptake and denitrification on uplands. However, accurately evaluating the effectiveness of these practices and setting regulations for nitrogen inputs requires an understanding of how human sources of nitrogen interact with ecohydrological systems. We evaluate how the distribution of nitrogen sources and the transport and transformation processes along hydrologic flowpaths controls nitrogen cycling, export, and the development of hot spots of nitrogen flux in suburban ecosystems. We chose a well-monitored suburban watershed, Baisman Run in Baltimore, Maryland, to evaluate patterns of in-stream NO_3^- concentrations and upland nitrogen-related processes in response to three common activities: irrigation, fertilization, and on-site sanitary

wastewater disposal (septic systems). We augmented a distributed ecohydrological model, RHESSys, with estimates of these additional loads to improve prediction and understanding of the factors generating stream NO_3^- concentrations and upland nitrogen cycling. The augmented model predicted discharge-weighted NO_3^- concentrations of 1.37 mg NO_3^--N/L, compared to observed 1.44 mg NO_3^--N/L and the model's prediction of 0.28 mg N/L without the additional loads. Estimated denitrification rates in grass lawns, a dominant land cover in suburban landscapes, were also in the range of measured values. Interestingly, the highest denitrification rates were downslope of lawn and septic locations in a constructed wetland, and at a sediment accumulation zone at the base of a gully receiving street drainage. These locations illustrate the development of hot spots and for nitrogen cycling and export in both planned and "accidental" retention features. Appropriate siting of best BMPs and the identification of spontaneously developed nutrient control points should be pursued to retain nutrients and improve water quality.

4.3 INTRODUCTION

Nitrogen (N) and carbon (C) are fundamental elements for ecosystem functions and are influenced by multiple factors including climate (Campo & Merino, 2016; Crowther et al., 2016), moisture and other soil properties (Pastor & Post, 1986; Wang et al., 2020), plant and microbial community composition (Chen et al., 2003), and human activities (Galloway et al., 2008). They are also influenced by the state and pattern of drainage flowpaths as different forms of C and N are mixed and transported to distinct edaphic conditions, potentially forming "hot spots" (McClain et al., 2003) that have a disproportionate influence on landscape and watershed scale biogeochemical cycling functions. Understanding mechanisms of C and N cycling and interactions with hydrologic processes is necessary to design and implement efficient ecosystem service restoration strategies. In urban and suburban ecosystems, human disturbance to biogeochemical cycling has led to air and water quality degradation and created a need for best management practices (BMPs) to improve local and downstream water quality, increase C and N retention, and promote ecosystem resilience to prepare for extreme weather events with changing climate. Therefore, gaining a comprehensive understanding of the ecohydrological behaviors and interactions between ecosystems and human activities can lay the foundation for effectively mitigating these environmental issues through well-conceived and sustainable management practices.

Several ecohydrological models have been developed to understand and quantify individual or integrated ecohydrological processes in unmanaged to highly managed ecosystems. Semi-distributed hydrologic models, such as the Storm Water Management Model (Lewis A Rossman, 2010) and the Soil Water Assessment Tool (Arnold et al., 1998), are widely used in studies of urban and mixed land use watersheds (Jayasooriya & Ng, 2014; Koltsida et al., 2023; Lee et al., 2018; L. A. Rossman, 2010; Samimi et al., 2020). These models simulate water balance based on subcatchment units with similar land cover and soil. Runoff from each subunit is based on curve numbers or infiltration excess, and are independently added to streamflow. However, these models lack hillslope water and nutrient mixing along hydrologic flowpaths that are important to simulate the formation of biogeochemical "hot spots", and the potential uptake and retention of water and nutrients downslope. Patch based ecosystem models, such as Biome-BGC (Running & Coughlan, 1988; Running & Gower, 1991), $\text{CENTURY}_{\text{NGAS}}$ (Parton et al., 1996) or the Community Land Model (Oleson et al., 2008) are designed to capture 1-dimensional patch-level water balance and biogeochemical processes affecting C and N, but also lack lateral drainage through topographically mediated flowpaths. Ignoring lateral redistribution of water and nutrients within terrestrial ecosystems may generate significant bias in estimating key hydrologic and biogeochemical processes (Band et al., 1993; Fan et al., 2019).

Fully distributed hydrology models, such as MIKE-SHE (Abbott et al., 1986a, 1986b), ParFlow (Maxwell, 2013), RTM-PiHM (Bao et al., 2017; Zhi et al., 2022) and RHESSys (Tague & Band, 2004) simulate coupled surface and subsurface hydrological processes to generate distributed surface runoff, recharge, soil moisture, evapotranspiration (ET), and other ecohydrological variables. Lateral surface and subsurface drainage redistribute precipitation, resulting in gradients of water availability within a watershed from ridge to riparian areas. These models include modules for biogeochemical reaction and transport processes, which can interact with the transport and storage patterns of soil water.

Additional inputs of water (e.g. lawn irrigation and septic effluent), C (e.g., mulch, lawn amendments) and N (e.g., septic system, lawn and garden fertilization, sanitary sewer leakage) occur on discrete land segments and can be significant or even dominant components of watershed mass budgets. Lawn fertilization can contribute more than half of the total N input in urban watersheds, even if it is only applied to 20 – 30% of the landscape (Groffman et al., 2004; Band et al. 2005, Hobbie et al. 2017). Atmospheric deposition and septic system wastewater N can comprise similar input amounts at the watershed scale, but septic input is concentrated over only 1-2% of the landscape, with a large, localized volume of wastewater sufficient to result in groundwater mounding and effluent plumes extending towards local streams (Cui et al., 2016). The concentrated inputs over limited areas by septic inputs and lawn fertilization with or without irrigation creates delivery or retention patterns of N "hot spots" that provide opportunities for targeting N mitigation strategies (Groffman et al., 2023).

A spatially explicit framework that simulates interactions between C, N, vegetation, water, and human activities has important advantages to understand and manage non-point source pollutants and "hot spots" (Groffman et al., 2009; Bernhardt et al., 2017) in urban watersheds. Fully distributed ecohydrological models should have the flexibility for users to design and evaluate the effectiveness of potential management scenarios (e.g., reforestation, green infrastructure, etc.) and regulations at the scale of human activity. Landscape management and treatment at these scales occur as part of residential, commercial, and institutional use spaces, and may require direct involvement of residents and other stakeholders. Therefore, the ability to represent processes at the scale of human perception can also provide information useful for decision making and community involvement. High-resolution simulations and visualization of spatially explicit water, nutrient cycling, and transport can facilitate understanding and communication of how human activity can alter terrestrial and aquatic ecosystem functions in urban ecosystems, and contribute to participatory planning.

The Regional Hydro-Ecological Simulator System (RHESSys, Tague & Band, 2004) is an ecohydrological model that simulates spatially distributed mass balances of water, C, and N of a watershed including hydrologic and biogeochemical stores and cycling. The hydrologic component in RHESSys routes water and solutes based on topographic and infrastructure surface water flowpaths, and two-dimensional subsurface flow based on shallow groundwater gradients. Biogeochemical process rates are then estimated with modules modified from Biome-BGC (Running & Hunt, 1993) and $CENTURY_{NGAS}$ (Parton et al., 1996) models. RHESSys is therefore capable of estimating spatiotemporal patterns of soil moisture, discrete human inputs of water, lateral redistribution, and evapotranspiration. The distributed soil water content and groundwater levels interact with biogeochemical processes, canopy evapotranspiration, and other ecosystem processes. RHESSys has the flexibility to simulate at resolutions commensurate with human perception of the landscape, facilitating assessment of small-scale human activity and modification to land cover and infrastructure.

In this study, we developed and used an augmented version of RHESSys to investigate the spatial and temporal distribution of hydrologic and biogeochemical C and N cycling and export in a low-density suburban watershed. Baisman Run (BARN) is in a suburban area of Baltimore County, with all households using septic systems and well water. We ran simulations with and without human additions of water and nitrogen and compared model results to field observations for streamflow, water chemistry, and soil nitrogen cycling processes to answer the following research questions:

1) What are the individual and interacting contributions of different watershed N sources to streamwater N export?

2) How do the spatially nested patterns of water and N inputs from human activities alter spatial patterns of a set of key ecohydrological processes (e.g., N retention, evapotranspiration, soil and groundwater levels and flows)?

3) What are the emergent patterns of N cycling and retention, including “hot spots and hot moments” at sites receiving direct additional N and downslope, offsite locations receiving transported N?

4.4 Methods

4.4.1 Study area

Our study watershed (Figure 4.1), Baisman Run (BARN), is located in Baltimore County, MD, outside of the urban service boundary. The 3.8 km^2 watershed is in the Piedmont physiographic province with a rolling, locally steep landscape. Mean elevation is 170.5 m, with average slope 7.8°. The downslope areas (i.e., vertical flow distance or Height Above Nearest Drainage < 10m, 25% of the watershed) are steeper than the upslope areas, with mean slopes 9.7° and 7.2°, respectively. Meteorological records from 2000 to 2018, collected by the Baltimore Ecosystem Study (BES) at the Oregon Park weather station adjacent to BARN, have mean annual maximum and minimum temperatures of 18.9 °C and 7.9 °C respectively, and mean annual precipitation of 1,024 mm. The discharge and gage height records of BARN have been monitored by USGS (Gage ID: 01583580) since 1999.

Soils in BARN range from silt clay loam to silt loam in the riparian areas to sandy loam on steeper slopes. Forested areas are dominated by approximately 100-year-old *Quercus spp*. (oaks) and *Carya spp.* (hickory). The entire watershed is underlain by the medium- to coarse-grained micaceous schist of the Loch Raven Formation, overlain by a weathered saprolite. The saprolite thickness is highest on ridges (up to 20m), thins (< 1 m) with some bedrock outcrops at steep midslope positions, and is 1–2 m in bottomland locations (Cleaves et al. 1970; St. Clair et al., 2015). Hydraulic conductivities of soils generally decrease with depth but may locally increase into the saprolite. The saprolite may store substantial amounts of moisture, and is drained through underlying bedrock fractures through a set of emergent springs on the valley sidewall-riparian area transition, providing a fairly steady baseflow (Putnam, 2018). Dominant land cover includes forest and lawns, covering 81.5% and 14.5% of the watershed, respectively. Impervious areas cover 4.0% of the watershed, including roofs of single-family houses, driveways and roads. Lawns are located in front and backyards of households in headwaters areas of BARN. Two natural gas supply lines cut through the watershed, creating two strips of herbaceous land. In this suburban watershed all households use groundwater wells for water supply and on-site septic systems to process wastewater. Lawn and garden fertilization is another major source of N input in BARN (Law et al., 2004). Septic and fertilization N and water additions are localized on lawns and septic drain fields near houses in the BARN headwaters. Irrigation and septic effluent are derived from well water, pumping deep groundwater to shallow soils. BARN is a useful watershed for examining the interactions between human activities and watershed ecohydrological response, as the sources and disposal of domestic water are on-site without external piped inputs and outputs.

