Environmental Processes (2025) 12:40
https://doi.org/10.1007/s40710-025-00782-2

RESEARCH

Understanding Nutrient Loading in a Hydrologically Sensitive Coastal Watershed. The Peace River Watershed, Florida, USA

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Received: 20 February 2025 / Accepted: 14 June 2025

Abstract

Nutrient pollution from anthropogenic activities threatens freshwater and estuarine ecosystems, exemplified by the Peace River Watershed (PRW) in Florida, where excess nutrients fuel harmful algal blooms (HABs) in Charlotte Harbor. This study utilized the Watershed Assessment Model (WAM) to analyze nutrient loading and identify source areas within the PRW, which is crucial for effective mitigation. WAM integrated land use, soil, rainfall, and wastewater data to create unique cells. Daily flows and loads were simulated using field-scale sub-models tailored to Florida conditions, routed through the river network, then calibrated and validated against observed data. Results showed an annual discharge of 1.9 cubic kilometers of water, 2,370 tons of total nitrogen (TN), and 1,025 tons of total phosphorus (TP) into Charlotte Harbor. Spatially, annual TN and TP offsite loads ranged from near zero to 116 and 38 kg/ha, respectively. Elevated TN levels were observed in urban areas with septic systems and agricultural lands lacking best management practices (BMPs). High TP loads were associated with phosphate mining and intensive agricultural areas. Groundwater nitrate was elevated in urban areas with septic systems. The TN: TP ratio at the outlet showed nitrogen limitation, although 33% of the source area was P-limited. Nitrogen reduction was attributed to attenuation processes during transport. Therefore, controlling HABs in the harbor requires targeted TN reduction in hotspot areas. This study established a water quality baseline, identified critical source areas, and will help in devising mitigation strategies in the PRW and similar coastal watersheds.

Highlights

- The WAM model simulated spatial nutrient loadings to identify source areas in the PRW, a coastal watershed in Florida.
- Urban areas with septic systems showed elevated total nitrogen levels in the groundwater.
- Phosphate mining lands generated elevated TP loads.
- Agriculture showed elevated levels of both TN and TP, particularly in areas lacking BMPs.

Extended author information available on the last page of the article

Published online: 20 June 2025

- While the PRW outlet (Charlotte Harbor) is primarily N-limited, 33% of the watershed source area exhibits P limitation.

Keywords Coastal watershed · Source area · Nutrient loading · Nitrogen · Phosphorus

1 Introduction

Increasing nutrient loading within coastal watersheds has become a pervasive issue, significantly threatening the quality of freshwater resources (Fu et al. 2020; Giri 2021). Although nutrient flux is a natural process, when enhanced by anthropogenic impacts such as an expansion of urban areas, intensification of agricultural practices, and disturbance by mining activities, it threatens the ecosystem functioning of a watershed (Kroeze et al. 2012; Tarabih et al. 2024; J. Zhang et al. 2022a, b; Rixon et al. 2024), jeopardizing the social and economic well-being of the community that relies on it (Gunko et al. 2022; McLennan 2022). This continuous increment of nutrient fluxes in coastal areas causes nutrient enrichment of downstream freshwater bodies, ultimately leading to eutrophication (Giri 2021; Wester et al. 2023). This nutrient enrichment is a major global impairment of fresh surface water resources such as lakes, reservoirs, and estuaries, triggering HABs, oxygen depletion, and a decline in biodiversity (Khan and Mohammad 2014; Schramm 2023; Zhang et al. 2022a, b; Brentjens and Bratt 2023). Effectively managing watershed nutrient flux in a changing global environment poses a significant challenge, necessitating advanced modeling tools that link source and sink areas.

Since human beings began to alter landscapes for increased productivity through agriculture, mining, and other activities, concerns about excess nutrient fluxes have plagued communities (Damania 2019; Wester et al. 2023). This alteration led to offsite effects, causing water quality degradation in freshwater resources (Desbureaux et al. 2022; Khan and Mohammad 2014; Wester et al. 2023). Nutrient levels in watershed systems became a major concern in the United States since the 1970s when air and water pollution became an issue to the environment, which led to the establishment of the Environmental Protection Agency (EPA) (Williams 1993). Water quality concerns have risen to a global scale as our population booms, industries expand, and climate change disrupts water cycles (Damania 2019; Desbureaux et al. 2022; Gunko et al. 2022). This growing water quality pressure on a finite resource threatens not only human health but also aquatic ecosystems (Gunko et al. 2022; Wester et al. 2023). Monitoring nutrient fluxes in a watershed requires significant investment and extensive spatial and temporal data collection. This makes the issue a global one, though the severity varies between countries.

Thus, developing successful nutrient management strategies hinges on acquiring a comprehensive understanding of the watershed's spatial and temporal patterns in hydrological and nutrient fluxes and processes (Fu et al. 2020; Yuan et al. 2020; Brentjens and Bratt 2023). Nutrient fluxes are assessed either through conventional measurements or modeling approaches (Kroeze et al. 2012). While field nutrient concentration measurements are reliable, the high cost and time investment required for comprehensive data collection often necessitate exploring alternative methods (Desbureaux et al. 2022). The modelling approach combined with limited observational flux data has been successful in quantifying nutrient loads in watersheds (Busari et al. 2023; Chinyama et al. 2014; Desbureaux et al. 2022; Fu et al. 2020). This modeling approach provides a reasonable estimate in better spatial and temporal resolutions using limited data availability for calibration and validation (Costa et al. 2021; Kroeze et al. 2012).

Based on the processes involved in the simulation, water quality models are classified as empirical, conceptual, and physical models (Chinyama et al. 2014; Costa et al. 2021; Jaiswal et al. 2020). Empirical models, while simple and data-driven, lack insight into the underlying processes. Conceptual models offer a broader picture but struggle to perfectly represent reality. Physical/process-based models represent watershed processes using numerical approaches employing mathematical-physical formulations, e.g., the Watershed Assessment Model (WAM) (SWET 2018) and Soil and Water Assessment Tool (SWAT) (Gassman et al. 2007). These models can be lumped, semi-distributed, or fully distributed based on how they handle spatial details (Costa et al. 2021; Jaiswal et al. 2020). Lumped models treat the entire watershed as uniform, ignoring spatial variations (Busari et al. 2023; Costa et al. 2021; Jaiswal et al. 2020). In contrast, distributed models account for these variations, making them more complex (Busari et al. 2023). Process-based models, though requiring significant data and expertise, provide the most detailed predictions (Jaiswal et al. 2020).

Scholars have applied process-based modeling to coastal watersheds to examine the anthropogenic impacts on groundwater nutrient loading (Berihun et al. 2025; Rixon et al. 2024; Brentjens and Bratt 2023), surface water nutrient enrichments (Vigiak et al. 2023; Howarth et al. 2021; Aboelnour et al. 2025), and their off-site effects (Medina et al. 2025) separately. However, recent studies have suggested considering surface and subsurface as one interconnected water system for future research, monitoring and management decisions in coastal watersheds (Douglas and Murgulet 2025). Furthermore, scholars have faced challenges in applying models to simulate nutrient levels tailored to varying soil and land use conditions (Palola et al. 2025; Costa et al. 2021). Consequently, assessing integrated surface and sub-surface nutrient levels specific to watershed conditions remains a challenge, particularly in hydrologically sensitive environments like Florida, where surface and subsurface interaction is frequently triggered by sea and ground level rises (Berihun et al. 2025; Douglas and Murgulet 2025).

Previous research concerning Florida's coastal watersheds has concentrated on flood hazards (Tsegaye et al. 2024), the consequences of phosphate mining (J. Zhang et al. 2022a, b), climate change effects, groundwater contaminations, agricultural influences (Wester et al. 2023), the development of HABs (Heil and Muni-Morgan 2021; Medina et al. 2024), review assessments on either surface or subsurface water quality (Costa et al. 2021; Howarth et al. 2021; Khare et al. 2021; Yuan et al. 2020). However, the coastal watersheds in Florida are hydrologically sensitive and characterized by shallow groundwater coupled with wetlands and ponds and have gone through several changes in land use including agricultural intensification, mining, and urban expansions (FDEP 2007; Lewelling et al. 1998; Medina et al. 2025) that necessitates a unique representation of surface and subsurface flows and nutrient processes. Several studies have addressed the response to the legacy of excess phosphorus fertilization and the effectiveness of BMPs in the Lake Okeechobee basin (Khare et al. 2021; Tarabih et al. 2024). The limited available research on PRW primarily focuses on hydrologic indicators of flow regimes through concentration-discharge curves (Onwuka et al. 2021), hypoxia formation and hydrodynamics of Charlotte Harbor (Chen 2020), and reports on hydrology and trends on water quality of Charlotte Harbor (Garcia et al. 2020; Medina et al. 2025). While these studies partially illustrate how hydrological alterations within the PRW can impact downstream water quality, their scope is often limited to specific portions of the watershed or solely to the harbor and doesn't address the anthropogenic implications on nutrient loading of the entire watershed. Thus, to comprehensively assess and mitigate nutrient loads across the entire hydrologically sensitive PRW, an integrated modeling approach is necessary to pinpoint specific source areas and understand the spatial variations of nutrient loading of the PRW for implementing effective mitigation measures.

