OPTIMIZATION OF RIPARIAN ZONE NITROGEN MANAGEMENT THROUGH THE DEVELOPMENT OF RIPARIAN MODEL

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OPTIMIZATION OF RIPARIAN ZONE NITROGEN MANAGEMENT THROUGH
THE DEVELOPMENT OF RIPARIAN MODEL

BY

MARZIA TAMANNA

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ABSTRACT

This thesis addresses the modeling approach to benefit the riparian zone nutrient management related to water quality in the Northeast and Midwest of USA. Nutrient (primarily Nitrogen (N)) loss from agricultural watersheds through runoff and drainage water continues to be a water quality concern of global importance. Since N is a crucial input for the sustainability of agriculture, the use of N has increased dramatically in recent decades and the excessive nutrient losses have increased too. Like global concern, agriculture (cropland, pasture, managed forest) is an important component of many watersheds of the USA Northeast where N flux to major estuaries is of substantial concern. In this circumstance, the finding from almost 30 years of research on riparian zone hydrology and biogeochemistry demonstrates that riparian zones can serve as best management practices (BMPs) to minimize the adverse agricultural impact on water quality.

Riparian zones have been used as one of the most important practices for water quality improvement in agricultural settings due to its ability to perform multi functions including reducing NO\textsubscript{3} concentrations in subsurface flow, trapping sediments and pesticides in overland flow, and control erosion. They are often characterized as “filters” or “buffers” and are vital elements in watershed management schemes for water quality maintenance and stream ecosystem habitat protection. Nevertheless, the buffering capacity of riparian zones (mostly for N) varies enormously due to the hydrogeomorphic setting such as topography, soil type, and
surficial geology of the riparian zone. Upland land use/land cover affects both the water quantity and quality of the water entering the riparian zone. Hydrogeomorphic setting can influence the flowpaths and hydrologic connections between upland sources of nitrate and the biologically active (i.e., upper 1-2 m) portions of the riparian zone. Thus, a number of key attributes related to location are critical in determining the potential impact of a riparian zone on water. These attributes are incorporated in models like the Riparian Ecosystem Management Model (REMM; Altier et al., 2002; Lowrance et al., 2000). Given the interest in expanding riparian zone BMPs, there is a critical need to advance our understanding of riparian functions at the site scale. Site-specific models can improve riparian zone management decisions that seek to place, restore and protect riparian zones more effectively.

REMM has been used to simulate managed riparian ecosystems in a number of settings in USA including Chesapeake Bay Watershed, Delaware, Mississippi, North Carolina, Georgia, California, and Puerto Rico. Globally, SWAT-REMM integration has been used in a glaciated landscape in New Brunswick, Canada by Zhang et al., 2017 to examine the effect of different levels of dividing up the watershed into sub-watershed for SWAT on the performance of the model. Liu et al., 2017 used REMM in China for the evaluation of riparian zones as BMP. However, REMM has not yet been integrated with AnnAGNPS model and applied to evaluate management at the field scale in the glaciated settings of the Northeast and Midwestern regions, even though the agricultural lands are linked to excessive nutrient pollution and
Riparian zones are widely used in these regions to mitigate N losses to streams. So, our focus on field scale analyses with AnnGNPS provides more insight into site scale behavior.

The objective of this work is to develop a set of Riparian Model parameters for the USA Midwest, USA Northeast to facilitate the use of REMM in these regions and improve its functionality with respect to N and N\(_2\)O. The work has been described in the following five manuscripts, as per the Graduate School Manual guidelines:

**Chapter 1. Manuscript I** (published in *Water*, 2020)

The objective of this work was to: (i) evaluate the performance of the AnnAGNPS model in simulating the runoff volume at three separate watersheds with glacial setting of Northeast and Midwest USA; (ii) improve the model’s runoff prediction capacity through calibration; (iii) validate the model’s runoff prediction with the improved calibrated parameters; (iv) conduct a parameter sensitivity analysis for runoff simulation; (v) conduct an analysis of the spatial distribution of runoff depth for three watersheds; (vi) provide a discussion of the model’s performance in order to estimate event peak discharge.

**Chapter 2. Manuscript II** (published in *Agriculture*, 2021)

The objective of this work was to test the application of REMM in formerly glaciated setting of Rhode Island (RI), USA for riparian zone nitrate dynamics.
Chapter 3. Manuscript III (In preparation for *Nutrient Cycling in Agroecosystems, 2021*)

The objective of this work was to test the ability of REMM model for riparian zone nitrogen simulation in two agricultural watersheds from the glacial setting of Indiana (IN), USA Midwest.

Chapter 4. Manuscript IV (In preparation for *Journal of Contaminant Hydrology, 2021*)

The objective of this work was to evaluate the potential of REMM model in a glaciated watershed of New York (NY), USA Northeast for riparian zone nitrogen estimation.


The objective of this work was to assess the climate change impact on runoff coming from field edge (upland) towards the riparian zone (stream edge) in the glaciated landscape of the Northeast and Midwest USA.

In conclusion, this study provides an evaluation of the ability of the REMM model for nutrient management in the glaciated setting of USA Northeast and USA Midwest and establishes a base of site specific parameters for water resources
managers. Model performance during calibration and validation phases shows that REMM model can be successfully coupled with upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and N data from multiple buffers. Both the hydrologic and nutrient testing of REMM showed that it captured well the daily measured data (WTDs and groundwater NO$_3$-N concentrations in stream edge) for both calibration and validation periods.
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Dedication

My parents, Late M. A. Wadud and Farida Begum, for whom I am here today.
PREFACE

Tamanna, Soni M. Pradhanang, Arthur J. Gold, Kelly Addy and Philippe G. Vidon is a manuscript in preparation for submission in the journal *Journal of Contaminant Hydrology*. The fifth chapter named as “Climate Change Impact on Runoff Prediction in the Glaciated Landscape of the Northeast and Midwest USA”, authored by Marzia Tamanna, Soni M. Pradhanang, Arthur J. Gold, Kelly Addy and Philippe G. Vidon is a manuscript in preparation for submission in the journal *Journal of Hydrologic Engineering - ASCE*. The sixth chapter contains primary conclusions, importance, and future research directions associated with this dissertation.
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CHAPTER 1

Manuscript I

Evaluation of AnnAGNPS Model for Runoff Simulation on Watersheds from Glaciated Landscape of USA Midwest and Northeast

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Abstract

Runoff modeling of glaciated watersheds is required to predict runoff for water supply, aquatic ecosystem management and flood prediction, and to deal with questions concerning the impact of climate and land use change on the hydrological system and watershed export of contaminants of glaciated watersheds. A widely used pollutant loading model, Annualized Agricultural Non-Point Source Pollution (AnnAGNPS) was applied to simulate runoff from three watersheds in glaciated geomorphic settings. The objective of this study was to evaluate the suitability of the AnnAGNPS model in glaciated landscapes for the prediction of runoff volume. The study area included Sugar Creek watershed, Indiana; Fall Creek watershed, New York; and Pawcatuck River watershed, Rhode Island, USA. The AnnAGNPS model was developed, calibrated and validated for runoff estimation for these watersheds. The daily and monthly calibration and validation statistics ($NSE > 0.50$ and $RSR < 0.70$, and $PBIAS \pm 25\%$) of the developed model were satisfactory for runoff simulation for all the studied watersheds. Once AnnAGNPS successfully simulated runoff, a parameter sensitivity analysis was carried out for runoff simulation in all three watersheds. The output from our hydrological models applied to glaciated areas will provide the capacity to couple edge-of-field hydrologic modeling with the examination of riparian or riverine functions and behaviors.

**Keywords:** Evaluation; AnnAGNPS model; runoff; simulation; watershed; glaciated landscape; USA
1. Introduction

Excess nutrient (primarily nitrogen and phosphorus) losses from agricultural watersheds in glaciated settings of the Midwest and Northeast of USA are one of the greatest water quality problems tied to modern agriculture [1–6]. These water quality problems include eutrophication, harmful algae blooms, and fish kills in the Gulf of Mexico, the Chesapeake Bay, the Hudson River Estuary, and other coastal areas [7–10]. A substantial body of research on riparian zone hydrology and biogeochemistry has shown that riparian zones can serve as efficient best management practices (BMPs) for nutrient removal [11,12].

The functional efficiency of nutrient removal in a riparian zone can vary widely depending on the characteristics within the riparian zone (e.g., vegetation, soil texture, depth to water table) and on its location in the landscape, timing, characteristics and extent of contaminant and hydrologic loading and its hydrogeomorphic setting [11–19]. Thus, it is significant to explore the relationships between the landscape-generated “edge-of-field” waterborne losses and riparian functioning to improve riparian design and to better quantify the extent of treatment within riparian zones [20]. Given the expense and time associated with empirical studies, simulation models offer the capacity to examine riparian zone performance across many different soils, topography settings, agricultural practices and climatic conditions.

The Riparian Ecosystem Management Model (REMM) [21,22] has been used in non-glaciated settings, primarily in a region extending from Texas to the Atlantic coast.
to explore the pollution abatement of riparian zones. In companion studies we are examining the efficacy of the REMM model for use in glaciated settings of the Northeast and Midwest, USA. Over the past two decades, the authors of this manuscript have generated an extensive, empirically derived data base on riparian zone structure and functions from three glaciated watersheds—located over a 1400 km east–west band encompassing about 3 degrees of latitude (39.72° to 42.44° N). All of these watersheds lack empirical data on edge-of-field overland runoff. In contrast to the piedmont and coastal plains where REMM has been used, the flux of water and water-borne contaminants in these glaciated regions are driven by differences in the magnitude and timing of snowmelt, the frequency of freeze–thaw phenomena, lower evapotranspiration and the varied soils and geomorphology (e.g., hilly, low permeability till that co-occur with flat, high permeability stratified drift landforms and narrow bands of alluvial soils) that often exist within small watersheds. [23] demonstrated that there were higher average annual runoff and higher seasonal (winter, spring, fall) runoff loads in northeastern glaciated watersheds than non-glaciated watersheds.

Here, we evaluate the AnnAGNPS (Annualized Agricultural Non-Point Source) [24,25] model for its efficacy to predict runoff from glaciated, upland areas to riparian buffers—a key requirement for riparian zone models such as the REMM model. AnnAGNPS is a distributed model that can assess the continuous hydrologic and water quality responses of watersheds to daily weather conditions in a variety of
soils, land uses and land covers. AnnAGNPS has been successfully used in several states of USA, such as Illinois [26], Indiana [27], Mississippi [28,29], Georgia [30], Kansas [31], New York [32]. Globally, AnnAGNPS has been effectively used in numerous parts of the world in recent years, including Brazil [33], Spain [34], Nigeria [35], Italy [36–38], Canada [39], Australia [40], Nepal [41], China [42], Belgium [43], Malaysia [44], and Saint Lucia [45]. AnnAGNPS has been implemented in all of these studies to predict runoff, sediment, and pollutant loadings under various environmental conditions representing different watersheds. AnnAGNPS model was selected for this study since the model can generate estimates of “edge-of-field” losses to specific downgradient cells, such as riparian zones, which enables the output of AnnAGNPS to serve as input to riparian buffer models. AnnAGNPS divides a watershed into a number of cells (characterized by similar land and soil properties) of various sizes, and runoff and contaminants are routed from these cells into the associated reaches, and the model either deposits pollutants within the stream channel system or transports them out of the watershed. The cell-based structure was not available in a number of other commonly used water quality models (e.g., Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS), Groundwater Loading Effects on Agricultural Management Systems (GLEAMS), Hydrological Simulation Program—FORTRAN (HSPF) and Soil & Water Assessment Tool (SWAT), which lack or are limited in the extent of spatially explicit simulations within and between locations at the field and watershed scale. For example, the SWAT
model divides a watershed into sub-watershed or hydrological response units (HRUs) exhibiting homogenous land, soil and slope characteristics instead of a fine-resolution grid network. This limits its incorporation of spatial variability (very crucial for riparian zone) in simulating the hydrologic dynamics that drive many functions of riparian buffers.

Based on the review of literature, we found that AnnAGNPS has not been applied in watersheds of glacial geomorphic settings, such as our studied watersheds, to model runoff generation and other hydrological processes. The objective of this study was to evaluate the suitability of the AnnAGNPS model in glaciated landscapes for the estimation of runoff quantity. The geomorphic setting, i.e., the glaciated landscape, makes the AnnAGNPS model application unique in this study. Our study area included Sugar Creek watershed in Indiana, Fall Creek watershed in New York and Pawcatuck River watershed in Rhode Island. All the watersheds are located in a glacial geomorphic setting.

Specifically, this paper aims to: (i) evaluate the performance of the AnnAGNPS model in simulating the runoff volume at three separate watersheds with glacial setting; (ii) improve the model’s runoff prediction capacity through calibration; (iii) validate the model’s runoff prediction with the improved calibrated parameters; (iv) conduct a parameter sensitivity analysis for runoff simulation; (v) conduct an analysis of the spatial distribution of runoff depth for three watersheds; (vi) provide a discussion of the model’s performance in order to estimate event peak discharge.
2. Materials and Methods

2.1. Study Area

Three watersheds from three different states were modeled for this study. Sugar Creek watershed (39°43′21″ N, 85°53′23″ W), a part of the White River watershed in central Indiana, is about 69 km² (Figure 1). The elevation of the watershed ranges from 241 m to 280 m, and the topography is nearly flat. The watershed consists largely of tile-drained agricultural lands (88% of the total watershed area, representative of agro-ecosystems of the glacial till plains from US Midwest [46]. This watershed is dominated by poorly drained soils where artificial drainage is usually used to lower the water table [47]. For the past 20 years, agricultural practices have been dominated by a corn/soybean rotation with either conventional or conservation tillage systems [48]. The temperature in the watershed is moderate, ranging from a 30-year (1982–2011) mean of 22.7 °C in summer to a mean of −1.4 °C in winter (Parameter-elevation Regressions on Independent Slopes Model (PRISM) Climate Group, accessed on 4 May 2019). The 30 years (1982–2011) average annual precipitation is approximately 1105.0 mm, about 51% of which occurs during the summer and the fall months. The 14 years (2000–2013) average annual snowpack is 32.3 mm.
Fall Creek watershed has an area of about 328 km² is located within the Finger Lakes region of New York State (42°28′ N, 76°27′ W) (Figure 1). The most extensive source of parent material is glacial till, with additional parent materials that consist of glacio-lacustrine sediments and glacio-fluvial (outwash) deposits. The watershed is a mixed land use landscape located at the southern terminus of the Wisconsin glaciation
The watershed is 4.8% urban/developed land use (residential, commercial and service, industrial, etc.), 45.3% forest (evergreen forestland, mixed forestland), and 49.4% agriculture (cropland and pasture, other agricultural land, shrub and brush rangeland) [50]. Soils in the watershed are dominated by Gravelly silt loam and Channery silt loam. These are typically very deep, well-drained soils. Elevations range from 270 m above mean sea level to 600 m [51]. The temperature in the watershed ranges from a 30-year (1982–2011) mean of 19.7 °C in summer to a mean of −3.6 °C in winter (PRISM Climate Group, accessed on 13 January 2019). The 30 years (1982–2011) average annual precipitation is approximately 930.3 mm, about 52.8% of which occurs during the spring and the fall months. The 14 years (2000–2013) average annual snowpack is 32.2 mm.

The Pawcatuck River watershed in Washington County is located in the New England Hydrologic Region of southern Rhode Island (41°32′30″ N, 71°35′ W) (Figure 1). The area of this watershed is about 258 km2. It consists mainly of forests (above 65% of the total watershed area) and agricultural fields (about 32% of the entire watershed area). The soil parent materials in the watershed are comprised mostly of glacial till, glacial outwash, and organic and alluvial deposit [52]. Agricultural lands (mostly turf farms) are predominately located on loess soils over glacial outwash. Forested settings are usually on till. The elevation of the watershed ranges from 16 m (shoreline) to 144 m (to gently rolling hills inland). It has a humid continental climate, with warm summers and cold winters. The temperature in the watershed ranges from
a 30-year (1982–2011) mean of 20.8 °C in summer to a mean of −0.4 °C in winter (PRISM Climate Group, accessed on 21 March 2019). The 30 years (1982–2011) average annual precipitation is approximately 1291.7 mm, about 50.8% of which occurs during the spring and the fall months. The 14 years (2000–2013) average annual snowpack is 44.8 mm.

2.2. Description of the AnnAGNPS Model

The Annualized Agricultural Non-Point Source Pollution Model (AnnAGNPS) [24] refers to a watershed scale, batch process, continuous and distributed simulation, daily time step, surface runoff, and pollutant loading computer model. The model has been designed to quantify and identify the source of pollutant loadings anywhere in the watershed for optimization and risk analysis. Hydrology, sediment, nutrient, and pesticide transportation are essential modeling components. This continuous version of the model is an improvement to the previously developed single-event Agricultural NonPoint Source model (AGNPS) watershed model [53]. The model uses and combines many modules of other commonly used models, such as Revised Universal Soil Loss Equation (RUSLE) [54], Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS) [55], Erosion Productivity Impact Calculator (EPIC) [56], and Groundwater Loading Effects on Agricultural Management Systems (GLEAMS) [57]. In this article, AnnAGNPS version 5.45 (United States Department of Agriculture (USDA)-Agricultural Research Service (ARS), National Sedimentation
Laboratory, Oxford, MS, USA) (Official Release-21 December 2016) was used for all simulations. A full description of this model and its associated components are available in [58].

2.3. Hydrological Modeling Component in AnnAGNPS

The main components within AnnAGNPS are the combination of the Soil Conservation Service (SCS) curve number (CN) technique [59] used to generate daily runoff and RUSLE 1.05 tool (USDA-ARS, Washington, DC, USA) [54] to produce daily sheet and rill erosion from fields [61]. AnnAGNPS divides the watershed into drainage areas called ‘cells’ that can have any shape, and each cell is assumed to have homogenous management and soil [28]. These cells portray the spatial variability of land use, soil, and topography within the watershed. These simulated cells are then integrated by simulated streams and rivers, which route the runoff and pollutants from every single homogeneous area downstream.

2.4. AnnAGNPS Data Input

For the execution of an AnnAGNPS model, the major input data are climate, land characteristics (e.g., topography, soils), field operations, chemical characteristics, and feedlot operations. Topography information about the three studied watersheds was acquired from the United States Geological Survey (USGS)—The National Map Viewer (TNM Viewer version 2.0, USGS, Washington, DC, USA) 7.5-min digital elevation models (DEM)—with a 10-m horizontal, 7-m vertical resolution. It was
used to obtain the necessary input data for running the TOPAGNPS (Topographic AGNPS) program version 5.45.a.011 (United States Department of Agriculture (USDA)-Agricultural Research Service (ARS), National Sedimentation Laboratory, Oxford, MS, USA), a Geographic Information System (GIS)-based landscape analysis component of AnnAGNPS that is used to generate the input parameters of the model. TOPAGNPS requires a user-selected watershed outlet location to produce the prerequisite model input files from the DEM dataset. The DEM was used to identify and measure the topographic features, to define surface drainage channels, to subdivide watersheds into cells along drainage divides and also to calculate representative cell parameters (cell area, slope, and length). The size of the cells depends on the values of the Critical Source Area (CSA) and Minimum Source Channel Length (MSCL) [34]. The CSA is defined as the minimum upstream drainage area required for a channel to form, while the MSCL is the minimum acceptable length of concentrated flow in a cell before a stream channel can be defined [60]. The CSA and MSCL values are critical to determining the extent of the stream network and resulting AnnAGNPS cells. Various combinations of CSA and MSCL values were applied until an accurate representation of the stream network and of the land use of the studied watersheds was acquired. For the three sites, CSA ranged from 5–170 ha and MSCL from 30–130 m (Figure 1). The number of cells per watershed ranged from 185 to 1800. The soil data are directly populated from United States Department of Agriculture (USDA)-Natural Resources Conservation Service (NRCS) Soil Survey
Center’s National Soil Information System (NASIS) data. NASIS data are associated with The Soil Survey Geographic (SSURGO) soil map. This soil map was overlaid onto the delineated watershed using the AGNPS GIS tool, and the dominant soil type for each subwatershed cell was determined. Then land use map obtained from National Land Cover Database (NLCD 2011)—United States Geological Survey (USGS) was also overlaid onto the delineated watershed using the AGNPS GIS tool. The six daily climate parameters needed for AnnAGNPS are (1) minimum air temperature; (2) maximum air temperature; (3) precipitation; (4) dew point; (5) solar radiation; and (6) wind speed. The data for three daily climate parameters—minimum air temperature, maximum air temperature, and precipitation—were acquired from the PRISM website at 4km spatial resolution (Parameter-elevation Regressions on Independent Slopes Model (PRISM) Climate Group, Oregon State University, created 4 February 2004). The remaining three daily climate parameters—dew point, solar radiation, and wind speed—were acquired from Texas A&M University’s global weather data site [61].

2.5. Observed Data

For stream flow data used in the calibration and validation, we used the daily observations from USGS gauging stations. These included USGS 03361650 Sugar Creek at New Palestine, Indiana (39°42'51" N, 85°53'08" W) for Sugar Creek watershed, the USGS 04234000 Fall Creek near Ithaca, New York (42°27'12" N, 76°28'22" W) for
the Fall Creek watershed, and the USGS 01117500 Pawcatuck River at Wood River Junction, Rhode Island (41°26′42″ N, 71°40′53″ W) for the Pawcatuck watershed. The traditional manual baseflow filtering approach was applied to the streamflow record to obtain runoff by removing baseflow from streamflow before comparison with AnnAGNPS output, as baseflow is not considered in the model [29].