The availability of several previously collected data sets allowed us to compare simulation results to field observations. Rich ecohydrological observations and lawn management surveys (Fraser et al., 2013; Law et al., 2004) from the BES are available in BARN. The BES collected weekly water chemistry concentration data at the BARN USGS gage since 1998. In addition, a fully forested subcatchment of BARN, Pond

Branch (POBR), is also monitored weekly by the BES and USGS (Gauge ID: 01583570). POBR serves as a forest control site without human water and nutrient additions. Finally, we have previously measured N stores and cycling rates, including lawn soil NO_3^- content and denitrification measurements of BARN (Suchy et al., 2023), the University of Maryland Baltimore County (Raciti et al., 2011), and other sites in the region (Groffman et al., 2009).

4.4.2 RHESSys setup and calibration

Our study period makes use of observed and simulated watershed processes from water year 2013 to 2017 (i.e., Oct. 1, 2012 to Sep. 30 2017), with a 30-year simulation spinup period to stabilize groundwater levels, C and N pools. Inspection of the spin-up storage of soil C and N showed they were asymptotic with stable C:N ratios. The watershed is delineated using 1-m digital elevation data accessed from Maryland GIS portal (https://data.imap.maryland.gov) and r.watershed from GRASS GIS (https://grass.osgeo.org/grass82/manuals/r.watershed.html). Streams are identified when accumulated drainage areas are above 10 ha (Figure 4.1),which approximates the extension of Baltimore County's hydrology lines dataset (https://opendata.baltimorecountymd.gov/datasets/hydrology-lines). Detailed land use information is derived from the 1-m high-resolution land use and land cover (LULC) data from the Chesapeake Conservancy (https://www.chesapeakeconservancy.org/conservation-innovation-center/high-resolution-data/lulc-data-project-2022). The dataset contains "roof" as a LULC class, from which we identified 249 spatially isolated clusters of roofs within BARN. Further comparison with the Baltimore County parcel dataset (https://opendata.baltimorecountymd.gov) and latest Google Earth satellite data, we filtered out detached garages and sheds, and determined the main building to be found in each parcel. There are 181 households identified, although a set of the homes are located on the watershed divide, providing some uncertainty to the effective number of septic systems. Note that this is a larger number of households compared to Law et al. (2004) as there has been more development since that survey, and the identification of additional households along the watershed divide with more precise delineation of the watershed. Patches in centroids of these 181 main buildings were identified as "drain-in" patches. Drain-in patches were paired with "drain-to" patches, which were identified to receive septic wastewater additions and will be discussed in detail in the next section (see Method 3.2: Septic system). The riparian areas in RHESSys were defined as areas with height above nearest drainage (HAND, Nobre et al., 2011) below 1.5 meters. These areas were set to receive additional drainage from the deep groundwater system. The start and end of the growing season are hardcoded in RHESSys and vary for different vegetation species, i.e., from May 5th to Oct 2nd for tree species and Mar 31st to Sep 2nd for grass (Lin et al., 2015; 2019). Sensitivity analysis of grass growing season showed negligible impacts on ecohydrological responses as temperature becomes a limiting factor.

RHESSys requires several parameters to simulate lateral and vertical water flows within soils and topography. In this study, we calibrated eight parameters (Table 4.1) for soil properties (i.e., lateral and vertical saturated hydraulic conductivities and their decay rates, pore size index, and air entry pressure) with initial estimates from the SSURGO soils dataset (https://data.nal.usda.gov/dataset/soil-survey-geographic-database-ssurgo) and deep groundwater features (i.e., bypass seepage from shallow saturated zones, and drainage rate to stream). The calibration was performed with daily USGS discharge records, and the parameter set yielding the highest Nash-Sutcliffe efficiency (NSE, Nash & Sutcliffe, 1970) was used to simulate ecohydrological processes in this study. Our meteorological forcing data were acquired from a local weather station operated by the BES at Oregon Ridge Park just outside the BARN watershed boundary.

4.4.3 Human additions of water and nitrogen

We included estimates of fertilization, onsite wastewater disposal from septic systems, and irrigation, as input to RHESSys to incorporate water and N management decisions and capture how such activities affect water and N cycling and export within the study watershed.

4.4.3.1 Fertilization

The lawn fertilization module in RHESSys allows users to determine the fertilization rate and when and where applications are applied to lawns. In this study, we assumed all households in BARN fertilize three times with a 60-day interval between applications from end of March on their lawns. This fertilization frequency is consistent with our prior household surveys and similar to results of surveys conducted in other suburban communities (Carrico et al., 2013; Martini et al., 2015). Law et al. (2004) and Fraser et al. (2013) conducted in-person household surveys in a set of neighborhoods in the Baltimore area, including BARN, and found mean annual total fertilization rates on lawns ranging from 3.7 to 13.6 g N/m^2. Both surveys were conducted during significant drought conditions (2002 and 2008) when lawncare was reduced due to groundwater supply concerns. Hence, we consider the survey results may be on the lower end of actual rates. In this study, we used the two rates reported in Law et al. (2004), 3.7 and 8.4 g

N/m^2 on lawns for BARN and for a denser suburban site (or 5.5 and 12.4 kg N/ha/year at watershed scale), respectively, to account for how uncertainties of fertilization rates impact the in-stream and upland N dynamics in BARN.

In the model, applied fertilizer is stored in an independent pool of each lawn patch, and each day we assumed a fixed fraction of available nutrients in the fertilizer pool leaching to other pools, of which 80% is dissolved to detention storage and 20% to soil. The daily fraction is determined by the fertilization interval (FI), following Equation 1:

$$Fraction = -\frac{\log 0.1}{FI} \qquad \text{(Equation 1)}$$

In our case study, our fertilization interval, 45 days, results in 2.2% of nutrients in the fertilization pool transported to other pools per day and then stored, consumed by vegetation, immobilized, or further transported to groundwater and downslope. In this study, we considered fertilizer input only contains NO_3^-, following sensitivity analysis that found varying NO_3^- and NH_4^+ proportion had negligible impacts. Phosphorous fertilizer is not considered as RHESSys currently does not simulate the phosphorous cycle.

4.4.3.2 Septic system

All households within BARN use septic systems to disperse wastewater. Wastewater from a house is released first to septic tanks for settling, then to drain fields which are typically placed downslope of the house. Therefore, soils in specified, downslope areas receive additional water and N input from septic effluents and may become biogeochemical cycling “hot spots”. Using prior studies, we estimated the nitrogen load from septic systems as 7.7 kg N/capita/year and water input as 110.5

m^3/capita/year (~80 gal/capita/day), resulting in a NO_3^- concentration 70 mg N/L which is comparable to Gold et al. (1990) and Lowe et al. (2009). We set the average number people per household as 3.3 for these single-family houses based on survey results from Law et al. (2004) and census information. Applying these water and NO_3^- loads for 181 houses in BARN results in an additional 2,541 kg N/ha/year of NO_3^- input on septic fields and total of 4,599 kg N/year into the watershed. At the watershed scale, 12.0 kg N/ha/year of NO_3^- and 0.08 mm/day (110,058 m^3/year) of demand for septic source water (SSW_{demand}) from drain-in patches level are added. Septic water and N loads are currently set to be evenly distributed every day.

Septic source water is drawn from drain-in patches (i.e., centroid patches of main buildings) and transported to detention storage in septic drain-to patches (Figure 4.2) which are the drain fields of septic systems, defined as the closest downslope lawn patches to drain-in patches. We regulated actual water withdrawal of septic source water (SSW_{actual}) to not exceed the available water in surface detention (pond) and groundwater storage:

$$SSW_{actual} = min[x \cdot SSW_{demand}, SD_{storage}] + min[(1 - x) \cdot SSW_{demand}, GW_{storage}]$$

(Equation 2)

where x is the user-defined fraction of water withdrawal from surface detention storage, and $SD_{storage}$ and $GW_{storage}$ are available water in surface detention and deep groundwater storage of hillslope at drain-in patches (Figure 4.2). Users can flexibly modify the fraction of water withdrawn from drain-in patches' surface detention and groundwater storage. There is only one pond in BARN that is occasionally used for irrigation, and we set $x = 0$ in BARN assuming all water is sourced from wells. The source water is added to septic drain-to patches (orange arrow in Figure 4.2), where it is further subject to hydrological and biogeochemical processes. Nutrients are also added to drain-to patches' storage, depending on deep groundwater concentrations and withdrawal rate.

4.4.3.3 Irrigation

Although irrigation practices and quantities vary significantly among households, irrigation is commonly applied during the growing season, and especially during dry and hot conditions. Therefore, we designed a mechanism to determine the total irrigation amount based on water stress of grass. Specifically, the amount of irrigation applied on lawns is determined by a water stress factor (*WSF*):

$$WSF = \frac{PET-ET}{PET} \qquad \text{(Equation 3)}$$

where PET and ET represent patch level potential and actual ET. During continuously hot and dry days, WSF would be high due to low soil water content (lower ET) and high atmospheric demand for water (higher PET). Our model then activates the irrigation function and calculates the demand of irrigation for patches modulated by water shortage. This function effectively modulates soil water conditions by the addition of groundwater sourced irrigation.

Unlike the septic source water (SSW_{demand}) which is fixed each day, the daily demand for irrigation source water (ISW_{demand}) for a lawn patch is then modified by the water stress factor (*WSF*) in Eq. 3:

$$ISW_{demand} = IR_{max} \cdot WSF \cdot lawn\% \qquad \text{(Equation 4)}$$

where IR_{max} is the user-defined maximum daily irrigation rate, and $lawn\%$ is the fraction of grass in an irrigated patch. We defined the maximum irrigation rate (IR_{max}) in

BARN as 4 mm/day in the current model, which can be modified based on the local practices or for sensitivity analysis. Like septic source water, withdrawal of irrigation source water cannot exceed available water in detention and groundwater storages. The actual irrigation source water is calculated identically as the actual septic source water based on Eq. 2, in which the SSW_{demand} is replaced by ISW_{demand}. The irrigation amount is pumped from surface detention and/or deep groundwater storages of drain-in patches (i.e., centroids of houses in Figure 4.2), with user-defined ratio from each storage, irrigating lawns around houses. We currently set no irrigation source water is withdrawn from surface detention (i.e., $x = 0$); instead, as with septic source water, irrigation water is fully withdrawn from deep groundwater storage in BARN. Irrigated lawns are identified within 50 m from houses, covering 33.7 ha (60.6%) out of 55.7 ha of lawns in BARN, consistent with observations of the proportion of lawns fertilized and irrigated.