For this study, we deployed a WAM application for the impacted PRW, which is the most important freshwater resource in southwestern Florida (Medina et al. 2025). WAM is a physically based hydrology and water quality model extensively applied in Florida and validated for different watersheds in the state (Bottcher et al. 2012; SWET 2018). It is a powerful tool for simulating surface and subsurface systems in complex watersheds (Tarabih et al. 2024). It simulates daily flow and loads from diverse natural and manmade landscapes, considering factors like impervious surfaces and on-site water and nutrient management practices (Graham et al. 2009). This detailed analysis allows for the evaluation of water quality at any location within the watershed and is useful for pinpointing source areas (Khare et al. 2021; Tarabih et al. 2024).

The objective is to implement surface and subsurface integrated modeling to assess nutrient fluxes at the source areas of a hydrologically sensitive coastal watershed. Nutrient loading from different land uses will be used to explain the offsite effects of impacted coastal watershed, the PRW, and its implications for Charlotte Harbor's HABs by identifying the limiting nutrient. We are particularly interested in investigating the spatial distribution of nutrient fluxes originating from diverse land use types within the watershed. This study provides a foundation for water quality managers to assess future research needs and evaluate potential water quality changes resulting from upcoming hydrological alterations and empower planners to proactively make decisions regarding current and future developments in PRW and other similar watersheds globally.

2 Materials and Methods

2.1 The Study Area

The Peace River Watershed (PRW) encompasses 608,650 hectares in southwest Florida (Fig. 1) and provides a critical source of water supply, agriculture, phosphate mining, and recreation while sustaining vital ecosystem services for the community (FDEP 2007; Lewelling et al. 1998; Onwuka et al. 2021). Originating in the Polk highlands, the river flows south through various physiographic regions, eventually discharging into Charlotte Harbor, where excess nutrient loading threatens the harbor's ecosystem, fueling HABs (Garcia et al. 2020; Hammett 1990; Heil and Muni-Morgan 2021; Kim et al. 2010; Metz and Lewelling 2009). Like many coastal watersheds, the PRW boasts a remarkable diversity of plant and animal life (FDEP 2007). The PRW contains hundreds of named lakes, ponds, rivers, and streams, and travels through swamps, flatwoods, hammocks, and marshes before reaching Charlotte Harbor and supplying drinking water for nearly one million residents across Polk, Hardee, Desoto and Charlotte counties (Metz and Lewelling 2009). The PRW is hydrologically sensitive due to the shallow groundwater, complex lakes, pond-river interaction and the dynamic land use pattern that affects the water quantity and quality of the basin.

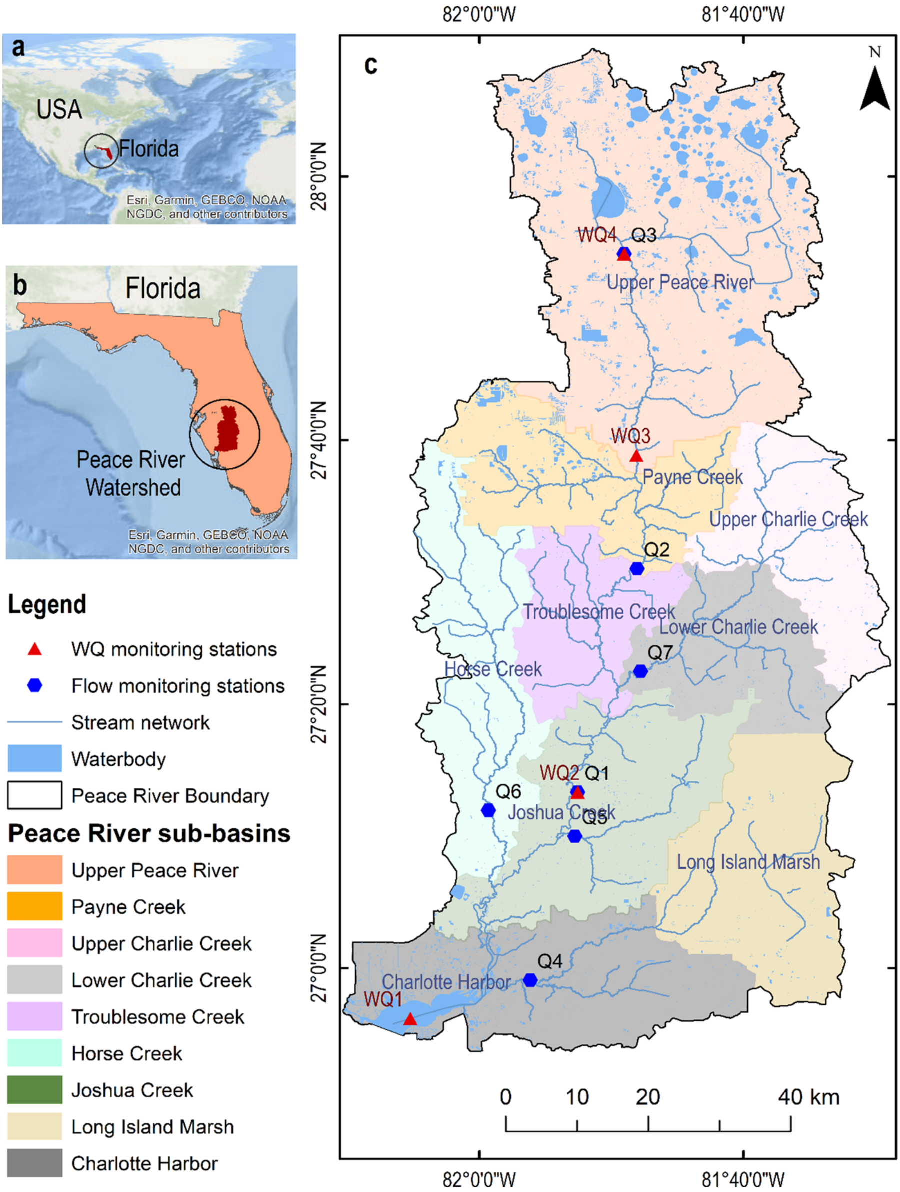

Fig. 1 Location of the study area. (**a**) location of Florida superimposed on a global map (**b**) The location of PRW superimposed on the state of Florida map, and (**c**) a detailed map of PRW including sub-basins, water bodies, river network, flow and water quality monitoring stations

Water quality degradation is a concern in the PRW. Excess nutrient loading (nitrogen and phosphorus) from point and nonpoint sources within the watershed, such as fertilizer runoff from farms and lawns, and discharges from wastewater treatment facilities, is leading to environmental problems (FDEP 2007; Metz and Lewelling 2009; Hammett 1990). Land use practices like phosphate mining, agricultural intensification and urban development contribute to this by increasing nutrient fluxes (FDEP 2007; Hammett 1990). Excessive nitrogen and phosphorus runoff contributes to HABs in Charlotte Harbor, the second-largest estuary in Florida, jeopardizing the environment and public health (Chen 2020; Garcia et al. 2020; Kim et al. 2010; Medina et al. 2025).

The Peace River relies on rainfall, averaging 135 centimeters annually, with most occurring in summer months. While some water evaporates or is transpired by plants, the rest replenishes the aquifer or flows into the Harbor through its tributaries (Lewelling et al. 1998). A significant amount of water is also withdrawn daily to support irrigation from the downstream part of the river and to supply water to the urban areas (Onwuka et al. 2021).