2.6. Model Assessment

The performance of model was evaluated by comparing observed and AnnAGNPS modeled data at the watershed outlet. The assessment of the model was accomplished for runoff on both daily and monthly time scales. Assessment of model performance for runoff included both qualitative and quantitative methods. Qualitative methods included comparing graphs of observed and modeled data. We followed the recommendation of [62] and used three quantitative statistics: Nash–Sutcliffe efficiency ($\text{NSE}$), percent bias ($\text{PBIAS}$), and ratio of the root mean square error to the standard deviation of measured data ($\text{RSR}$), along with the graphical techniques, to model performance evaluation. Generally, model simulation can be judged as satisfactory when $\text{NSE} > 0.50$ and $\text{RSR} < 0.70$, and also when $\text{PBIAS} \pm 25\%$ for streamflow [62]. We also used the coefficient of determination ($R^2$) for quantitative evaluations; $R^2$ represents the variation in measured data explained by the model [62]. Values can range from 0 to 1, with 1 indicating that all variations in the measured data are explained by the model. Values greater than 0.5 are normally considered
acceptable [62]. According to Nash and Sutcliffe [63], $NSE$ is a normalized statistic that defines the relative magnitude of the residual variance when compared to the variance in the measured data. The statistic denotes how well the observed data fit the modeled data in the 1:1 line. The $NSE$ value ranges from $-\infty$ to 1 with 1 representing a perfect fit. Values between 0 and 1 are considered an acceptable performance level for the model [62].

$NSE$ is computed as shown in Equation (1):

$$NSE = 1 - \left[ \frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - Y_{mean})^2} \right]$$

(1)

Where $Y_i^{obs}$ is the $i$th observation for the constituent being evaluated, $Y_i^{sim}$ is the $i$th simulated value for the constituent being evaluated, $Y_{mean}$ is the mean of observed data for the constituent being evaluated, and $n$ is the total number of observations.

$PBIAS$ is computed as shown in Equation (2):

$$PBIAS = \left[ \frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim}) \times 100}{\sum_{i=1}^{n}(Y_i^{obs})} \right]$$

(2)

$RSR$ is computed as shown in Equation (3):

$$RSR = \left[ \frac{\sqrt{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}}{\sum_{i=1}^{n}(Y_i^{obs} - Y_{mean})^2} \right]$$

(3)

2.7. Model Calibration, Validation and Sensitivity Analysis for Runoff Simulation
The SCS-CN, the most important parameter in the AnnAGNPS model for simulating runoff, is utilized in many studies to calibrate runoff [34–37]. For that reason, the SCS curve number was also used to calibrate runoff in this study. For Sugar Creek watershed, the AnnAGNPS model was calibrated for runoff from 1 January 2000 to 31 December 2007 (average annual 1178.1 mm precipitation) and validated from 1 January 2008 to 31 December 2013 (average annual 1199.2 mm precipitation). For Fall Creek watershed, the AnnAGNPS model was calibrated for runoff from 1 January 2000 to 31 December 2007 (average annual 994.4 mm precipitation) and validated from 1 January 2008 to 31 December 2013 (average annual 969.3 mm precipitation). For Pawcatuck watershed, the AnnAGNPS model was calibrated for runoff from 1 January 2000 to 31 December 2004 (average annual 1247.1 mm precipitation) and validated from 1 January 2008 to 31 December 2013 (average annual 1340.6 mm precipitation). Before performing the watershed simulation, the model was initialized for two years.

In this study, we also evaluated the sensitivity value of the most sensitive parameter (SCS-CN) used for runoff estimation. The sensitivity analysis for CN was performed by the modification or adjustment of the curve number within the recommended range (30–100). The lower numbers indicate low runoff potential, whereas higher numbers signify increasing runoff potential. We utilized the integration of a local method into a global sensitivity method (the random one-factor-at-a-time) design proposed by [64]. This method consists of repetitions of a local
method whereby the derivatives are calculated for each parameter by adding a small change to the parameter. The change in model outcome can then be measured by some lumped measure such as total mass export, sum of squares error between modeled and observed values or sum of absolute errors. The sensitivity analysis and the calibration of streamflow for the AnnAGNPS model were manually calibrated as in other studies [36].

Based on [65], the selection of initial SCS CNs for the different land use types was completed. Sugar Creek watershed consists of various land uses like cropland (only corn), cropland (corn–soybean rotation), fallow land, forested and urban area. Initially, the CN for a straight row crop with poor hydrological conditions was used for both corn and corn–soybean rotation during the growing season, while the CN for a fallow field with crop residue and good hydrological conditions was used after harvest during the non-growing season. The CN for woods with good hydrological conditions was used for forested areas. The CN for urban areas with 85% impervious cover was used for urban areas (Table 1). The sensitivity analysis was done only for croplands. The CN for Row Crop (SR—Poor) was adjusted by running the model a number of times and by relatively changing the value of CN from its initial value by ±9.8% to ±2.8% during the calibration phase.
Fall Creek watershed consists of various land uses like urban (developed), forested, some cropland (tall grass, squash, potato), and hay/pasture. At the start of the calibration, the CN for a straight row crop with poor hydrological conditions was used for potato, while the CN for a contoured row crop with poor hydrological conditions was used for tall grass and squash fields. The CN for woods with poor hydrological conditions was used for forested areas. The CN for urban land use with the newly graded condition was used for urban areas (Table 2). For this watershed, we focused our sensitivity analysis on the croplands (tall grass and squash fields). The CN for Row Crop (C—Poor) was adjusted by running the model for a number of times by relatively changing the value of CN from its initial value by ± 13.6% to ± 2.6% during the calibration phase.

Table 1 Curve numbers (CN) used for model calibration, Sugar Creek watershed.

<table>
<thead>
<tr>
<th>Cover Description</th>
<th>Curve Number for Hydrological Soil Groups</th>
<th>Initial Values</th>
<th>Values After Calibration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>A</td>
<td>B</td>
</tr>
<tr>
<td>Row Crop (SR—Poor)</td>
<td></td>
<td>72</td>
<td>81</td>
</tr>
<tr>
<td>Fallow (CR—Good)</td>
<td></td>
<td>74</td>
<td>83</td>
</tr>
<tr>
<td>Woods (Good)</td>
<td></td>
<td>30</td>
<td>55</td>
</tr>
<tr>
<td>Urban (85% imp)</td>
<td></td>
<td>89</td>
<td>92</td>
</tr>
</tbody>
</table>

SR—straight row, CR—crop residue cover.
Table 2 Curve Numbers (CN) used for model calibration, Fall Creek watershed.

<table>
<thead>
<tr>
<th>Cover Description</th>
<th>Curve Number for Hydrological Soil Groups</th>
<th>Initial Values</th>
<th>Values After Calibration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>A</td>
<td>B</td>
</tr>
<tr>
<td>Row Crop (SR—Poor)</td>
<td></td>
<td>72</td>
<td>81</td>
</tr>
<tr>
<td>Urban (Newly graded)</td>
<td></td>
<td>77</td>
<td>86</td>
</tr>
<tr>
<td>Woods—Grass (Poor)</td>
<td></td>
<td>57</td>
<td>73</td>
</tr>
<tr>
<td>Fallow (CR—Poor)</td>
<td></td>
<td>76</td>
<td>85</td>
</tr>
<tr>
<td>Row Crop (C—Poor)</td>
<td></td>
<td>70</td>
<td>79</td>
</tr>
</tbody>
</table>

SR—straight row, CR—crop residue cover, C—contoured.

The Pawcatuck watershed consists mostly of forested areas, some urban areas and agricultural fields (turf). At the beginning of the calibration the CN for woods with good hydrological conditions was used for forested areas. The CN for a straight row crop with good hydrological conditions was used for turf during the growing season, while the CN for crop residue cover with good hydrological conditions was used after harvest during the non-growing season. The CN for urban area with 85% impervious cover was used for urban areas (Table 3). We performed our sensitivity analysis for cropland and forested area for this watershed. The CN for Row Crop (SR—Good), Row Crop (C—Poor) and Woods (Good) were adjusted by running the model a number of times by relatively changing the value of CN from its initial value by ± 10% to ± 30% during the calibration phase.
Table 3 Curve Numbers (CN) used for model calibration, Pawcatuck River watershed.

<table>
<thead>
<tr>
<th>Cover Description</th>
<th>Curve Number for Hydrological Soil Groups</th>
<th>Initial Values</th>
<th>Values After Calibration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>A</td>
<td>B</td>
</tr>
<tr>
<td>Row Crop (SR—Good)</td>
<td></td>
<td>67</td>
<td>78</td>
</tr>
<tr>
<td>Row Crop (C + CR—Good)</td>
<td></td>
<td>64</td>
<td>74</td>
</tr>
<tr>
<td>Urban (85% imp)</td>
<td></td>
<td>89</td>
<td>92</td>
</tr>
<tr>
<td>Woods (Good)</td>
<td></td>
<td>30</td>
<td>55</td>
</tr>
</tbody>
</table>

SR—straight row, CR—crop residue cover, C—contoured.

After the initial run of the model without calibration, the model was calibrated to support a better estimation of runoff. The model performance improved for both daily and monthly runoff calculations after calibration. The results were evaluated using both graphical and statistical methods [64] until the best simulation results were obtained. For runoff validation, all model parameters after calibration were kept the same, and the simulated data were compared with the observed runoff data. Following calibration in the Pawcatuck watershed, the CN for the turf crop was substantially lower than for a straight row crop with good hydrological conditions, reflecting the higher infiltration rates of turf grass (Table 3).

3. Results

3.1. Runoff Calibration and Validation

According to the classification tabulated in [31] for model correlations and efficiencies modified from [62], our calibrated model for the Sugar Creek watershed
predicted the daily runoff volume of the watershed with good correlation and good agreement ($R^2 = 0.57$, $NSE = 0.57$ for daily and $R^2 = 0.67$, $NSE = 0.63$ for monthly calibration) between daily observed and daily modeled runoff volume (Table 4, Figure 2a). The calibrated model, when applied to the same watershed for the validation phase, predicted a daily runoff volume with good correlation and good agreement for both daily and monthly scales ($R^2 = 0.58$, $NSE = 0.57$ for daily and $R^2 = 0.72$, $NSE = 0.68$ for monthly) (Table 4, Figure 2b). Total runoff estimation by the model during the calibration phase differed from the observed runoff by only about 6.44%, whereas it differed by about 20.5% during validation. The calculated $PBIAS$ value for calibration was less than 10, which indicated an excellent calibration performance rate. The model was biased to overestimate runoff volume during both calibration and validation phases. The observed runoff volumes from January 2000 to December 2013 at the watershed outlet were used for model calibration and validation at daily and monthly scales. The model over-predicted some runoff volumes during the drier months (December to February), whereas it under-predicted some during the wetter months (May to August) (Figure 2).

Table 4 Runoff calibration and validation results for Sugar Creek watershed.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Calibration Period (1 January 2000 to 31 December 2007)</th>
<th>Validation Period (1 January 2008 to 31 December 2013)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Daily</td>
<td>Monthly</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.57</td>
<td>0.67</td>
</tr>
<tr>
<td>$NSE$</td>
<td>0.57</td>
<td>0.63</td>
</tr>
<tr>
<td>$PBIAS$</td>
<td>$-6.44%$</td>
<td>$-6.47%$</td>
</tr>
<tr>
<td>$RSR$</td>
<td>0.66</td>
<td>0.61</td>
</tr>
</tbody>
</table>
Figure 2. Graphical comparison between daily modeled and observed runoff (a) after calibration and (b) validation phase for Sugar Creek Watershed.

In the case of Fall Creek watershed, the statistical evaluation of model performance for calibration and validation is presented in Table 5. The value of NSE for both daily and monthly time scales is greater than 0.5, so the model calibration performance can be rated as good. The positive PBIAS indicated the overall underestimation of runoff by the model compared to the observed runoff volume.
(calibration phase) and the negative PBIAS indicated the overall overestimation of runoff by the model compared to the observed runoff volume (validation phase). Total runoff estimation by the model during the calibration phase differed from the observed runoff by about 16.5%, whereas it differed by about 5.5% during validation. The calculated PBIAS value for validation was less than 10, which pointed to an excellent model performance. Figure 3a,b graphically illustrates observed and modeled daily runoff volume at the USGS 04234000 for calibration and validation phase, respectively, for Fall Creek watershed. The model over-predicted some runoff volumes during the months in which less precipitation occurred (December to March), whereas it under-predicted some during the months in which more precipitation occurred (April to August) (Figure 3). This tendency of the model could be improved by adjusting the evaporation rate associated with the interception of precipitation events.

Table 5 Runoff calibration and validation results for Fall Creek watershed.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Calibration Period</th>
<th>Validation Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1 January 2000 to 31 December 2007)</td>
<td>(1 January 2008 to 31 December 2013)</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>Monthly</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.54</td>
<td>0.64</td>
</tr>
<tr>
<td>$NSE$</td>
<td>0.51</td>
<td>0.60</td>
</tr>
<tr>
<td>$PBIAS$</td>
<td>16.48%</td>
<td>16.59%</td>
</tr>
<tr>
<td>$RSR$</td>
<td>0.70</td>
<td>0.63</td>
</tr>
</tbody>
</table>
Figure 3. Graphical comparison between daily modeled and observed runoff after (a) after calibration and (b) validation phase for Fall Creek Watershed.

In the case of Pawcatuck River watershed, graphical comparisons of daily observed and modeled runoff volumes at the USGS 01117500 were presented in Figure 4a,b for the calibration and validation phase, respectively. The statistical evaluation of model performance for calibration and validation is presented in Table 6. The range of NSE (0.51 to 0.54 for daily and 0.68 to 0.83 for monthly) showed a good agreement between daily observed and modeled runoff volume. The positive PBIAS
indicates the overall underestimation of runoff by the model compared to the observed runoff volume for both calibration and validation. Total runoff estimation by the model during the calibration phase differed from the observed runoff by about 21.5%, whereas it differed by about 5.8% during validation. The calculated PBIAS value for validation was less than 10, which pointed to an excellent model performance. The results show a general tendency for AnnAGNPS to overestimate spring (March–May) and summer (June–August) runoff volumes compared to observed data for both calibration and validation periods (Figure 4).

**Figure 4.** Graphical comparison between daily modeled and observed runoff (a) after calibration and (b) validation phase for Pawcatuck River Watershed.
Table 6 Runoff calibration and validation results for Pawcatuck River watershed.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Calibration Period</th>
<th>Validation Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1 January 2000 to 31 December 2004)</td>
<td>(1 January 2008 to 31 December 2013)</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>Monthly</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.62</td>
<td>0.75</td>
</tr>
<tr>
<td>NSE</td>
<td>0.51</td>
<td>0.68</td>
</tr>
<tr>
<td>PBIAS</td>
<td>21.56%</td>
<td>21.15%</td>
</tr>
<tr>
<td>RSR</td>
<td>0.70</td>
<td>0.56</td>
</tr>
</tbody>
</table>

The estimated RSR values varied from 0.65 to 0.70 for daily (fair) and 0.41 to 0.63 for monthly (very good to fair) for all three watersheds during calibration and validation periods.

3.2. Sensitivity Analysis

After the sensitivity analysis, it was quite clear how sensitive the CN was for runoff simulation in AnnAGNPS model. The sensitivity analysis demonstrated differences between sites in the response of modeled runoff to changes in CN. In Sugar Creek, the percent change in runoff volume (2000–2007 or 7-year average) from its initial condition was found to be up to 5.5 due to 9.8% changes in the CN (Figure 5).
Figure 5. Sensitivity analysis for CN for a straight row crop with poor hydrological conditions; Sugar Creek watershed.

In the case of Fall Creek watershed, during the entire phase of modification of CN, the percent change in runoff volume (2000–2007 or 7-year average) from its initial condition ranged from 11 to 149 due to 2.6% to 13.6% changes in the CN (Figure 6).

Figure 6. Sensitivity analysis for CN for a contoured row crop with poor hydrological conditions; Fall Creek watershed.
In the case of Pawcatuck River watershed, during the entire phase of modification of CN for Row Crop (SR—Good), the percent change in runoff volume (2000–2004 or 5-year average) from its initial condition ranged from 1.4 to 180 due to −30% to 30% changes in the CN (Figure 7a). The percent change in runoff volume (2000–2004 or 5-year average) from its initial condition ranged from 0.41 to 9.5 due to −10% to 30% changes in the CN for Row Crop (C + CR—Good) (Figure 7b). The percent change in runoff volume (2000–2004 or 5-year average) from its initial condition ranged from 9 to 82 due to −10% to 30% changes in the CN for Woods (Good) (Figure 7c).

Figure 7. Sensitivity analysis for (a) CN for Row Crop (SR—Good), (b) CN for Row Crop (C + CR—Good), (c) CN for Woods (Good), Pawcatuck River Watershed.
3.3. Spatial Distribution of Runoff Depth

Since the AnnAGNPS model is able to provide landscape spatial variability by representing a watershed with a number of land areas (cells), we also evaluated the average annual runoff depth for all the watersheds (Figure 8). In this figure, the different shades of color indicate different average annual runoff depths in mm/year for each individual cell in a watershed. The darker shades in the figure represent higher runoff depths. At the outlet of the Sugar Creek Watershed, the average annual runoff depth is 755.1 mm and 830.5 mm for calibration period (2000–2007) and validation period (2008–2013), respectively. The runoff depth for each cell (mm/year) for this watershed ranged from 1.45 to 955 mm/year. At the outlet of the Fall Creek Watershed, the average annual runoff depth is 174.0 mm and 185.5 mm for calibration period (2000–2007) and validation period (2008–2013), respectively. The runoff depth for each cell (mm/year) for this watershed ranged from 3.51 to 326 mm/year. At the outlet of the Pawcatuck River Watershed, the average annual runoff depth is 97.1 mm and 135.9 mm for calibration period (2000–2004) and validation period (2008–2013), respectively. The runoff depth for each cell (mm/year) for this watershed ranged from 0.0 to 475.8 mm/year.
Figure 8. Spatial distribution of average annual runoff depth in mm/year for: (a) calibration period, Sugar Creek Watershed; (b) validation period, Sugar Creek Watershed; (c) calibration period, Fall Creek Watershed; (d) validation period, Fall Creek Watershed; (e) calibration period, Pawcatuck River Watershed; (f) validation period, Pawcatuck River Watershed.
3.4. Model Performance to Estimate Event Peak Discharge

After the successful calibration and validation of runoff volumes, the model performance for the estimation of event peak discharge was done. For this purpose, we first looked at the gage height data for the selected USGS gauging stations. Then, we picked only the events when the gage height exceeded the existing flood stage (determined by USGS) for the particular station. The existing flood stage is 2.4 meters, 1.8 meters and 1.5 meters at USGS 03361650, USGS 04234000 and USGS 01117500, respectively. Figure 9 presents a graphical comparison between observed and model simulated event peak discharge (cubic meter per second) for only those selected events for all three watersheds. The model generally underestimated the peak discharge compared to few overestimations (calibration period) for Sugar Creek Watershed. On the other hand, in the case of Fall Creek Watershed, the model performed well to capture the highest peak discharge from tropical storm Lee in September 2011 based on the entire simulation period (2000–2013). Pawcatuck River Watershed showed a similar model performance. The record peak discharge in Rhode Island from the historic flood in March 2010 was captured by the model.
AnnAGNPS performed satisfactorily for runoff prediction and simulation. Both daily and monthly calibration and validation statistics ($\text{NSE} > 0.50$ and $\text{RSR} < 0.70$, and
PBIAS ± 25% based on [62]) of the developed AnnAGNPS model was satisfactory for runoff simulation for all the watersheds in the study. The model is categorized as satisfactorily performing when the range of the NSE value falls between 0.36 and 0.75 [66].

This study tested the applicability of the AnnAGNPS model on a glaciated landscape, which is why the three chosen watersheds are located in a glacial geomorphic setting. The area of the studied watersheds varied from 69 to 328 km². After the evaluation of developed model performance, it was learned that the range of NSE varied from 0.51 to 0.57 and 0.60 to 0.83 for daily and monthly scale, respectively. This range is quite similar or even better than the 0.69 to 0.75 (on a monthly scale) found by [34] for a Mediterranean agricultural watershed (2.07 km²) in Spain, 0.73 (on a monthly scale) by AnnAGNPS found by [26] for a 289.3 km² watershed in Illinois, USA, 0.65 (on a monthly scale) by SWAT and 0.48 to 0.58 (on a monthly scale) by AnnAGNPS found by [36] for a Mediterranean watershed (506 km²) in Southern Italy, 0.53 (on a daily scale) by SWAT found by [67] for a large 1110 km² agricultural watershed in southwest France, 0.67 to 0.84 (on a monthly scale; calibration phase) found by SWAT for a large 4000 km² watershed in the North Carolina coastal plain [68], and the 0.53 to 0.62 (on a daily scale) found by SWAT for two watersheds in a semiarid region of Iraq [69]. Even, if we look at the value of R² from our study, it ranged from 0.54 to 0.66 (daily) and 0.64 to 0.86 (monthly). This range is also better than the one (0.50 to 0.80 by AnnAGNPS and 0.62 to 0.81 by SWAT
on a monthly scale) found by [31] for an agricultural watershed in south-central Kansas. We also compared our developed AnnAGNPS model evaluation results with the results found from another water quality model, GLEAMS, in a study by [70] for agricultural watersheds in Indiana. The runoff calibration results were reported by [70] as $\text{NSE} = 0.62$ and $R^2 = 0.70$ for a monthly scale, which is quite similar to our results ($\text{NSE} = 0.63$ and $R^2 = 0.67$ for Sugar Creek; $\text{NSE} = 0.60$ and $R^2 = 0.64$ for Fall Creek; $\text{NSE} = 0.68$ and $R^2 = 0.75$ for Pawcatuck) for our studied watersheds.