4.4.4 Scenarios and "hot spots"

We focus on evaluating changes of NO_3^- dynamics in stream and upland areas when additional NO_3^- is added from fertilization and/or septic systems with four scenarios (Table 4.2): *none* (no fertilization or septic inputs), *fertilization only*, *septic only*, and *both* (fertilization and septic inputs) – to our study watershed. Irrigation is activated in all scenarios, including our reference control scenario "*none*" to emphasize NO_3^- dynamics without residential N inputs. With the current setting, scenario *both* receives a total of 35 kg N/ha/year N deposition, with 11 (31.4%), 12 (34.3%), and 12 (34.3%) kg N/ha/year from atmospheric deposition, fertilization, and septic effluents, respectively, expressed at the watershed level. We resampled the daily simulated NO_3^-

concentration from RHESSys to weekly averages for comparison with the sampled weekly water chemistry from BES for BARN.

We further evaluated changes in ecohydrological processes at potential on-site “hot spots” (e.g., residential lawns and septic drainage fields) receiving direct human water and nitrogen inputs as well as “off-site” potential “hot spots” located downslope areas that receive human water and nitrogen inputs added upslope (e.g., riparian areas and wetlands). Lawns are identified as patches with more than 50% of grass, and downstream forests are patches with more than 50% of forest in the residential area of BARN. One offsite location is a constructed wetland (upper red circle in Figure 4.1), while the other spontaneously developed as an “accidental wetland” (Palta et al., 2017) in an area receiving road drainage and gully sedimentation, and is referred to as a “sedimentation accumulation zone” (lower red circle in Figure 4.1).

4.5 RESULTS

4.5.1 Ecohydrological responses

Calibration of streamflow simulations (Figure 4.3) with irrigation, fertilization and septic input (scenario *both*) produced a maximum Nash-Sutcliffe value of 0.70 from water year 2013 to 2017 (Oct 1st, 2012 to Sep 30th, 2017) with calibrated parameter values listed in Table 4.1. The mean of simulated streamflow was 1.13 mm/day, which is slightly lower than the 1.16 mm/day mean observed runoff at the USGS gage. Our model tended to underestimate the lowest flows compared to streamflow observations, with mean simulated growing season (from May to September) streamflow of 0.90 mm/day which is 0.19 mm/day lower than the USGS records, 1.08 mm/day.

Mean streamflow was decreased by only 0.01 mm/day by adding septic processes as this addition increased ET during the growing season (comparing to scenario *none,* Figure 4.4 - upper). The increase in ET was associated with an increase in net photosynthetic rates during the growing season of 0.01 g C/m^2 (comparing scenario *none*, Figure 4.4 - lower). No change of streamflow or ET (< 0.01 mm/day) was found when only fertilization was activated.

4.5.2 Improved prediction of NO_3^- export

Turning fertilization and septic processes on and off in the model produced variation in in-stream NO_3^- concentration and load simulations (Figure 4.5). In our 5 - year study period, the mean streamflow-weighted NO_3^- concentrations for scenarios *none* and *septic only* were 0.29 and 0.72 mg NO_3^--N/L. With the two reported fertilization rates from Law et al. (2004), the mean streamflow-weighted NO_3^- concentrations for scenarios *fertilization only* and *both* were 0.53 and 1.01 mg NO_3^--N/L at the lower fertilization rate of 3.7 g NO_3^--N/m^2, and 0.84 and 1.37 mg NO_3^--N/L at the higher rate of 8.4 g NO_3^--N/m^2, respectively. All results and discussion from this point would be reported based on the higher fertilization rate.

Compared to the mean streamflow-weighted long-term observed NO_3 concentration of 1.44 mg NO_3^--N/L at the BARN USGS gauge, the simulated mean in-stream NO_3 concentration after considering fertilization at the higher rate and septic loads is only 5% lower. Specifically, after considering both N inputs from fertilization and septic processes (i.e., scenarios *both*), we substantially reduced the daily bias to -0.07 (-5%) mg NO_3^--N/L. In-stream NO_3^- load followed a similar trend as concentration. For scenarios *none*, *fertilization only*, *septic only*, and *both*, mean annual NO_3^- loads (Table

4.3) were 1.2, 3.6, 3.1, and 5.9 kg NO_3^--N/day compared to the 7.1 kg NO_3^--N/day estimated from measurements at the gage. The NO_3^- retention rate varied across different scenarios ranging from a high of 89% is scenario *none* (atmospheric deposition only) to a low of 84% in scenario *both*. In scenario *septic only*, retention rate was 87%, and in scenario *fertilization only*, retention was 85%.

At seasonal scales (Figure 4.6), the model produced similar ranges of NO_3^- concentrations in spring and winter but a greater range of NO_3^- concentration in summer and fall compared to the BES observations. The mean simulated NO_3^- concentrations of scenario *both* were lower in all seasons by -0.28 (-18.7%), -0.50 (-31.3%), and -0.30 (-19.1%) mg NO_3^--N/L in spring, summer, fall, and winter than BES weekly records (Table 4.3), respectively. For simulated NO_3^- load, scenario *both* underestimated in all seasons by -2.53 (-24.3%), -1.88 (-33.5%), -0.38 (-8.4%), and -0.09 (-1.1%) kg NO_3^--N/day in spring to winter compared to the load records calculated from BES concentration and USGS discharge observations (Table 4.3).

4.5.3 Ecohydrological and biogeochemical responses at "hot spots"

In our simulations, fertilizer is slowly released to soil and surface detention and transported downslope. This transport is augmented by irrigation and septic fields. As a result, water and NO_3^- are redistributed through other patches along subsurface hydrological flowpaths, providing "off-site" ecohydrological and biogeochemical responses downslope and across the whole watershed.

4.5.3.1 Soil Moisture and ET

The average water table depth (Figure 4.7) in scenario *none* was 4.75 m during the study period. Fertilization had negligible effects on soil moisture or water table depth compared to the base (*none*) scenario. However, septic processes decreased water table depth to 4.68 m (by -0.06 m, -1.3%) by groundwater mounding, which increases shallow groundwater flow to surrounding patches along connected flowpaths. Specifically in septic drainage field patches, the mean water table depth decreased to 3.64 m (-0.77 m, -17.4%) in scenarios *both* and *septic only* compared to the mean depth of 4.41 m, in scenarios *none* and *fertilization only* (Figure 4.7). Setting hillslope groundwater as the only source for septic process, we found groundwater withdrawal resulted in drier conditions (i.e., increase of water table depth) in riparian areas where the mean water table depth increased to 0.22 m (+0.01 m, +4.7%) in scenarios *both* and *septic only* compared to 0.21 m depth in scenarios *none* and *fertilization only*.

The watershed-scale ET were 42.1 mm/month in scenarios *none* and *fertilization only*, and 42.2 mm/month in scenarios *septic only* and *both*. As the result of higher soil moisture level after activating septic processes in scenario *both*, ET in lawn patches and septic drainage fields increased to (by) 39.3 (+0.2, 0.5%) and 40.7 (+7.7, 23.3%) mm/month, compared to the levels in scenarios *none* or *fertilization only*, respectively. ET at riparian areas were 54.0 and 54.1 mm/month in scenarios *none* and *septic only*; With fertilization activated in scenarios *fertilization only* and *both*, riparian ET dropped slightly by about 0.1 (0.1%) mm/month, possibly due to the greater vegetation growth and higher ET at upland areas.

4.5.3.2 Denitrification

Our model suggested significant changes in denitrification after including additional NO_3^- Inputs from fertilization and septic processes. The mean annual rates (Figure 4.8) of denitrification at the watershed scale were 12, 12.4, and 14 kg N/ha/year in scenarios *fertilization only*, *septic only*, and *both*, respectively, increasing by 20.8%, 24.5%, and 41.3% compared to the rate of 9.93 kg N/ha/year without fertilization and septic processes (i.e., scenario *none*). There were a few locations with reduced denitrification after adding fertilization and septic processes, but only 0.57% of patches (220 out of 38,263 patches) of the watershed experienced notable decreases (> 5%), with mean rate dropping from 4.6 to 4.1 kg N/ha/year. From these patches, we further identified 19 near-stream patches (i.e., HAND < 3 m) and found that they all experienced substantial water table drops (11 mm average reduction) with septic processes due to upland groundwater extraction.

Denitrification rates increased significantly in "hot spots" – lawn, septic drainage field, and riparian areas (Table 4.4). Compared to scenario *none*, scenario *fertilization only* had higher denitrification rates than scenario *septic only* in lawns and riparian areas, except for septic drainage patches where the annual denitrification rate with only septic processes (i.e., scenario *septic only*) was almost 3-fold higher (+210%) than the reference scenario *none.* There was a 40% increase with scenario *fertilization only* compared to the reference scenario *none*.

Fertilization and septic processes added more than 20 kg N/ha/year load at the watershed level and concentrated at the upland residential areas. These additions increased mean denitrification rates in forest patches in and below residential areas (i.e., excluding patches in Pond Branch) by 45.2% (Table 4.4). The annual denitrification rates in the sedimentation accumulation zone (upper red circle in Figure 4.1) showed a significant increase after activating fertilization and septic processes, reaching values of 73.8 kg N/ha/year before and 99.2 (+25.4, 34.4%) kg N/ha/year after activation.

Similarly, denitrification rates in the constructed wetland (lower red circle in Figure 4.1) increased from 79.3 kg N/ha/year before to 101.4 (+22.2, 28.0%) kg N/ha/year after activation.

Changes in denitrification varied among seasons (Table 4.4). At the watershed scale and in all "hot spots", "hot moments" of denitrification rates were generally found in spring and summer, followed by fall, and lowest in winter. The greatest increases (%) in denitrification at all locations were in spring when fertilizer is applied to lawns. Riparian areas had significant increase of denitrification increase in winter when the watershed receives sustained NO_3^- input from septic effluents.

Our modeled denitrification rates are consistent with measurements from field studies in Baltimore. Assuming 210 days (~7 months) that denitrification would occur, Raciti et al. (2011) reported a denitrification rate of 204 kg N/ha/year at 20 °C from saturated soil samples from fertilized lawns at the University of Maryland Baltimore County. At the same temperature, Suchy et al. (2023) reported a higher rate, 744 kg N/ha/year, when lawn soil samples collected from BARN were saturated. We further interpolated the two rates based on the method from Raciti et al. (2011), assuming 5% storm (i.e., saturated soil) and 95% dry (i.e., low-soil-moisture) days with 2.95 kg N/ha/year rate in a year. The projected annual denitrification rates were 13 and 40 kg N/ha/year with rates at saturated conditions from Raciti et al. and Suchy et al, which are very similar to estimates of annual denitrification from our simulated scenarios (Figure 4.8). Particularly, the 25 and 85 percentiles of annual denitrification rate on lawn in scenario *both* are 2.94 to 31.6 kg N/ha/year, respectively, which are reasonably consistent to the range of empirical measurements at dry to moist soil conditions.