The watershed primarily consists of agricultural lands (36.5%), followed by wetlands (19%), forest and brush (16%) and urban areas (13%) (Fig. 2c). Phosphate mining in the northwestern side of PRW (Khare et al. 2021), intensive agriculture in the central and easter parts of the basin (about 70% enrolled in BMPs), and urban development in the upper and lower PRW have historically changed the natural landscape and land use pattern in PRW (FDEP 2007; Lewelling et al. 1998; Metz and Lewelling 2009). The majority of PRW (60%) is covered by poorly drained soils, primarily flatwood soils, followed by 14% of the area underlain by very poorly drained soils including muck, wetland soils, and mining lands (Fig. 2c). In contrast, well-drained soils, encompassing moderately well-drained, well-drained, and excessively drained types, account for only about 12% of the watershed, primarily concentrated in the Upper Peace River, Payne Creek, and Upper Charlie Creek basins (Figs. 1 and 2c).

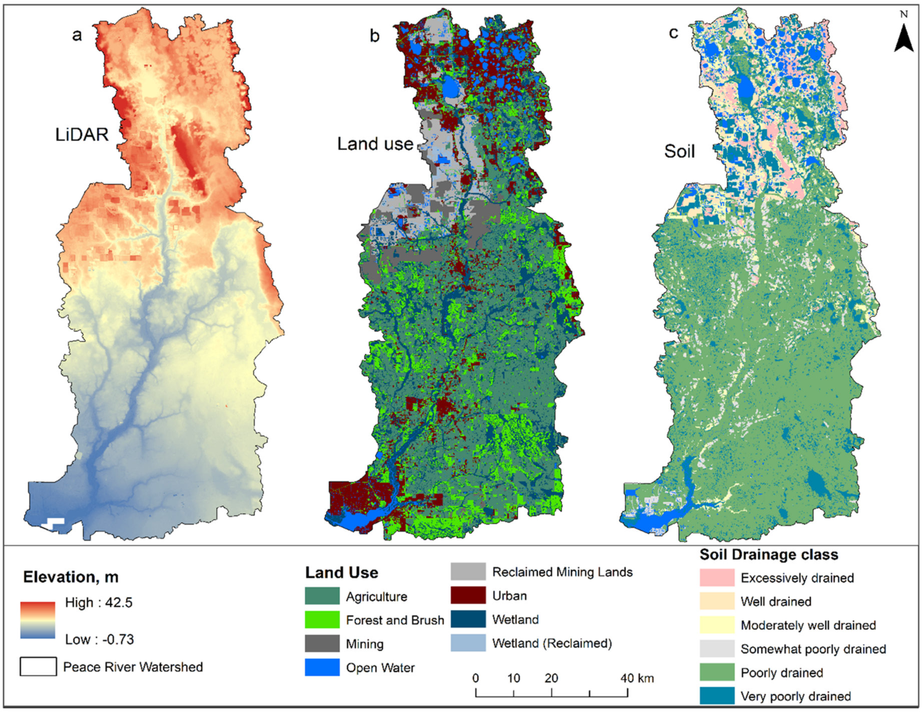

Fig. 2 Peace river watershed characteristics: (**a**) Topography derived from 2018 Lidar DEM (**b**) A land use type developed in 2021, and (**c**) A soil drainage class mapped in 2019

2.2 Datasets

All necessary data required to assess point and nonpoint source loadings were collected from available sources as described in Table 1. The downloaded datasets include the PRW boundaries, the drainage network, climate data, land use, soil data mapped by the National Resource Conservation Service (NRCS), digital elevation models (DEMs) based on LiDAR data (Fig. 2a), and NEXRAD rainfall data from the Southwest Florida Water Management District (SWFWMD) site (Fig. 2; Table 1). Rainfall also contributes nutrient deposition into the watershed. Hence, atmospheric loadings for N and P are represented in WAM by multiplying the rainfall volume by TN and TP concentration of 0.2 and 0.007 mg/l, respectively, derived from previous studies done by the South Florida Water Management District

Table 1 List of datasets used for the PRW nutrient modeling

Dataset	Dataset Name	Parameter(s)	Resolution and/or data type	Time period
PRW Boundaries	*The USGS Watershed Boundary Dataset*	Watershed and sub-basin HUC	HUC-8 & 10, Feature Class	2022
Hydrologic Network Data	*National Hydrography Dataset Plus*	Waterbodies	Feature class	2018
Soils	*National Resource Conservation Service Web Soil Survey*	Soil classifications	Feature class	2019
Land Use / Land Cover	*Florida Department of Environmental Protection (FDEP)*	Land Use Designations	Feature class	2021
Topographic	*The USGS Lidar and DEM data*	Lidar and DEM Topographic data	2.5ft grid cells (resampled into a hectare grid), Raster	2018
Weather	*FAWN - Florida Automated Weather Network (ufl.edu)*	Temperature, Wind, Solar Radiation	15-minute interval data (aggregated into monthly), Tabular	1996–2023
Rainfall	*NEXRAD Daily Rainfall Data processed by SFWMD*	Daily Rainfall	2 km x 2 km of 15-minute interval (aggregated into daily), Temporal Raster	1996–2023
Aquifer Recharge Areas	*Florida Geographic Data Library Data Catalog (RCHARG_DEC03, AQDRIN)*	Estimated recharge to the Floridan aquifer	Feature class	1996–2023
Wastewater utilities	*Florida Water Management Inventory Septic/Sewer Systems FDEP Wastewater Facilities*	Parcel information for septic/sewer. FDEP link is to wastewater treatment facilities	Variable times series data, Feature class	1996–2023
Irrigation Sources	*SFWMD Groundwater Well Catalog*	Listing of well data with lat/long,	Tabular	1996–2023
Hydraulic Structures	*SWFWMD Structure Operational Guidelines*	Protocols for structure operations, also a live status map	Daily structure operations summary of gate status and water levels, Feature class	1996–2023
Point Sources	*EPA ECHO (Filter by HUC 031001)*	Discharge from permitted facilities	Annual, Feature class	1996–2023
Streamflow	*The USGS Current Water Data*	Discharge, Stage Height,	Daily, Time series	1996–2023
Nutrients	IWR Database	TN & TP	Variable times series data, Time series	1996–2023

(SFWMD) in the Lake Okeechobee Watershed (LOW) (Faridmarandi et al. 2021). Wastewater treatment data detailing which urban areas receive either septic or sewer treatment were downloaded from the water management inventory, collected by the Florida Department of Health (Table 1). Wastewater Treatment Facilities (WWTFs) were obtained from the Florida Department of Environmental Protection (FDEP). Additionally, a dataset indicating sources for the withdrawal of irrigation water and locations of major hydraulic structures with their operational protocols, primarily situated in the upper PRW, was obtained from SWFWMD (Table 1). Lastly, observed streamflow data were obtained from USGS gage locations within the domain, and nutrient data acquired from FDEP's publicly available Impaired Waters Rule (IWR) and Watershed Information Network (WIN) database were downloaded and used.

2.3 Modeling Approach

Understanding the runoff process and the nutrient flux in hydrologically sensitive coastal watersheds like PRW demands the representation of an integrated approach for both surface and subsurface flows. This involves processing spatial data for input using ArcGIS, unique cell creation, surface and sub-surface attenuation processes and routing to stream networks and receiving water bodies (Fig. 3). Integrating ArcGIS streamlines spatial data preparation and visualization, enabling user-friendly input/output and customizable data display. It facilitates the organization dataset for DEM, land use/cover, soil, rain, utility zone, climatic, and hydraulic structures with their operation protocols. In addition, spatially distributed point sources were represented as a time series of inflows or outflows of any reach within the reach network.