Simulated runoff followed a similar trend (seasonal fluctuation) to observed runoff. In general, the model performed better in capturing event peak discharge for Fall Creek and Pawcatuck River watersheds rather than Sugar Creek Watershed. This poor performance of the model at the Sugar Creek Watershed could be improved by testing the effect of different storm types for rainfall distribution. As the regression coefficients for calculating the unit peak discharge are determined by storm type, the storm type within the AnnAGNPS model significantly influences peak discharge [71]. Simulated peak runoff was underestimated during some flood periods such as a major January 2005 flood event, record December 2013 flooding in Sugar Creek watershed, and a major April 2005 flood in Fall Creek watershed. In most cases, the model could not accurately simulate the flood runoff when the river overflowed. This characteristic of a hydrological model such as, AnnAGNPS is similar to that found in studies carried out by other studies (AnnAGNPS and SWAT performed poorly due to several extraordinary floods recorded during the study period in [36]; the SWAT model
underestimated the largest flood in the study of [67]). However, the model was able to capture the historic September 2011 flood caused by tropical storm Lee in the case of Fall Creek watershed. In addition to that, the model even successfully captured the peak runoff volume (3815724.36 m$^3$ on 30 March 2010) generated from the historic 2010 flood in Rhode Island during the validation phase. Daily simulated runoff was also overestimated for some periods. Larger errors occurred when simulated peak runoff and average runoff differed significantly from the observed runoff volume. The spatial variability of the runoff depth (Figure 8) could be attributed to the differences in land use, topography, soil type, and soil physical characteristics [58]. The output (spatial variability, i.e., cell wise runoff depth within the watershed) of the AnnAGNPS model contributes the riparian model field input data required for further research.

5. Conclusions

Model performance during calibration and validation phases shows that AnnAGNPS can be successfully used to predict runoff from watersheds in the glaciated settings of the Northeast and Midwest United States. This provides the capacity to couple edge-of-field hydrologic modeling with models that examine riparian or riverine functions and behaviors. The AnnAGNPS model effectively estimated runoff volume and portrayed the seasonal pattern of runoff in all the studied watersheds. The developed AnnAGNPS model could not capture some peak
runoff events during wet periods and formed some unnecessary over-prediction of runoff during dry periods of the year. This characteristic of the model could be improved by the readjustment of the evaporation rate in association with the interception of precipitation events. The sensitivity analysis was limited to one specific contributing land use only, which could be extended by considering all possible combinations of CN based on the mixed land use of the watershed.

**Author Contributions:** M.T. was responsible for the conceptualization, methodology, data processing, software, calibration, validation, formal analysis, writing—original draft preparation, and editing; S.M.P. was responsible for the conceptualization, funding acquisition, methodology, model/software supervision, and writing—review and editing; A.J.G., K.A. and P.G.V were responsible for the conceptualization, funding acquisition, and writing—review and editing; R.L.B. was responsible for model/software supervision, as well as writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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CHAPTER 2

Manuscript II

Riparian Zone Nitrogen Management through the Development of the Riparian Ecosystem Management Model (REMM) in a Formerly Glaciated Watershed of the USA Northeast

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Abstract

The Riparian Ecosystem Management Model (REMM) was developed, calibrated and validated for both hydrologic and water quality data for eight riparian buffers located in a formerly glaciated watershed (upper Pawcatuck River Watershed, Rhode Island) of the US Northeast. The Annualized AGricultural Non-Point Source model (AnnAGNPS) was used to predict the runoff and sediment loading to the riparian buffer. Overall, results showed REMM simulated water table depths (WTDs) and groundwater NO$_3$-N concentrations at the stream edge (Zone 1) in good agreement with measured values. The model evaluation statistics showed that, hydrologically REMM performed better for site 1, site 4, and site 8 among the eight buffers, whereas REMM simulated better groundwater NO$_3$-N concentrations in the case of site 1, site 5, and site 7 when compared to the other five sites. The interquartile range of mean absolute error for WTDs was 3.5 cm for both the calibration and validation periods. In the case of NO$_3$-N concentrations prediction, the interquartile range of the root mean square error was 0.25 mg/L and 0.69 mg/L for the calibration and validation periods, respectively, whereas the interquartile range of $d$ for NO$_3$-N concentrations was 0.20 and 0.48 for the calibration and validation period, respectively. Moreover, REMM estimation of % N-removal from Zone 3 to Zone 1 was 19.7%, and 19.8% of N against actual measured 19.1%, and 26.6% of N at site 7 and site 8, respectively. The sensitivity analyses showed that changes in the volumetric water content between field capacity and saturation (soil porosity) were driving water table and denitrification.
Keywords: Riparian zone, REMM model, Nitrate (NO$_3$-N), Glaciated, Rhode Island, New England
1. Introduction

Riparian zones occur at the interface of terrestrial and aquatic components of the landscape. They regularly receive and process large amounts of excess nitrogen (N), draining out of agricultural fields towards open water bodies. They are often characterized as “filters” or “buffers” and are vital elements in watershed management schemes for water quality maintenance and stream ecosystem habitat protection [1–3].

Agriculture (cropland, pasture, managed forest) is an important component of many watersheds of the USA Northeast where N losses to major estuaries is of substantial concern. Decades of research on riparian zone hydrology and biogeochemistry has shown that riparian zones can serve as best management practices (BMPs) to mitigate the impact of agriculture (excessive leaching of nutrients, mostly N) on the quality of our waters [4–6]. Nevertheless, the buffering capacity of riparian zones (mostly for N) varies enormously due to the hydrogeomorphic setting such as topography, depth to water table, soil type, and surficial geology of the riparian zone [7–12]. Upland land use/land cover affects both the water quantity and quality of the water entering the riparian zone. Hydrogeomorphic setting can influence the flow-paths and hydrologic connections between upland sources of nitrate and the biologically active (i.e., upper 1–2 m) portions of the riparian zone [7,13,14]. Thus, a number of key attributes related to location are critical in determining the potential impact of a riparian zone on water. These attributes are
incorporated in models such as the Riparian Ecosystem Management Model (REMM) [15,16]. Given the interest in expanding riparian zone BMPs, there is a critical need to advance our understanding of riparian functions at the site scale. Site-specific models can improve riparian zone management decisions that seek to place, restore and protect riparian zones more effectively.

Despite the acknowledged value of riparian zones in mitigating N pollution, only a limited number of numerical models or landscape-based approaches have been developed that can improve the use and management of riparian zones to achieve water quality improvements in physiographic settings associated with landscapes that were formed by glaciation. Several past studies include: statistical models to develop functional relationships between riparian characteristics and N removal [17–19]; conceptual models to generalize riparian zone functions [7,14]; landscape-based approaches for the estimation of the riparian width required for achieving a 90% nitrate removal [20]; spatially distributed model for estimating nitrogen removal [21]; 3-D high-resolution reactive transport modeling to investigate the spatial and temporal variability of nitrogen fluxes in the riparian zone [22]; GIS-based tools to assess and target riparian buffers placements [23] or identify the connection between the upslope area runoff and the storage capacity of riparian buffer; overall potential estimates of riparian zones for N removal at the landscape-scale [6,24,25]; study of long term nitrate removal in stream riparian zones [26–28].
Added to these approaches, a process-based model, REMM has been used to simulate hydrology, carbon and nutrient dynamics, and plant growth in riparian zones [15,16]. REMM has been used to simulate managed riparian ecosystems in a number of settings in USA including Chesapeake Bay Watershed [29]; Delaware [30]; Mississippi [31]; North Carolina [32–34]; Georgia [12,35,36]; California [37]; and Puerto Rico [38]. Globally, SWAT-REMM integration has been used in a glaciated landscape in New Brunswick, Canada by [39] to examine the effect of different levels of dividing up the watershed into sub-watersheds, for SWAT on the performance of the model. Reference [40] used REMM in China for the evaluation of riparian zones as BMP. However, REMM has not yet been integrated with the AnnAGNPS model and applied to evaluate management at the field scale in the glaciated settings of the Northeast region, even though the agricultural lands are linked to excessive nutrient pollution and riparian zones are widely used in these regions to mitigate N losses to streams. Therefore, our focus on field scale analyses with AnnGNPS provides more insight into site scale behavior.

Although REMM offers users the potential for quantitative assessments of riparian functions at the site scale, it requires a considerable amount of site-specific information to parameterize and run, including information on water and nutrient flux from source areas that contribute to the riparian zone. The absence of site specific data frequently results in users relying on default parameters. REMM simulations are also not bounded by maximum or minimum values, which can lead to unrealistic
simulation results if the model is poorly parameterized or not validated adequately with empirical data. Therefore, we suggest there exists a critical need to determine the usability of REMM in glaciated settings of the USA Northeast that is informed by (1) field data to offer an independent way of generalizing riparian function in these regions; (2) site-specific estimates of water flux and nutrient loading from uplands to the riparian zones.

The goals of this field scale study were to test the ability of the REMM model in formerly glaciated setting of Rhode Island (RI), USA for riparian zone nitrate dynamics. Specifically, this modeling study demonstrates these aspects via evaluation of the REMM model’s ability to simulate the basic hydrologic (water table depths or WTDs) and water quality (groundwater nitrate (NO$_3$-N) concentrations) parameters by using site-specific field data. This process involved (i) REMM model set-up, including site-specific inputs from uplands to a number of monitored riparian sites in RI, (ii) improvements to the model’s capacity for water table depth and groundwater nitrate concentration simulation through calibration of the developed model by means of comparing model outputs with field data collected from eight buffer sites in RI, (iii) validation of model’s output with the improved calibrated parameters, (iv) conduct a parameter sensitivity analysis. Ultimately, this approach will facilitate the use of this model in this region and improve its functionality with respect to nitrogen transformations and flux.
2. Materials and Methods

2.1 Site Description

Our study focused on eight riparian sites from the state of RI, USA. All the sites are located in upper Pawcatuck River Watershed, Washington County, set in the New England Hydrologic Region of southern RI (41°32′30″ N, 71°35′ W) (Figure 1) and all were monitored for hydrology and water quality. The area of this watershed is about 258 km². It consists mainly of forests (above 65% of the total watershed) and agricultural fields (about 32%). The soil parent materials in the watershed are comprised mostly of glacial till, glacial outwash, and organic and alluvial deposit [13]. Agricultural lands (mostly turf farms) are predominately located on loess soils over glacial outwash. Forested settings are usually on till. The elevation of the watershed ranges from 16 m (shoreline) to 144 m (gently rolling hills inland). It has a humid continental climate, with warm summers and cold winters. The temperature in the watershed ranges from a 30-year (1982–2011) mean of 20.8 °C in summer to a mean of −0.4 °C in winter (PRISM Climate Group, accessed on 21 March 2019). The 30 years (1982–2011) average annual precipitation is approximately 1290 mm, about 50.8% of which occurs during the spring and the fall months. The 14 years (2000–2013) average annual snowfall was 79 mm.

All sites were forested riparian wetlands dominated by red maple (*Acer rubrum* L.). Half the sites were located on first or second streams, one site was located along a pond, two sites were on a 4th order stream and one riparian site bordered an
intermittent stream. The upland land use at seven of the sites was for commercial turf operations and only one site had forested uplands. Irrigation was routinely applied on a number of turf farms. Details on the soils, slope and buffer dimensions are provided in Tables 1 and 2.

Figure 1. Location of Riparian Sites in upper Pawcatuck River Watershed, Rhode Island (Dataset Sources: USA States Shapefile was obtained from United States Census Bureau via their cartographic boundary files—shapefiles [41]; stream lines—shapefile was downloaded from open source Rhode Island Geographic Information System [42]; Pawcatuck River Watershed boundary was generated by a subset of TOPAZ, TOPAGNPS (the set of TOPAZ modules used for AGNPS)).
2.2 Description of the REMM Model

REMM is a field-scale process-based, two dimensional, daily time-step model that simulates interactions between hydrology, nutrient dynamics, sediment transport, and vegetation growth. REMM computes the loading of water, sediments, carbon, and nutrients coming from the upland into the riparian buffer. The model was designed such that water and total N are transported from upland to field edge (Zone 3), field edge to mid-buffer (Zone 2), mid-buffer to stream edge (Zone 1), and ultimately from stream edge to open water body by means of surface runoff, seep flow, and subsurface flow [15,16].

REMM file version 0.1.1.46 (United States Department of Agriculture (USDA)-Agricultural Research Service (ARS)) was used for all simulations. A broad description of the REMM model is available in several publications, including [12,15,16,35]. Briefly, REMM is a computer simulation model of riparian forest buffer systems. The structure of REMM is consistent with the three zone riparian system as mentioned in [4]. REMM was originally field tested using a five-year hydrologic and nutrient dataset collected from an experimental riparian buffer site in Tifton, Georgia [12,35]. We used this model, and prepared the model set-up for each riparian site, parameterized, calibrated and validated for both hydrologic and nutrient simulation.

Within the model, the riparian system is considered to consist of three zones (parallel to a stream) between the field and the water body. However, zone 2 is not visibly distinguished at the riparian sites in this study, as they tend to move abruptly
from zone 1 to zone 3 (Figure 2). Each zone includes litter and three soil layers (through which the vertical and horizontal movement of water takes place) that terminate at the bottom of the plant root system, and a plant community that can include six plant types in two canopy levels. The riparian system characterized in REMM was originally designed to represent increasing levels of management away from the stream [15]. REMM is written in the C++ programming language.

**Figure 2.** Cross-section of riparian buffer system at all the sites as simulated in REMM.

In the REMM module, movement of water and storage is defined by several processes, i.e., interception, evapotranspiration (ET), infiltration, vertical drainage, surface runoff, subsurface lateral flow, upward flux from the water table in response to ET, and seepage or exfiltration. These processes are simulated for Zone 3, Zone 2, and Zone 1. The water movement and storage between the zones is based on a
A combination of mass balance and rate-controlled approaches. Equation (1) presents the mass balance of water within each soil layer:

\[
SM(t) = SM(t-1) + Q_{V-in(t)} - Q_{V-out(t)} + Q_{H-in(t)} - Q_{H-out(t)} - ET(t) \quad (1)
\]

where \(SM(t)\) (mm) is the soil moisture on day \(t\), \(SM(t-1)\) (mm) is the soil moisture from the preceding day, \(Q_{V-in}\) (mm) is the addition of water as a result of infiltration in case of the upper soil layer, or drainage from upper soil layer for intermediate soil layers, \(Q_{V-out}\) (mm) is drainage out of the layer, \(Q_{H-in}\) (mm) is contribution because of lateral subsurface flow, \(Q_{H-out}\) (mm) is the outflow of water to lateral subsurface downslope flow, and \(ET\) (mm) is evapotranspiration [12]. REMM hydrologic outputs generated from the water balance simulation include daily surface and subsurface losses to the water body, evapotranspiration and deep seepage. Deep seepage is specified by the user for each zone as input only, such that the water that is lost through the deep seepage never comes back into REMM computations [16,35].

Reference [35] described the equation used in REMM model to simulate denitrification. Denitrification is calculated as the function of the interaction of factors representing the degree of anaerobiosis, temperature, nitrate—N, and available carbon:

\[
\text{Nitrate-N}_{\text{denitrification},t} = \text{Minimum of } \left\{ \text{Nitrate-N}_t \text{ or } \left( K_d \times S_d \times A_f \times T_{\text{denitrification},t} \times \left( \alpha \times N_f + C_f \right) \right) \right\} \quad (2)
\]

where \(K_d\) is the rate of denitrification under optimal conditions (kg cm\(^{-3}\) ha\(^{-1}\)), \(S_d\) is the depth of the soil layer (cm), \(A_f\) is the scaler factor representing the effect of anaerobiosis.
on denitrification (0–1), $T_{\text{denitrification}}$ is the scaler factor representing the effect of temperature on denitrification (0–1), $N_f$ is the scaler factor representing the effect of nitrate–N on denitrification (0–1), $C_f$ is the scaler factor representing the effect of available carbon on denitrification (0–1), and $\alpha$ is a coefficient determining the influence of nitrate on denitrification set at 0.19.

According to [43], REMM has been developed as a hillslope-scale mechanistic model to predict how the width and composition of riparian habitats impact material loadings to streams. Particularly, when loadings of sediment and nutrients to the riparian zone are known, then REMM can be used to simulate the effect of riparian buffers on stream chemistry. Similar to other mechanistic models, the application of REMM is limited by its complexity and large needs for input data. REMM operates at the hillslope rather than catchment scale. It cannot forecast effects on instream ecological endpoints.

### 2.3 REMM Model Input Data

Parameters are input into the model through four basic files that include: (1) contributions of daily outputs from the field draining into the riparian system including surface runoff and associated eroded sediment, organic material and plant nutrients (*.FIN), (2) weather data (*.WEA), (3) vegetation (*.VEG) characteristics, and (4) soil physical and chemical parameters (*.BUF). The last two of these describe in detail the morphology of the buffer system being modeled. One of the key reliable
inputs needed to simulate riparian system in the REMM model is the water and nutrients originating from the upland source area. These can be field measured or simulated using models such as AnnAGNPS (AGricultural Non-Point Source) and APEX (Agricultural Policy/Environmental eXtender). We had to rely on AnnAGNPS simulated upland data due to lack of field measured data (surface runoff and sediment data) at the edge of the field. We calibrated and validated the AnnAGNPS generated runoff against USGS measured streamflow at the watershed outlet and used as field input to REMM. These calibrated inputs will increase the reliability of REMM buffer simulations.

**2.3.1 Upland Inputs (*.FIN file)**

Upland inputs comprise the daily flux of upland water, associated sediment, sediment-borne chemicals, and dissolved chemicals entering the upper side of the buffer system during the period of simulation. REMM requires daily subsurface and surface flow data from the field or upland area. In general, this information is missing from most REMM studies, which can hinder the ability of REMM to properly predict riparian functions at the site scale [39]. To offset this issue, a field-scale hydrological model, Annualized AGRicultural Non-Point Source (AnnAGNPS), was used to predict the runoff and sediment loading to the riparian buffer [44,45]. AnnAGNPS [46,47] is a daily time step, watershed scale, pollutant-loading, distributed model developed to simulate long-term runoff, sediment, nutrients, and pesticide transport from
The AnnAGNPS model defines cells of various sizes; contaminants are routed from these cells into the associated reaches, and the model either deposits pollutants within the stream channel system or transports them out of the watershed [47]. TOPAZ (one of the modules of AnnAGNPS) is the TOpographic PArameteriZation program which generates cell and stream network information from the watershed digital elevation model (DEM). It also provides all of the topographic related information for AnnAGNPS. DEM was acquired from the United States Geological Survey (USGS)—The National Map Viewer (TNM Viewer version 2.0, USGS, Washington, DC, USA) 7.5-min digital elevation models (DEMs)—with a 10-m horizontal, 7-m vertical resolution [51]. The soil map [52] and land use map [53] were also incorporated for watershed delineation. The simulated runoff and sediment loading from AnnAGNPS was calibrated by comparing with observed data (USGS gauge). The calibrated daily runoff and the sediment loading were used as input data in .FIN (Field Data File). The rationale behind the use of AnnAGNPS model rather than other hydrological models, such as SWAT, is that AnnAGNPS divides the entire watershed into a number of cells and generates cell wise upland input data file compatible to *.FIN file format in REMM model. For this study, AnnAGNPS was used with a cell size of 1.8–0.4 km$^2$ (interquartile range, IQR = $Q_3 - Q_1$, of cell size among a total of 185 cells in the watershed) to simulate input cells representing upland inputs to each of the eight riparian buffers within upper Pawcatuck River watershed. Details of the application of AnnAGNPS model on the study area can be found in our
companion study [45]. Table 1 represents the upland characteristics for all the riparian sites. The sites are all in glacial outwash with sandy loam or fine sandy loam soil.

2.3.2 Weather Data (*.WEA file)

REMM requires daily rainfall, maximum and minimum air temperature, solar radiation, wind speed, and dew point temperature as its climate input files. All these data are obtained from the closest available national weather service cooperative observer program (NWS COOP) weather stations [54,55]. The data for three daily climate parameters, i.e., minimum air temperature, maximum air temperature, and precipitation, were acquired from PRISM website at 4 km spatial resolution (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 4 February 2004).

2.3.3 Vegetation Data (*.VEG file)

The vegetation (.VEG) data file contains plant specific information. Regional data sets are being developed that define plant characteristics typical of a region. The vegetation database created by REMM developers (USDA-ARS) separately for northeastern region was used during the simulation period. It is also accessible to other users. Maximum rooting depths (MRDs) influenced water uptake and plant transpiration which influenced ET and simulated WTDs. A maximum rooting depth of 200 cm was used for all zones for all plant species, as all riparian sites were forested,
with hardwood red maple (Acer rubrum L.) the dominant species. This was the default value used by the model developers and also by the study of [33]. Therefore, whenever no local data were available, literature values were used. Initially we looked at the influence of MRDs on the simulated outputs, but found WTDs and nitrate concentration did change at a very low rate, so we did not include MRDs in our sensitivity part. The use of MRD as 200 cm in our study provided reasonable output for our area of study, so we did not change this parameter in our model. [15] also did not include MRDs as a parameter change input in their sensitivity analysis for streamflow, total N out, denitrification and N uptake. Besides, [56] stated from their REMM sensitivity study that REMM’s comparatively low sensitivity to vegetation parameters supports the use of regional vegetation datasets that would make model implementation simple without compromising results. A specific leaf area of 0.0045 ha/kg C was used for all zones for the buffer in all sites.

Table 1 Upland Characteristics for Riparian Sites.

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Geology</th>
<th>Soil</th>
<th>Land Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site 1</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 2</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Forested</td>
</tr>
<tr>
<td>Site 3</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 4</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 5</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 6</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 7</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
<tr>
<td>Site 8</td>
<td>Outwash</td>
<td>Sandy loam</td>
<td>Agricultural</td>
</tr>
</tbody>
</table>

2.3.4 Site Characteristics (*.BUF file)
Most of the modification required for simulation was performed within the main data file (.BUF). Plant, litter and soil layer information are given in this data file. Soil characteristics for each of the three zones with three soil layers are entered in BUFFER DATA FILE (*. BUF). The characteristics include S10 Fraction, bubbling pressure, pore size distribution index, layer thickness, wilting point, field capacity, soil porosity, permeability (cm/hr), % sand, % silt, % clay, bulk density, pH, base saturation, etc. The data for most of these soil characteristics were obtained from REMM user manual based on the soil texture [57]. Also whenever there is no data available default value or value from published literature was used. The soil layers in the model are intended to correspond with horizons in the soil profile. According to [35], REMM keeps the soil physical properties constant during the simulation period. Soil parameters (saturated hydraulic conductivity, bulk density) which might change during conversion of cropland to permanent buffers may be changed at model user defined points in the simulation or can be set as intermediate point between field and mature buffer conditions. [15] stated that REMM is designed to be used at a hillslope scale to simulate the effects of buffer systems on edge-of-field loadings. REMM takes upland outputs supplied by the user and calculates loadings of water, nutrients, sediment, and carbon based on actual area of the zones of a buffer system. Similarly for plant and litter information, modifications were based on the guidelines described in the REMM User’s Manual [57]. Whenever no local data were available, literature values were used. Table 2 shows the site characteristics of the modeled buffer for all riparian
sites. All sites were located on hydric soils with alluvial or glacial outwash parent materials. Although the land uses varied among watersheds, all riparian sites were forested, with red maple (*Acer rubrum* L.), the dominant species [58]. More details about land use are available in [11,58].