4.6 DISCUSSION

In BARN, the household water use from wells transport roughly 0.08 mm/day of water from groundwater to septic systems, expressed at the watershed level. However, the conversion of groundwater to septic usage showed only negligible changes in hydrological responses. Specifically, the simulated streamflow was slightly decreased compared to the condition without septic water input. Inspecting growing season phenology, we found both ET and net photosynthesis (Figure 4.4) were elevated with septic input. This may be due to the local increases in septic water and nutrients increasing ET during the growing season reducing groundwater recharge, and lowered groundwater storage reducing watershed baseflow. We also noted that our model tended to underestimate the lowest streamflows during the growing season. Several potential reasons could cause this discrepancy: 1) Higher transpiration estimates caused by uncertainties in vegetation ecophysiological parameters of RHESSys controlling vegetation water use or phenology; 2) Underestimation of groundwater recharge and release to streams during growing season; and 3) A lack of human modulation of groundwater use during dry periods. During our prior surveys (Law et al., 2004; Fraser et al., 2013) residents stated they had reduced their water use during the droughts. Additional empirical data about water flux, groundwater processes, and household water management are crucial to perform further calibrations of RHESSys parameters and human-mediated inputs, enhancing the model prediction accuracy of hydrological processes especially during growing season.

Activating fertilization and septic modules in RHESSys improved the simulations of in-stream NO_3^- concentration and load dynamics compared to the original RHESSys model. Compared to the weekly BES observations, our model underestimated the mean in-stream NO_3^- concentration and overestimated the seasonal variability. The underestimation of mean concentration could be attributed by uncertainties of N inputs. Firstly, though we used the mean values from previous studies, the actual N inputs from fertilization and septic effluents also have considerable variations. Secondly, BARN used to have extensive agricultural activities which may result in the accumulation of legacy N in the groundwater. Spinning up the model for 30 years may still be insufficient to account for the N export from groundwater, which possibly caused the lower simulated mean NO_3^- concentration compared to BES empirical estimates. Furthermore, we found the model yielded a stronger seasonality of N export, with simulated concentrations with fertilization and septic processes lower during the growing season but spiking right at the end of growing season. Again, uncertainty in RHESSys vegetation parameters and phenology may contribute to these differences, where the sudden ending of growing season caused quick mobilization of NO_3^- into streams. Also, the lower estimation of streamflow during growing season could increase residence time and retention, and reduce N export from upland and groundwater to streams, causing the underestimation of NO_3^- concentration and load in these periods.

Simulated mean NO_3^- concentration from scenario *none* is significantly greater than POBR observed estimates. The NO_3^- concentration records at POBR (Table 4.3) provide a reference of forest conditions of watersheds in the area. The higher estimation of NO_3^- concentration at BARN could be explained by the land use difference between the two. Specifically, there are more impervious areas and lawns in the upland of BARN than in POBR which is fully forested (with the exception of a regional gasline cut with herbaceous vegetation). This result implies that, even in the absence of additional NO_3^- input from human activities, the water quality in urban watersheds is unlikely to fully recover to pre-urbanization levels due to altered hydrology and differences in vegetation.

In addition to improving predictions of in-stream NO_3^- concentration, the simulated denitrification rates in lawns fell in the range of empirically estimated rates at BARN (Suchy et al., 2023) and other areas in Baltimore (Raciti et al., 2011). Among all "hot spots", the constructed wetland and sediment accumulation zone at the base of the gully exhibited the highest denitrification rates within the entire watershed, both before and after considering fertilization and septic processes. These rates were comparable to other wetland denitrification measurements: Groffman and Hansen (1997) estimated the denitrification rate ranged from 1 to >130 kg N/ha/year at several wetlands in Rhode Island; Poe et al. (2003) reported the rate ranging between 19 to 191 kg N/ha/year at a constructed wetland receiving agricultural runoff; Harrison et al. (2011) also found the rates as 89 and 158.41 kg N/ha/year at two wetlands of Minebank Run in Baltimore. In BARN, these wetlands were located in low-slope downstream areas and advertently or inadvertently treat runoff originating from roads and upstream households. Unlike lawns which may not maintain high soil moisture levels, these areas remain consistently wet throughout most of the year. These features create ideal conditions for promoting denitrification and effectively retaining nitrogen loads that would otherwise be transported to streams. This discovery highlights the significance of strategically selecting the locations for LID projects in future restoration efforts, and assessing the ecosystem services of spontaneously generated features.

Lastly, there are further improvements we recommend for our model. For septic processes, we assumed septic fields of houses located on the southern divide of BARN contribute drainage inside BARN. More detailed survey of septic system location relative to the drainage system is necessary. Our current setup assumed a uniformly daily NO_3^- input and wastewater volume of septic effluents for all houses and fixed fertilization amounts for lawns adjusted by application interval (Eq. 1). These parameters are subjected to further calibration when more observations are available or switching to other study watersheds. For fertilization, our model distributed the estimated total fertilization amount uniformly to all lawns in the watershed, at rates modulated by the proportion of lawns fertilized estimated by Law et al. (2004) and Fraser et al. (2013). In addition, fertilization rate and frequency could vary significantly in different lawns. Variable space and time patterns of fertilization rates could result in N “hot spots” where that exceed retention capacity relative to variable transport rates. For irrigation, our model may apply irrigation close to its maximum (4 mm/day) when water stress is high, but in reality, residents may not irrigate their lawns at these rates during drought to conserve groundwater. Current settings of our model could introduce excessive depletion of groundwater during droughts, and likely underestimations of baseflow and in-stream NO_3^- concentrations. Therefore, we note further detailed survey about water usage habits and observations between meteorological factors and groundwater storage are needed in the future to improve our current dynamics of water withdrawal in RHESSys.

4.7 CONCLUSIONS

This research investigated the impacts and feedbacks of spatially nested additions of water and nitrogen in a low density residential watershed with patterns of groundwater level, soil moisture, N cycling, and stream discharge, and NO_3^- concentrations and loads. We used long term and experimental data along with numerical simulation with the RHESSys model to assess the contributions of irrigation, fertilization, and septic wastewater on ecohydrological N cycling and export processes.

We set up our current model using information for all sources of water and N inputs derived from two decades of data collection and analysis by the BES, including two door-to-door surveys of household fertilizer use. Mean annual inputs of N to the watershed from all sources were estimated as 35 kg N/ha/year, with 31.4% of this input from atmospheric deposition, 34.3% from lawn and garden fertilization, and 34.3% from septic effluent. While atmospheric deposition is ubiquitous, the input of lawn fertilization and irrigation water, and septic effluent volume and nitrogen load are concentrated in limited areas of the watershed. Nitrogen and water inputs to our model are scaled by climate information, and declining rates of N deposition over time. Results showed that our model with these settings was able to simulate streamflow patterns realistically, and reasonably estimated patterns of in-stream NO_3^- concentrations and loads with NO_3^- inputs from fertilizer and septic effluents beyond atmospheric deposition. Model estimates of lawn denitrification are also consistent with our prior field and lab measurements.

We address three questions in succession, using results from numerical experimental scenarios with different combinations of human N inputs:

1) What are the individual and interacting contributions of different watershed nitrogen sources to streamwater nitrogen export?

Simulations with solely septic or fertilization inputs increased NO_3^- export by 1.9 and 2.4 kg NO_3^--N/ha/year individually, while including both sources increased export by 4.7 kg NO_3^--N/ha/year, compared to the base scenario's 1.2 kg NO_3^--N/ha/year with only atmospheric deposition.

2) How do the spatially nested patterns of water and N inputs from human activities alter spatial patterns of a set of key ecohydrological processes (e.g., N retention, evapotranspiration, soil and groundwater levels and flows)?

Simulation results indicate septic systems using deep groundwater as the water resource disposed to shallow soils, resulted in systematic shallow water table increases within upland residential areas and small drops in riparian areas of residential subcatchments. Results show how on-site extraction of water could alter the hydrological conditions of both "on-site" locations where septic effluent is directly disposed, and "off-site" locations. These results occur because while the septic effluent is depleted by evapotranspiration, the deeper groundwater that emerges in riparian areas is unaffected by evapotranspiration. Thus, extraction of water for domestic use lowers riparian water tables even when this water is ultimately discharged back into the environment via a septic system. Likewise, the spatial pattern of denitrification showed increases not only in sites receiving N inputs directly (i.e., lawns and septic drainage fields) but also in "off-site" downstream areas receiving transported NO_3^- from upland zones.

3) What are the emergent patterns of nitrogen cycling and retention, including “hot spots” and “control points” at sites receiving direct additional N and downslope, offsite locations receiving transported N?

In the residential subcatchments of the watershed, riparian zones, constructed and accidental wetlands were found to be “hot spots” of denitrification. These areas have the combination of subsidized supplies of water and NO_3^-, providing mixing zones with conditions promoting denitrification that are more consistent than fertilized lawn areas with variable soil moisture. These results show that effective siting of BMPs and a careful assessment of spontaneously existing (accidental) retention zones can be used to achieve environmental goals for developed watersheds, by leveraging naturally occurring and built features providing ecosystem services.

The improved simulations with more complete, spatially nested inputs of water and nitrogen highlight the importance of the structured spatial heterogeneity of human impacts to fully understand ecohydrological processes in developed watersheds. Oversimplified model structures and input could introduce significant bias that are inapplicable to formulate future water improvement plans. The spatially distributed inputs and RHESSys model structure may provide a reliable framework to evaluate current coupled water, carbon and nitrogen cycles, but also understand and predict effectiveness of ecosystem restorations to improve water quality and ecosystem health in developed watersheds.