A grid-based approach, utilizing the Basin Unique Cell Shell (BUCSHELL) sub-module, was employed to divide the watershed into 608,650 grid cells, one hectare in size. Based on land use and soil conditions, appropriate field-scale models were selected for each grid cell. The field-scale models used are the Everglades Agricultural Area Model (EAAMOD) for agricultural and urban land uses on high water table soils (Bottcher et al. 1998) and the Groundwater Loading Effect of Agricultural Management System (GLEAMS) for well-drained soils (Knisel 1993). For wetlands, open water, and mining areas, a special-case model utilizing a water balance approach together with user-defined Event Mean Concen-

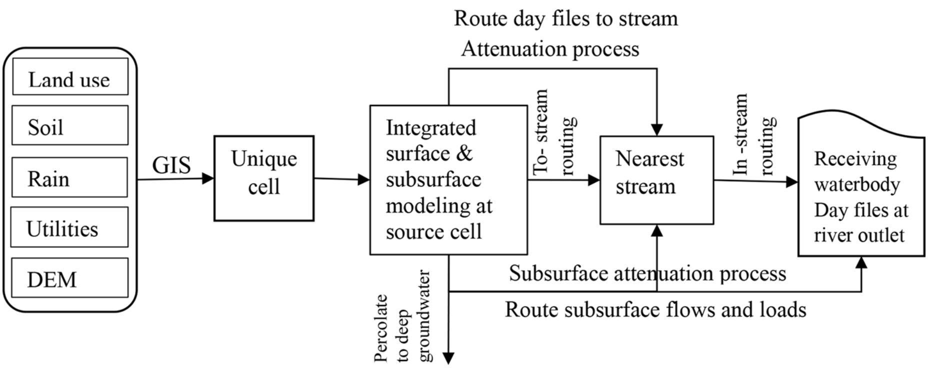

Fig. 3 Conceptual representation of load and flow simulations in PRW, Florida, using the Watershed Assessment Model (WAM)

tration (EMC) values was applied to calculate runoff and nutrient concentrations (SWET 2018). Daily estimates of surface and subsurface flows, along with estimated nutrient loads generated by the field-scale models, were post-processed to account for the impacts of impervious surfaces, wastewater systems, stormwater features, and management practices.

The daily output of water and nutrient loads from the field scale models was routed to the destination stream reaches (to-stream routing) and subsequently through the stream network to the watershed outlet (in-stream routing). This step uses the Basin Land Area to Stream Routing (BLASRoute) sub-module. To-stream routing was based on delayed unit hydrographs, defined separately for surface and sub-surface flows whereas in-stream routing was based on a modified time-varying linear reservoir routing technique. Flow and nutrient assimilation processes are applied during the source cell-to-stream and in-stream phases. These attenuation processes are based on the distance the flows and nutrient loads pass over different landscape features, such as overland, varying wetland types or through groundwater. The attenuation process from the source cell to the target conveyance reach is more influenced by flow rate/velocity and distance of travel (Eq. 1). To-stream attenuation process is represented by:

$$C = (C_0 - C_b)\, e^{-a \cdot x \cdot q - b \cdot x \cdot d} + C_0 \tag{1}$$

where C (mg/l) is the final concentration in the stream reach at end of the time step or the concentration of overland flow when it enters the nearest stream; C_0 (mg/l) is the concentration in the stream reach at the start of a time step or the concentration of the overland flow as it is leaving the source cell; C_b (mg/l) is the user-specified background concentration; a and b (both dimensionless) are user-specified attenuation exponents; q (m^3/s/ha) is the flow rate leaving the source cell during that day and d (m) is the flow distance.

The in-stream attenuation method assumes the rate of constituent exchange between the stream and the stream bottom is proportional to the concentration difference between the stream and the user-specified background concentration. Thus, in-stream attenuation is better correlated with the wetted perimeter of a reach and residence time within the reach (Eq. 2). The in-stream attenuation algorithm is applied at each time step, with the concentration C_t at time t a function of the background concentration C_{bg}, the concentration at the previous time step C_{t-1}, the hydraulic radius R, the time step Δt and the coefficient a is given by:

$$C_t = C_{bg} + (C_{t-1} - C_{bg}) \cdot \exp\left(\frac{a\Delta t}{R}\right) \tag{2}$$

2.4 Model Calibration and Validation

To evaluate the model's ability to represent real-world conditions, four statistical metrics were used: the Nash-Sutcliffe Efficiency (NSE), Root Mean Squared Error (MSE), Kling-Gupta Efficiency (KGE), and percentage of bias. These are commonly used goodness-of-fit measures in hydrology to assess the accuracy of hydrological and water quality models by comparing simulated and observed data, considering factors like bias, correlation, and variability (Althoff and Rodrigues 2021). NSE directly measures how well-simulated values match observed values, with one (1) being a perfect fit and zero (0) indicating no better than the mean (Eq. 3). A NSE > 0.5 for flow and > 0.35 for TN/TP is generally considered satisfactory (Moriasi et al. 2015). MSE measures the average squared difference between simulated and observed values, providing a quantitative measure of overall error (Eq. 4). Predictors closer to zero indicate higher accuracy. KGE breaks down MSE into components reflecting bias, variability, and correlation, offering a comprehensive assessment of model performance (Eq. 5). The percentage of bias calculates the relative difference between the means of simulated and observed values, indicating systematic over/or underestimation (Eq. 6). A PBAIS within ± 15% for runoff and ± 30% for TN and TP at a daily scale is often considered acceptable (Althoff and Rodrigues 2021; Moriasi et al. 2015). The detailed calibration and validation procedures of WAM are provided in the supplemental material Table SM2.

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n} (O_i - S_i)^2}{\sum_{i=1}^{n} (O_i - \bar{O})^2} \right] \tag{3}$$

$$RMSE = \frac{1}{n} \sqrt{\sum_{i=1}^{n} (O_i - S_i)^2} \tag{4}$$

$$KGE = 1 - \sqrt{(r-1)^2 + (\alpha - 1)^2 + (\beta - 1)^2} \tag{5}$$

$$\mathrm{PBIAS} = \frac{\sum_{i=1}^{n} 100 (O_i - S_i)}{\sum_{i=1}^{n} O_i} \tag{6}$$

Where n represents the number of simulated or measured data; i represent the sequence of the simulated or measured data; O_i and S_i represent the i^{th} observed and simulated values, $\bar{O}$and $\bar{S}$ are the average of the observed and simulated values, r is the Pearson correlation coefficient between observed and simulated values, α is the variability ratio (standard deviation of simulated values / standard deviation of observed values), and β is the bias ratio (mean of simulated values/mean of observed values).

As the rainfall-runoff process generates nutrient yields, calibration and validation were done first for runoff, followed by TN and TP loads. Stream flows were assessed at the United States Geological Survey (USGS) gage locations in the domain, and water quality results were assessed at water quality sampling stations of the Impaired Waters Rule (IWR) and/or Watershed Information Network (WIN). Calibration and validation stations were chosen based on data reliability and watershed coverage (Fig. 1 for the locations). The standard calibration process ensures that the model's parameters accurately represent the physical processes involved in the simulation at each major step. This process is divided into three main parts: source cell nutrient load and flow generation, cell-to-stream routing, and in-stream routing, as detailed in the supplemental material Table SM2.

2.5 Statistical Methods

Daily loads and flows were organized and analyzed using Microsoft Excel 365. Descriptive statistics were determined, and spatial analysis, including zonal statistics, was conducted using ArcGIS 10.8.2. Calibration and validation model performance metrics goodness-of-fit visualized/plotted using R Studio. Spatial and temporal distributions, as well as hotspot maps, were generated from the WAM model output and were reorganized and mapped using ArcMap 10.8.2.

3 Results

3.1 Model Calibration and Validation Results

Figure 4; Table 2 show comparisons between the measured flow data (black dotted lines) and the flow values simulated using WAM (solid red line) over the 10-year calibration period (Fig. 4). The model simulations were conducted using historical data from 1996 to 2023. The first three years of the source-cell model output were used as a spin-up period to ensure the model's internal processes reached a steady state before analyzing the results. Consequently, the overland and stream routing model was run from 1999 through the end of 2023 for a total of 25 years. Simulation results were calibrated at multiple watershed outlet stations leveled from Q1-Q7 for flow and WQ1 to WQ4 for TN and TP (Fig. 1). The goodness of fit values at the stations along the mainstem of the Peace River and main tributaries