2.3.5 Collection of Field Data and Other Essential Inputs for REMM

All original or field data (including site characteristics) was collected by a team led by the authors of this manuscript, with decades of published research on riparian zones in glaciated settings of the USA Northeast regions, for the development and calibration of the REMM model. Water table levels were recorded in water table wells in the riparian zone, biweekly during spring and fall when water table depths were expected to change most rapidly, and bi-monthly during summer and winter. A network of mini-piezometer nests was installed across the riparian zone from the upland to the stream. The mini-piezometers allowed the collection of nitrate samples at discrete depths and the examination of groundwater denitrification in situ using the push–pull method [59]. The information regarding in-situ groundwater denitrification capacity measurement in the study sites 1, 2 [11] and sites 3,4 [58] were based on the studies done by [11,58]. For sites 1 & 2, the in-situ groundwater denitrification rates measured are within the range reported by previous studies [59]. For sites 3 & 4, in situ groundwater denitrification capacity measured in the shallow wells was not significantly different from that measured in deep wells; thus, shallow
and deep wells were pooled (i.e., combined) for statistical analysis. A soil pit was dug and soil samples were taken from all soil horizons and analyzed for carbon content. Particle size distribution (percentage sand, silt, and clay) and soil carbon were determined for samples taken from the soil pits. The percentage sand, silt, and clay were used as inputs into REMM. The depth of the stream as 0.305 m has been used for all the sites [6].

Groundwater samples were analyzed for NO$_3$–N using the SM 4500 NO$_3$ F automated cadmium reduction method on an Alpkem RFA 300 Rapid Flow Auto-analyzer (O.I. Analytical, Wilsonville, Oregon, USA) [11, 58]. Soil samples were also examined for denitrification enzyme activity (DEA). The DEA is the potential denitrification measured under fully anaerobic conditions and excess NO$_3$–N and available carbon. The denitrification rate constant ($K_d$) is based on the denitrification potential measurement [60]. This was the only user input in REMM for simulating denitrification in all zones and layers within the buffer. For sites (1 and 2) and sites (3 and 4), field data (depth to water table, groundwater NO$_3$–N concentrations) were collected for a five-year period (1999–2003) and a two-year period (2004–2005), respectively [11,58,61]. For sites (5, 6, 7 and 8), a simplified methodology (three-well approach) had been followed according to procedures described in [62] for the collection of field data (WTDs and NO$_3$–N concentrations in the groundwater) for year 2018–2019.
The erosion factors, topsoil condition, Manning’s $n$, soil structural characteristics, soil pH and temperature, permeability class (referring to the saturated hydraulic conductivity in the soil profile), surface condition, inter-rill roughness, soil and litter characteristics, including physical properties related to soils by soil texture (bulk density, porosity, field capacity, wilting point), pore size distribution index and bubbling pressure were obtained from the REMM user’s manual based on soil texture of riparian sites [55]. Additional buffer topographic inputs consisting of buffer width, slope, and length and stream depth were obtained from a detailed topographic survey conducted in the field at the beginning of the study.
Table 2. Site Characteristics of the modeled Buffer

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Buffer Area (m²)</th>
<th>Buffer Length (m)</th>
<th>Riparian Zone Width (3, 2, and 1) (m)</th>
<th>Zone Slope (%)</th>
<th>Manning’s n for Zone (3, 2, and 1)</th>
<th>Soil (Zone 1)</th>
<th>Soil Drainage Class (Zone 1)</th>
<th>Geomorphology</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site 1</td>
<td>10,000</td>
<td>100</td>
<td>100 (33, 35, 32)</td>
<td>&lt;3</td>
<td>(0.046, 0.046, 0.046)</td>
<td>Sandy, mesic Aeric Endoaquete</td>
<td>somewhat poorly drained</td>
<td>Outwash</td>
</tr>
<tr>
<td>Site 2</td>
<td>1,200</td>
<td>100</td>
<td>12 (4, 5, 3)</td>
<td>&lt;3</td>
<td>(0.074, 0.074, 0.012)</td>
<td>Sandy, mesic Terric Haplosaprist</td>
<td>somewhat poorly drained to very poorly drained</td>
<td>Outwash</td>
</tr>
<tr>
<td>Site 3</td>
<td>6,000</td>
<td>100</td>
<td>60 (20, 20, 20)</td>
<td>&lt;3</td>
<td>(0.074, 0.074, 0.074)</td>
<td>Coarse-loamy Fluvuquertic Humaquete</td>
<td>very poorly drained</td>
<td>Alluvium</td>
</tr>
<tr>
<td>Site 4</td>
<td>6,500</td>
<td>100</td>
<td>65 (21.67, 21.67)</td>
<td>&lt;3</td>
<td>(0.074, 0.074, 0.012)</td>
<td>Sandy Typic Humaquete</td>
<td>very poorly drained</td>
<td>Alluvium</td>
</tr>
<tr>
<td>Site 5</td>
<td>3,300</td>
<td>100</td>
<td>33 (11, 11, 11)</td>
<td>&lt;3</td>
<td>(0.046, 0.046, 0.046)</td>
<td>Coarse-loamy Fluvuquertic Humaquete</td>
<td>very poorly drained</td>
<td>Outwash</td>
</tr>
<tr>
<td>Site 6</td>
<td>3,750</td>
<td>100</td>
<td>37.5 (12.5, 12.5)</td>
<td>&lt;3</td>
<td>(0.046, 0.046, 0.046)</td>
<td>Sandy, mesic Terric Haplosaprist</td>
<td>somewhat poorly drained to very poorly drained</td>
<td>Outwash</td>
</tr>
<tr>
<td>Site 7</td>
<td>5,800</td>
<td>100</td>
<td>58 (19.33, 19.33)</td>
<td>&lt;3</td>
<td>(0.046, 0.046, 0.046)</td>
<td>Sandy, mesic Terric Haplosaprist</td>
<td>somewhat poorly drained to very poorly drained</td>
<td>Outwash</td>
</tr>
<tr>
<td>Site 8</td>
<td>15,600</td>
<td>100</td>
<td>156 (52, 52, 52)</td>
<td>&lt;3</td>
<td>(0.046, 0.046, 0.046)</td>
<td>Sandy, mesic Terric Haplosaprist</td>
<td>somewhat poorly drained to very poorly drained</td>
<td>Outwash</td>
</tr>
</tbody>
</table>

2.4 Model Assessment

The performance of the model was evaluated by comparing field collected/measured data and REMM modeled data for both WTD and groundwater NO₃-N concentration. The assessment of the model was accomplished for daily WTD and groundwater NO₃-N concentration. Assessment of model performance for WTD
and groundwater NO$_3$-N concentration included both qualitative and quantitative methods. Qualitative methods included comparing graphs of measured and modeled data.

We used the mean absolute error (MAE) to statistically compare the simulated and measured WTDs by quantitatively assessing the goodness-of-fit between simulated and measured WTDs. Equation (3) was used to determine the MAE.

\[
MAE = \frac{\sum |W_m - W_s|}{n}
\]  

where $W_m$ is measured WTD (cm), $W_s$ is simulated WTD (cm), and $n$ is the number of observations.

The simulated groundwater NO$_3$–N concentrations were compared against the field measured by using the root mean square error (RMSE) and MAE. The RMSE was computed as:

\[
RMSE = \sqrt{\frac{\sum (M_i - S_i)^2}{n}}
\]  

where $M_i$ is the measured NO$_3$-N (mg/L), $S_i$ is the simulated NO$_3$-N (mg/L), and $n$ is the number of observations.

The Willmott’s index of agreement ($d$) between simulated and measured data (WTDs and groundwater NO$_3$-N concentration) was also calculated. The value of $d$ is dimensionless and varies between 0 and 1, with an index of 1 corresponding to perfect agreement between simulated and measured data [63]. Equation (5) was used to compute Willmott’s index of agreement:
\[ d = 1 - \left[ \frac{\sum (M_i - S_i)^2}{\sum (|S_i - M|) + (|M_i - M|)^2} \right] \]  

where \( d \) is the Willmott’s index of agreement, \( M_i \) is the measured data, \( S_i \) is the simulated data, and \( M \) is the mean of the measured data.

2.5 REMM Model Calibration, Validation and Sensitivity Analysis

The Riparian Model was developed and evaluated first by testing the hydrologic component (measured WTDs) followed by the nutrient cycling component (measured groundwater \( \text{NO}_3 \)-N concentrations). The model was calibrated and validated for both hydrology and nutrient cycling in zone 1 and its three layers (similar to the procedures defined in [15,33]. The simulation period varies by site due to different field data collection periods. Simulation period ranged from 1999 to 2005 for sites 1 and 2 [11] and sites 3 and 4 [58] and from 2018 to 2019 for sites 5–8. The calibration and validation dates for each site is shown in tabulated form in Table 3.
Table 3. Calibration and Validation Period for WTDs and Groundwater NO$_3$-N concentrations in Zone 1 for all Riparian Sites.

<table>
<thead>
<tr>
<th>Site</th>
<th>WTD Simulation Period (mm/dd/yy)</th>
<th>NO$_3$-N Simulation Period (mm/dd/yy)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration Validation</td>
<td>Calibration Validation</td>
</tr>
<tr>
<td>Site 1</td>
<td>5 October 1999 to 14 December 2001</td>
<td>6 February 2002 to 5 September 2003</td>
</tr>
<tr>
<td>Site 3</td>
<td>12 March 2004 to 3 September 2004</td>
<td>1 October 2004 to 15 August 2005</td>
</tr>
<tr>
<td>Site 5</td>
<td>26 May 2018 to 17 July 2018</td>
<td>15 August 2018 to 15 November 2018</td>
</tr>
<tr>
<td>Site 6</td>
<td>29 May 2018 to 29 July 2018</td>
<td>14 September 2018 to 8 November 2018</td>
</tr>
<tr>
<td>Site 8</td>
<td>19 June 2018 to 18 July 2018</td>
<td>17 August 2018 to 1 November 2018</td>
</tr>
</tbody>
</table>

The model was manually calibrated by changing the values of input parameters one at a time. The range of input parameters were either defined by field/laboratory measurements or obtained from literature or the REMM user’s manual.
• We used field measured daily WTDs in order to calibrate and validate the hydrologic component of REMM.

• Soil inputs of the upland area were first calibrated, but buffer parameters were kept constant. Soil parameters (soil porosity, field capacity, and wilting point) were then modified within recommended ranges consistent with the soil texture to reduce difference between simulated and measured WTDs.

• We needed to adjust the soil layer thickness so that REMM generated buffer runoff and AnnAGNPS calibrated runoff, and simulated and measured WTDs were in close agreement.

• For the improvement of REMM predictions of WTDs, saturated hydraulic conductivities were also adjusted. The REMM permeability class of 2 (saturated hydraulic conductivities ranging from 42–141 \( \mu \text{m/s} \)) was used for all the sites. Hydraulic conductivities significantly affected horizontal water movement between riparian zones and the vertical gravity drainage between soil layers [16].

• The simulated WTDs were also sensitive to deep seepage from the bottom of the third layer (especially when the simulated water table was within layer 3) and were adjusted to improve model predictions of WTDs. Potential deep seep of 0.2 mm/day and 0.1 mm/day were used for all zones for site 5 and site 7, respectively. However, the other six sites had no potential deep seep in the model.
• After the hydrologic calibration, the litter and soil carbon and nitrogen pools needed to be stabilized. Otherwise REMM might calculate irrational drop in soil organic carbon and associated high N mineralization. Followed by [64], several 35-year simulations (a period selected based on available local historical weather data) were performed by varying percentage of active, slow, and passive pools. Using the initial residue and humus pools, simulations were run and the carbon and nitrogen pools at the end of the period were then used as initial pool values for new simulations. The model was again rerun for another 35-year period which helped to stabilize the carbon and nitrogen pools. After stabilizing these pools, the denitrification rate constant (K_d) was modified to improve the goodness-of-fit between simulated and measured NO_3-N concentrations in groundwater. The calibrated soil physical buffer inputs, and calibrated K_d inputs are available in supplemental Tables S1 and S2, respectively.

• The calibrated model that achieved the best goodness of fit with observed conditions for both WTDs and groundwater NO_3-N concentrations had previously been saved. All the calibrated parameters were used without further changes to validate the model for the validation period. Model assessment guidelines defined in Section 2.4 were used to judge goodness of fit for WTDs and groundwater NO_3-N concentrations in both calibration and validation phases.
Sensitivity analyses were performed to determine the effects of changing a number of key parameters associated with plant growth, nutrient cycling, surface runoff, and soil physical properties for REMM’s hydrological and nutrient simulation in Zone 1. We evaluated the sensitivity value of the most sensitive parameters (soil porosity, field capacity, wilting point) used for WTD estimation. In addition, we also evaluated the sensitivity value of the most sensitive parameters (soil porosity, field capacity, Kd) used for ground water nitrate concentration estimation. Each parameter was changed by +10% and −10% from the values used as the best estimates for each riparian site during calibration as described in [15]. Field capacity was always kept less than soil porosity during the change of these parameters. We utilized the integration of a local method into a global sensitivity method (the random one-factor-at-a-time) design proposed by [65]. This method consists of repetitions of a local method whereby the derivatives are calculated for each parameter by adding a small change to the parameter. The change in model outcome can then be measured by some lumped measure such as total mass export, sum of squares error between modeled and observed values or sum of absolute errors. The following equation has been used to perform sensitivity test for each parameter change—

\[
\text{% change (in daily average WTD or Nitrate)} = \frac{t-c}{t} \times 100\%
\]  

(6)
where \( I \) is the initial calibrated daily average WTD or Nitrate, and \( C \) is the changed daily average WTD or Nitrate after parameter change.

The sensitivity analysis and the calibration of REMM model were manually calibrated as in other studies [15,64,66,67].

3. Results

3.1 Water Table Depths Calibration and Validation

Field measured and REMM simulated daily WTD (cm below surface) dropped from field edge Zone 3 to Zone 1. Figure 3 displays the field measured and REMM simulated daily mean WTD (cm below surface) coming from field edge (Zone 3) to stream edge (Zone 1) for site 5, site 6, site 7, and site 8. Due to lack of measured field edge (Zone 3) data, we could not compare the Zone 3 REMM simulated WTDs for site 1–4. Simulated daily Water Table Depths (WTDs) were compared with those measured in the field in Zone 1 (closest to the stream) of the riparian buffer for all the sites. Simulated and measured WTDs in Zone 1 for the calibration and validation periods are shown in Figure 4. Simulated and field measured WTDs in Zone 1 were compared using mean absolute error (MAE) and Willmott’s index of agreement (\( d \)). In order to measure the variability of MAE and \( d \) among the eight riparian sites, the interquartile range (IQR) is shown as the difference between 75th and 25th percentiles, \( IQR = Q_3 - Q_1 \) (Table 4).
In general, among the riparian sites, the first Quartile ($Q_1$), the second Quartile or median ($Q_2$), and the third Quartile ($Q_3$) of the MAE for WTDs in Zone 1 was 5.5 cm, 7.5 cm, and 9 cm for the calibration period, respectively. Likewise, for the validation period, the first Quartile ($Q_1$), the second Quartile or median ($Q_2$), and the third Quartile ($Q_3$) of the MAE for WTDs in Zone 1 was 5 cm, 7 cm, and 8.5 cm, respectively. In case of Willmott’s index of agreement ($d$) between daily measured and simulated WTDs, the $Q_1$, the $Q_2$, and the $Q_3$ of the $d$ for WTDs in Zone 1 was 0.12, 0.34, and 0.62 for the calibration period, respectively. Similarly, for the validation period, the $Q_1$, the $Q_2$, and the $Q_3$ of the $d$ for WTDs in Zone 1 was 0.33, 0.64, and 0.75, respectively. The value of $d$ was within the acceptable limit between 0 and 1 for both the calibration and validation periods.

Figure 3. Measured and simulated daily mean WTD (cm below surface) coming from field edge (Zone 3) to stream edge (Zone 1) for site 5, site 6, site 7, and site 8 (mean is calculated for the simulation period specific to each site).
Figure 4. (a–p) Measured and Simulated WTDs in Zone 1 of the Buffer during the Calibration and Validation Period for eight riparian sites (Site 1, Site 2, Site 3, Site 4, Site 5, Site 6, Site 7, Site 8)

Table 4. Statistical Comparison between Measured and Simulated WTDs in Zone 1 for all sites

<table>
<thead>
<tr>
<th>MAE (cm)</th>
<th>Willmott’s index of agreement (d)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration</td>
</tr>
<tr>
<td>Minimum</td>
<td>5</td>
</tr>
<tr>
<td>Interquartile Range</td>
<td>9–5.5</td>
</tr>
<tr>
<td>Maximum</td>
<td>26</td>
</tr>
</tbody>
</table>
3.2 Groundwater NO$_3$-N concentrations Calibration and Validation

We noticed a decline in field measured and REMM simulated daily groundwater NO$_3$-N concentrations in Zone 1 coming from the field edge of Zone 3. Figure 5 presents the field measured and REMM simulated daily mean NO$_3$-N concentration (mg/L) coming from field edge (Zone 3) to stream edge (Zone 1) for site 5, site 6, site 7, and site 8. Hence, we could obtain an estimate of percentage N-removal from Zone 3 to Zone 1. In particular, for site 5, site 6, site 7, site 8, REMM showed 4.3% increase, no change, 19.7% removal, 19.8% removal of N against actual measured 46.7% removal, 28.9% removal, 19.1% removal, and 26.6% removal of N, respectively. Due to lack of measured field edge (Zone 3) nitrate data, we could not compare the Zone 3 REMM simulated nitrate concentration for site 1–4. Yet, it can be mentioned that the REMM simulated daily mean NO$_3$-N coming from Zone 3 to Zone 1 were 8.4 mg/L to 0.4 mg/L, 25.7 mg/L to 1.1 mg/L, 8.8mg/L to 3.9 mg/L, and 10.9 mg/L to 1.4 mg/L for site 1, site 2, site 3, and site 4, respectively. So the approximate simulated percentage N-removal rate becomes 95.2%, 95.7%, 55.7%, and 87.2% for site 1, site 2, site 3, and site 4, respectively. It is noted that the mean value is calculated only for the simulation dates for each site. Simulated and measured daily groundwater NO$_3$-N concentrations in Zone 1 during the calibration and validation period are shown in Figure 6 and are compared statistically using the root mean square error (RMSE), the mean absolute error (MAE), and the Willmott’s index of agreement ($d$). In order to measure the variability of RMSE, MAE, and $d$ among the eight riparian sites, the interquartile range
(IQR) is shown as the difference between 75th and 25th percentiles, $IQR = Q_3 - Q_1$ (Table 5).

![Daily Mean Nitrate Concentration](image)

**Figure 5.** Measured and simulated daily mean nitrate concentration (mg/L) coming from field edge (Zone 3) to stream edge (Zone 1) for site 5, site 6, site 7, and site 8 (mean is calculated for the simulation period specific to each site).

Overall, for the calibration period, the $Q_1$, the $Q_2$, and the $Q_3$ of the RMSE for groundwater NO$_3$-N concentrations in Zone 1 were 0.50 mg/L, 0.63 mg/L, and 0.75 mg/L among the riparian sites, respectively, whereas a slightly higher value of RMSE was found for the validation period, including the $Q_1$, the $Q_2$, and the $Q_3$ of the RMSE for NO$_3$-N in Zone 1 as 0.38 mg/L, 0.65 mg/L, and 1.07 mg/L, respectively. For the calibration period, the $Q_1$, the $Q_2$, and the $Q_3$ of the MAE for groundwater NO$_3$-N concentrations in Zone 1 were 0.45 mg/L, 0.56 mg/L, and 0.68 mg/L among the riparian sites, respectively. For the validation period, the $Q_1$, the $Q_2$, and the $Q_3$ of the MAE for
NO$_3$-N in Zone 1 were 0.36 mg/L, 0.61 mg/L, and 0.99 mg/L among the riparian sites, respectively. In case of Willmott’s index of agreement ($d$) between daily measured and simulated groundwater NO$_3$-N concentrations, the Q$_1$, the Q$_2$, and the Q$_3$ of the $d$ for NO$_3$-N in Zone 1 were 0.40, 0.44, and 0.60 for calibration period, respectively. Similarly, for the validation period, the first Quartile (Q$_1$), the second Quartile or median (Q$_2$), and the third Quartile (Q$_3$) of the $d$ for WTDs in Zone 1 was 0.17, 0.52, and 0.65, respectively. The value of $d$ was within the acceptable limit between 0 and 1 for both calibration and validation periods.
Figure 6. (a–p) Measured and Simulated Groundwater NO$_3$-N concentrations in Zone 1 of the Buffer during the Calibration and Validation Period for eight riparian sites (Site 1, Site 2, Site 3, Site 4, Site 5, Site 6, Site 7, Site 8).