4.8 FIGURES AND TABLES

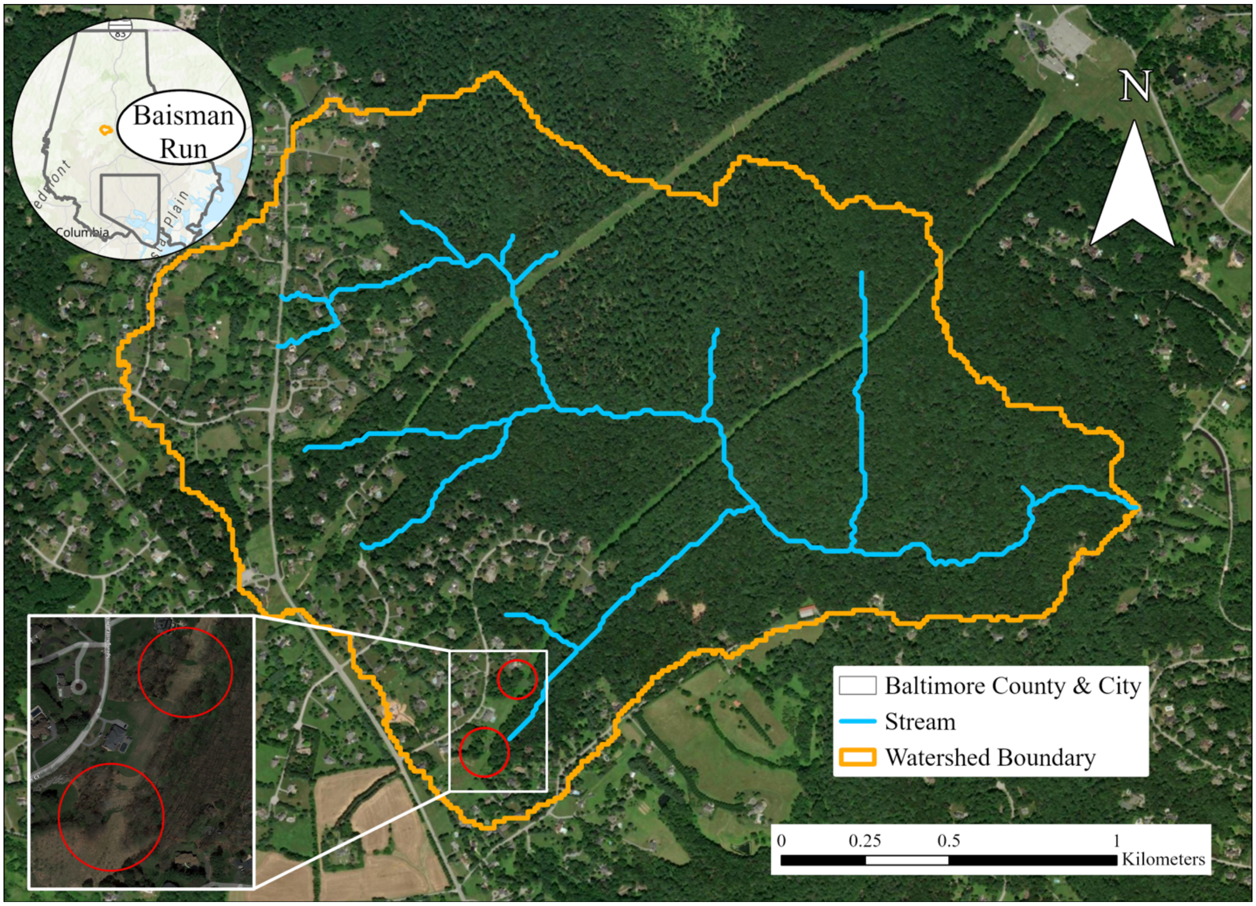

Figure 4.1. Study watershed Baisman Run (BARN) in suburban Baltimore County, Maryland. The white box highlights two "hot spots": A sediment accumulation zone (upper circle) receiving drainage from roads and constructed wetland (lower circle) in BARN, all of which have high nutrient retention capacity to reduce N from upland residential areas from being transported to streams.

Table 4.1. Calibrated multipliers for RHESSys parameters generating the highest NSE for streamflow

Sensitivity Parameter	Name	Details	Value
s	m	decay of hydraulic conductivity with depth	0.924
	K	hydraulic conductivity at the surface	0.707
	depth	soil depth	4.835
sv	m	vertical decay of hydraulic conductivity with depth	0.659
	K	hydraulic conductivity at the surface	1.601
svalt	po	pore size index	1.798
	pa	air entry pressure	0.509
gw	sat_to_gw_coeff	bypass fraction to deep groundwater	0.010
	gw_loss_coeff	groundwater storage/outflow parameters	0.916

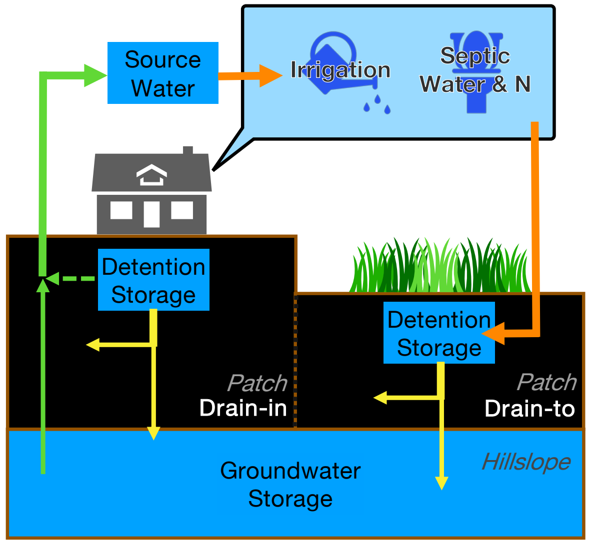

Figure 4.2. Groundwater well and surface source water for irrigation and septic systems augmented to RHESSys model. The source water (green arrows) is extracted from surface detention (e.g., ponds) and groundwater storages to house centroids (i.e., drain-in patches) and redistributed to surface detention of downslope lawns patches after usage (i.e., drain-to patches, see Method 3) for septic and irrigation purposes (orange arrows). After redistribution of source water, infiltration (yellow arrows) to soil and percolation to hillslope groundwater would follow the original procedure of RHESSys

Table 4.2. Scenarios evaluated in BARN and corresponding combinations of new features

Scenario Name	Irrigation	Fertilization	Septic Processes
None	✓		
Fertilization Only	✓	✓	
Septic Only	✓		✓
Both	✓	✓	✓

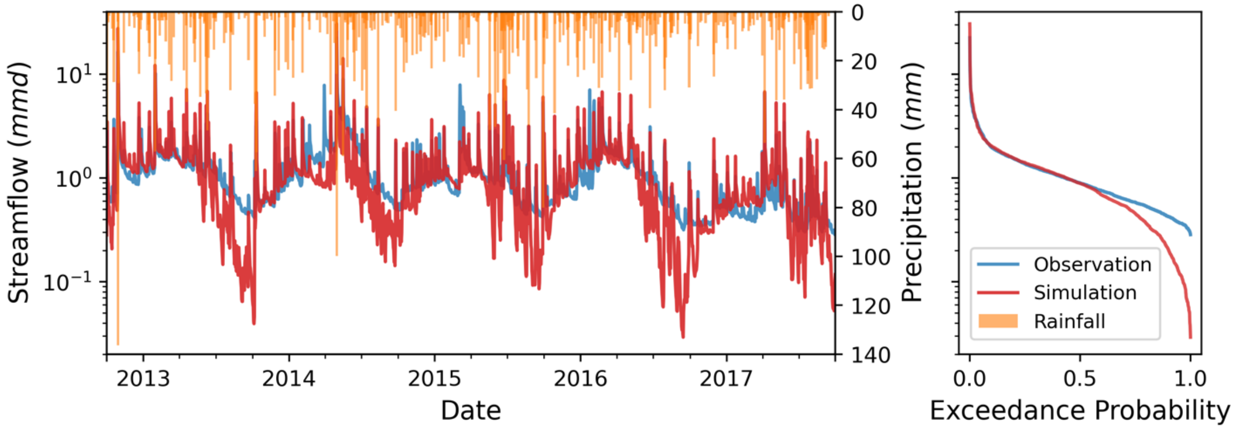

Figure 4.3. Comparison of streamflow time series (left) and duration curve (right) between USGS (blue) observations and RHESSys simulations (red) with irrigation, fertilization, and septic components on (i.e., scenario *both*)

Table 4.3. Mean daily NO3– concentration (mg NO3–-N/L) and load (kg NO3–-N/day) for each season and the entire study period from BES weekly records (all letters capitalized, BARN and POBR) and RHESSys simulations (first letter capitalized, Both, Fert. only, Sept. only, and None) from water year 2013 to 2017

	Season	Scenario					
		BARN	**Both**	**Fert. only**	**Sept. only**	**POBR**	**None**
Concentration	**Spring**	1.5	1.22	0.72	0.63	0.02	0.23
	Summer	1.6	1.1	0.72	0.56	0.07	0.26
	Fall	1.57	1.27	0.86	0.66	0.06	0.32
	Winter	1.75	1.48	0.91	0.78	0.01	0.31
	Mean	1.61	1.27	0.80	0.66	0.04	0.28
Load	**Spring**	10.43	7.90	4.55	4.19	0.01	1.47
	Summer	5.61	3.73	2.32	1.89	0.02	0.76
	Fall	4.50	4.12	2.64	2.20	0.01	0.96
	Winter	7.99	7.90	4.79	4.23	0.01	1.67
	Mean	7.13	5.91	3.58	3.13	0.01	1.22

Note: Nitrate load at POBR is estimated from BES observed nitrate concentration and USGS discharge records. Discharge for BARN is greater than in POBR.

Table 4.4. Seasonal and annual denitrification rate (kg N/ha/year) of locations/patches of interest and watershed under four scenarios. Absolute and relative changes (all positive) from scenario none are reported in parentheses below denitrification rates. Rates for forest excluded Pond Branch patches where no household exists

Location	Season	Scenario: None	Fertilization Only	Septic Only	Both
Lawn	Spring	13.52	18.34 (4.82, 35.7%)	16.6 (3.08, 22.7%)	20.3 (6.77, 50.1%)
	Summer	18.99	24.04 (5.05, 26.6%)	21.7 (2.71, 14.3%)	25.72 (6.73, 35.4%)
	Fall	14.27	17.44 (3.16, 22.2%)	16.6 (2.33, 16.3%)	19.11 (4.84, 33.9%)
	Winter	9.81	11.69 (1.88, 19.2%)	11.84 (2.03, 20.7%)	13.15 (3.34, 34.1%)
	Annual	14.15	17.88 (3.73, 26.4%)	16.68 (2.53, 17.9%)	19.57 (5.42, 38.3%)
Drain-to	Spring	5.62	8.21 (2.59, 46.1%)	19.83 (14.22, 253.0%)	19.84 (14.23, 253.2%)
	Summer	7.68	9.31 (1.63, 21.2%)	21.23 (13.55, 176.4%)	21.3 (13.62, 177.3%)
	Fall	6.65	7.63 (0.98, 14.7%)	20.88 (14.23, 213.8%)	20.92 (14.27, 214.5%)
	Winter	5.11	6.23 (1.12, 22.0%)	15.77 (10.66, 208.6%)	15.75 (10.64, 208.3%)
	Annual	6.27	7.85 (1.58, 25.2%)	19.43 (13.16, 210.1%)	19.45 (13.19, 210.5%)
Riparian	Spring	12	18.99 (6.99, 58.2%)	19.56 (7.56, 63.0%)	24.62 (12.62, 105.1%)
	Summer	13.53	17.99 (4.47, 33.0%)	17.65 (4.13, 30.5%)	21.41 (7.89, 58.3%)
	Fall	10.17	14.39	13.67	17.19