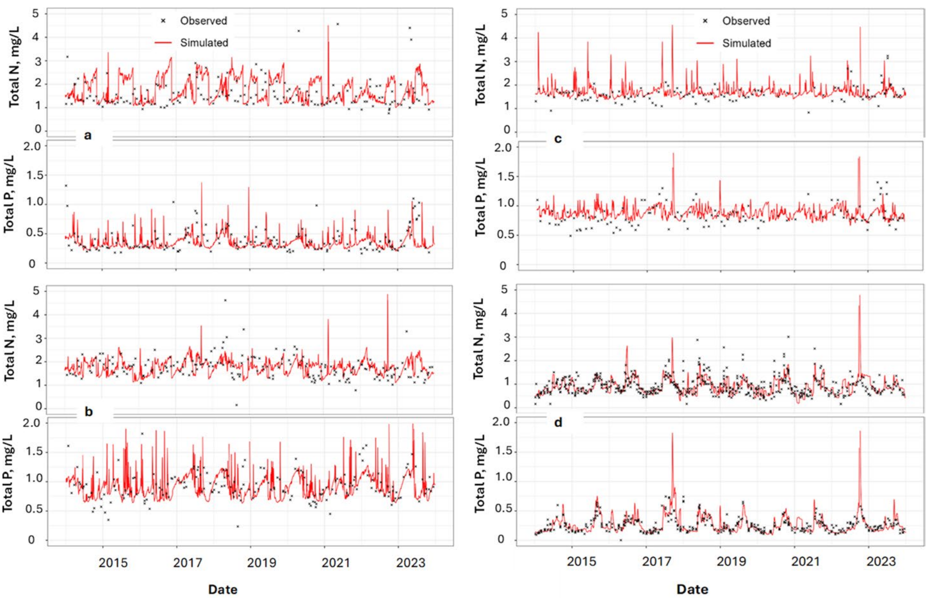

Fig. 4 Comparison of measured and simulated TN and TP concentrations along the main stem of the Peace River (Fig. 1 shows the location of the stations) (**a**) Peace River at Bartow (WQ4), (**b**) Peace River at Arcadia (WQ2), (**c**) Peace River at county line Rd (WQ3) and (**d**) Peace River Estuary Lower Segment, the watershed outlet (WQ1)

Table 2 Performance evaluation statistics over the 2014–2023 calibration period for flow, TN and TP

Flow at seven selected USGS gage locations along the main stem of the Peace River and main tributaries						
Gage location			NSE	RSME	KGE	% Bias
Q1(Peace River at Arcadia, 2296750)			0.90	19.6	0.93	4.1
Q2 (Peace River at Zolfo Springs, 2295637)			0.86	10.17	0.91	2.3
Q3 (Peace River at Bartow, 2294655)			0.79	4.58	0.77	2.5
Q4 (Shell Creek near Punta Gorda, 2298202)			0.78	8.60	0.86	5.1
Q5 (Joshua Creek at Nocatee, 2297100)			0.73	3.89	0.83	4.3
Q6 (Horse Creek near Arcadia, 2297310)			0.75	6.49	0.87	4.4
Q7 (Charlie Creek near Gardner, 2296500)			0.83	7.82	0.90	3
Nutrients at four water quality sampling locations along the main stem of the Peace River						
Gage location	Nutrient	Observed Mean	Simulated Mean	Number of observations	RMSE	% Bias
Peace River Estuary Lower Segment (WQ1)	TN	0.93	0.95	333	0.34	2
	TP	0.27	0.27	206	0.11	1.6
Peace River at Arcadia (WQ2)	TN	1.62	1.68	109	0.42	3.6
	TP	0.88	0.88	109	0.60	-0.1
Peace River at county line Rd (WQ3)	TN	1.75	1.75	177	0.55	-0.1
	TP	0.94	0.92	180	0.28	-1.8
Peace River at Bartow (WQ4)	TN	1.75	1.80	161	1.04	2.6
	TP	0.42	0.34	163	0.30	-19

showed an excellent fits over the calibration period with NSE values ranging from 0.73 to 0.9, root-mean-square error (RMSE), Kling-Gupta Efficiency (KGE), and percent bias (% bias) are within the acceptable limits (Table 2 and Fig. SM1 in the supplemental material).

The calibration period used for TN and TP was the same as the flow calibration, i.e., calibration over the 10-year period from the beginning of 2014 through the end of 2023. Observed and simulated results over the calibration period at four locations distributed along the main stem of the Peace River are shown below (Fig. 4). These four locations are arranged from the upper end of the Peace River (at State Road 60 at Bartow) to the watershed outlet (the Peace River Estuary lower segment) where it discharges into Charlotte Harbor. Given the limited observations at these stations, goodness-of-fit values reasonably agreed with measurements (Table 2). However, simulated results showed extreme values during hurricanes, indicating substantial nutrient generation and transport to the Peace River water. While hurricane-induced flow agreed with simulated values (Fig. SM1), the lack of water quality samples during these periods prevents comparison. It is likely, though, that nutrient levels were elevated during hurricanes, as indicated in our simulations.

3.2 Nutrient Loading from Source Cells

3.2.1 Surface Loading

Figure 5 indicates a long-term average of total nitrogen and total phosphorus loading of PRW. The annual averages were calculated over the full modeling period, from 1999 to 2023. The darkest blue/blue represents those cells with the lowest estimated nutrient loading values, followed by the green representing intermediate estimated nutrient loads, and increasing until the red/darkest red represents the upper estimated loads (Fig. 5a, b). These estimated values represent attenuated loadings from each cell to the destination reach in the stream network. Thus, the estimated flows and loads shown take into account overland attenuation processes between the source cell and the destination stream reach.

Spatially, estimated annual TN varies from near zero to about 116 kg/ha, while TP ranges from near zero to 38 kg/ha. Estimated loads of both TN and TP on open water (e.g., lakes) represent atmospheric loading, calculated by multiplying the rainfall volume by 0.2 and 0.007 mg/L for TN and TP, respectively. TN was the highest in urban areas (Table 3) particularly urban areas that rely on septic waste treatment systems, including the upper PRW and the lower PRW in Charlotte County (Fig. 5a; Table 3). TN was also greater in agricultural areas that utilize fertilizer, such as row crops, nurseries, or citrus that have not implemented BMPs (Table SM1 in the supplemental material). Some intensive agricultural lands produced very high nitrogen loads but were of a spatially limited extent (Table SM1).

Phosphorus was highest on phosphate mining lands, in particular, on non-mandatory reclaimed lands, followed by agricultural areas (Table 3; Fig. 5b). While the area was small, dairy outer pasture, cattle feeding operations and peach and pecan orchards produced the greatest TP loading. Citrus and improved pasture with and without fertility BMP have also greater TP loading (Table SM1). Urban areas, wetlands, forest and brush areas have the lowest TP loading (Table 3).

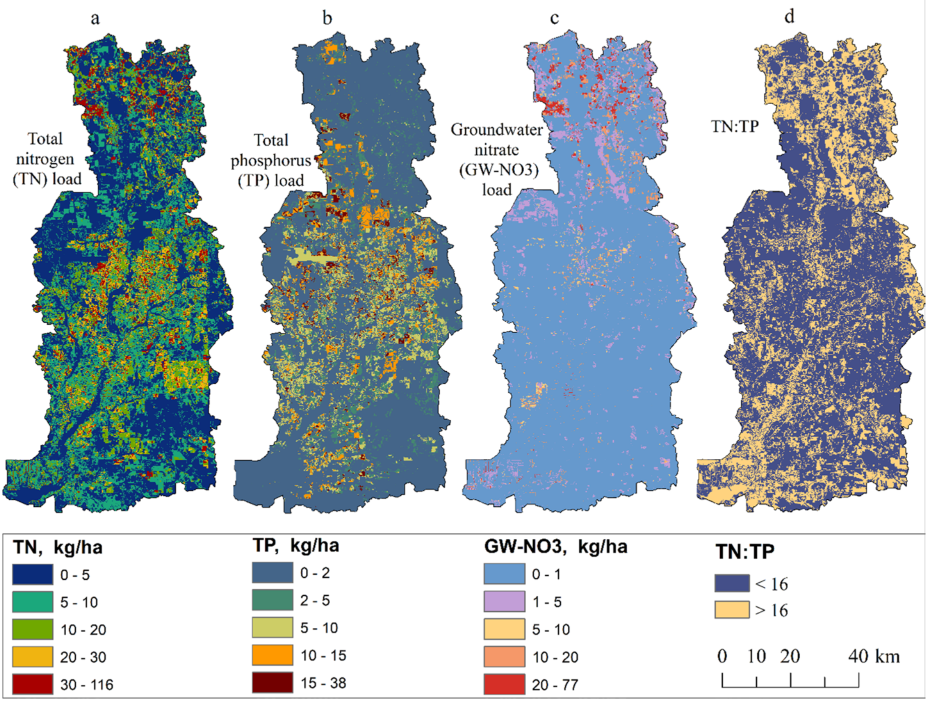

Fig. 5 Estimated annual average nutrient loading from source cells to destination reaches of PRW (**a**) total nitrogen (TN), (**b**) total phosphorus (TP), (**c**) groundwater nitrates, and (**d**) Nitrogen to phosphorus ratio (TN: TP)

Table 3 Land use type with area coverage, annual TN, groundwater nitrate (GW NO3) and TP loading of PRW

Land use category	Hectares	% of Area	TN, kg/ha	GW (NO3), kg/ha	TP, kg/ha	TN: TP
Urban	80,973	13.3	15.4	6.1	0.5	29.4
Agriculture	222,118	36.5	12.4	1.8	4.5	2.7
Mining	28,996	4.8	2.4	1	6.6	0.4
Reclaimed Mining Lands	36,629	6	3.8	0.8	4.6	0.8
Wetland (Reclaimed)	4,840	0.8	3.2	0.5	1.4	2.3
Forest and Brush	93,346	15.3	5.3	0.4	0.4	12
Wetland	113,999	18.7	4	0.3	0.6	6.4
Open Water	27,982	4.6	2.3	0.5	0.2	9.5
Average			6.1	1.4	2.4	7.9

Total nitrogen concentrations along the main stem showed a decreasing trend from the upper PRW,1.80 mg/L, to the lower PRW, 0.95 mg/L (Table 2). Whereas, total phosphorus concentration peaked at stations WQ2 & WQ3, the middle part of the watershed (Table 2). Both nitrogen and phosphorus showed minimum concentration at the lower part of the PRW.