Table 5. Statistical Comparison between Measured and Simulated Groundwater NO$_3$-N concentrations in Zone 1 for all the sites

<table>
<thead>
<tr>
<th>Willmott’s index of agreement ($d$)</th>
<th>RMSE (mg/L)</th>
<th>MAE (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calibration</td>
<td>Validation</td>
<td>Calibration</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.22</td>
<td>0.11</td>
</tr>
<tr>
<td>Interquartile Range</td>
<td>0.60–0.40</td>
<td>0.65–0.17</td>
</tr>
<tr>
<td>Maximum</td>
<td>0.83</td>
<td>0.76</td>
</tr>
</tbody>
</table>

3.3 Parameter Sensitivity Analysis for Water Table Depths and Groundwater NO$_3$-N concentrations Simulation

The sensitivity analysis indicated that simulated predictions of Water Table Depths and groundwater NO$_3$-N concentrations could be very sensitive to selected soil physical properties and the denitrification rate constant, respectively. Change in soil porosity caused the greatest change in WTD in all the sites except Site 3 and Site 5 (where change in field capacity caused slightly higher percentage change in WTD.
than that caused by change in soil porosity; Figure 7). Several of the sites did not exhibit >±10% in response to a 10% change in the input parameters.

In Figure 8, the percentage change in daily average groundwater $\text{NO}_3$-N concentrations from initial for calibration period is shown to be due to corresponding parameter change. Not only change in soil porosity and field capacity but also change in the denitrification rate constant, $K_d$ greatly affected modeled denitrification and simulated and measured $\text{NO}_3$-N concentrations.

4. Discussion

The MAE between measured and simulated daily WTDs is comparable to the average absolute error of 14 to 36 cm achieved in two previous REMM modeling studies using three to five years of field data from a riparian site in Georgia [12] and two sites in North Carolina [33,34]. The Willmott’s index of agreement ($d$) can be compared to the same statistic found in [33], as 0.72 to 0.92 (yearly scale). In more than 50% of the calibrations and validations, the value of $d$ was greater than 0.5, indicating good agreement with measured data.

The simulated WTDs generally followed seasonal patterns (deeper during drier months, June to November and rising to the surface during wet months, December to May) of measured WTDs for all the sites. As for sites 1–4, REMM simulated WTDs were not under-predicted or over-predicted in a constant manner throughout the entire study period. Conversely and relatively, sites 5–8 showed a particular pattern.
At site 5 and site 8, REMM over-predicted the WTDs than the measured from the late spring to summer while under-predicted from late summer to fall season. On the other hand, REMM over-predicted the summer WTDs than the measured but under-predicted the fall WTDs at site 6 and site 7. However, there were some discrepancies between simulated and measured WTDs during some time periods as a result of the spatial variability in rainfall between sites and the considered PRISM climate station, overestimation of ET by model etc. Summer field conditions were modified by irrigation of the uplands at sites 7 and 8. During field sampling, sites 5 and 6 were in unirrigated hay production (with deeper roots systems than turf). As a result, the daily water table depths were significantly deeper at sites 5 and 6 compared to other six sites.
Figure 7. Parameter Sensitivity Analysis for Water Table Depths Simulation in Zone 1 of the Buffer for (a) Site 1, (b) Site 2, (c) Site 3, (d) Site 4, (e) Site 5, (f) Site 6, (g) Site 7 and (h) Site 8. WP = Wilting Point in Z1L1; FC = Field Capacity in Z1L1; SP = Soil Porosity in Z1L1; Z1L1 = Zone 1 Soil Layer 1.
Figure 8. Parameter Sensitivity Analysis for Groundwater NO₃-N concentrations Simulation in Zone 1 of the Buffer for (a) Site 1, (b) Site 2, (c) Site 3, (d) Site 4, (e) Site 5, (f) Site 6, (g) Site 7 and (h) Site 8. FC = Field Capacity in Z1L1; SP = Soil Porosity in Z1L1; Kd = Denitrification rate constant; Z1L1 = Zone 1 Soil Layer 1.

The RMSE between measured and simulated daily groundwater NO₃-N concentrations is comparable relatively to the 1.05 to 1.50 mg/L obtained in the field testing study using 5 years of data from a riparian site in North Carolina Coastal Plain [33]. The range of \( d \) is also close (0.34 to 0.68) to what was found by [33] from their study on the North Carolina Coastal Plain. The mean absolute error (MAE) for NO₃-N concentrations in Zone 1 for calibration and validation periods was reasonably similar to the less than 1 mg/L of absolute error found by [15]. The difference between measured and simulated NO₃-N concentrations during some time periods was most
likely due to the low frequency of field data collection. This was caused by dry groundwater wells, especially during the summer months, and submergence of one of the groundwater wells in the river during heavy rainfall in November 2018. These constraints resulted in a lower Willmott’s index of agreement in case of Site 3, Site 4, and Site 8. In addition, in the case of Site 1, the NO$_3$-N concentration was very low (beyond the detection limit of 0.02 mg/L) for several times, which restricted the frequency of data collection.

The sensitivity analyses showed that changes in the volumetric water content between field capacity (FC) and saturation (i.e., soil porosity) was driving the water table and denitrification dynamics. This observation is consistent with the relationships of [68] which show that percentage saturation (the percentage of water-filled pore space, as determined by water content and total porosity) is closely related to denitrification. The lower the difference between FC and soil porosity, the more the changes in water table response to precipitation events [69] and, the larger this difference, the less responsive water tables will be to infiltration—and these two parameters also influence percentage of water-filled pore space. With a high FC and low porosity, the percentage of water-filled pore space at FC might be high enough to regularly generate denitrification.

In general, the results from the sensitivity analyses demonstrated that the percentage change in WTDs (−36% to 25%) was comparatively less than that of groundwater nitrate concentration (−60% to 60%) when responding to the input
parameter change (Figures 7 and 8). The percentage change in WTDs was least in site 1 and site 6 (−7% to 7%) while site 2, site 7, and site 8 had the greatest percentage change in WTDs (−36% to 25%). In terms of NO$_3$-N concentration, percentage change was the least in site 5 and site 7 (−4.5% to 3.5%) whereas site 2 and site 4 had the most percentage change (−60% to 60%).

5. Conclusions

In this study, we successfully calibrated and validated the REMM model by coupling upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and N data from multiple buffer sites located in a formerly glaciated watershed of Rhode Island. Both the hydrologic and nutrient estimation of REMM showed that it captured well the daily measured WTDs and groundwater NO$_3$-N concentrations in Zone 1 and in Zone 3 for the study periods. The sensitivity analyses demonstrated that changes in the volumetric water content between field capacity and saturation (soil porosity) was directing water table and denitrification dynamics.

This modeling study indicated the suitability of REMM to simulate the basic hydrologic and nutrient cycling processes happening in real-world buffers, particularly in glacial geomorphic settings (where REMM applicability has not yet been tested). The use of distributed model AnnAGNPS provided better estimates of upland inputs. The calibrated parameters and model outputs of this study establishes the base for site-specific parameters required to evaluate management and design of
riparian buffers effectively in other sites of a similar setting. The site specific and design parameters are soil porosity, field capacity, wilting point, denitrification rate constant, riparian width, vegetation type, etc. The riparian zone hydrologic and nutrient quantification through REMM also contributes to keeping a check on the rates of change occurring in the essential ecological processes (water cycle, biogeochemical or nutrient cycling, the flow of energy, etc.) in ecosystems.

Supplementary Materials: The following are available online at https://www.mdpi.com/article/10.3390/agriculture11080743/s1, Table S1: Calibrated and Validated Parameters (soil physical properties) used in the REMM simulations, Table S2: Calibrated and Validated Parameter (Denitrification rate constant, Kd) for the Riparian Buffer.

Author Contributions: M.T. was responsible for the conceptualization, methodology, data processing, software, calibration, validation, formal analysis, writing—original draft preparation, and editing; S.M.P. was responsible for the conceptualization, funding acquisition, methodology, model/software supervision, and writing—review and editing; A.J.G., K.A. and P.G.V. were responsible for the conceptualization, funding acquisition, and writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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**Institutional Review Board Statement:** Not applicable.

**Informed Consent Statement:** Authors ensure that data shared are in accordance with consent provided by participants on the use of confidential data.

**Data Availability Statement:** The data presented in this study are available on request from the corresponding author. The data are currently not publicly available due to the logistic reasons.

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**Conflicts of Interest:** The authors declare no conflict of interest.

**References**


## Supplementary Tables

### Table S1. Calibrated and Validated Parameters (soil physical properties) used in the REMM simulations.

<table>
<thead>
<tr>
<th>Parameters</th>
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</thead>
<tbody>
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<td></td>
<td>Site Name</td>
</tr>
<tr>
<td>Site 1</td>
<td>(25, 50, 80)</td>
</tr>
<tr>
<td>Site 2</td>
<td>(60, 50, 80)</td>
</tr>
<tr>
<td>Site 3</td>
<td>(55, 60, 90)</td>
</tr>
<tr>
<td>Site 4</td>
<td>(55, 50, 80)</td>
</tr>
<tr>
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<tr>
<td>Site 6</td>
<td>(250, 300, 390)</td>
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<tr>
<td>Site 7</td>
<td>(42, 92, 132)</td>
</tr>
<tr>
<td>Site 8</td>
<td>(80, 120, 150)</td>
</tr>
</tbody>
</table>

### Table S2. Calibrated and Validated Parameter (Denitrification rate constant, $K_d$) for the Riparian Buffer.

<table>
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<td>(cm)</td>
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<tr>
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<td>(0.07, 0.0103, 0.003)</td>
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<tr>
<td>Site 3</td>
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<td>Site 4</td>
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<tr>
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<td>(0.02, 0.0103, 0.002)</td>
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<tr>
<td>Site 7</td>
<td>(0.02, 0.0103, 0.002)</td>
</tr>
<tr>
<td>Site 8</td>
<td>(0.02, 0.0103, 0.002)</td>
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</table>
CHAPTER 3

Manuscript III

Riparian Zone Nitrogen Prediction for Agricultural Watersheds in the Glaciated Landscape of the Midwestern USA Using REMM

In preparation for *Nutrient Cycling in Agroecosystems*, August 2021

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Abstract

The Riparian Ecosystem Management Model (REMM) was developed, calibrated and validated for both hydrologic and water quality data for two riparian buffers located in the glaciated landscape (Sugar Creek Watershed and Eagle Creek Watershed, Indiana) of the US Midwest. The Annualized AGricultural Non-Point Source model (AnnAGNPS) was used to predict the runoff and sediment loading to the riparian buffer. The calibrated daily runoff and the sediment loading were used as input data into the REMM model. The REMM model was then developed and evaluated first by testing the hydrologic component (measured daily water table depths (WTDs)) followed by the measured daily groundwater NO$_3$-N concentrations). Overall, results showed simulated WTDs and NO$_3$-N concentrations in good agreement with measured values. The value of mean absolute error for WTDs was between 6 cm to 32 cm during the calibration and validation periods. The value of Willmott’s index of agreement (d) as ≥ 0.5 for both the sites indicated a fair agreement between measured and simulated daily WTDs during calibration. The value of d ranged between 0.05 to 0.53 to show the agreement between daily measured and simulated groundwater NO$_3$-N concentrations during both calibration and validation periods at the two riparian sites. The sensitivity analyses showed that the % water-filled pore space (100* volumetric moisture content/porosity) associated with the volumetric water content between field capacity (FC) and saturation (i.e., soil porosity) was driving water table and nitrogen dynamics.
**Keywords:** Riparian zone, REMM model, Nitrate (NO$_3$-N), Glaciated, Indiana
1. Introduction

Nitrogen is one of the most crucial nutrients for crop production (Balasubramanian et al. 2004). However, Nitrate (NO$_3$-N) losses from agricultural lands in the USA Midwest flow into the Mississippi River Basin and ultimately contribute substantially to hypoxia in the Gulf of Mexico (Kladivko et al. 2014). The USA Midwest offers some of the most productive agricultural soils in the world. The use of subsurface (tile) drainage is very common throughout these regions as a means of supporting agriculture by removing excess soil water as quickly as possible. This drainage system not only rapidly transports soil water but also agrochemicals, including Nitrate (NO$_3$-N) (Davis et al. 2000). Thus, it is a greater matter of concern that an increase of agricultural production could result in larger export of nitrate from cultivated fields into surface water bodies. This is primarily true of agricultural landscapes in the midwestern USA, a region characterized by intensive corn (Zea mays L.) production systems receiving large amounts of N fertilizer (150–200 kg N/ha/year). This land management practice, blended with a humid climate, generates an environment where considerable loads of cropland-derived N can be conveyed to riparian zones (Fisher et al. 2014).

Riparian zones are now well recognized for removal of nutrients like nitrogen (N) from upland sources. As a result of their removal capacity and their appearance as thin borders between streams and agricultural fields, they are frequently described as “filters” or “buffers”. They play a key role in watershed management schemes...
(Groffman et al. 2000; Jacinthe et al. 2003; Vidon et al. 2010). It has been well-established by the decades of research on riparian zone hydrology and biogeochemistry that riparian zones can function as Best Management Practices (BMPs) to control the agricultural runoff before entering the stream (Dosskey 2001; Kellogg et al. 2010; Welsch 1991). However, the buffering capacity of riparian zones fluctuates in response to the hydrogeomorphic setting such as soil type, topography, depth to water table, and surficial geology of the riparian zone (Gold et al. 2001; Hill 2000; Inamdar et al. 1999; Kellogg et al. 2005; Lowrance et al. 1997; Vidon and Hill 2004).

Upland land use alters both the water quantity and quality of the water entering the riparian zone. Hence, several key attributes related to location are critical in determining the potential impact of a riparian zone on water.

Despite the acknowledged value of riparian zones in mitigating nutrient pollution, only a limited number of numerical models or landscape-based approaches have been developed that can improve the use and management of riparian zones to achieve greater water quality in physiographic settings associated with landscapes that were formed by glaciation. In this situation in growing riparian zone BMPs, there is a critical need to advance our understanding of riparian functions at the site scale. Site-specific models can improve riparian zone management decisions that seek to place, restore and protect riparian zones more effectively.
The Riparian Ecosystem Management Model (REMM) (Altier et al. 2002; Lowrance et al. 2000) integrates many site attributes to simulate hydrology, carbon and nutrient dynamics, and plant growth in riparian zones. REMM has been used to simulate managed riparian ecosystems in a number of settings in USA including Chesapeake Bay Watershed (Graff et al. 2005); Delaware (Allison et al. 2006); Mississippi (Langendoen et al. 2009); North Carolina (Tilak et al. 2014; Tilak et al. 2017); Georgia (Bhat et al. 2007; Inamdar et al. 1999); California (Graff et al. 2008); and Puerto Rico (Williams et al. 2016). REMM was originally field tested using a five-year hydrologic and nutrient dataset collected from an experimental riparian buffer site in Tifton, Georgia (Inamdar et al. 1999).

A major challenge in using the REMM model is the requirement for information on water and nutrient flux from source areas that contribute to the riparian zone. The absence of site specific data frequently results in users relying on default parameters. REMM simulations are also not bounded by maximum or minimum values, which can lead to unrealistic simulation results if the model is poorly parameterized or not validated adequately with empirical data. To partially address this challenge, a SWAT-REMM integration approach has been used in a glaciated landscape in New Brunswick, Canada by Zhang et al. 2017. They examined the effect of different sub-watershed areas for estimating edge-of-field losses and the performance of the REMM model. However, there is a paucity of finer scale analyses that match the scale of field losses to the scale of site-specific conditions within
riparian zones. The ANNAGNPS model holds promise. This study is the first to integrate REMM with AnnAGNPS model in the glaciated, highly productive agricultural settings of the Midwest region. Before this study, our companion study Tamanna et al. 2021 applied similar approach but for USA northeastern State, Rhode Island. Riparian zones are widely used in these regions to mitigate nutrient losses to streams and field scale analyses can provide lessons for management.

Therefore, we point to a critical need to determine the usability of REMM in glaciated settings of the USA Midwest that is informed by i) field data to offer an independent way of generalizing riparian function in these regions; ii) sites-specific estimates of water flux and nutrient loading from uplands to the riparian zones.

In this study, we explored the potential of REMM model for riparian zone nitrogen simulation in two watersheds from the glacial setting of Indiana (IN), USA Midwest. Specifically, this modeling study compared the simulation of riparian zone nitrogen through the REMM model between two watersheds. This process involved (i) REMM model set-up, including site-specific inputs from uplands to two monitored riparian sites in IN, (ii) improvements to the model’s capacity for water table depths and groundwater nitrate concentrations simulation through calibration of the developed model by means of comparing model outputs with field data collected from two buffer sites in IN, (iii) validation of model’s output with the improved calibrated parameters, (iv) conduct a parameter sensitivity analysis.
2. Materials and Methods

2.1 Site Description

Our study focused on two riparian sites from the state of IN, USA. Both the sites were monitored for hydrology and water quality. The first riparian site named as Leary Weber Ditch (LWD) is located in Sugar Creek watershed (39°43′21″ N, 85°53′23″ W), a part of the White River watershed in central Indiana (Fig. 1). The area of the watershed is about 69 km². The elevation of the watershed ranges from 241 m to 280 m, and the topography is nearly flat. The watershed consists largely of tile-drained agricultural lands (88% of the total watershed area, representative of agro-ecosystems of the glacial till plains from USA Midwest. This watershed is dominated by poorly drained soils where artificial drainage is usually used to lower the water table. For the past 20 years, agricultural practices have been dominated by a corn/soybean rotation with either conventional or conservation tillage systems. The temperature in the watershed is moderate, ranging from a 30-year (1982–2011) mean of 22.7 °C in summer to a mean of −1.4 °C in winter (Parameter-elevation Regressions on Independent Slopes Model (PRISM) Climate Group, accessed on 4 May 2019). The 30 years (1982–2011) average annual precipitation is approximately 1105.0 mm, about 51% of which occurs during the summer and the fall months. The 14 years (2000–2013) average annual snowfall is 91.6 mm (Tamanna et al. 2020).
The LWD site is located 30 km east of Indianapolis in the Tipton Till Plains. Vegetation consists of a mixture of various grass species and shrubs. This site represents the narrow riparian zones (20–30 m wide) predominant along tile-drained corn (*Zea mays*) (2009, 2011) and soybean (*Glycine max*) (2010) fields in glacial till plains of the USA Midwest. Detailed site information can be found in Fisher et al. 2014; Liu et al. 2014; Vidon and Cuadra 2010.
The second riparian site named as Scott Starling Nature Sanctuary (SSNS) is located in Eagle Creek Watershed in Central Indiana, USA, about 16 km northwest of downtown Indianapolis (Fig. 2). The drainage area of this watershed is about 428 km$^2$. This drainage area drains into the Eagle Creek Reservoir, a significant source of drinking water for Indianapolis and the surrounding region. The topography is comparatively flat to undulating, with some dissection near Eagle Creek reservoir. Soil type contains productive soils developed in glacial till and loess. Agriculture is the dominant land use in the watershed (approximately 60% of the watershed area), with corn and soybeans being the principal crops. But, high and low-density land use is also now increasing as a result of the increasing Indianapolis population and associated increases in urban/suburban infrastructure developments (Babbar-Sebens et al. 2013). The elevation of the watershed ranges from 226 m to 292 m. It has a predominantly temperate continental and humid climate. The temperature in the watershed ranges from a 30-year (1982–2011) mean of 23.0°C in summer to a mean of −1.3 °C in winter (PRISM Climate Group, accessed on 21 March 2019). The 30 years (1982–2011) average annual precipitation is approximately 1179 mm, about 56.6% of which occurs during the spring and the fall months. The average (2010–2011) annual snowfall was 72.5 mm.

The SSNS riparian zone comprises a restored wetland area and a near-stream alluvium area located in a glacial till valley along Fishback Creek near Indianapolis, Indiana, USA. Our modeling study focuses on the alluvium area only. Till thickness
is approximately 50-55 m in the upland and about 20-25 m thick underlying the riparian zone. Topography in the riparian site is a steep concave topography with a flat area directly adjacent to the stream. Vegetation in the riparian zone is herbaceous except near the stream where hardwood species dominate. Land use in the upland is dominated by low-density housing surrounded by forest. Septic systems in the upland and the use of lawn fertilizer could potentially affect water quality in the riparian zone. Before the restoration of the wetland area in 1999, both the wetland and alluvium areas were used for row crop production until 1991. In the alluvium area, the vegetation involves mature trees, including red maple (*Acer rubrum*), silver maple (*Acer saccharinum*), swamp white oak (*Quercus bicolor*), and American sycamore (*Platanus occidentalis*). Detailed site information can be found in Vidon and Smith 2007 and Vidon et al. 2014.

Details on the soils, slope and buffer dimensions are provided in Tables 1 and 2.

**Table 1 Upland Characteristics for Riparian Sites**

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Geology</th>
<th>Soil</th>
<th>Land Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>LWD</td>
<td>Glacial Till</td>
<td>Silty clay loam</td>
<td>Fertilized Cropland</td>
</tr>
<tr>
<td>SSNS</td>
<td>Alluvium</td>
<td>Loam</td>
<td>Forested Residential</td>
</tr>
</tbody>
</table>


Fig. 2 Location of Riparian Site (SSNS) in Eagle Creek Watershed, Indiana

Table 2 Site Characteristics of the modeled Buffer

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Buffer Area (m²)</th>
<th>Buffer Length (m)</th>
<th>Riparian Zone Width (3, 2, and 1) (m)</th>
<th>Zone Slope (%)</th>
<th>Manning’s n for Zone (3, 2, and 1)</th>
<th>Soil (Zone 1)</th>
<th>Soil Drainage Class (Zone 1)</th>
<th>Geomorphology</th>
</tr>
</thead>
<tbody>
<tr>
<td>LWD</td>
<td>2,500</td>
<td>100</td>
<td>25 (8.33, 8.33, 8.33)</td>
<td>&lt;1</td>
<td>(0.018, 0.018, 0.018)</td>
<td>Silty clay loam</td>
<td>poorly drained</td>
<td>Glacial Till</td>
</tr>
<tr>
<td>SSNS</td>
<td>20,000</td>
<td>100</td>
<td>200 (66.67, 66.67, 66.67)</td>
<td>&lt;1</td>
<td>(0.018, 0.018, 0.018)</td>
<td>Sandy loam</td>
<td>Well-drained</td>
<td>Alluvium</td>
</tr>
</tbody>
</table>

2.2 Description of the REMM Model
REMM is a process-based, two-dimensional, daily time-step model that assesses the fate of nutrients and sediment coming from the edge of an agricultural field, through a three-zone riparian area, up to the edge of a stream. REMM takes upland inputs and computes loading of water, sediments, nutrients, and carbon into the buffer where water and total N are transported from upland to Zone 3 (field edge), Zone 3 to Zone 2 (mid-buffer), Zone 2 to Zone 1 (near the stream) and finally from Zone 1 to stream via surface runoff, seep flow, and subsurface flow (Altier et al. 2002; Lowrance et al. 2000).