			(4.22, 41.6%)	(3.5, 34.5%)	(7.02, 69.1%)
	Winter	8.66	13.48 (4.81, 55.6%)	13.37 (4.71, 54.4%)	16.48 (7.82, 90.3%)
	Annual	11.09	16.21 (5.12, 46.2%)	16.06 (4.97, 44.9%)	19.93 (8.84, 79.7%)
Forest	**Spring**	12.52	15.57 (3.05, 24.3%)	16.93 (4.41, 35.2%)	19.27 (6.75, 53.9%)
	Summer	9.34	11 (1.66, 17.8%)	11.3 (1.96, 21.0%)	12.91 (3.57, 38.2%)
	Fall	9.51	11.11 (1.92, 22.6%)	11.41 (1.9, 20.0%)	12.85 (3.34, 35.1%)
	Winter	8.49	10.41 (1.92, 22.6%)	11.42 (2.93, 34.4%)	12.81 (4.32, 50.9%)
	Annual	9.97	12.02 (2.06, 20.6%)	12.77 (2.8, 28.1%)	14.46 (4.49, 45.1%)
Watershed	**Spring**	11.88	14.84 (2.96, 24.9%)	15.59 (3.71, 31.2%)	17.86 (5.98, 50.4%)
	Summer	10.17	12.13 (1.96, 19.3%)	12.04 (1.87, 18.4%)	13.81 (3.64, 35.8%)
	Fall	9.69	11.34 (1.64, 16.9%)	11.46 (1.77, 18.2%)	12.89 (3.2, 33%)
	Winter	8.18	9.89 (1.71, 20.9%)	10.66 (2.48, 30.3%)	11.9 (3.71, 45.4%)
	Annual	9.98	12.05 (2.07, 20.8%)	12.44 (2.46, 24.6%)	14.11 (4.13, 41.4%)

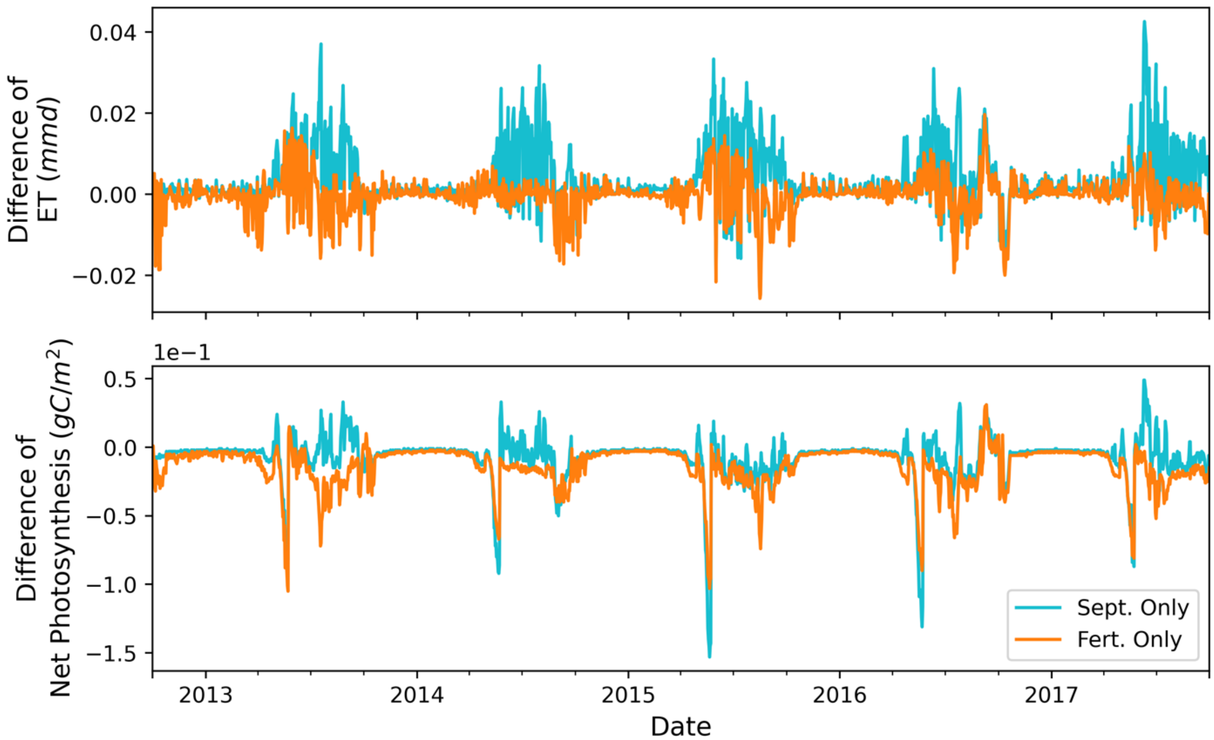

Figure 4.4. Difference of ET (upper) and net photosynthesis (lower) after adding septic or fertilization processes solely into RHESSys from the baseline of scenario *none*

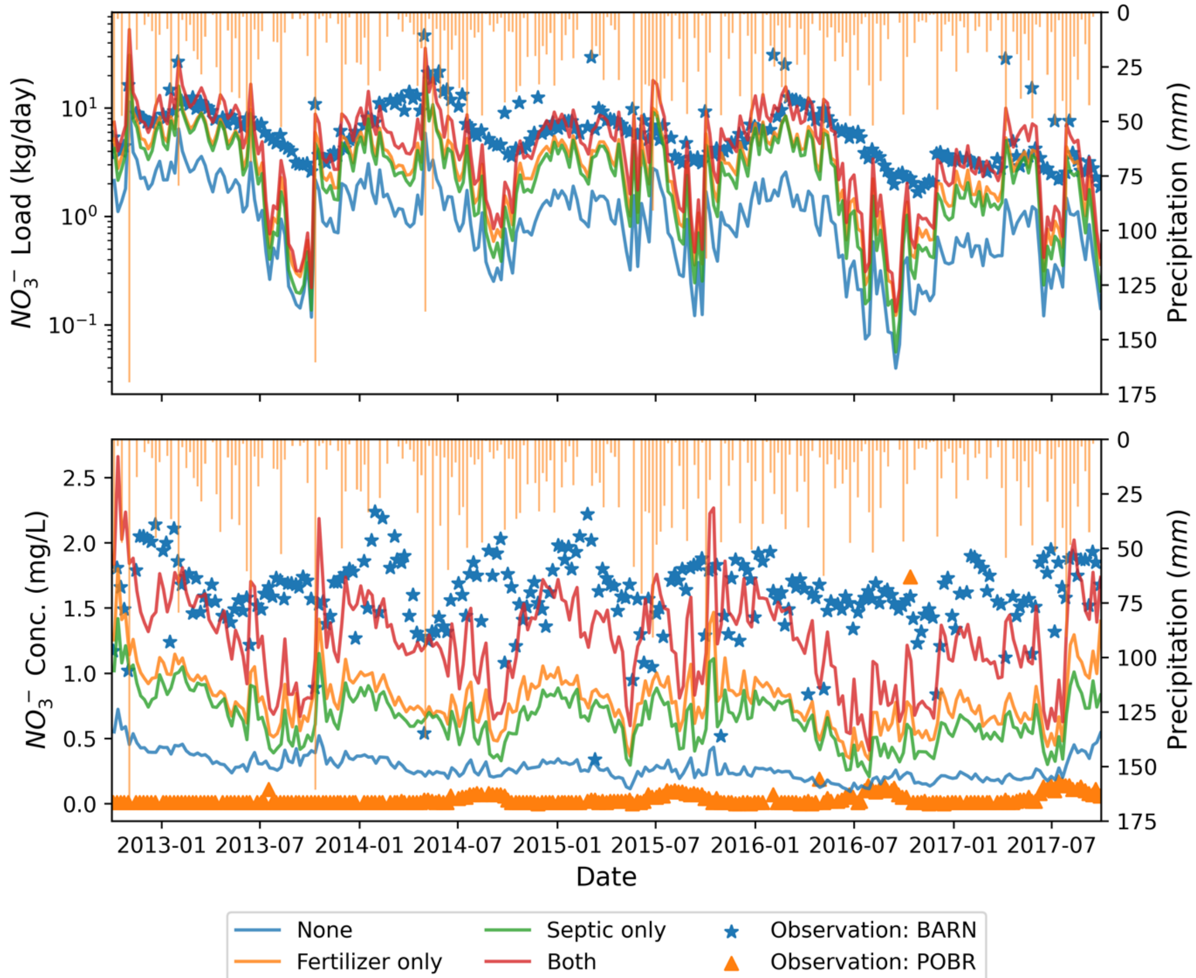

Figure 4.5. Mean weekly NO_3^- load (upper) and concentration (lower) from RHESSys scenarios none, fertilization only, septic only, and both fertilization and septic processes and BES weekly NO_3^- concentration records from water year 2013 to 2017. Fertilization rate was 83.7 kg NO_3^--N/ha/year. Observed BARN load records are calculated using BES concentration and USGS discharge records, and POBR load records are not included

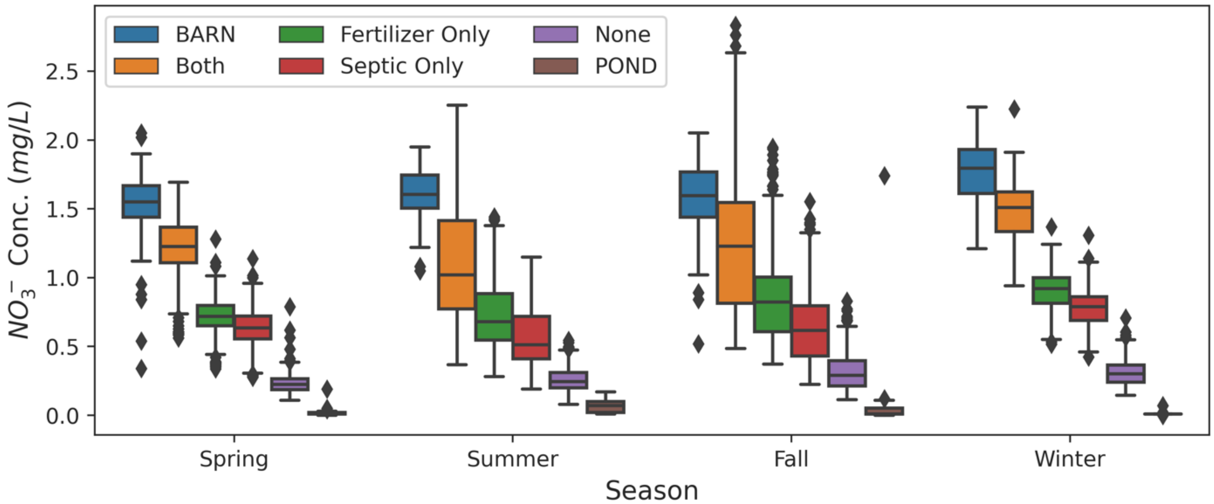

Figure 4.6. Seasonal distributions of in-stream NO_3^- concentrations from BES observations at BARN and POBR and RHESSys simulations with combinations of fertilization and septic features