3.2.2 Groundwater Nitrate Loading

The groundwater nitrate was elevated in the urban areas associated with septic systems (Table 3). Spatially, the estimated annual GW-NO3 varies from near zero to about 77 kg/ha (Fig. 5c). The upper PRW showed the greatest nitrogen loading (Fig. 5c). In addition, urban areas in Hardee, Desoto and Charlotte counties indicated elevated concentrations, particularly the upper PRW showed a hotspot area for GW-NO3. Medium-density residential areas were the highest contributor of nitrates, followed by high-density residential areas (Table SM1). Additionally, reclaimed mining land, dairies, row crops and tree nurseries showed elevated values next to the urban areas (Table SM1). Wetlands, mining, forest and brush areas have the lowest nitrate loading to the groundwater (Table 3).

3.2.3 Estimated Nutrient Loading at Gauge Locations and Sub-Basin Outflows

Simulation results presented in Table 4 provide estimated annual average water volume and nutrient loads calculated from WAM simulation output at gage locations (gauge locations are shown in Fig. 1). In this table, the estimated volumes and loads at each location on the PRW include all of the aggregated upstream contributions, e.g., the discharge at the downstream Payne Creek basin (Q2 Peace River at Zolfo Springs, 2295637 location) includes the inflow from the Upper Peace River basin. Therefore, the total annual average outflow of 1.9 cubic kilometers of water, 2,370 tons of nitrogen and 1,025 tons of phosphorus was discharged to Charlotte Harbor (Table 4). In comparison, the upper part of PRW has the greatest nitrogen and hence the highest nitrogen to phosphorus ratio.

In contrast, Table 5 shows the estimated volume of flows and loads at the outlet of each sub-basin (sub-basins are shown in Fig. 1) with upstream contributions removed. For example, the estimated discharge at the Payne Creek basin outlet was calculated using the value in Table 4 and subtracting the contribution from the Upper PRW. The flows and loads for each sub-basin were calculated similarly by deducting the contribution from any upstream contributing sub-basin. The result reveals, the Upper Charlie Creek, Payne Creek, and Trou-

Table 4 Estimated water discharge, TN, and TP loads at selected monitoring stations

Location	Volume (Mm3/y)	TN (ton/y)	TP (ton/y)	TN: TP
Q1 (Peace River at Arcadia, 2296750)	996	1905	1089	1.8
Q2 (Peace River at Zolfo Springs, 2295637)	551	1028	565	1.8
Q3 (Peace River at Bartow, 2294655)	201	405	68	5.9
Q4 (Shell Creek near Punta Gorda, 2298202)	300	174	48	3.6
Q5 (Joshua Creek at Nocatee, 2297100)	1,352	2331	1343	1.7
Q6 (Horse Creek near Arcadia, 2297310)	191	260	171	1.5
Q7 (Charlie Creek near Gardner, 2296500)	262	549	325	1.7
WQ1 (Peace River Estuary Lower Segment)	1,882	2370	1025	2.3

Table 5 Estimated water discharge, TN, and TP loads for each sub-basin

Sub-basin name	Area	Volume	TN	TP	flow	TN	TP	TN: TP
	Hectares	(Mm3/y)	(ton/y)	(ton/y)	(cm/y)	(kg/ha)	(kg/ha)	Ratio
Upper Peace River	165,988	342.1	632	351	20.6	3.8	2.1	1.8
Payne Creek	55,795	208.4	396	214	37.3	7.1	3.8	1.9
Upper Charlie Creek	45,744	129.1	330	153	28.2	7.2	3.4	2.1
Lower Charlie Creek	41,404	133.2	220	171	32.3	5.3	4.1	1.3
Horse Creek	59,297	191.2	260	171	32.3	4.4	2.9	1.5
Long Island Marsh	64,249	213.6	263	51	33.3	4.1	0.8	5.3
Troublesome Creek	36,633	116.7	254	140	31.8	6.9	3.8	1.8
Joshua Creek	70,419	231.4	237	142	32.8	3.4	2	1.7
Charlotte Harbor	70,856	316.4	-223	-368	44.7	-3.1	-5.2	0.6

blesome Creek sub-basins exhibited the highest TN loading per acreage, while Lower Charlie Creek, Troublesome Creek and Payne Creek sub-basins exhibited the highest TP loading (Table 4). Conversely, Charlotte Harbor sub-basin, Upper Peace River, Joshua Creek, and Long Island Marsh sub-basins displayed the lowest TN loading per acreage, and Charlotte Harbor sub-basin, Long Island Marsh and Upper Peace River exhibited the lowest TP loading per hectare (Table 5). The TN and TP load differences in Charlotte Harbor are negative, indicating outflow TN and TP loads from the basin are lower than the inflow loads.

Considering the total area of the PRW (the sum of all areas in Table 5), the overall area-averaged nutrient loading, calculated from the total load at the watershed outlet (WQ1 station in Table 4), is 3.9 kg/ha of TN and 1.7 kg/ha of TP.

3.2.4 Nitrogen To Phosphorus Ratio

The estimated nitrogen-to-phosphorus ratios at the gauge locations and sub-basin outlets ranged from 1.5 to 5.9 and 0.6 to 5.3, respectively, with an average of 2.3 at the watershed outlet (Table 4). Long Island Marsh sub-basin had the greatest estimated TN: TP ratio,

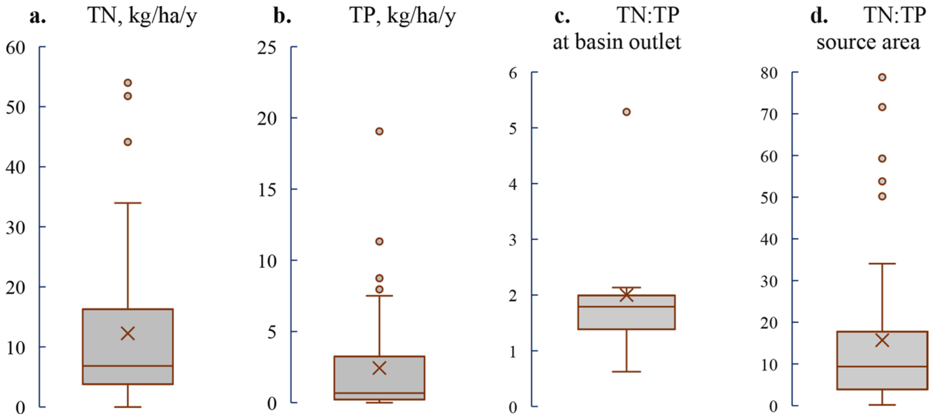

Fig. 6 Box plot of average nutrient loading at the source cells of Peace River watershed by land use category (**a**) total nitrogen (**b**) total phosphorus (**c**) nitrogen to phosphorus ratio at sub-basin outlets and (**d**) total nitrogen to total phosphorus ratio from the source cells

which was almost threefold that of the other sub-basins (Table 5). The Upper Charlie Creek, Payne Creek, Upper Peace River and Troublesome Creek sub-basins followed the Long Island Marsh sub-basin. Charlotte Harbor sub-basin had the lowest, followed by Lower Charlie Creek and Horse Creek.