REMM file version 0.1.1.46 (United States Department of Agriculture (USDA)-Agricultural Research Service (ARS)) was used for all simulations. A broad description of the REMM model is available in several publications, including (Altier et al. 2002; Inamdar et al. 1999; Lowrance et al. 2000). Briefly, REMM is a computer simulation model of riparian forest buffer systems. The structure of REMM is consistent with buffer system specifications recommended by the U.S. Forest Service and the USDA-Natural Resources Conservation Service as national standards (Welsch et al. 1991; NRCS 1995).

Within the model, the riparian system is considered to consist of three zones (parallel to a stream) between the field and the water body (Fig. 3). Each zone includes litter and three soil layers (through which the vertical and horizontal movement of water takes place) that terminate at the bottom of the plant root system and a plant community that can include six plant types in two canopy levels. The riparian system
characterized in REMM was originally designed to represent increasing levels of management away from the stream (Lowrance et al. 2000). REMM is written in the C++ programming language.

![Cross-section of riparian buffer system at the sites as simulated in REMM](image)

**Fig. 3** Cross-section of riparian buffer system at the sites as simulated in REMM

The details of mass balance of water movement within each soil layer and the equation used in REMM model to simulate denitrification is available in Tamanna et al. 2021.

### 2.3 REMM Model Input Data

Four basic input files for REMM execution include: (1) contributions of daily outputs from the field draining into the riparian system including surface runoff and associated eroded sediment, organic material and plant nutrients (*.FIN), (2) weather data (*.WEA), (3) vegetation (*.VEG) characteristics, and (4) soil physical and chemical parameters (*.BUF). A field-scale hydrological model, Annualized AGricultural Non-
Point Source (AnnAGNPS), was used to predict the upland input (runoff and sediment loading) to the riparian buffer (Tamanna et al. 2020; Yongping et al. 2007). The AnnAGNPS model defines cells of various sizes; contaminants are routed from these cells into the associated reaches, and the model either deposits pollutants within the stream channel system or transports them out of the watershed. The simulated runoff and sediment loading from AnnAGNPS was calibrated by comparing with observed data (USGS gauge). The calibrated daily runoff and the sediment loading were used as input data in .FIN (Field Data File). The size of the cells depends on the values of the Critical Source Area (CSA) and Minimum Source Channel Length (MSCL) (Tamanna et al. 2020). The values of (CSA, MSCL) were (5 ha, 30 m) and (100 ha, 150 m) for Sugar Creek watershed and Eagle Creek watershed, respectively. For this study, AnnAGNPS was used with a cell size of 0.05 km$^2$ (interquartile range, IQR $= Q_3 - Q_1$, of cell size among 1788 no. of cells in the watershed) for Sugar Creek watershed (69 km$^2$) and a cell size of 0.88 km$^2$ (interquartile range, IQR $= Q_3 - Q_1$, of cell size among 528 no. of cells in the watershed) for Eagle Creek watershed (428 km$^2$) to simulate input cells representing upland inputs to LWD and SSNS riparian buffers respectively. Details about the application of AnnAGNPS model on the Sugar Creek watershed can be found in our companion study (Tamanna et al. 2020). The development, calibration and validation of runoff via AnnAGNPS model for Eagle creek watershed is presented in this article.
For stream flow data used in the calibration and validation, we used the daily observations from USGS gauging station USGS 03353460 Eagle Creek at Clermont, Indiana (39°48'52" N, 86°18'19" W) for Eagle Creek watershed (Figure 2). The traditional manual baseflow filtering approach was applied to the streamflow record to obtain runoff by removing baseflow from streamflow before comparison with AnnAGNPS output, as baseflow is not considered in the model (Yasarer et al. 2018).

The SCS Curve Number (CN) was used to calibrate runoff in this study. For Eagle Creek watershed, the AnnAGNPS model was calibrated for runoff from 1 January 2008 to 31 December 2009 (average annual 1253.9 mm precipitation) and validated from 1 January 2010 to 31 December 2011 (average annual 1178.2 mm precipitation). Before performing the watershed simulation, the model was initialized for two years.

Based on Cronshey et al. 1985, the selection of initial SCS CNs for the different land use types was completed. Eagle Creek watershed consists of various land uses like cropland (only corn), cropland (corn–soybean rotation), fallow land, forested and urban area. Initially, the CN for a straight row crop with good hydrological conditions was used for corn and the CN for a straight row crop with poor hydrological conditions was used for corn–soybean rotation during the growing season, while the CN for a fallow field with crop residue and good hydrological conditions was used after harvest during the non-growing season. The CN for woods with good hydrological conditions was used for forested areas. The CNs for residential areas with 12% and 20% impervious cover were used for urban areas (Table 3).
Table 3 Curve numbers (CN) used for model calibration, Eagle Creek watershed

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<td>Initial Values</td>
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<td>Row Crop (SR—Good)</td>
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<td>Row Crop (SR—Poor)</td>
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<tr>
<td>Woods (Good)</td>
<td></td>
<td>30</td>
<td>55</td>
<td>70</td>
<td>77</td>
<td>30</td>
<td>42</td>
<td>58</td>
<td>68</td>
</tr>
<tr>
<td>Residential (12% imp)</td>
<td></td>
<td>46</td>
<td>65</td>
<td>77</td>
<td>82</td>
<td>30</td>
<td>50</td>
<td>60</td>
<td>70</td>
</tr>
<tr>
<td>Residential (20% imp)</td>
<td></td>
<td>51</td>
<td>68</td>
<td>79</td>
<td>84</td>
<td>41</td>
<td>61</td>
<td>71</td>
<td>80</td>
</tr>
</tbody>
</table>

SR—straight row, CR—crop residue cover

The performance of model was evaluated by comparing observed and AnnAGNPS modeled data at the watershed outlet. The assessment of the model was accomplished for runoff on both daily and monthly time scales. Assessment of model performance for runoff included both qualitative and quantitative methods. Qualitative methods included comparing graphs of observed and modeled data. We followed the recommendation of Moriasi et al. 2007 and used three quantitative statistics: Nash–Sutcliffe efficiency (NSE), percent bias (PBIAS), and ratio of the root mean square error to the standard deviation of measured data (RSR), along with the graphical techniques, to model performance evaluation. Generally, model simulation can be judged as satisfactory when $NSE > 0.50$ and $RSR < 0.70$, and also when $PBIAS$
± 25% for streamflow. We also used the coefficient of determination ($R^2$) for quantitative evaluations; $R^2$ represents the variation in measured data explained by the model. Values can range from 0 to 1, with 1 indicating that all variations in the measured data are explained by the model. Values greater than 0.5 are normally considered acceptable. The $NSE$ value ranges from $-\infty$ to 1 with 1 representing a perfect fit. Values between 0 and 1 are considered an acceptable performance level for the model.

**Table 4** Runoff calibration and validation results for Eagle Creek watershed

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Calibration Period</th>
<th>Validation Period</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1 January 2008 to 31 December 2009)</td>
<td>(1 January 2010 to 31 December 2011)</td>
</tr>
<tr>
<td></td>
<td>Daily</td>
<td>Monthly</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.52</td>
<td>0.83</td>
</tr>
<tr>
<td>$NSE$</td>
<td>0.50</td>
<td>0.81</td>
</tr>
<tr>
<td>$PBIAS$</td>
<td>12.16%</td>
<td>12.19%</td>
</tr>
<tr>
<td>$RSR$</td>
<td>0.71</td>
<td>0.43</td>
</tr>
</tbody>
</table>

According to the classification tabulated in Parajuli et al. 2009 for model correlations and efficiencies modified from Moriasi et al. 2007, our calibrated model for the Eagle Creek watershed predicted the daily runoff volume of the watershed with a good correlation and good agreement ($R^2 = 0.52$, $NSE = 0.50$ for daily and $R^2 = 0.83$, $NSE = 0.81$ for monthly calibration) between daily observed and daily modeled runoff volume (Table 4, Fig. 4a). The calibrated model, when applied to the same watershed for the validation phase, predicted a daily runoff volume with a fair
correlation and fair agreement for both daily and monthly scales ($R^2 = 0.48 \sim 0.5$, NSE = 0.46 \sim 0.5 for daily and $R^2 = 0.71$, NSE = 0.67 for monthly) (Table 4, Fig. 4b). Total runoff estimation by the model during the calibration phase differed from the observed runoff by only about 12.16%, whereas it differed by about 23.39% during validation. The calculated PBIAS value for calibration was between ±11\% \sim ± 15, which indicated a very good calibration performance rate. The model was biased to underestimate runoff volume during both calibration and validation phases.

![Graphical comparison between daily modeled and observed runoff](image)

**Fig. 4** Graphical comparison between daily modeled and observed runoff (a) after calibration and (b) validation phase for Eagle Creek Watershed
The weather input, vegetation data and site characteristics were included into the REMM model following the procedures mentioned in Tamanna et al. 2021. Table 2 and section 2.1 shows the site characteristics of the modeled buffer for two riparian sites. All field data was collected by a team led by the authors of this manuscript with decades of published research on riparian zones in glaciated settings of the USA Midwest regions for the development and calibration of REMM model. For LWD site, water table levels were recorded in water table wells in the riparian zone once a month between October 2009 and August 2011. A network of 5–6 piezometer nests were installed across the riparian zone from the upland to the stream. Details of field data collection is available in Liu et al. 2014. The percent sand (15%), silt (50%), and clay (35%) were used as inputs into REMM according to soil type (silty clay loam) information from REMM’s user manual. For SSNS site, water table levels were recorded in water table wells in the riparian zone once a month between October 2009 and August 2011. A network of 14 piezometer nests were installed across the riparian zone from the upland to the stream. Details of field data collection is available in Vidon et al. 2014. Groundwater samples were collected and analyzed for NO\textsubscript{3}–N using a photometric analyzer (Aquakem 20, EST Analytical, Fairfield, OH) for both the sites. The percent sand (60%), silt (25%), and clay (15%) were used as inputs into REMM according to soil type (sandy loam) information from REMM’s user manual. The depth of the stream as 2.5 m has been used for both the sites in the model. The other
essential input data were included into the REMM model following the procedures mentioned in Tamanna et al. 2021.

2.4 Model Assessment

The performance of the model was evaluated by comparing field collected/measured data and REMM modeled data for both WTD and groundwater NO₃–N concentration. The evaluation statistics (the mean absolute error, MAE; the root mean square error, RMSE; the Willmott’s index of agreement, d) used for this study are the similar used in Tamanna et al. 2021.

2.5 REMM Model Calibration, Validation and Sensitivity Analysis

The Riparian Model was developed and evaluated first by testing the hydrologic component (measured WTDs) followed by the nutrient cycling component (measured groundwater NO₃–N concentrations). The model was calibrated and validated for both hydrology and nutrient cycling in zone 1 and its three layers (similar to the procedures defined in Lowrance et al. 2000 and Tilak et al. 2014). The simulation period is from October 2009 to August 2011 for both LWD and SSNS sites. The model was manually calibrated by changing the values of input parameters one at a time. The range of input parameters were either defined by field/laboratory measurements or obtained from literature or REMM user’s manual. Model calibration and sensitivity analysis procedure details are available in Tamanna et al. 2021. The REMM permeability class of 4 (saturated hydraulic conductivities ranging from 4 – 14 μm/s)
was used for both the sites. The simulated WTDs were also sensitive to deep seepage from the bottom of the third layer (especially when the simulated water table was within layer 3) and were adjusted to improve model predictions of WTDs. Vidon and Smith 2007 observed seeps along the valley at the interface between the two till units contributing water to the riparian zone at site SSNS. So, potential deep seep as 0.2 mm/day was used for all zones for site, SSNS. However, the other site, LWD had no potential deep seep in the model. The calibrated soil physical buffer inputs, and calibrated $K_d$ inputs are available in supplemental Tables S1.

3. Results

3.1 Water Table Depths Calibration and Validation

Simulated daily Water Table Depths (WTDs) were compared with those measured in the field in Zone 1 (closest to the stream) of the riparian buffer for both the sites. Simulated and measured WTDs in Zone 1 for the calibration and validation periods are shown in Fig. 5. Simulated and field measured WTDs in Zone 1 were compared using mean absolute error (MAE) and Willmott’s index of agreement ($d$) (Table 5).

In case of site LWD, the MAE for WTDs in Zone 1 was 6 cm, and 23 cm for the calibration and validation period, respectively. Whereas, the MAE for WTDs in Zone 1 was 10 cm, and 32 cm for the calibration and validation period, respectively for site SSNS. In case of Willmott’s index of agreement ($d$) between daily measured and
simulated WTDs, the value of $d$ was 0.50 and 0.11 for the calibration and validation period, respectively for LWD site. The value of $d$ was 0.65 and 0.31 for the calibration and validation period, respectively for SSNS site. The value of $d$ was within the acceptable limit between 0 and 1 for both the calibration and validation periods for both sites.

![Graphs of WTDs](image)

**Fig. 5** Measured and Simulated WTDs in Zone 1 of the Buffer during the Calibration and Validation Period for two riparian sites (LWD and SSNS)

**Table 5** Statistical Comparison between Measured and Simulated WTDs in Zone 1 for two sites

<table>
<thead>
<tr>
<th>Site Name</th>
<th>MAE (cm)</th>
<th>Willmott’s index of agreement ($d$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration</td>
<td>Validation</td>
</tr>
<tr>
<td>LWD</td>
<td>6</td>
<td>23</td>
</tr>
<tr>
<td>SSNS</td>
<td>10</td>
<td>32</td>
</tr>
</tbody>
</table>
3.2 Groundwater NO$_3$–N concentrations Calibration and Validation

We noticed a decline in REMM simulated daily groundwater NO$_3$–N concentrations in Zone 1 coming from the field edge zone 3. The simulated daily mean NO$_3$–N concentrations dropped from 1.21 to 0.20 mg/L and 0.91 to 0.66 mg/L during 2009 - 2011 in Site LWD and Site SSNS respectively. Simulated and measured daily groundwater NO$_3$–N concentrations in Zone 1 during the calibration and validation period are shown in Figure 4 and are compared statistically using the root mean square error (RMSE), the mean absolute error (MAE), and the Willmott’s index of agreement ($d$) (Table 6).

In case of site LWD, the MAE for groundwater NO$_3$–N in Zone 1 was 0.38 mg/L and 0.44 mg/L for the calibration and validation period. Whereas, a relatively higher MAE for groundwater NO$_3$–N in Zone 1 (0.76 mg/L and 0.63 mg/L) was found for the calibration and validation period, respectively for site SSNS. In case of Willmott’s index of agreement ($d$) between daily measured and simulated groundwater NO$_3$–N, the value of $d$ was 0.53 and 0.50 for the calibration and validation period, respectively for LWD site. In contrast, quite low value of $d$ (0.05 and 0.19) was found for the calibration and validation period, respectively for SSNS site. The value of $d$ was within the acceptable limit between 0 and 1 for both calibration and validation periods. In case of site LWD, the RMSE for groundwater NO$_3$–N in Zone 1 was 0.45 mg/L and 0.50 mg/L for the calibration and validation period. Whereas, the RMSE for
groundwater NO$_3$–N in Zone 1 was 0.98 mg/L, and 0.81 mg/L for the calibration and validation period, respectively for site SSNS.

![Calibration and Validation Period](image)

**Fig. 6** Measured and Simulated Groundwater NO$_3$–N concentrations in Zone 1 of the Buffer during the Calibration and Validation Period for two riparian sites (LWD and SSNS)

**Table 6** Statistical Comparison between Measured and Simulated Groundwater NO$_3$–N concentrations in Zone 1 for two sites

<table>
<thead>
<tr>
<th>Site Name</th>
<th>MAE (mg/L)</th>
<th>Willmott’s index of agreement ($d$)</th>
<th>RMSE (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration</td>
<td>Validation</td>
<td>Calibration</td>
</tr>
<tr>
<td>LWD</td>
<td>0.38</td>
<td>0.44</td>
<td>0.53</td>
</tr>
<tr>
<td>SSNS</td>
<td>0.76</td>
<td>0.63</td>
<td>0.05</td>
</tr>
</tbody>
</table>
3.3 Parameter Sensitivity Analysis for Water Table Depths and Groundwater NO$_3$–N concentrations Simulation

The sensitivity analysis indicated that simulated predictions of Water Table Depths and groundwater NO$_3$–N concentrations could be very sensitive to selected soil physical properties. Change in soil porosity caused the greatest change in WTD in site SSNS whereas change in field capacity caused the greatest change in WTD in site LWD (Fig. 7). Both the sites did not display >±15% in response to a 10% change in the input parameters.

In Fig. 8, the percent change in daily average groundwater NO$_3$–N concentrations from actual for calibration period is shown due to corresponding parameter change. Change in soil porosity, field capacity and denitrification rate constant (K$_d$) caused considerable changes in NO$_3$–N concentrations.

4. Discussion

The MAE between measured and simulated daily WTDs is quite comparable to the average absolute error of 14 to 36 cm obtained in two previous model testing studies using three to five years of data from a buffer sites in Georgia (Inamdar et al. 1999) and two sites in North Carolina (Dukes and Evans 2003; Tilak et al. 2014). The value of Willmott’s index of agreement ($d$) as ≥ 0.5 for both the sites indicated a fair agreement between measured and simulated daily WTDs during calibration.
The simulated WTDs generally followed the seasonal trends (deeper during drier months, July to November and rising to the surface during wet months, December to June) of measured WTDs. However, there were some discrepancies between measured and simulated WTDs during some time periods due to the spatial variability in rainfall between sites and the considered PRISM climate station, overestimation of ET by model etc.

**Fig. 7** Parameter Sensitivity Analysis for Water Table Depths Simulation in Zone 1 of the Buffer for (a) LWD, and (b) SSNS. **WP** = Wilting Point in Z1L1; **FC** = Field Capacity in Z1L1; **SP** = Soil Porosity in Z1L1; **Z1L1** = Zone 1 Soil Layer 1
Fig. 8 Parameter Sensitivity Analysis for Groundwater NO$_3$–N concentrations Simulation in Zone 1 of the Buffer for (a) LWD, and (b) SSNS. FC = Field Capacity in Z1L1; SP = Soil Porosity in Z1L1; Kd = Denitrification rate constant; Z1L1 = Zone 1 Soil Layer 1

In case of site LWD, simulated groundwater NO$_3$–N concentrations were very low for most of the times except some high values during spring months (March – May). On the other hand, for site SSNS, the model could simulate several higher groundwater NO$_3$–N concentrations during both calibration and validation periods when compared to the field measured values. The RMSE between measured and
simulated daily groundwater NO\textsubscript{3}-N concentrations (0.45 \textasciitilde 0.98 mg/L) is comparable somewhat to the value (1.05 to 1.50 mg/L) found by Tilak et al. 2014 using 5 years of data from a riparian site in North Carolina Coastal Plain. The range of \(d\) (0.05 to 0.53) is quite less or close to what (0.34 to 0.68) was found by Tilak et al. 2014 from their study. The mean absolute error (MAE) for NO\textsubscript{3}-N concentrations in Zone 1 (0.38 to 0.76 mg/L) for calibration and validation periods was fairly similar to the less than 1 mg/L of absolute error found by Lowrance et al. 2000. The difference between measured and simulated NO\textsubscript{3}-N concentrations during some time periods was because of the low frequency of field data collection. Especially during the summer months, the dry groundwater wells affected the data collection.

The sensitivity analyses showed that the volumetric water content between field capacity (FC) and saturation (i.e., soil porosity) was driving water table and nitrogen dynamics. Moreover, the smaller the difference between FC and soil porosity, the further the changes in water table response to precipitation events (Heliotis et al. 1987), and the higher this difference, the fewer responsive water tables will be to infiltration - and these two parameters also impact % water-filled pore space. In addition to that, the changes (10\% increase) in the soil porosity caused the greatest change in NO\textsubscript{3}-N concentrations in both the sites. This demonstrates that the model is very sensitive to the soil porosity in terms of NO\textsubscript{3–N} concentrations estimation.

5. Conclusions
In this study, we successfully calibrated and validated the REMM model by coupling upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and nitrate data from two buffer sites located in two glaciated watersheds of Indiana. This study expands the application of REMM for riparian zone nitrate prediction after our companion study (Tamanna et al. 2021). In addition to that, the findings from this modeling study revealed the suitability of REMM to simulate the basic hydrologic and nutrient cycling in glacial geomorphic settings of midwestern USA (where REMM applicability has not been tested yet).

**Author Contributions:** M.T. was responsible for the conceptualization, methodology, data processing, software, calibration, validation, formal analysis, writing—original draft preparation, and editing; S.M.P. was responsible for the conceptualization, funding acquisition, methodology, model/software supervision, and writing—review and editing; A.J.G., K.A. and P.G.V were responsible for the conceptualization, funding acquisition, and writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflicts of interest.