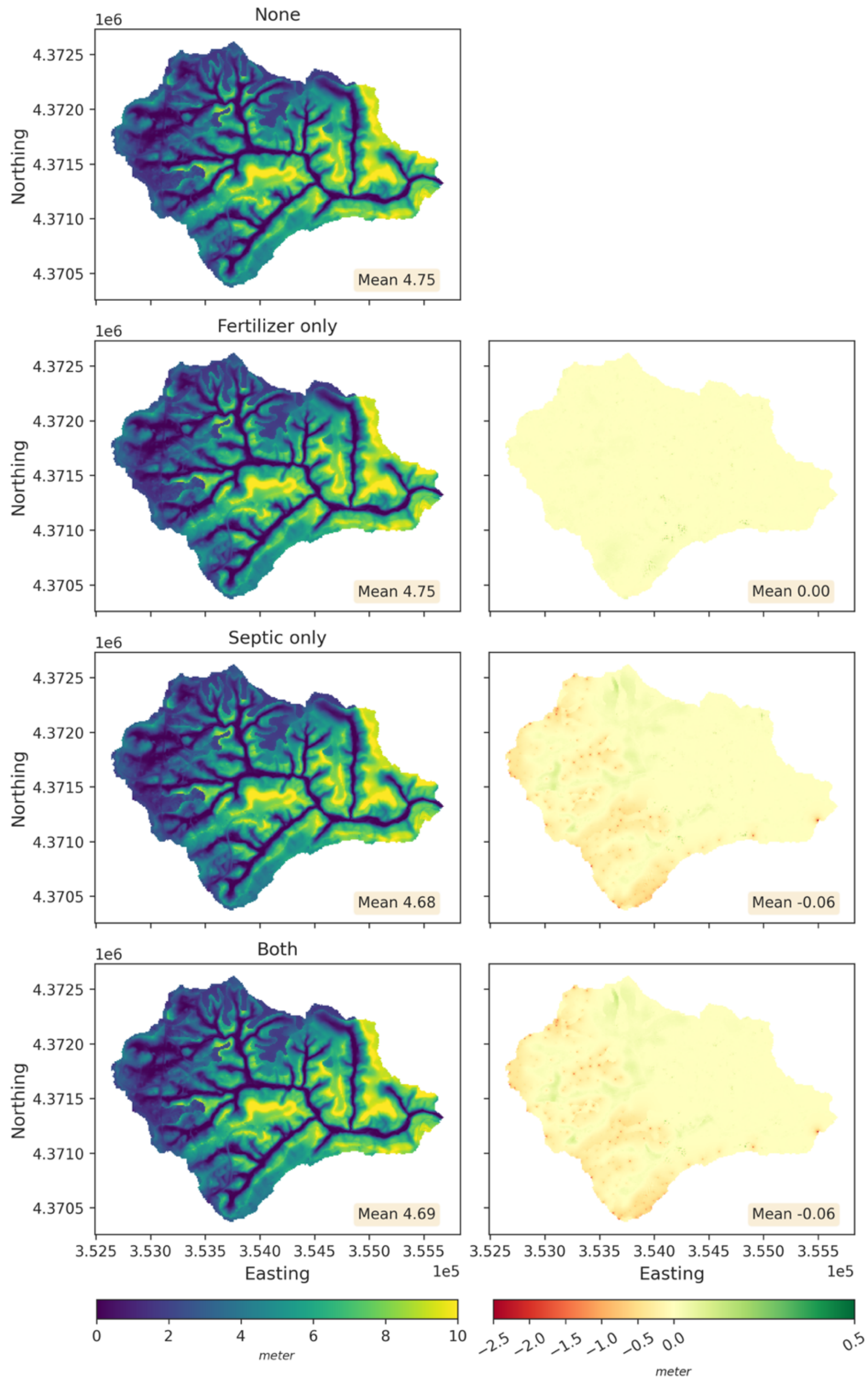

Figure 4.7. Water table depth (m, left) before and after considering fertilization, septic, or both inputs (i.e., scenarios none, fertilization only, septic only, and both) and corresponding changes in depth (right). Isolated and dark red dots in the right panels generally represent individual septic fields. Irrigation is applied for all scenarios showed.

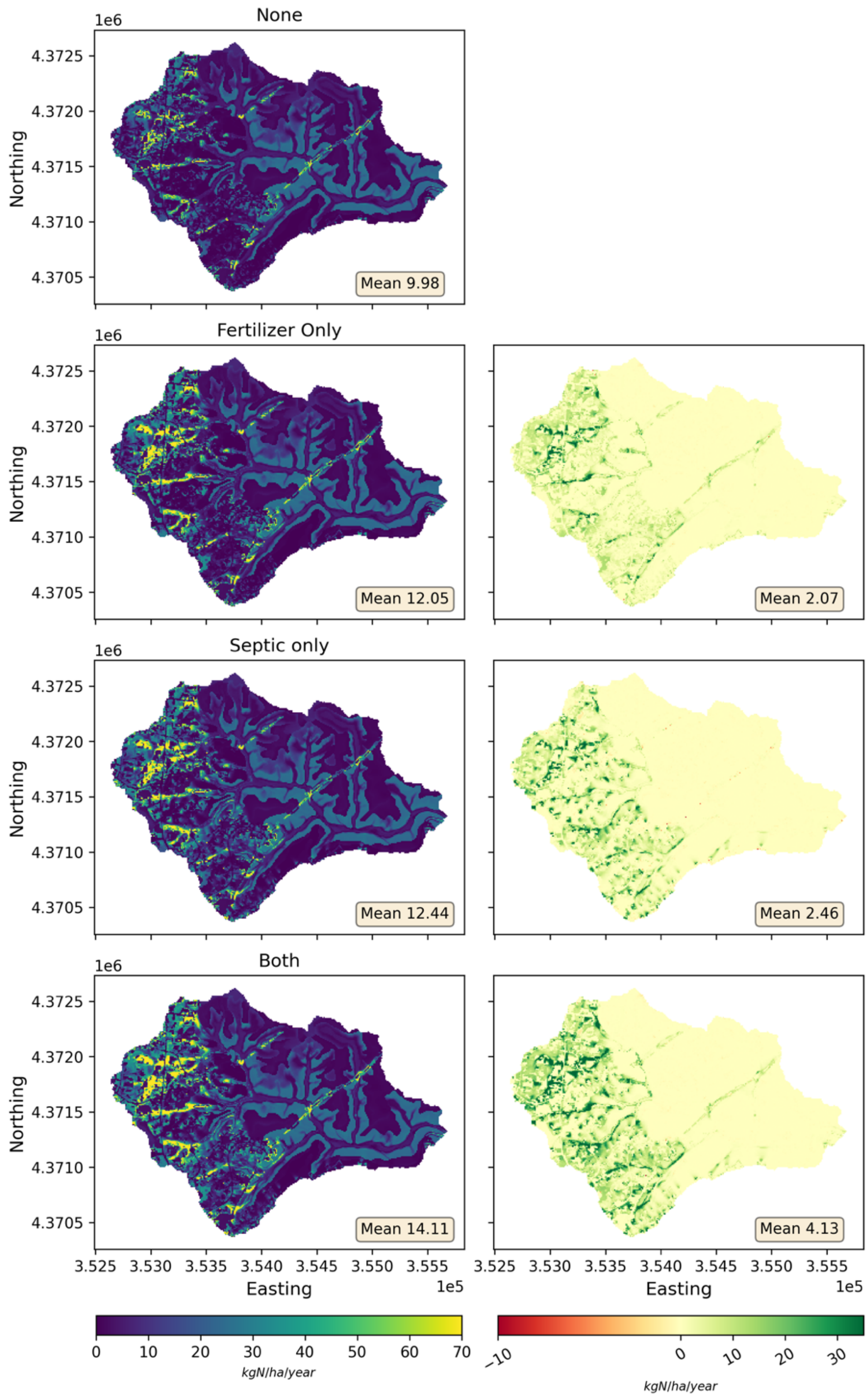

Figure 4.8.Denitrification (kg N/ha/year, left) after adding fertilization, septic, or both features and corresponding changes (right). Irrigation is applied for all scenarios showed.

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Chapter 5: CONCLUSIONS

The primary objective of this dissertation is to establish an analytic framework that accomplishes two main goals: 1) a predictive understanding of the interactions between human activities and ecohydrological processes in urban ecosystems, and 2) quantifying the environmental and socioeconomic benefits derived from various types of ecosystem restoration. Urbanization has not only led to the creation of impervious areas but has also amplified nutrient inputs into terrestrial ecosystems through human activities like lawn irrigation and fertilization, septic effluents, and sanitary sewer leakage. Consequently, urban streams often have flashier hydrographs, characterized by higher peak flow after storm events, and export excessive nutrient loads to downstream waterbodies. In essence, numerous urban areas face threats of frequent floods and degradation of aquatic ecosystems. To address these environmental concerns, ecosystem restoration practices such as green infrastructures, stream restoration, and reforestation have been widely implemented in urban watersheds, with the aim to restore urban ecosystems towards their pre-urbanization conditions. However, the effectiveness of these restoration practices in mitigating hydrograph flashiness and nutrient export can vary significantly among watersheds, and there is currently no systematic process-based framework for quantifying the expected outcomes at the watershed scale resulting from different upland ecosystem restoration plans.

In addition to the environmental benefits, socioeconomic benefits brought by ecosystem restoration practices should also be assessed. For example, urban forestation can alleviate urban heat island effects and improving living conditions for disadvantaged communities. Strategic spatial planning of restoration practices presents an opportunity to bridge the gap in environmental justice among different demographic groups within urban areas, ultimately creating a better living environment for all residents in the region.

5.1 SIGNIFICANCE

This dissertation attempts to establish process-based and statistical models to estimate and understand outcomes of water quantity and quality from various scenarios of upland terrestrial and in-stream aquatic restoration practices to aid decision making procedures. To evaluate the proposed frameworks, we conducted case studies in two urban watersheds in Baltimore, MD, and compared the simulation outputs with streamflow and water chemistry observations. The results highlight a few crucial takeaways for urban ecosystem and restoration:

5.1.1 Asynchrony between environmental and economic benefits of stream restoration

In Chapter 2, we built a statistical model for Baltimore metro areas to evaluate projected nitrogen reduction from stream restoration for every 1,000-ft stream reach, based on Baltimore Ecosystem Study's long-term water chemistry observations and detailed process-based simulations. To our knowledge, this is the first attempt to quantify environmental benefits of stream restoration efforts for all stream reaches in a large region. Meanwhile, Rosenberg et al. (2023) estimated the willingness to pay (WTP) of all households in Baltimore, reflecting local residents' marginal preference to nearby stream restoration projects and allowing an assessment of the relationship between environmental and economic benefits for the whole region. The optimal expectation would be both benefits are in phase or positive relationship, which means stream restoration at reaches with high nitrogen reduction could also bring high co-benefits to nearby residents. In other words, maximizing environmental benefits would results in high socioeconomic benefits in Baltimore region. However, our results suggested the reverse that streams with high expected nitrate reduction are found in areas with low economic benefits, which are commonly located in wealthier and suburban communities, while where stream restoration can bring high economic benefits is always found in low-income and urban communities, where associated nitrate reduction potential is generally low.