The upper part of PRW (just the area upstream of the Q3 station, not the entire sub-basin) has the greatest nitrogen and hence the highest nitrogen to phosphorus ratio (5.9), while the overall ratio at the outlet is 2.3 (Table 4; Fig. 1 shows the location of WQ1). The TN: TP ratios at the source cell level had an average of 15.7 and ranged from 0.2 to 29.4 (Table 3). About 33% (73,922 hectares) of the source cells had an estimated TN: TP ratio greater than 16, while the remaining had an TN: TP ratio less than 16 (Fig. 5d). The urban residential areas showed the highest ratio (Table 3; Fig. 5d) followed by row crops (with and without BMPs), sod farming, mixed wetland hardwoods, hardwood conifer mixed, coniferous plantations, scrub and brushland, and unimproved pastures (Table SM1). Conversely, all phosphate mining areas, cypress, dairies, field crops, freshwater marshes, hardwood conifer mixed, hardwoods, and improved pasture (with and without BMPs) indicated the lowest TN: TP ratio in the source cells (Table SM1).

The average TN and TP values exhibited an upward skew with outliers (Fig. 6a & b). Conversely, the TN: TP ratio at the basins displayed a downward skew with one outlier value (Fig. 6b). The TN: TP ratio at the source cells followed a similar pattern to the distribution of TN and TP (Fig. 6d). All plots indicated the availability of hotspots for nitrogen and phosphorus loading with in the watershed.

4 Discussion

4.1 Nutrient Loading in the Watershed

The simulated flow and nutrient loads of the watershed's source cells and reaches provide information on the hydrological and water quality characteristics of the PRW and Charlotte

Harbor. The simulated nutrient loading at the source cell level in the PRW had an annual average of 6.1 kg/ha of TN, 1.4 kg/ha of GW-NO3, and 2.4 kg/ha of TP (Table 3) compared to the basin average loading of 3.9 kg/ha of TN and 1.7 kg/ha of TP at the outlet of the watershed (Table 4). TN was highest in urban areas on septic systems and in areas that allow landscape fertilization, such as in the Upper PRW (Table 3, Table SM1 and Fig. 5a). TN also tends to be higher in agricultural areas that utilize fertilizer, such as row crops, nurseries, and citrus that have not implemented BMPs (Table 3, Table SM1 and Fig. 5a). Estimated TP loads were highest on mining lands, and in particular non-mandatory reclaimed lands (for mining practices before 1975) (Table 3). Phosphate mining activities in the PRW have led to land disturbance, reduced water quantity, and elevated levels of TP in the river water (Zhang et al. 2022a, b).

Agricultural land uses in the PRW, which is increasing over time (Garcia et al. 2020), are dominated by pasture lands and citrus, including row crops, nurseries, dairies, and sod farms (Lewelling et al. 1998). These agricultural land uses may generate significant amounts of TN and TP (Wester et al. 2023). Agricultural BMPs play a key role in reducing nutrient loading (Tarabih et al. 2024), and it is evidenced in our simulation too (Table SM1). Scrub and brushland have the largest acreage of the native land uses (Fig. 2b), followed by several wetland categories that are more conducive to water quality due to their natural filtration capabilities (Fisher and Acreman 2004). Those wetlands dominate along the course of the Peace River and other tributary channels such as Payne Creek and Charlie Creek and act to buffer pollutants (Fig. 1). Open water has the largest native land cover acreage in the Upper Peace River and Charlotte Harbor sub-basins and facilitates the retention of flows and loads that trigger the percolation of nutrients to the subsurface level, particularly in the upper part of PRW (Lewelling et al. 1998; Metz and Lewelling 2009). Consequently, the greatest concentration of TN and GW-NO3 was observed in the upper PRW compared to the decreased concentration in the lower part of the watershed (Tables 1 and 3).

The watershed characteristics, such as land use, vary from basin to basin and hence have varying nutrient contributions to the Peace River. The Upper Peace River and Payne Creek basins have significant reclaimed mining land and active mining areas, thus generating a high TP load from those lands compared to other basins. However, Payne Creek has little urban development and thus the highest estimated TN loads are produced by pasturelands and citrus, which combined have nearly the acreage of reclaimed lands (Table 3). Mining land extends to the northernmost part of the Horse Creek basin, adjacent to the mining areas in the westernmost part of the Payne Creek basin (Fig. 1). Reclaimed mining land in the remaining basins covers only minor amounts of land. As a result, TP concentration is greater at WQ2 and WQ3 stations, which are relatively located close to these mining areas (Table 2; Figs. 1, 2 and 5).

Urban land use showed greater TN and GW-NO3 loading from the source cells (Table 3; Fig. 5). These elevated values are likely due to septic systems, which are a critical challenge in Florida (Badruzzaman et al. 2012; Lapointe et al. 2017). Septic systems, while generally effective, can pose risks to water quality over time (Brewton et al. 2022). The associated risk is that when the septic tank is damaged or improperly installed, it can leak wastewater into the surrounding soil, contaminating groundwater which is a primary source of drinking water for many people (Zhu et al. 2016). Overloading a septic system can also lead to the discharge of untreated or partially treated wastewater into the environment, again potentially contaminating nearby water bodies (Badruzzaman et al. 2012). While regular maintenance is crucial for a septic system's proper functioning, neglecting inspections and pumping can lead to clogs or damage, compromising water quality (Brewton et al. 2022).

The lower nitrogen to phosphorus ratio (2.3:1) at the PRW outlet is an indication that nitrogen is a limiting constituent for the Harbour's HABs, which is similar to most of the coastal watersheds in Florida (Howarth et al. 2021; Redfield 1958). However, a relatively higher TN: TP ratio is shown at the source cell level than at the outlet of the PRW and sub-basins (Fig. 6). The reason for this is that nitrogen loading into receiving water bodies is influenced by attenuation processes due to nitrification and flow to groundwater within the soil while the phosphorus is often bound to soil particles, making it less likely to leach (Nevins et al. 2020). The extent of this attenuation in the PRW varies widely due to factors such as soil drainage, organic matter content, and water table depth (Nevins et al. 2020; SWET 2018). Well-drained sandy soils in the Upper PRW (Figs. 1 and 2) are generally well-aerated, promoting nitrification of ammonium (Metz and Lewelling 2009). However, these soils often lack sufficient organic matter for denitrification, limiting their ability to reduce nitrate loading to groundwater. As a result, a significant amount of nitrate moves through soil into groundwater resulting in higher nitrate loading to groundwater in the areas underlain by well-drained soils in the upper PRW (Fig. 5). In addition, moderately well-drained soils may exhibit some denitrification if a fluctuating water table creates anoxic conditions. Poorly drained soils with high organic matter content that cover the majority of the PRW can provide a near-surface anoxic zone, ideal for denitrification, but may hinder nitrification (Hazen and Sawyer 2009). Spodosols (poorly drained), common in the PRW (Fig. 2), may have higher denitrification potential due to organic matter concentration at lower depths serving as an energy source for bacteria. Saturated zones, whether below the water table or as perched water, can restrict reaeration and promote denitrification while inhibiting nitrification (Zhu et al. 2016). A shallow, fluctuating water table can create conditions for periodic denitrification in soils like spodosols which is a very common phenomenon in the PRW (Hazen and Sawyer 2009; Zhu et al. 2016). Seasonal fluctuations allow for ammonium adsorption during wet periods and subsequent nitrification during dry periods (Zhu et al. 2016). If organic matter is present, nitrate can be denitrified when the soil becomes saturated again.

4.2 Comparison with Other Studies

While studies specifically focused on nutrient loading within the PRW remain limited, our findings offer valuable insights when compared to broader regional and international assessments. Our estimated TN load of 2,370 tons/yr at the watershed outlet (Table 4) is greater than the 1,820-1,827 tons/yr reported by the SWFWMD between 1985 and 1991 and 2009–2015 (Garcia et al. 2020). This discrepancy is likely due to the difference in methodology and the spatial extent of the analysis. The SWFWMD study by Garcia et al. (2020) relied on data from four gauged stations representing the upstream area of the watershed near Arcadia, whereas this study encompassed the entire watershed and calculated the total load at the watershed outlet. This broader scope of analysis might explain a reasonable increase in the estimated nitrogen loading compared to the SWFWMD study, as their analysis may have excluded a significant portion of the nitrogen load contributed during hurricane events, which were accounted for in this study. Furthermore, the higher estimates in this study could also be due to increased human-induced impacts in recent years across agricultural, mining, and urban lands (Garcia et al. 2020; J. Zhang et al. 2022a, b).