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Supplementary Table

**Table S1.** Calibrated and Validated Parameters (soil physical properties and Kₐ) used in the REMM simulations.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>Soil Layer Thickness (cm)</th>
<th>Soil Porosity</th>
<th>Field Capacity</th>
<th>Wilting Point</th>
<th>Permeability (cm/hr)</th>
<th>Kₐ</th>
</tr>
</thead>
<tbody>
<tr>
<td>LWD</td>
<td>(50, 90, 180)</td>
<td>(0.47, 0.47, 0.47)</td>
<td>(0.324, 0.324, 0.324)</td>
<td>(0.20, 0.20)</td>
<td>(1.016, 1.016, 1.016)</td>
<td>(0.02, 0.0103, 0.002)</td>
</tr>
<tr>
<td>SSNS</td>
<td>(40, 80, 170)</td>
<td>(0.36, 0.36, 0.36)</td>
<td>(0.22, 0.22, 0.22)</td>
<td>(0.08, 0.08)</td>
<td>(1.016, 1.016, 1.016)</td>
<td>(0.02, 0.0103, 0.002)</td>
</tr>
</tbody>
</table>
CHAPTER 4

Manuscript IV

Riparian Zone Nitrogen Prediction for a Mixed-land Use Watershed in the Glaciated Landscape of the USA Northeast Using REMM

In preparation for Journal of Contaminant Hydrology, August 2021

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Abstract

The Riparian Ecosystem Management Model (REMM) was developed to quantify water quality benefits of riparian buffers in a glaciated watershed of New York. The REMM model was successfully calibrated and validated by coupling upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and nutrient data from three buffer sites located in a glaciated watershed of New York. Both the hydrologic and nutrient testing of REMM showed that it captured well the daily measured data (WTDs and groundwater NO\textsubscript{3}-N concentrations in Zone 1) for both calibration and validation periods. The value of mean absolute error for WTDs was between 3 cm to 12 cm during the calibration and validation periods. Besides, the value of Willmott’s index of agreement \( d \) between daily measured and simulated groundwater NO\textsubscript{3} concentrations ranged between 0.14 - 0.86 during both calibration and validation periods at the three riparian sites. The sensitivity analyses demonstrated that changes in the volumetric water content between field capacity and saturation (soil porosity) was directing water table and denitrification dynamics.

Keywords:
Riparian Zone; REMM model; nitrate; water table depth; New York
1. Introduction

Nutrient (primarily Nitrogen (N)) loss from agricultural watersheds through runoff and drainage water continues to be a water quality concern of global importance. Since N and P are crucial inputs for the sustainability of agriculture, the use of both inputs has increased dramatically in recent decades and the excessive nutrient losses have increased as well (Ding et al., 2020; Pathak et al., 2010; Schröder et al., 2004; Spiess, 2011; Vidon et al., 2017; Wang et al., 2014). Agriculture (cropland, pasture, managed forest) is an important component of many watersheds of the USA Northeast where N flux to major estuaries is of substantial concern. In this circumstance, the finding from almost 30 years of research on riparian zone hydrology and biogeochemistry demonstrates that riparian zones can serve as best management practices (BMPs) to minimize the adverse agricultural impact on water quality (Dosskey, 2001; Ice, 2004; Kellogg et al., 2010; Welsch, 1991).

Riparian zones have been used as one of the most important practices for water quality improvement in agricultural settings due to their ability to perform multiple functions including reducing NO₃⁻ concentrations in subsurface flow, trapping sediments, and pesticides in overland flow, and control erosion (Dosskey, 2001; Vidon et al., 2019; Welsch, 1991). Because of their vital role in watershed management schemes for water quality maintenance and stream ecosystem habitat protection, they are regularly considered as “filters” or “buffers” (Groffman et al., 2000; Jacinthe et al., 2003; Vidon et al., 2010; ). The hydrogeomorphic setting such as topography, soil type,
and surficial geology of the riparian zone significantly affect the buffering capacity of riparian zones (mostly for N) (Gold et al., 2001; Hill, 2000; Inamdar et al., 1999; Kellogg et al., 2005; Lowrance et al., 1997; Vidon and Hill, 2004). Upland land use and land cover affects both the water quantity and quality of the water entering the riparian zone. As a result, several crucial characteristics related to location are essential in defining the potential effect of a riparian zone on water quality. Riparian Ecosystem Management Model (REMM; Altier et al., 2002; Lowrance et al., 2000) combines all these characters together. In such context in enlarging riparian zone BMPs, there is a vital need to advance our understanding of riparian functions at the site scale. Site-specific models can enhance riparian zone management decision capacity to place, restore and protect riparian zones more efficiently.

REMM was originally field tested using a five-year hydrologic and nutrient dataset collected from an experimental riparian buffer site in Tifton, Georgia (Inamdar et al. 1999). So far REMM has been used to simulate managed riparian ecosystems in a number of settings in USA including Chesapeake Bay Watershed (Graff et al. 2005); Delaware (Allison et al. 2006); Mississippi (Langendoen et al. 2009); North Carolina (Tilak et al. 2014; Tilak et al. 2017); Georgia (Bhat et al. 2007; Inamdar et al. 1999); California (Graff et al. 2008); and Puerto Rico (Williams et al. 2016). Other than that, a SWAT-REMM integration approach has been used in a glaciated landscape in New Brunswick, Canada by Zhang et al. 2017. Before this study, our companion study Tamanna et al. 2021 applied AnnAGNPS-REMM integration approach for USA
northeastern State, Rhode Island. This study is the first to integrate REMM with AnnAGNPS to model the glaciated agricultural watersheds from New York state, USA Northeast. Riparian zones are commonly used in these regions to alleviate nutrient losses to streams and field scale analyses can provide examples for water resources management.

This study endeavored to evaluate the REMM model’s capacity for riparian zone nitrogen estimation in a glaciated watershed of New York (NY), USA Northeast. Specifically, this modeling exercise demonstrates the details on testing of the REMM model when using site-specific field data to evaluate the model’s ability to simulate the basic hydrologic and environmental parameters such as water table depths (WTDs) and groundwater nitrate (NO$_3$-N) concentrations. This process involved (i) REMM model set-up, including site-specific inputs from uplands to three monitored riparian sites in NY, (ii) improvements to the model’s capacity for WTDs and groundwater NO$_3$-N concentrations simulation through calibration of the developed model by means of comparing model outputs with field data collected from three buffer sites in NY, (iii) validation of model’s output with the improved calibrated parameters, (iv) conduct a parameter sensitivity analysis. Eventually, this approach will aid the use of this model in this region and improve its functionality with respect to N transformations and flux.

2. Materials and Methods

2.1 Site Description
Tamanna et al. 2020 stated that Fall Creek watershed with an area of about 328 km² is located within the Finger Lakes region of New York State (42° 28′ Latitude, 76° 27′ Longitude) (Fig. 1). The most extensive source of parent material is glacial till, with additional parent materials that consist of glaciolacustrine sediments and glaciofluvial (outwash) deposits. The watershed is a mixed land-use landscape located at the southern terminus of the Wisconsin glaciation. The watershed is 4.8% urban/developed landuse (residential, commercial and service, industrial, etc.), 45.3% forest (evergreen forestland, mixed forestland), and 49.4% agriculture (cropland and pasture, other agricultural land, shrub and brush rangeland. Soils in the watershed are dominated by Gravelly silt loam and Channery silt loam. These are typically very deep, well-drained soils. Elevations range from 270 m above mean sea level to 600 m. The temperature in the watershed ranges from a 30 years (1982-2011) mean of 19.7°C in summer to a mean of -3.6°C in winter (PRISM Climate Group). The 30 years (1982-2011) average annual precipitation is approximately 930.3 mm, about 52.8% of which occurs during the spring and the fall months. The 14 years (2000-2013) average annual snow fall is 111.0 mm. Three riparian sections with contrasting physical attributes were identified along Fall Creek. These riparian zones are categorized on the basis of stream evolution stage.
The first riparian site is located within an inner meander (IM). The IM contains an unmaintained strip of tall grasses 16 m long measuring outwards from Fall Creek, and a mowed section 12 m long measuring outwards from the unmaintained strip to cropland. Tall grasses dominate vegetation in the unmaintained portion. Squash was grown on the cropland at the time of this study (Rook, 2012).
The second riparian site is located along a straight section (SS) of Fall Creek. This site consists of an unmaintained strip 35 m wide, and a mowed section 6 m long measuring outwards from the unmaintained strip to cropland. Tall grasses dominate vegetation in the unmaintained portion. This cropland was left fallow at the time of this study (Rook, 2012).

The third site is located along an outer meander / oxbow formation (OX) section of Fall Creek. This site consists of an unmaintained strip 29 m long measuring outwards from Fall Creek, and a mowed section 5 m long measuring outwards from the unmaintained strip to cropland. This riparian zone contains an oxbow depression, which is incised 1.75 m at the outer edge of the unmaintained portion. Tall grasses dominate the vegetation between the oxbow formation and Fall Creek while the short grasses dominate the deep depression. Potatoes were grown on the cropland at the time of this study (Rook, 2012).

Details on the soils, slope and buffer dimensions are provided in Tables 1 and 2.

2.2 Description of the REMM Model

REMM is a field-scale process-based, two dimensional, daily time-step model that simulates hydrology, nutrient cycling, and plant growth in a riparian buffer zone. REMM takes upland inputs and computes loading of water, sediments, nutrients, and carbon into the buffer where water and total N are transported from upland to Zone 3 (field edge), Zone 3 to Zone 2 (mid-buffer), Zone 2 to Zone 1 (near the stream) and
finally from Zone 1 to stream via surface runoff, seep flow, and subsurface flow (Altier et al. 2002; Lowrance et al. 2000).

REMM file version 0.1.1.46 (United States Department of Agriculture (USDA)-Agricultural Research Service (ARS)) was used for all simulations. A broad description of the REMM model is available in several publications, including (Altier et al. 2002; Inamdar et al. 1999; Lowrance et al. 2000). Briefly, REMM is a computer simulation model of riparian forest buffer systems. The structure of REMM is consistent with buffer system specifications recommended by the U.S. Forest Service and the USDA-Natural Resources Conservation Service as national standards (Welsch et al. 1991; NRCS 1995).

Within the model, the riparian system is considered to consist of three zones (parallel to a stream) between the field and the water body (Figure 2). Each zone includes litter and three soil layers (through which the vertical and horizontal movement of water takes place) that terminate at the bottom of the plant root system and a plant community that can include six plant types in two canopy levels. The riparian system characterized in REMM was originally designed to represent increasing levels of management away from the stream (Lowrance et al. 2000). REMM is written in the C++ programming language.
The details of mass balance of water movement within each soil layer and the equation used in REMM model to simulate denitrification is available in Tamanna et al. 2021.

**Table 1** Upland Characteristics for Riparian Sites.

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Geology</th>
<th>Soil</th>
<th>Land Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>IM</td>
<td>Glacial Till</td>
<td>Sandy loam</td>
<td>Organic Cropland</td>
</tr>
<tr>
<td>SS</td>
<td>Glacial Till</td>
<td>Sandy loam</td>
<td>Organic Cropland</td>
</tr>
<tr>
<td>OX</td>
<td>Glacial Till</td>
<td>Sandy loam</td>
<td>Organic Cropland</td>
</tr>
</tbody>
</table>

**2.3 REMM Model Input Data**

Four basic input files for REMM execution include: (1) contributions of daily outputs from the field draining into the riparian system including surface runoff and associated eroded sediment, organic material and plant nutrients (*.FIN), (2) weather data (*.WEA), (3) vegetation (*.VEG) characteristics, and (4) soil physical and chemical parameters (*.BUF). A field-scale hydrological model, Annualized AGricultural Non-Point Source (AnnAGNPS), was used to predict the upland input (runoff and
sediment loading) to the riparian buffer (Tamanna et al. 2020; Yongping et al. 2007). Details about the application of AnnAGNPS model on the Fall Creek watershed can be found in our companion study (Tamanna et al. 2020). Table 1 shows the upland characteristics for all riparian sites. Table 2 and section 2.1 shows the site characteristics of the modeled buffer for three riparian sites. The weather input, vegetation data and site characteristics were included into the REMM model following the procedures mentioned in Tamanna et al. 2021. All field data were collected by a team led by the authors of this manuscript with decades of published research on riparian zones in glaciated settings of the USA Northeast regions for the development and calibration of REMM model. For, Water table (WT) measurements, each site was instrumented with a dense network of shallow monitoring wells, piezometers, and static chambers and also a PVC tube inserted into each well and piezometer. Routine sampling was conducted on 8 occasions from May 31, 2011 to November 03, 2011. All samples were tested for nitrate (NO$_3$-N) with a Bran and Luebbe Autoanalyzer 3. Soil samples were also analyzed for denitrification enzyme activity (DEA). The DEA is the potential denitrification measured under fully anaerobic conditions and excess NO$_3$-N and available carbon. The denitrification rate constant ($K_d$) is based on the denitrification potential measurement (Tiedje, 1982). This was the only user input in REMM for simulating denitrification in all zones and layers within the buffer. Detailed field data collection and sampling information is available in Rook, 2012. The stream depth as 1.6 m has been used for this study. The other
essential input data were included into the REMM model following the procedures mentioned in Tamanna et al. 2021.

**Table 2** Site Characteristics of the modeled Buffer.

<table>
<thead>
<tr>
<th>Riparian Site</th>
<th>Buffer Area (m²)</th>
<th>Buffer Length (m)</th>
<th>Riparian Zone Width (3, 2, and 1) (m)</th>
<th>Zone Slope (%)</th>
<th>Manning’s n for Zone (3, 2, and 1)</th>
<th>Soil Drainage Class (Zone 1)</th>
<th>Soil (Zone 1)</th>
<th>Geomorphology</th>
</tr>
</thead>
<tbody>
<tr>
<td>IM</td>
<td>3,000</td>
<td>100</td>
<td>30 (10, 10, 10)</td>
<td>&lt;1</td>
<td>(0.032, 0.032, 0.032)</td>
<td>Silt loam</td>
<td>Well-drained</td>
<td>Glacial Till</td>
</tr>
<tr>
<td>SS</td>
<td>5,000</td>
<td>100</td>
<td>50 (16.67, 16.67, 16.67)</td>
<td>&lt;1</td>
<td>(0.032, 0.032, 0.032)</td>
<td>Silt loam</td>
<td>Well-drained</td>
<td>Glacial Till</td>
</tr>
<tr>
<td>OX</td>
<td>5,000</td>
<td>100</td>
<td>50 (16.67, 16.67, 16.67)</td>
<td>&lt;1</td>
<td>(0.032, 0.032, 0.032)</td>
<td>Silt loam</td>
<td>Well-drained</td>
<td>Glacial Till</td>
</tr>
</tbody>
</table>

2.4 Model Assessment

The performance of the model was evaluated by comparing field collected/measured data and REMM modeled data for both WTD and groundwater NO₃–N concentration. The evaluation statistics (the mean absolute error, MAE; the root mean square error, RMSE; the Willmott’s index of agreement, d) used for this study are the similar used in Tamanna et al. 2021.

2.5 REMM Model Calibration, Validation and Sensitivity Analysis

The Riparian Model was developed and evaluated first by testing the hydrologic component (measured WTDs) followed by the nutrient cycling component (measured groundwater NO₃-N concentrations). The model was calibrated and validated for both hydrology and nutrient cycling in zone 1 and its three layers (similar to the procedures defined in Lowrance et al. 2000 and Tilak et al. 2014). The calibration
period is from 5/31/2011 to 8/3/2011 and the validation period is from 8/23/2011 to 11/3/2011 for all the sites except site OX with a different calibration period as from 6/21/2011 to 8/3/2011. Because, OX site WTD data was absent during the first sampling round. The model was manually calibrated by changing the values of input parameters one at a time. The range of input parameters were either defined by field/laboratory measurements or obtained from literature or REMM user’s manual. The REMM permeability class of 1 (saturated hydraulic conductivities ranging from \( > 141 \ \mu \text{m/s} \)) was used for all the sites. Model calibration and sensitivity analysis procedure details are available in Tamanna et al. 2021. The calibrated soil physical buffer inputs, and calibrated Kd inputs are available in supplemental Tables S1 and S2, respectively.

3. Results

3.1 Water Table Depths Calibration and Validation

Simulated daily Water Table Depths (WTDs) were compared with those measured in the field in Zone 1 (closest to the stream) of the riparian buffer for all the sites. Simulated and measured WTDs in Zone 1 for the calibration and validation periods are shown in Fig. 3. Simulated and field measured WTDs in Zone 1 were compared using mean absolute error (MAE) and Willmott’s index of agreement (\( d \)) (Table 3).
Table 3 Statistical Comparison between Measured and Simulated WTDs in Zone 1 for all the sites.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>MAE (cm)</th>
<th>Willmott’s index of agreement (d)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration</td>
<td>Validation</td>
</tr>
<tr>
<td>IM</td>
<td>11</td>
<td>11</td>
</tr>
<tr>
<td>SS</td>
<td>12</td>
<td>7</td>
</tr>
<tr>
<td>OX</td>
<td>10</td>
<td>3</td>
</tr>
</tbody>
</table>

In case of site IM, the MAE for WTDs in Zone 1 was 11 cm for both the calibration and validation period. Whereas, the MAE for WTDs in Zone 1 was 12 cm, and 7 cm for the calibration and validation period, respectively for site SS. For site OX, the MAE for WTDs in Zone 1 was 10 cm, and 3 cm for the calibration and validation period, respectively. In case of Willmott’s index of agreement (d) between daily measured and simulated WTDs, the value of d was 0.39 and 0.33 for the calibration and validation period, respectively for IM site. The value of d was 0.42 and 0.92 for the calibration and validation period, respectively for SS site. The value of d was 0.41 and 0.94 for the calibration and validation period, respectively for OX site. The value of d was within the acceptable limit between 0 and 1 for both the calibration and validation periods for all sites.
Fig. 3. Measured and Simulated WTDs in Zone 1 of the Buffer during the Calibration and Validation Period for three riparian sites (IM, SS, OX).

3.2 Groundwater NO$_3$-N concentrations Calibration and Validation

Simulated and measured daily groundwater NO$_3$-N concentrations in Zone 1 during the calibration and validation period are shown in Fig. 4 and are compared statistically using the root mean square error (RMSE), the mean absolute error (MAE), and the Willmott’s index of agreement ($d$) (Table 4).
Table 4 Statistical Comparison between Measured and Simulated Groundwater NO$_3$-N concentrations in Zone 1 for the Calibration and Validation Periods.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>MAE (mg/L)</th>
<th>Willmott’s index of agreement (d)</th>
<th>RMSE (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calibration</td>
<td>Validation</td>
<td>Calibration</td>
</tr>
<tr>
<td>IM</td>
<td>0.20</td>
<td>1.67</td>
<td>0.78</td>
</tr>
<tr>
<td>SS</td>
<td>0.07</td>
<td>0.09</td>
<td>0.86</td>
</tr>
<tr>
<td>OX</td>
<td>0.06</td>
<td>0.41</td>
<td>0.84</td>
</tr>
</tbody>
</table>

In site IM, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.20 mg/L for the calibration period. Whereas, for the validation period, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 1.67 mg/L. For site SS, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.07 mg/L for the calibration period. While, for the validation period, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.09 mg/L. In regard to site OX, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.06 mg/L for the calibration period. Whereas, for the validation period, the MAE for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.41 mg/L.

In respect to Willmott’s index of agreement (d), at site IM, the value of d for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.78 for the calibration period. Whereas, for the validation period, the value of d for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.14. Site SS, on the other hand, the value of d for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.86 for the calibration period. Whereas, for the validation period, the value of d for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.75. As for site OX, the
value of $d$ for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.84 for the calibration period. Whereas, for the validation period, the value of $d$ for groundwater nitrate (NO$_3$-N) in Zone 1 was 0.48. The value of $d$ was within the acceptable limit between 0 and 1 for both the calibration and validation periods for all sites.

### 3.3 Parameter Sensitivity Analysis for Water Table Depths and Groundwater NO$_3$-N concentrations Simulation

The sensitivity analysis indicated that simulated predictions of Water Table Depths and groundwater NO$_3$-N concentrations could be very sensitive to selected soil physical properties and the denitrification rate constant, respectively. Change in field capacity caused the greatest change in WTD in all the sites except Site OX (where change in soil porosity caused higher % change in WTD than caused by change in field capacity; Fig. 5). All the sites did not exhibit $>\pm 25\%$ in response to a 10% change in the input parameters.

In Fig. 6, the percent change in daily average groundwater NO$_3$-N concentrations from actual for calibration period is shown due to corresponding parameter change. Change in soil porosity, field capacity and $K_d$ caused substantial changes in NO$_3$-N concentrations.
Fig. 4. Measured and Simulated Groundwater NO$_3$-N concentrations in Zone 1 of the Buffer during the Calibration and Validation Period for three riparian sites (IM, SS, OX).

4. Discussion

The MAE between measured and simulated daily WTDs is quite comparable to the average absolute error of 14 to 36 cm obtained in two previous model testing
studies using three to five years of data from a buffer sites in Georgia (Inamdar et al. 1999) and two sites in North Carolina (Dukes and Evans 2003; Tilak et al. 2014). The Willmott’s index of agreement (\(d\)) can be compared to the statistic found in Tilak et al., 2014 as 0.72 to 0.92 (yearly scale). The value of \(d\) was very high as 0.92 and 0.94 during the validation period at site SS and site OX respectively and indicated an excellent agreement between measured and simulated daily WTDs. On the other hand, the value of \(d\) was between 0.25 to 0.49 and pointed towards a fair agreement between measured and simulated daily WTDs in all other cases (Parajuli et al., 2009).