The asynchrony in environmental and economic benefits of stream restoration suggests an important message to decision makers, that planning future stream restoration projects for maximizing environmental gains would likely miss the opportunity to use stream restoration to bring green features to disadvantaged communities and fill the gap of environmental justice among communities with different demographic characteristics. However, we find that there are a small sub-population of stream reaches that could bring both high environmental and economic benefits (Figure 2.4). Therefore, we recommend decision makers to perform our analytic framework to identify future stream restoration locations that can meet local environmental and social goals.

5.1.2 Effectiveness of GI to mitigate stormwater and stream nitrate loads

Chapter 3 focused on evaluating the effectiveness of different restoration scenarios aimed at mitigating the stormwater and nitrogen export in Scotts Level Branch Watershed. The scenarios we considered involved reforestation of lawns, the construction of roadside bioswales, and the conversion of roads into pervious pavements. An important aspect of these scenarios is that they were carried out without increases in stormwater storage. While the spatial extent of restoration in these scenarios may be ambitious and challenging to implement in reality, we found that their impacts on reducing peak flows during extreme storms and in-stream nitrogen loads were low to moderate. Specifically, the infiltration-based roadside bioswales and permeable roads were able to partially modify hydrograph flashiness by increasing infiltration and baseflow in SLB, resulting in a decrease in peak flow and a gentler recession limb. However, these measures fell short of fully restoring the watershed to its pre-urbanization conditions, consistent with the findings suggested by Jefferson et al. (2017). Surprisingly, the increase in baseflow caused by infiltration-based LID elevated in-stream nitrate export by mobilizing subsurface sources, contradicting the intended purpose of these LID practices to improve water quality. On the other hand, reforestation demonstrated the greatest effectiveness in mitigating high flows during storms, significantly increase nutrient uptake in upland areas and reduce in-stream nitrate load. Therefore, it is crucial to consider a variety of ecosystem restoration practices in urban watersheds to strike a balance between the potential increase in nutrient export resulting from elevated infiltration and the improved nutrient retention in upland areas with expanded tree cover. This study also explored the relationship between upland restoration and aquatic nitrate reduction efficiency, considering the impact of terrestrial restoration on flow regimes. In Chapter 2, we used the nitrate duration curve as an indicator to measure the effectiveness of stream restoration in reducing nitrate levels. The results indicated only minor changes in the nitrate duration curve (Figure 3.10) after upland restoration, primarily because the impervious surfaces in SLB were not effectively addressed by terrestrial and aquatic restoration efforts. This finding emphasizes the need to combine green infrastructures with other grey infrastructures, such as detention storage and sewer network repair, to effectively mitigate flashy flow regimes and control nutrient inputs in urban watersheds. It suggests that solely relying on GI/LID may not be sufficient to resolve the current water quantity and quality issues resulting from urban ecosystem degradation.

5.1.3 Interactions between ecohydrological processes and human activities

In Chapter 4, our study enhanced the existing RHESSys model by incorporating detailed nitrogen input and irrigation practices in an exurban watershed. This improvement was crucial for accurately simulating upland ecological processes and in-stream nitrate dynamics in urban watersheds. Urban ecosystems often receive significant nitrogen inputs from human activities, which can surpass atmospheric deposition. Based on lawn fertilization surveys (Fraser et al., 2013; Law et al., 2004) and empirical data on septic systems, we conducted a case study at Baisman Run – a low-density suburban watershed with water supplied from domestic wells – to assess whether the addition of N loads from fertilization and septic effluents improved the simulation of in-stream nitrate concentration and upland denitrification when compared to observations from the Baltimore Ecosystem Study (BES). Our results demonstrated that the average simulated nitrate concentration increased from 0.43 to 1.48 mgN/L, which better aligned with the observed concentration of 1.61 mgN/L from BES. Furthermore, we estimated the denitrification rate at lawns to be 22.76 kgN/ha/year, falling within the range of two empirical measurements conducted on lawns in the BES (Raciti et al., 2011; Suchy et al., 2023). As a result, our augmented RHESSys model, which incorporated surveyed N inputs from human activities, substantially improved the simulation of nitrogen cycling and dynamics in urban watersheds. This enhancement contributes to a better understanding of the current state of nitrogen in urban ecosystems and facilitates the development of targeted practices aimed at reducing nitrogen export and addressing issues stemming from excessive nutrient inputs.

The spatial simulation capabilities of our RHESSys model enabled the identification of N hotspots characterized by high rates of nitrogen sink processes, such as denitrification, within the watershed. Riparian areas, which typically receive groundwater subsidies from upland areas, exhibited higher denitrification rates compared to the watershed average. Additionally, two areas downslope residential nitrogen sources (lawns and septic spreading fields) had the highest denitrification rates within the watershed. One of these areas was a constructed wetland, while the other was an "accidental" wetland that received drainage from upland streets. The unique geomorphology and locations of these two areas facilitated the accumulation of nitrogen and water from upland households, resulting in consistently anoxic and nitrogen-rich conditions that sustained high rates of denitrification. Consequently, future restoration practices could consider incorporating planned and unplanned wetland-like retention features to promote denitrification in upslope areas and reduce nitrogen export in streams.

5.1.4 Implications for solving urban ecosystem syndrome

However, across the three urban ecosystem studies presented in this dissertation, a common conclusion emerges: returning to pre-urbanization conditions after human disturbances is difficult, despite extensive restoration efforts. In Chapter 2, significant nitrate reduction resulting from stream restoration was only observed in less-developed watersheds, which raises concerns about the cost-effectiveness and environmental equity of restoration constructions. In Chapter 3, large-scale reforestation efforts demonstrated limited success in mitigating flood quantities and nitrogen export, which may not meet desired environmental outcomes. In Chapter 4, even in Baisman Run, where only a small portion of the area is impervious due to residential uses (such as roofs, driveways, and roads), the nitrate concentration is significantly higher than the forested reference watershed, Pond Branch, partially due to introduced N sources in limited areas, creating "hot spots" (Bernhardt et al., 2017; McClain et al., 2003). This finding suggests that even minimal human activities or alterations can significantly change ecosystem behavior. Failing to understand the current status of an urban watershed and evaluate potential outcomes of ecosystem restoration puts local agencies at risk of failing to achieve their planned environmental goals despite investing significant funds. It is important to note that all projects within this dissertation focused on retrofitting already developed watersheds rather than new developments. We hope that our framework can also assist in assessing the ecohydrological and biogeochemical status, guiding the design of future developments, and enhancing downstream ecosystem health in the future.

5.1.5 Important messages to urban ecohydrology

There are several surprising yet crucial findings and implications that need to be conveyed to decision makers and researchers:

- Alterations of vegetation types (i.e., reforestation on lawns) could change the seasonality of soil moisture, creating a wetter period before the trees' growing season, which could increase flood risks. Specifically, after reforesting lawns, the start of transpiration is delayed, resulting in an extended period where soil moisture remains high between the two vegetation types' starts of the growing season. Storms during this period can trigger higher surface runoff due to the elevated soil moisture and potentially cause flooding in urban watersheds.
- Infiltration-based LIDs may not mitigate but increase nutrient export may be unexpectedly in urban watershed. Greater infiltration promoted by roadside

bioswale, permeable road, and similar LIDs in urban watersheds could increase soil moisture and nitrate load through higher nitrification, and the higher groundwater table could mobilize nutrients from sanitary sewer leakages.

- Prioritizing maximal environmental gains for future restoration practices could likely result in low socioeconomic gains, and balancing both gains is essential for addressing challenges beyond environmental issues in urban areas. Specifically, ecosystem restoration practices provide opportunities to address social equity challenges, such as improve the gaps of living environments and access to green and recreational spaces between wealthy and disadvantaged communities.
- There are still many physical processes and human behaviors in urban hydrology that are not sufficiently observed and studied (e.g., sanitary sewer leakage of water and nutrients and fertilization quantity and frequency). More data and modeling efforts are needed to understand the current environment conditions of urban watersheds and project future ecohydrological dynamics with greater confidence.

5.2 FUTURE RESEARCH

5.2.1 Beyond environmental benefits of ecosystem restoration

The economic and environmental benefits of stream restoration were spatially assessed in Chapter 2. We evaluated the potential environmental outcomes from other types of terrestrial restoration (i.e., reforestation, roadside bioswale, and permeable road), but their corresponding economic benefits to the residents could also be estimated in the future.

This comprehensive evaluation would enable a balanced consideration of both the environmental and economic benefits associated with various types of ecosystem restoration spatially. Additionally, it presents an opportunity to introduce green features to highly impervious regions and address equity issues of living environment in disadvantaged communities. Collaborating with economists to quantify the economic benefits and engaging with residents are critical components of this process. By involving residents, we can better understand and address the localized issues that matter most to them.

5.2.2 Improvements of existing frameworks

This dissertation relies on using the spatially explicit RHESSys model to address the environmental research questions, with the presence of long term and field experimental data. We include human-induced practices (i.e., irrigation, fertilization, septic processes) into simulations of ecosystems, but there are other processes which are not fully understood yet and should be modeled in RHESSys to better understand ecohydrological and biogeochemical dynamics of urban ecosystems. Specifically, many urban watersheds contain complicated piped sanitary and stormwater drainage networks. The infiltration and inflow (I&I) to sanitary sewer networks significantly impacts not only discharge quantity (Bhaskar & Welty, 2012; Bhaskar et al., 2015) but also levels of in-stream pollutants from leakage of sanitary sewers (Delesantro et al., 2022). Further data collection efforts are necessary to better comprehend I&I and leakage processes within urban watersheds. These data would then need to be integrated and replicated within the current RHESSys model, enabling a more comprehensive representation of urban watershed dynamics with reduced uncertainty. By incorporating these additional processes, we can improve the model's ability to capture the complex dynamics of urban ecosystems and provide more reliable insights into their ecohydrological and biogeochemical behavior.

5.2.3 Extrapolations with data-based methods

This dissertation primarily relies on biophysically process-based models for assessments urban ecohydrological dynamics. While these models offer appealing interpretability, they require extensive calibration, which in turn demands observations that may not be universally available and involves time-consuming simulations. Conversely, machine learning models are gaining popularity in environmental science and are widely utilized for predicting diverse physical and ecological processes in timely manners.

Given that our frameworks provide detailed and varied simulations of ecohydrological processes at fine spatial and temporal resolutions, there exists potential to train machine learning models using our results. This would allow us to extrapolate our current understanding of urban ecosystems from gauged to ungauged watersheds. For instance, convolutional neural networks could be employed to discern the influence of topographic, land-use, human, and meteorological factors on denitrification rates at specific locations within a gauged watershed. Subsequently, this model could be used to identify areas with high denitrification rates in other neighboring regions or to evaluate how planned restoration practices within a watershed might modify denitrification rates following modifications to vegetation or topographic features. Preliminary experimentation with these methods is encouraging, but beyond the scope of the current dissertation.

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