In comparing the nutrient loading rates per unit area, our estimated average annual TN load for the PRW (calculated as 3.9 kg/ha based on the total watershed area) is slightly lower than the 2012 USGS SPARROW model's statewide average of 4.5 kg/ha for TN (US EPA 2013). However, our TP estimates for the PRW (1.7 kg/ha) appear higher than the USGS SPARROW model's statewide TP estimates of 0.6 kg/ha (US EPA 2013). The average TN load for the adjacent Myakka River basin, which also discharges to Charlotte Harbor, was reported between 3.1 and 4.1 kg/ha and TP loading between 1.4 and 1.7 kg/ha (Garcia et al. 2020), placing our PRW estimates within a comparable range for both nutrients.

In contrast, simulated nutrient loading in PRW appears lower than observed in the Lake Okeechobee Watershed (LOW) in Florida (Tarabih et al. 24; Faridmarandi et al. 2021; Khare et al. 2021). Faridmarandi et al. (2021) reported a long-term average annual nitrogen inflow loading to Lake Okeechobee, 6.8 kg/ha over 45 years (1974–2018), which is greater than our estimated watershed average of 3.9 kg/ha for TN, in PRW. Similarly, Tarabih et al. (24) and Faridmarandi et al. (2021) used a 1995–2018 data range, 24 years and found a maximum of 4.9 kg/ha, still suggesting a lower nitrogen loading in the PRW based on our findings. The highest average TP load reported in the LOW was 1.33 kg/ha (Faridmarandi et al. 2021; Khare et al. 2021), which is lower than our estimated TP loading in the PRW (1.7 kg/ha).

Our nutrient loading estimates in the PRW also appear lower than those reported by Aboelnour et al. (2025) using the SWAT model in two sub-watersheds in New York. The study in the Niles and Paw Paw sub-watersheds (11,064 and 1112 km², respectively) showed significantly higher annual average TN and TP loadings at the source level (106 and 79 kg/ha, respectively for TN and 106 and 79 kg/ha for TP) compared to our watershed-scale estimates for the PRW. Furthermore, when comparing to a European marine region, our TN estimates for the PRW are smaller, while our TP estimates are greater than those reported by Vigiak et al. (2023). The application of the GREEN model (R open source GREENeR) for a 5.7 Mkm² area showed average TN estimates of 8.4 kg/ha and TP estimates of approximately 1 kg/ha for the period of 1990 to 2018.

These comparisons highlight the regional variability in nutrient loading and underscore the importance of considering watershed-specific characteristics, methodological approaches, and temporal scales when interpreting and comparing nutrient loading estimates. While direct comparisons with limited PRW-specific studies suggest a potentially higher TN load in our analysis due to broader spatial coverage and inclusion of extreme events, comparisons with larger regional and international studies provide a broader context for understanding the relative magnitude of nutrient loading in the PRW.

4.3 Implications of Algae and Red Tide Blooming, Recommendations

The Peace River Watershed (PRW) presents a significant environmental challenge due to excess nutrient loading, primarily nitrogen and phosphorus to Charlotte Harbor (Metz and Lewelling 2009). These nutrients discharged into Charlotte Harbor lead to the growth of algae and red tides, negatively impacting the ecosystem and human health (FDEP 2007; Hammett 1990; Lewelling et al. 1998). Annually the PRW discharges and estimated 2,370 tons of nitrogen and 1,025 tons of phosphorus into Charlotte Harbor. This excessive nutrient load is primarily generated from urbanization, agricultural intensification, and phosphate mining activities within the watershed (Badruzzaman et al. 2012), fueling the growth of HABs in the harbor (Garcia et al. 2020; Kim et al. 2010; Medina et al. 2025).

Based on the Redfield ratio (TN: TP = 16:1), which represents the optimal nutrient balance for phytoplankton growth in the ocean, the current TN: TP ratio at the watershed's outlet is 2.3 indicating that nitrogen is the limiting nutrient for the growth of HABs in Charlotte Harbor (Redfield 1958). However, about 33% of the PRW have TN: TP greater than 16, indicating phosphorus is limiting at a significant amount of source lands (Table 3, Table SM1 and Fig. 5d). Therefore, a substantial amount of nitrogen is leached into the groundwater, particularly from urbanized areas in the upper PRW (Fig. 5). While the transport of groundwater nitrates in Florida is a complex issue with significant environmental implications, those nitrates may eventually make their way to the coastal water of Charlotte Harbor triggering HABs.

Developing effective management strategies for Charlotte Harbor requires an understanding of the sources and spatial variations of nitrogen inputs within the watershed. Thus, effective implementation of management strategies to reduce nitrogen inputs is crucial for improving water quality by controlling HABs (Medina et al. 2025). According to the SWFWMD study, a five-year annual total nitrogen load of 1,800 tons is suggested as a targeted strategy, which requires targeted mitigation interventions (Garcia et al. 2020). Reduction strategies for nitrogen in hotspot areas, mainly urban and agricultural land uses, could lower the potential for the growth of HABs in the Harbor. However, it is not clear that much nutrient load reduction could be achieved by implementing these mitigation strategies. A planned study on scenario–hotspot–based water quality mitigation might provide the answer. Furthermore, the effect of hurricane-induced mixing and dilution of the Harbor and its implications on spreading the HABs needs further investigation.

5 Conclusions

The study explores nutrient loading within a hydrologically sensitive coastal watershed, the PRW, aiming to pinpoint critical source areas and assess the nutrient fluxes originating from various land uses. By understanding the spatial distribution of these fluxes, identifying the limiting nutrients, and the governing hydrological and nutrient processes, the study seeks to provide insights useful for effective water quality mitigation measures in coastal watersheds. The Watershed Assessment Model was applied to the PRW to simulate the hydrology and water quality of the watershed. WAM divided the watershed into grids to determine water flows and nutrient loads from each source cell and route them to the nearest reach, followed by instream routing to the watershed outlet. Model performance was assessed by comparing measured and predicted flows and loads at several monitoring locations. The results showed that estimated annual TN loads at the source cell level varied spatially from near zero to about 116 kg/ha, while TP was lower, ranging from near zero to a high of about 38 kg/ha. TN was highest in urban areas, particularly urban areas relying on septic waste treatment systems, including the upper PRW and the lower PRW in Charlotte County. TN was also greater in agricultural areas utilizing higher fertilization rates, such as row crops, nurseries, or citrus, that had not implemented BMPs. Similarly, groundwater nitrate was elevated in urban areas associated with septic systems. Spatially, the upper PRW showed the greatest nitrogen loading. TP was highest on mining lands, particularly non-mandatory reclaimed lands, followed by agricultural areas. The TN: TP ratio at the watershed outlet was 2.3, indicating nitrogen as the limiting nutrient based on the Redfield ratio, suggesting that control of Harbor's HABs should focus on nitrogen reduction. However, the TN: TP ratio in the source cell averaged 15.7 and ranged from 0.2 to 29.4, indicating that certain areas are also phosphorus-limiting. The nitrogen reduction at the outlet is attributed to the attenuation processes, such as denitrification and nitrate percolation to the subsurface level. Compared to conventional methods, the model output provides a cost-effective way to assess nutrient loading across the entire watershed, potentially saving significant resources that could be spent on collecting nutrient concentrations in several locations. This study provides a foundation for water quality managers to assess future research needs and evaluate potential water quality changes resulting from upcoming hydrological alterations. These findings are critical for water quality managers to make informed decisions regarding targeted mitigation strategies for current and future watershed developments.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s40710-025-00782-2.

Acknowledgements The authors would like to gratefully acknowledge the SAS team members for their follow-ups on the overall Peace River Watershed project. We would also like to acknowledge the technical guidance, assistance, and quality control support provided by FDEP.

Author Contributions M.K. Contributed to the initial draft, methodology, data analysis and interpretation, and incorporated revisions based on comments and suggestions from co-authors. S.T. significantly contributed to conceptualization, methodology, manuscript revisions, and project administration. A.J. contributed to data collection, analysis, methodology, writing the initial project report, reviewing and editing. A.B. and K.K.A. contributed to the methodology and manuscript review. R.R. contributed to the project conceptualization, project supervision, writing review and editing, and overall guidance. All authors reviewed and approved the final manuscript.

Funding The research was funded by a grant from the Florida Department of Environmental Protection (FDEP) Grant GR00514T8 awarded through the legislative fund to support improving the water quality of Peace River Watershed, Florida.

Data Availability No datasets were generated or analysed during the current study.

Declarations

Competing Interests The authors declare no competing interests.

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