The simulated WTDs generally followed the seasonal pattern (deeper during drier months, June to November and rising to the surface during wet months, December to May) of measured WTDs. However, there were some discrepancies between measured and simulated WTDs during some time periods due to the spatial variability in rainfall between sites and the considered PRISM climate station, overestimation of ET by model etc. Besides, OX site WTD data was absent during the first sampling round.
Fig. 5. Parameter Sensitivity Analysis for Water Table Depths Simulation in Zone 1 of the Buffer for (a) IM, (b) SS, (c) OX. WP = Wilting Point in Z1L1; FC = Field Capacity in Z1L1; SP = Soil Porosity in Z1L1; Z1L1 = Zone 1 Soil Layer 1.
Fig. 6. Parameter Sensitivity Analysis for Groundwater NO$_3$-N concentrations Simulation in Zone 1 of the Buffer for (a) IM, (b) SS, (c) OX. FC = Field Capacity in Z1L1; SP = Soil Porosity in Z1L1; Kd = Denitrification rate constant; Z1L1 = Zone 1 Soil Layer 1.
We noticed a decline in REMM simulated daily groundwater NO$_3$-N concentrations in Zone 1 coming from the field edge zone 3 in case of site SS and site OX. The simulated daily mean NO$_3$-N concentrations dropped from 0.10 mg/L to 0.008 mg/L and 3.1 mg/L to 0.03 mg/L for sites SS and OX respectively during the simulation period (May 2011 to November 2011). On the contrary, at site IM, the simulated daily mean NO$_3$-N concentrations was higher (0.25 mg/L) in Zone 1 (stream edge) than that (0.06 mg/L) of in field edge (Zone 3). These model observations are consistent with the field measured data. Rook, 2012 mentioned in his study that, at site IM, NO$_3$-N mean value was highest at the center of the riparian zone (1.82 mg/L, +/- 2.87mg/L), and lowest at the field edge (0.64 mg/L, +/- 0.87mg/L); at site SS, mean NO$_3$-N concentrations decreased through the riparian zone, with elevated concentrations at the field edge (0.16 mg/L, +/- 0.33 mg/L) and depressed mean concentrations at the stream edge (0.05 mg/L, +/- 0.07 mg/L); at site OX, at the field edge, mean NO$_3$-N concentrations were at their highest (0.33 mg/L, +/- 0.89 mg/L) while at the stream edge, mean NO$_3$-N concentrations were at their lowest (0.10 mg/L, +/- 0.26 mg/L).

The RMSE between measured and simulated daily groundwater NO$_3$-N concentrations is relatively comparable to that of 1.05 to 1.50 mg/L obtained in the field testing study using 5 years of data from a buffer site in North Carolina Coastal Plain (Tilak et al., 2014). The range of $d$ (0.48 to 0.86) is also close to what (0.34 to 0.68) found by Tilak et al., 2014 from their study in North Carolina Coastal Plain except one low (0.14) $d$ value found during validation at site IM. The mean absolute error (MAE)
(0.07 to 1.67) for NO$_3$-N concentrations in Zone 1 for calibration and validation periods was reasonably similar to less than 1 mg/L of absolute error found by Lowrance et al 2000. Yet, the difference between measured and simulated NO$_3$-N concentrations during some time periods was probably due to the low frequency of field data collection.

The sensitivity analyses showed that changes in the volumetric water content between field capacity (FC) and saturation (i.e., soil porosity) was driving water table and denitrification dynamics. This observation is consistent with the relationships of (Linn and Doran, 1984) that shows that % saturation (the % of water-filled pore space, as determined by water content and total porosity) is closely related to denitrification. When compared to field capacity and Kd, the change (10% decrease) in soil porosity caused the greatest change in nitrate concentration in site IM and site OX. Whereas in case of site SS, both soil porosity and denitrification rate constant caused significant changes in nitrate concentration.

5. Conclusions

We successfully calibrated and validated the REMM model by coupling upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and nitrate data from three buffer sites located in a glaciated watershed of New York. Both the hydrologic and nutrient testing of REMM showed that it captured well the daily measured data (WTDs and groundwater NO$_3$-N concentrations in Zone 1) for both
calibration and validation periods. The sensitivity analyses demonstrated that changes in the volumetric water content between field capacity and saturation (soil porosity) was directing water table and denitrification dynamics. This study added another application of REMM for riparian zone nitrate estimation in glacial geomorphic settings of northeastern USA after our companion study (Tamanna et al. 2021).

**Author Contributions:** M.T. was responsible for the conceptualization, methodology, data processing, software, calibration, validation, formal analysis, writing—original draft preparation, and editing; S.M.P. was responsible for the conceptualization, funding acquisition, methodology, model/software supervision, and writing—review and editing; A.J.G., K.A. and P.G.V were responsible for the conceptualization, funding acquisition, and writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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USDA-ARS, National Sedimentation Laboratory, Mississippi, USA, for the AnnAGNPS model assistance.

Conflicts of Interest: The authors declare no conflicts of interest.

References


and nutrient cycling component for a coastal plain riparian system. Trans. ASAE 42, 1691.


## Supplementary Tables

**Table S1** Calibrated and Validated Parameters (soil physical properties) used in the REMM simulations.

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Zone 1 (Layer 1, Layer 2, Layer 3)</th>
<th>Zone 2 (Layer 1, Layer 2, Layer 3)</th>
<th>Zone 3 (Layer 1, Layer 2, Layer 3)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Site Name</strong></td>
<td><strong>Soil Layer Thickness (cm)</strong></td>
<td><strong>Soil Porosity</strong></td>
<td><strong>Field Capacity</strong></td>
</tr>
<tr>
<td>IM</td>
<td>(90, 110, 152)</td>
<td>(0.40, 0.40, 0.40)</td>
<td>(0.22, 0.22, 0.22)</td>
</tr>
<tr>
<td>SS</td>
<td>(90, 120, 190)</td>
<td>(0.47, 0.47, 0.47)</td>
<td>(0.22, 0.22, 0.22)</td>
</tr>
<tr>
<td>OX</td>
<td>(71, 81, 152)</td>
<td>(0.42, 0.40, 0.40)</td>
<td>(0.22, 0.22, 0.22)</td>
</tr>
</tbody>
</table>

**Table S2** Calibrated and Validated Parameter (Denitrification rate constant, $K_d$) for the Riparian Buffer.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Denitrification rate constant, $K_d$</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Site Name</strong></td>
<td><strong>Zone 1 (Layer 1, Layer 2, Layer 3)</strong></td>
</tr>
<tr>
<td>IM</td>
<td>(0.3, 0.3103, 0.02)</td>
</tr>
<tr>
<td>SS</td>
<td>(0.02, 0.0103, 0.52)</td>
</tr>
<tr>
<td>OX</td>
<td>(0.02, 0.0103, 0.002)</td>
</tr>
</tbody>
</table>
CHAPTER 5

Manuscript V

Climate Change Impact on Runoff Prediction in the Glaciated Landscape of the Northeast and Midwest USA

In preparation for *Journal of Hydrologic Engineering - ASCE*, August 2021

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Abstract

The study evaluated the performance of AnnAGNPS model in assessing the climate change impact on runoff coming from field edge (upland) towards the riparian zone (stream edge) in the glaciated landscape of the Northeast and Midwest USA.

AnnAGNPS was first calibrated and validated runoff for the historical period (1980 to 2009) for all three watersheds, Pawcatuck River watershed from Rhode Island, Fall Creek watershed from New York, and Sugar Creek watershed from Indiana to provide a baseline for comparing future projections. After AnnAGNPS successfully simulated runoff, a comparison of the runoff generation was carried out for short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG) scenarios along with the historical period (1980–2009) for all three watersheds. Our developed model well presented the seasonal pattern of runoff fluctuation for both historical and future periods for both low and high emission scenarios by indicating wet or dry trends in months.

Keywords:

AnnAGNPS; Runoff; Prediction; Glaciated; Climate change
1. Introduction

All aspects of the hydrologic cycle can be significantly impacted by climate change. Changes in climate will be magnified as a greater change in runoff. Various modeling studies have been carried out in approximately every part of the world to investigate climate change impact on runoff (Teng et al. 2012). Besides, global warming is now strongly evident. According to all past research and evidence, the Intergovernmental Panel on Climate Change (IPCC), 2007 concluded that the observed increase in anthropogenic greenhouse gas concentrations had caused most of the global surface air temperature increases since the mid-20th century. Eventually, global warming will bring changes in rainfall and other climate variables in the atmosphere, which will be amplified in the runoff (Chiew et al. 2009). Loss of nutrients from agricultural watersheds and the subsequent deterioration in water quality are of general concern, possibly affecting drinking water supplies and recreational values along with ecosystem health (Carpenter et al. 1998). Bosch et al. 2014 stated that it is obvious that high runoff years bring about very high nutrient loads, and wetter years are going to become more frequent under future climates. In this context, it is important to first quantify the runoff coming from the agricultural watersheds during future climate periods.

This study describes the modeling approach to assess the climate change impact on magnitude and peak flow of runoff coming from agricultural fields (croplands) in the glaciated landscape of the Northeast and Midwest USA. In
particular, the hydrological model AnnAGNPS is used to quantify the climate change impact on runoff generating from three different watersheds.

2. Materials and Methods

2.1 Study Area

For runoff prediction, three watersheds from three different states of Northeast and Midwest of USA were modeled for this study – (i) Upper Pawcatuck River watershed from Rhode Island (RI), (ii) Fall Creek watershed from New York (NY), and (iii) Sugar Creek watershed from Indiana (IN). Detailed information on land cover, soil texture, topography, weather and other relevant attributes for each watershed are mentioned in Tamanna et al. (2020).

2.1 Description of AnnAGNPS Model

Annualized AGricultural Non-Point Source (AnnAGNPS), was used to predict the runoff (Tamanna et al. 2020) from the three selected watersheds. AnnAGNPS (Bingner and Theurer 2001; Geter and Theurer 1998) is a daily time step, watershed scale, pollutant-loading, distributed model developed to simulate long-term runoff, sediment, nutrients, and pesticide transport from agricultural watersheds (Parajuli et al. 2009; Pradhanang 2010; Pradhanang and Briggs 2014). The AnnAGNPS model defines cells of various sizes; contaminants are routed from these cells into the associated reaches, and the model either deposits pollutants within the stream
channel system or transports them out of the watershed (Geter and Theurer 1998). The simulated runoff from AnnAGNPS was calibrated by comparing with observed data (USGS gauge). Detailed model set-up, calibration and validation for runoff simulation for those three watersheds are available in Tamanna et al. (2020).

2.3 Climate Change Variables

In case of Sugar Creek watershed, the projected change in average air temperature and precipitation over short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG) scenarios were incorporated and compared to the historical period (1980–2009). The basis for low and high greenhouse gas emission scenarios are the (Representative Concentration Pathway) RCP4.5 and RCP8.5 respectively adopted by the International Panel on Climate Change (IPCC) fifth Assessment Report (AR5). The RCPs are consistent with a wide range of possible changes in future anthropogenic GHG emissions, and aim to represent their atmospheric concentrations. RCP 4.5 assumes that global annual GHG emissions (measured in CO2 equivalents) peak around 2040, then decline whereas in RCP 8.5, emissions continue to rise throughout the 21st century.

For Fall Creek watershed, the projected change in maximum and minimum air temperature and precipitation over short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG)
scenarios were incorporated and compared to the historical period (1980–2009). The basis for low and high greenhouse gas emission scenarios are the (Representative Concentration Pathway) RCP4.5 and RCP8.5 respectively adopted by the International Panel on Climate Change (IPCC) fifth Assessment Report (AR5). The RCPs are consistent with a wide range of possible changes in future anthropogenic GHG emissions, and aim to represent their atmospheric concentrations. RCP 4.5 assumes that global annual GHG emissions (measured in CO₂ equivalents) peak around 2040, then decline whereas in RCP 8.5, emissions continue to rise throughout the 21st century.

In regard to upper Pawcatuck River watershed, the projected change in average air temperature and precipitation over short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG) scenarios were incorporated and compared to the historical period (1980–2009). The basis for low greenhouse gas emission scenarios is the 2007 International Panel on Climate Change SRES B1 scenario. And the high emissions are based on the SRES A1fi scenario. The B1 scenario is a circumstance where rapid economic growth is incorporated with a clean, resource efficient technology and GHG emissions levels return to pre-industrial concentrations, estimated at CO₂ levels of 300 parts per million (ppm). The high-emission scenario (A1fi) is a scenario emphasized on fossil fuel intensive technologies for rapid economic growth resulting in CO₂ levels reaching 940 ppm.
For all the watersheds, the CSV (Comma Separated Values File) input files for climate was modified by the climate variables without any change in the previously established calibration settings of AnnAGNPS model. The climate variables used for Sugar Creek watershed, Indiana are based on values published by Hamlet et al. (2019), which were generated from statistically-downscaled climate change scenarios for the State of Indiana. The basis for this approach is Global Climate Model (GMC) simulations from the Coupled Model Intercomporion Project phase 5 (CMIP5) associated with the IPCC Fifth Assessment Report on global climate change. The statistical downscaling uses the Hybrid Delta (HD) approach developed at the University of Washington (Hamlet et al. 2013 and Tohver et al. 2014) (Supplementary Table S1). The climate variables used for Fall Creek watershed, New York are based on values published by ACIS, Northeast Regional Climate Center and Cornell University. The basis for this approach is the General Circulation Model (GCM) projections from the 32 Climate Model Intercomparison Project Phase 5 (CMIP5) (Taylor et al. 2012). The projections have been downscaled to a spatial resolution of 1/16 degree (approximately 6 km x 6 km) using the Localized Constructed Analog (LOCA) method of (Pierce et al. 2014). Supplementary Table S2 presents the climate change variables adopted and modified from ACIS, Northeast Regional Climate Center for Tompkins County, New York. The climate variables used for upper Pawcatuck River watershed, Rhode Island are based on values published by Wake et al. (2014) at the University of New Hampshire, which were
generates from four global climatic models downscaled to the New England region. Supplementary Table S3 referred by Chambers et al. (2017) presents the climate change variables adopted and modified from Wake et al. (2014) for Kingston, Rhode Island.

For all the watersheds, in order to get the relative comparison between the runoff quantity during historical period (1980-2009) and runoff amount over short term, medium term and long term time frame for low and high emission scenarios, historical (1980-2009) climate data obtained from PRISM website at 4km spatial resolution (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 4 Feb 2004) was incorporated in AnnAGNPS model and model was run to simulate runoff for the historical period. After that, the model was run to simulate runoff for short term (2010–2039), medium term (2040–2069) and long term (2070–2099) along with historical period.

The calibrated model for each watershed was first run over the entire 30-year period (1980–2009) to get the understanding about the model simulation performance for the historical period. In order to do the evaluation, the observed streamflow data (1980-2009) from each individual USGS gauge was obtained and then calculated the runoff amount by using the previously mentioned manual baseflow filtering approach. The data simulated from 1980 to 2009 provide a baseline for comparing future projections (Table 1). These approaches are similar to the one followed by Chiew et al. (2009); Teng et al. (2012).
Table 1  AnnAGNPS Model runoff simulation for historical period (Daily Scale)

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<tr>
<th>Watershed</th>
<th>Mean Observed Runoff (mm)</th>
<th>Mean Modeled Runoff (mm)</th>
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3. Results and Discussions

3.1 Historical Conditions vs Future Projections for runoff

In case of Sugar Creek watershed, for low emission scenarios, runoff volume over short term will decrease than that over historical period for all the months except February. While, runoff volume will drop during early spring, summer and fall seasons and rise during winter and late spring over medium term. Over long term, overall runoff decline will be observed for all seasons except slight increase in winter season. For high emission scenarios, the runoff volume will drop for all seasons except slight rise during winter and late spring over both short term and medium term. Besides, the runoff volume will decrease than that from historical period for all the year round except the month of January during long term (Fig. 1). In general, the trend is noticeably drier summer and fall over short, medium and long term when compared to historical period.
Winter and spring seasons are wetter than summer and fall seasons. These observations are in agreement with the future trends mentioned in Hamlet et al. (2019).
From Fig. 2, it is observed that in case of Fall Creek watershed, for both high and low emission scenarios, runoff volume will go up over short, medium and long term than the historical period for fall, winter and spring except summer (June, July, August). The total annual runoff volume will increase by 15.2% and 12.9% over 2010 – 2039 for low and high emission scenarios respectively. Over 2040 – 2069, the total annual runoff volume will increase by 11.4% and 9.5% for low and high emission scenarios respectively. An increase by 4.4% and a decrease by 7.4% will be
experienced over 2070 – 2099 in terms of total annual runoff volume for low and high emission scenarios respectively. This prediction interprets that in future winter would be drier than before.

**FIG. 3.** Monthly Mean Runoff volume for Low and High Emission Scenarios (Pawcatuck River Watershed)

In case of Pawcatuck River watershed, Fig. 3 shows that for low emission scenarios, average runoff volume will increase over short, medium and long term for all seasons except early spring (March) in comparison to historical period. For high emission scenarios, runoff volume will increase over short term for all the months
except November than the historical period. On the other hand runoff volume will drop over medium and long term for all the seasons from the historical monthly mean runoff volume.

4. Conclusions

The projected change in average air temperature and precipitation over short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG) scenarios were incorporated and compared to the historical period (1980–2009) for all three watersheds. Our developed model well presented the seasonal pattern of runoff fluctuation (rise in winter then a drop starting from late spring continuing over summer and fall and again starting rise from late fall to winter) for both historical and future time periods for both low and high emission scenarios in all three watersheds.

The Sugar Creek watershed will experience a trend of relatively drier summer-fall and wetter winter–spring over 2010 – 2099 for both low and high emission scenarios. In general, an increase in total annual runoff will occur for both high and low emission scenarios over 2010-2039, 2040-2069 and 2070-2099 in Fall Creek watershed. A slight decrease for high emission scenarios over 2070 – 2099 could occur. Because sometimes the rate of evapotranspiration is not significantly low enough to generate substantial runoff inside the model for a longer period. Average runoff volume will increase over 2010 - 2099 for all seasons except early spring
(March) for low emission scenarios in Pawcatuck River watershed. For high emission scenarios, runoff volume will increase over short term for all the months except November than the historical period. On the other hand runoff volume will drop over medium and long term for all the seasons from the historical monthly mean runoff volume.

This study will aid the water resources managers, decision makers and stakeholders to be able to take effective management decisions. Because they will gain a better understanding of both historical trend and future runoff quantification under climate change scenarios. This study will also be beneficial for the study of runoff impact on erosion and other water quality monitoring.

Author Contributions: M.T. was responsible for the conceptualization, methodology, data processing, software, calibration, validation, formal analysis, writing—original draft preparation, and editing; S.M.P. was responsible for the conceptualization, funding acquisition, methodology, model/software supervision, and writing—review and editing; A.J.G., K.A. and P.G.V were responsible for the conceptualization, funding acquisition, and writing—review and editing. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflicts of interest.

References


Future; Sustainability Institute at the University of New Hampshire: Durham, NH, USA, 2014. Available online: https://scholars.unh.edu/sustainability/ (accessed on 1st October 2018).
Supplement Information

Table S1 (a) presents projected seasonal temperature changes (°C) over Indiana. The value is the spatially averaged, ensemble mean temperature change. (b) presents projected seasonal precipitation changes (%) over Indiana. The value is the spatially averaged, ensemble-mean, percent change in $P$. Seasons are considered as Winter (December, January, February); Spring (March, April, May); Summer (June, July, August) and Fall (September, October, November) for this study.

(a)

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(b)

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Table S2 Climate change variables adopted and modified from ACIS, Northeast Regional Climate Center for Tompkins County, New York. Temperatures listed as degree (°C) increase, for both maximum ($T_{\text{max}}$) and minimum ($T_{\text{min}}$) temperatures.

Precipitation (Precip) values listed as a relative change computed based on the published values.

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Table S3 Climate change variables adopted and modified from Wake et al., 2014 for Kingston, RI. Low emissions based on SRES A1fi scenario and high emissions based on SRES B1 scenario. Temperatures listed as degree (°C) increase, averaged from the published minimum and maximum temperatures. Precipitation values listed as a relative change computed based on the published values.

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CHAPTER 6
Conclusions

In this dissertation, we successfully calibrated and validated the REMM model by coupling upland inputs from a distributed model (AnnAGNPS) with field-measured hydrologic and groundwater nitrate data from thirteen buffer sites located in four different watersheds from USA Northeast and Midwest. All these riparian sites are located in glacial geomorphic setting. For runoff prediction, four watersheds from three different states of Northeast and Midwest of USA were modeled for this study – (i) Upper Pawcatuck River watershed from Rhode Island (RI), (ii) Fall Creek watershed from New York (NY), and (iii) Sugar Creek watershed from Indiana (IN), (iv) Eagle Creek watershed from Indiana (IN). Whereas, in regard to riparian zone nutrient prediction, total thirteen riparian sites were considered for this study – eight sites in upper Pawcatuck River watershed, three sites in Fall Creek watershed, one site in Sugar Creek watershed and one site in Eagle Creek watershed. Both the hydrologic and nutrient testing of REMM showed that it captured well the daily measured data (WTDs and groundwater NO$_3$-N concentrations in Zone 1) for both calibration and validation periods. The parameter sensitivity analyses demonstrated that changes in the volumetric water content between field capacity and saturation (soil porosity) was directing water table and nitrogen dynamics.

This modeling study showed the suitability of REMM to simulate the basic hydrologic and biogeochemical processes occurring in real-world buffers, particularly
in glacial geomorphic settings (where REMM applicability has not been tested yet). In addition to that, the use of distributed model AnnAGNPS provided better estimates of upland inputs. As such, the output (appropriate calibrated parameters) of this study establishes the base for site-specific parameters (soil porosity, field capacity, wilting point, denitrification rate constant etc.) required to evaluate management and design of riparian buffers effectively (e.g., riparian width, vegetation type) in other sites of a similar setting.

Additionally, after AnnAGNPS successfully simulated runoff, a comparison of the runoff generation was carried out for short term (2010–2039), medium term (2040–2069) and long term (2070–2099) time frame for low and high greenhouse gas (GHG) scenarios along with the historical period (1980–2009) for all three watersheds. Our developed model well presented the seasonal pattern of runoff fluctuation for both historical and future periods for both low and high emission scenarios by indicating wet or dry trends in months.

For future analysis if we could add more degree of change (i.e., ±20%, ±30%, etc. of input parameter change) into the sensitivity analysis, this addition would be able to bring more scenarios in terms of riparian hydrologic and nutrient estimation corresponding to various input parameter modification. In future, the riparian model REMM can be used to assess the climate change impact on riparian zone nutrient concentration in multiple buffers located in those four watersheds.